The Influence of Urban Soil Rehabilitation on Soil Carbon Dynamics, Greenhouse Gas Emission, and Stormwater Mitigation

by

Yujuan Chen

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Susan D. Day, Chair
W. Lee Daniels
Kevin J. McGuire
Brian D. Strahm
P. Eric Wiseman

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Abstract

Global urbanization has resulted in rapidly increased urban land. Soils are the foundation that supports plant growth and human activities in urban areas. Furthermore, urban soils have potential to provide a carbon sink to mitigate greenhouse gas emission and climate change. However, typical urban land development practices including vegetation clearing, topsoil removal, stockpiling, compaction, grading and building result in degraded soils. In this work, we evaluated an urban soil rehabilitation technique that includes compost incorporation to a 60-cm depth via deep tillage followed by more typical topsoil replacement. Our objectives were to assess the change in soil physical characteristics, soil carbon sequestration, greenhouse gas emissions, and stormwater mitigation after both typical urban land development practices and post-development rehabilitation. We found typical urban land development practices altered soil properties dramatically including increasing bulk density, decreasing aggregation and decreasing soil permeability. In the surface soils, construction activities broke macroaggregates into smaller fractions leading to carbon loss, even in the most stable mineral-bound carbon pool. We evaluated the effects of the soil rehabilitation technique under study, profile rebuilding, on soils exposed to these typical land development practices. Profile rebuilding incorporates compost amendment and deep tillage to address subsoil compaction. In the subsurface soils, profile rebuilding increased carbon storage in available and aggregate-protected carbon pools and microbial biomass which could partially offset soil carbon loss resulting from land development.

Yet, urban soil rehabilitation increased greenhouse gas emissions while typical land development resulted in similar greenhouse gas emissions compared to undisturbed soils. Additionally, rehabilitated soils had higher saturated soil hydraulic conductivity in subsurface soils compared to other practices which could help mitigate stormwater runoff in urban areas. In our study, we found urban soil management practices can have a significant impact on urban ecosystem service provision. However, broader study integrating urban soil management practices with other ecosystem elements, such as vegetation, will help further develop effective strategies for sustainable cities.

Dedication

This dissertation is dedicated to my parents and my brothers for their love, support and encouragement.

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List of Abbreviations

C: carbon;

CH_4: methane;

CO_2: carbon dioxide;

CO_2eq: carbon dioxide equivalent;

ET: enhanced topsoil;

GHG: greenhouse gas;

GWP: global warming potential;

MBC: microbial biomass carbon;

N_2O: nitrous oxide;

PR: profile rebuilding;

SOC: soil organic carbon;

SOM: soil organic matter;

TOC: total organic carbon;

TP: typical urban land development practice;

UN: undisturbed soil.

Chapter 1
Introduction

1.1 Background

More than half of the world's population lives in urban areas and the number of urban residents continues to grow, accompanied by rapid and ongoing conversion of land to urban uses (Seto et al., 2012), and the consequent degradation of urban soils (Craul, 1992, 1994; Jim, 1993, 1998a, b). Urban sprawl has resulted in vegetation removal, increased grey infrastructure (conventional piped drainage and water treatment systems, i.e., pipes, tanks) and impervious surface, and severely altered soils due in large part to grading and compaction. In consequence, urbanization leads to impaired urban ecosystem services and functions, e.g., elevated atmospheric carbon dioxide (CO_2), heat islands, and stormwater. Thus, there is a need to evaluate whether degraded urban soils can be rehabilitated via management practices to restore some of these urban ecosystem services. Soil plays a significant role in carbon (C) sequestration (Batjes, 1998; Lal, 2004) and urban ecosystem have potential to store more C than other ecosystems (Edmondson et al., 2012). Thus, optimizing C sequestration in urban soil could be very beneficial since global urban land cover is projected to reach 1.4% in the next decades (Seto et al., 2012). This is especially of interest if this optimization also results in enhanced provision of other ecosystem services such as stormwater mitigation. Techniques that incorporate compost into subsoils via deep tilling and focus on addressing subsoil compaction may be particularly appropriate for soil rehabilitation in urban areas where subsoil compaction is prevalent. Urban soil rehabilitation offers the potential to store more C in urban soils while simultaneously improving soil conditions and, in time, providing better ecosystem services. However, research about urban soil is limited, and most studies focus on agricultural soil. Few studies have documented urban soil problems and landuse conversion effects, and little is known about how to address these urban soil issues. This research proposes to examine how urban soil rehabilitation affects the capacity of urban soils for C storage, water transmission, and greenhouse gas (GHG) emission.

1.2 Literature Review

1.2.1 Increase in urbanized land

By the end of 2011, over half of the world's population was living in urban areas. The world's urban population is projected to increase from 3.6 billion in 2011 to 6.3 billion in 2050 (United Nations, 2011). Rapid urbanization has resulted in significant land use change. Rural and forested lands have been converted to urbanized land that is characterized by impervious surfaces and damaged soil structure. Urban land cover will increase by 1.2 million km^2 by 2030, which will nearly triple the global urban land area extant in 2000 (Seto et al., 2012). Because of the dramatic increase in urbanized land, the role of urban lands at both regional and international levels is of increasing interest.

1.2.2 What are typical urban land development practices?

Vegetation clearing, topsoil removal and replacement, soil stockpiling, grading, compacting, and building are typical components of land development for urban uses.

Construction activities and associated human-derived artifacts also have potential to affect urban soil forming processes (Effland and Pouyat, 1997). Furthermore, landforms including soil surfaces and slopes are massively modified by building construction activities, for example, dumping, filling, spilling, digging, and planting (Pickett and Cadenasso, 2009). As a result, urban soils are typically highly compacted and severely modified by human activities (Craul, 1992; Effland and Pouyat, 1997).

1.2.3 Typical urban land development practices lead to degraded urban soils

Broadly defined, urban soil includes both disturbed and undisturbed soils in urban areas. While undisturbed urban soils are influenced by their urban location, disturbed urban soils are directly, and sometimes intensely, manipulated by human activities related to a variety of urban land development practices. Typical urban land development practices lead to degraded urban soils with low vegetative cover, high bulk densities, low infiltration rates, disrupted aggregation, and disturbed C cycles (Craul, 1992; Jim, 1998a, b; Kaye et al., 2006; Woltemade, 2010).

Urban soils are typically highly compacted. In an early study of the soils at the Mall in Washington DC, Short et al. (1986) found mean bulk densities of 1.61 and 1.74 Mg m^{-3} at the soil surface and at 0.3 m soil depth which demonstrated compaction occurring in urban areas. Other studies conducted in Hong Kong (Jim, 1993, 1998a, b) reported compacted urban soils along streets. However, soil bulk density did not consistently decrease along an urban-rural gradient in New York City due to the high variation among landuses (Pouyat, 2002). The age of the urban land development is another factor that can affect soil bulk density, as newer urban soils in the United States have been found to be more compacted than older urban soils (Park et al., 2010; Scharenbroch et al., 2005).

Typical urban land development practices not only influence soil compaction, but also affect soil aggregation and structure. In highly disturbed roadside soils in Hong Kong, the proportion of aggregates was very low (Jim, 1998a). However, soil structure and aggregation are critical to soil organic matter (SOM). Aggregates directly influence SOM by physical protection (Tisdall and Oades, 1982) and indirectly affect SOM by regulating microbes, water, oxygen, and nutrients in the soils which all can in turn influence soil C dynamics (Six et al., 2004). But different size aggregates have different effects on soil C dynamics. Microaggregates (53-250 μm) protect SOM in the long term, while macroaggregate (250-2000 μm) turnover influences the stabilization of SOM (Six et al., 2004). Soil aggregates are sensitive to management practices (Six et al., 1998). Thus urban land development practices could lead to aggregate disruption which will results in soil C loss.

1.2.4 The effects of degraded urban soils on carbon sequestration

Globally, soils store about three times more C than the world's vegetation and twice the C that is present in the atmosphere (Sheikh et al., 2009). Thus, small changes in soil organic carbon (SOC) may affect the global C budget and climate system. Terrestrial ecosystem management is a critical tool for climate regulation because terrestrial ecosystems can store large quantities of C in living vegetation and in soils which together function as a significant global C sink (Schimel, 1995).

The transformation of landscapes from non-urban to urban land uses has the potential to greatly modify soil C pools and fluxes. Physical disturbances and inputs of various materials by humans can greatly alter the amount C stored in urban soils. In the year 2000, C storage in human settlements of the conterminous United States was estimated to be 18 Pg of C which represented 10% of the total land C storage, and, in urban areas, 64% of this C was stored in soils (Churkina et al., 2010). Pouyat et al. (2006) estimated the SOC density for all urban soils in the conterminous United States to be 7.7 ± 0.2 kg m^{-2}. But SOC densities vary widely among different soil, landuse and landcover types. Whole-ecosystem studies in a range of cities are needed to better understand C storage and balance at both continental and global scales (Pataki et al., 2006). Don and Freibauer (2010) conducted a meta-analysis on land use change effects on SOC based on more than 210 publications over ten years. They found conversion of forest into different agricultural systems always caused SOC losses. From 1945 to 2007, urbanization resulted in 0.21 Pg C loss in the Southern United States (Zhang et al., 2012). Moreover, the C loss might be greater when the difference between open space and impervious surfaces is considered (Raciti et al., 2012). On the other hand, in Front Range, Colorado, Golubiewski (2006) found old urban lawns (50 years old) had higher TOC than new urban lawns(less than 10 years old) which implies that urban lawns could have accumulated soil C over time. The author also found urban lawns could store more C than local prairie or agricultural fields. However, the soil C data of grassland and cultivated land used in this study is from other literature. Moreover, the comparison is within 30 cm soil depth. Thus, further information is needed to better understand soil C dynamics over time after lawn establishment. Pouyat et al. (2006) observed that soils of residential lawns appear to have the highest density of C in urban landscapes. Age of land development can also influence soil C storage. Sun et al. (2010) examined the content and densities of SOC in urban soil in different landuse types in Kaifeng, China, and found new urban areas had the greatest amount of SOC, followed by old urban areas, and finally suburbs. In the United States, research on residential landscapes in Idaho and Washington State (Scharenbroch et al., 2005) and in school yards in Ohio (Park et al., 2010) both with silt loam soils found that soil C storage typically increases with age since development. This might be due to C inputs from vegetation and intensive management practices, such as mulching, composting and fertilization. These differences appear to relate to historic land uses. Kaifeng, for example, is an ancient capital city in China, and older portions of the city have been covered with densely spaced buildings for thousands of years. Soil C content based on land use history and it is difficult to predict soil C storage in urban areas based on current land use alone.

Nonetheless, many urban studies have focused primarily on differences in C dynamics among landuse types. For example, studies have focused on quantifying soil C storage in urban areas (Pouyat, 2002; Pouyat et al., 2006) and comparing soil C storage in urban ecosystems with nonurban ecosystems (Pouyat et al., 2009; Qian and Follett, 2012). Recently, there is also increasing interest in residential lawn C storage (Guertal, 2012; Qian and Follett, 2002; Selhorst and Lal, 2013; Zirkle et al., 2011). Urban lawns have higher C storage in the soil compared to rural grasslands because of intensive management practices, however, there may also be increased irrigation, fertilization, and mowing costs for maintaining these lawns depending on climate and region. On the other hand, there are few studies on the potential for increasing urban soil C storage through management practices that improve degraded urban soils with potential benefits through both increased direct and indirect ecosystem service provision.

At a larger temporal scale, studies carried out in the southern United States from 1895 to 2007 (Tian et al., 2012a; Zhang et al., 2012) showed that pre-urbanization vegetation type, time since land conversion and elevated atmospheric CO_2 concentration were three factors that influence C dynamics and sequestration during and post-urbanization. Based on a study estimating C dynamics of urbanized/developed lands in the Southern United States from 1945 to 2007 using a process-based model (Dynamic Land Ecosystem Model), Zhang et al. (2012) proposed that soil C storage rapidly decreases at the beginning of land conversion to urban uses, and then gradually increases. This suggests that there is potential for changes in soil C management to have significant impacts over the long term.

Physical disturbance has been recognized as one of critical factors affecting C storage in urban landscapes. One very common type of urban soil disturbance is the scraping and stockpiling of surface soils followed by compaction of the lower horizons through extensive grading during urban land development. However, it is unclear how this type of disturbance affects the soil C cycle (Pouyat et al., 2010). Soil compaction is pervasive on disturbed sites and has been well documented as one of limiting factors to urban vegetation establishment and growth (Day et al., 1995; Fleming et al., 2006; Jim, 1993) primarily because of root impedance and restricted air and water movement within compacted soil. Furthermore, soil compaction can also affect the soil C cycle by directly influencing C input and storage, and indirectly affecting soil physical properties such as bulk density, strength and porosity (Brevik et al., 2002; Deurer et al., 2012; Nawaz et al., 2012). At a military training site at Fort Benning, Georgia, a series of studies has been conducted to evaluate the effects of military training related disturbance (e.g., foot traffic, tank training exercises) on soil C storage. Silveira et al. (2010) found the high level of disturbance due to foot traffic and tank training exercises (that resulted in near total loss of understory vegetation) reduced total soil C compared to low-impact sites. In a companion study, Debusk et al. (2005) observed the reduction in SOM content of surface horizons with increasing levels of disturbance and proposed that chronic physical disturbance of topsoil might accelerate decomposition of the more recalcitrant SOM fractions. Furthermore, decreased microbial biomass carbon (MBC) suggested that labile C also decreased with increasing level of disturbance (Silveira et al., 2010). However, it is still unclear how compaction and other disturbance affect individual C pools.

1.2.5 The effects of degraded urban soils on microbial biomass carbon

In addition to its direct effect on soil carbon, compaction can also affect the amount (Beylich et al., 2010; Busse et al., 2006; Silveira et al., 2010) and distribution (Santrucková et al., 1993) of MBC and thus soil carbon sequestration. Soil MBC is a critical component in the soil C cycle that may mediate movement of C into more stable pools (Dalal, 1998). Compaction can have a positive, negative or no effect on MBC. The effects vary with landuse type, the level of compaction, the duration of compaction, site history, and management practices. Some authors suggest that bulk density values above 1.7 Mg m^{-3} will decrease microbial biomass (Beylich et al., 2010; Busse et al., 2006), presumably because compaction may alter the availability of water and air as well as microbial food resources. Compost amendments can provide C and energy sources to soil microbes, and thus can increase soil MBC (Bastida et al., 2012; Pascual et al., 1999; Wiseman et al., 2012). A meta-analysis (Kallenbach and Grandy, 2011) indicated that even small amounts of organic amendments can rapidly increase microbial biomass in agricultural systems. If this hypothesis is valid in urban systems as well, urban soil rehabilitation by compost amendment has potential to increase MBC, accelerating C sequestration in urban soils. The microbial response to typical urban land development practices and post-development rehabilitation can provide direct and accurate information to test this hypothesis. It has been documented that MBC has a positive relationship with TOC (Carter et al., 1999) and with the labile C pool (Murphy et al., 2011). There is a need to better understand the link between C dynamics and microbial biomass changes to understand the process of C turnover (Kuzyakov, 2010). However, because the fate of organic C is complex, the relationship between MBC and different soil C pools will be influenced by numerous site factors, such as soil type, plant communities, and management practices (Carter et al., 1999). Hence, how soil C pools and MBC respond to grading, scraping, and compaction will help elucidate how urban land development affects C cycles. In addition, increases in MBC during soil restoration may be indicators of the potential of rehabilitated soils to sequester additional C in more stable forms.

Topsoil removal, stockpiling and replacement are common processes during urban land development and can also affect MBC and soil C pools. Harris et al. (1989) found microbial biomass was decreased in stored topsoils after stockpiling during opencast mining. Similar observations were reported in a uranium wellfield study (Stahl et al., 2002). Wick et al. (2009) investigated the effect of topsoil stockpiling at a mining site on different soil C pools and found reductions in the available C pool (free light fraction) in surface soil after 3 years. However, few studies have examined stockpiling and other topsoil handling practices in the context of urban development. Schindelbeck and van Es (2012) conducted a case study to evaluate right-of-way construction (installation of a natural gas delivery pipeline) impacts on soil properties and found organic matter content on right-of-way corridors was lower than off right-of-way at adjacent sites (2.5% and 5%, respectively). Both TOC content and active C were negatively affected. Effects on organic C content also vary by the length of stockpiling period (Kundu and Ghose, 1997). Unlike mining or other sites, topsoil stockpiled at urban construction sites that is later reapplied on site is typically held only for short durations (less than 1 year), although soil moved off site and later sold may be stockpiled for longer periods. Thus, there is a need to assess the effects of typical topsoil handling at urban development sites on C storage, and, in particular, on different C pools. By documenting changes in each of these C pools, we can better understand how urban development and soil management practices affect C storage in soils.

1.2.6 The effects of degraded urban soils on greenhouse gas emission

By 2030, GHG emission is projected to increase by 25-90% compared to 2000 (IPCC, 2007). Global CO_2 emissions, the greatest overall contributor to global warming, have increased by 3.4 Pg C from 1990 to 2011 with average annual growth rates of 1.9% per year in the 1980s, 1.0% per year in the 1990s, and 3.1% per year since 2000 (Peters et al., 2012). Thus, there are compelling reasons to attempt to mitigate climate change by offsetting GHG emissions across a broad range of human activities. Furthermore, modern climate change is dominated by human influences, and land use change resulting from urbanization is one important driving force (Karl and Trenberth, 2003). In addition, the gases from urban areas are the dominant anthropogenic sources, the GHG emissions from the construction and operation of cities, in particular, are large and increasing (Grimmond, 2007). By 2050, 70% of the world's population will live in urban areas (Seto and Shepherd, 2009). Maximum average per capita GHG emission could be more than 15 tonnes CO_2 equivalent (CO_2eq) (Hoornweg et al., 2011). Thus, urban areas play an important role in global GHG emissions and mitigation.

Urban soils have been suggested as a cost-effective approach for sequestering large amounts SOC and consequently mitigating GHG emissions (Lal, 2003; Lorenz and Lal, 2009; Pouyat et al., 2006). Carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4) are three major GHGs contributing to global climate change. North America is a net atmospheric CO_2 source even though photosynthesis could offset about 35% fossil-fuel-based CO_2 emissions by transferring C to vegetation, soils, and wood products (King et al., 2012). In particular, atmospheric CO_2 levels in urban areas are much higher than nonurban areas. For example, in Phoenix, Arizona, the CO_2 levels in the metropolitan region were 50% greater than the surrounding non-urban areas. Eighty percent was from anthropogenic sources (Koerner and Klopatek, 2002) and only 16% was from soils. However, it is still unclear whether urban soils are a net CO_2 source or sink. In Fort Collins, Colorado, CO_2 emission from urban lawns were more than five times that from native grasslands (Kaye et al., 2005). In contrast, in Phoenix, Arizona, CO_2 fluxes was lower in urban sites compared to rural sites along an urban-rural gradient in a desert landscape (Koerner and Klopatek, 2010). In Chicago, Grimmond et al. (2002) observed negative values of CO_2 flux from urban soils. Urban lawns have been studied intensively recently and may contribute significantly to regional soil-atmosphere gas exchange even though they represent a small portion of the landscape cover (Kaye et al., 2004). However, indirect CO_2 emissions associated with turfgrass and other land management activities have potential to deplete SOC sequestration effects. In addition, CO_2 fluxes from urban grasslands may differ from those of urban forests (Groffman et al., 2009). However, most studies only compare the difference of CO_2 fluxes between urban and nonurban systems, few studies explored how to diminish this difference via soil management practices.

During 1990-2010, worldwide biogenic N_2O emissions ranged from 0.80 to 1.02 Pg CO_2eq yr^{-1}. Projected N_2O emissions will increase 157-227% by the end of this century (Tian et al., 2012b). Previous studies have documented that urbanization increased N_2O emissions compared to natural lands. In Fort Collins, Colorado, urban lawns had more than 10 times the N_2O emissions of natural grasslands (Kaye et al., 2004). In Phoenix, Arizona, urban lawns had higher N_2O emissions compared to natural landscapes (Hall et al., 2008). In southern California, N_2O emissions from urban landscapes were approximately equal to or greater than agricultural emissions in urbanized areas (Townsend-Small et al., 2011). Fertilizer, soil moisture, soil temperature, frequency of disturbance, and soil amendment have been reported as factors associated with increased N_2O emissions from urban soils (Bijoor et al., 2008; Groffman et al., 2009; Huang et al., 2004; Maggiotto et al., 2000). However, the heterogeneity of urban soils leads to uncertainties concerning the significance of these factors and their interactions.

During 1990-2010, biogenic CH_4 emissions ranged from 0.16 to 0.50 pg of CO_2eq per year. Projected CH_4 emission will increase by 137-151% by the end of this century (Tian et al., 2012b). Globally, soils can potentially take up about 40 Tg CH_4 from the atmosphere annually. Atmospheric CH_4 consumption is especially sensitive to anthropogenic disturbances, which typically decrease CH_4 consumption (King, 1997). It has been documented that urbanization reduced soil CH_4 uptake capacity. Goldman et al. (1995) observed urban forests had lower CH_4 consumption rates than rural forests along an urban to rural land-use gradient. In Fort Collins, Colorado, CH_4 uptake was reduced by about half in urban lawns compared to natural grasslands (Kaye et al., 2004). Moreover, in Baltimore, Maryland, CH_4 uptake capacity was almost completely eliminated in urban lawns compared to rural forests (Groffman and Pouyat, 2009). It is unknown if soil management in urban areas can restore this eliminated or reduced CH_4 uptake capacity.

Mitigation of GHG emissions is a long term mission for international, national and local governments (IPCC, 2007; Krause, 2011). Management practices could enhance political strategies at the technical level. In agriculture, some practices (e.g., reduced tillage, improved silviculture, producing woody bioenergy crops) are already being implemented. However, limited information is available for urban soils. Other experimental techniques such as biochar production and application, deep soil C sequestration, and modification of plant characteristics via biotechnology have been presented (Post et al., 2012). Thus, there is considerable interest in opportunities to mitigate GHG emission via soil management practices. A cost-efficient and effective approach to manage urban soils to mitigate climate change is desirable.

1.2.7 The effects of degraded urban soils on stormwater mitigation

Urbanization can increase short-term peak discharge and total stormwater runoff (Leopold, 1968) and lead to nonpoint source pollution (Novotny et al., 1985). Urbanization also increases impervious surface area. In 2000 and 2001, Elvidge et al. (2007) found that 83,337 km^2 of the United States land area is impervious surface area (e.g., roads, parking lots, buildings) which is 1.05% of the total land. A higher proportion of impervious surfaces are found in urbanized areas and is associated with increased stormwater runoff leading to flooding, flashy streams, and degraded surface water quality (Paul and Meyer, 2001). The U.S Environmental Protection Agency (EPA, 2003) reported that impervious surfaces can generate more than 5 times as much runoff as a forested area. In urbanized settings, even open soil areas (i.e., without pavement or buildings) designated for landscaping, tree lawns, or parks, may be nearly impervious due to grading, scraping, and soil compaction during and after urbanization (Gregory et al., 2006; Pitt et al., 2008). Urban soils are typically highly compacted (Jim, 1993, 1998a, b; Short et al., 1986) that severely impede water infiltration (Gregory et al., 2006; Pitt et al., 2008). Pitt et al. (2008) demonstrated that compaction significantly reduced soil infiltration rates based on laboratory and field tests, especially in sandy soils. They also found steady-state infiltration rates decreased as the level of compaction increased. In sandy soils in North Central Florida, average water infiltration rates were 37.7- 63.4 cm hr^{-1} in undeveloped non-compacted natural forest soil and 0.8-17.5 cm hr^{-1} in post-development compacted yard soils (Gregory et al., 2006). The water infiltration decreases were clearly due to home construction activities even with high variation among sites. Similarly, Hamilton and Waddington (1999) reported that the average infiltration rate of home lawns in central Pennsylvania was between 0.4 and 10.0 cm hr^{-1}. They also concluded that construction processes, e.g., excavation procedures and lawn establishment methods, affected infiltration rates. Furthermore, the age of residential site can affect infiltration. In another Pennsylvania study, pre-2000 constructed residential sites had higher mean infiltration rates (9.0 cm hr^{-1}) than those constructed post-2000 (2.8 cm hr^{-1}) (Woltemade, 2010). Whether this is due to changes in construction practices or increases in infiltration that occur due to soil aggregation over time is unknown.

There is increasing interest in environmentally sensitive stormwater management practices that take into account the influence of site and soil variables on watershed hydrology (Pitt and Clark, 2008). Low impact development (LID), in particular, has been presented as alternative stormwater management and site design approach to minimize construction impacts on hydrology (Williams and Wise, 2006). A wide range of best management practices (BMPs) have been studied (Bartens et al., 2008; Collins et al., 2010; Xiao and McPherson, 2011). However, all these practices are mainly focused on creating a facility to collect and store stormwater. An integrated stormwater mitigation technique is needed to utilize existing landscapes to optimize stormwater interception and mitigation as well as enhance other ecosystem services, such as biomass production.

Tillage is commonly used to loosen compacted agricultural soils, but is relatively rare in urban settings. One study, however, found deep tillage (up to 60 cm) alone slightly improved soil infiltration at three parks in the Minneapolis Metropolitan area (Olson et al., 2013). Compost amendment has also been reported as an effective practice to increase infiltration rates in disturbed urban soils (Pitt et al., 1999). Surface compost amendment of 7 cm, for example, was found to increase soil saturated hydraulic conductivity (K_{sat}) by 2.7-5.7 times that of the control (Olson et al., 2013). However, it's unclear whether K_{sat} will be enhanced with deep tillage accompanied by compost amendment. In addition, urban vegetation can help reduce peak discharge and stormwater runoff (Sanders, 1986). It is well documented that tree canopy can intercept rainfall to reduce peak discharge (Xiao et al., 1998; Xiao et al., 2000). In a recent study conducted in Manchester, UK, the role of urban trees in reducing surface water runoff has been highlighted (Armson et al., 2013). In this study, trees and their associated tree pits reduced runoff from asphalt by 62%. Furthermore, tree roots can play important role in stormwater mitigation by increases in permeability due to root penetration. Bartens et al. (2008) demonstrated that tree roots increased average K_{sat} 27-fold compared to unplanted controls. Thus, an urban stormwater mitigation system is needed to integrate these potential mitigation components.

1.2.8 The effects of degraded urban soils on tree growth

Urban trees are a significant component of urban vegetation and generate a wide range of ecosystem services including aesthetic value, removing air pollutants, mitigating stormwater, saving energy, C sequestration, noise buffering, and providing wildlife habitat and economic benefits (Bolund and Hunhammar, 1999; McPherson and Simpson, 2002, 2003). In particular, urban trees can sequester C and reduce CO_2 in the atmosphere. The above-ground parts of urban trees in the conterminous USA stored 7.0×10^{11} Mg of C ($14,300 million value) with a gross C sequestration rate of 2.28×10^{10} Mg C yr^{-1} ($460 million year^{-1}) in 2002 (Nowak and Crane, 2002). Carbon stored in biomass formed by the urban forest amounted to about 0.2×10^9 Mg in Beijing, China (Yang et al., 2005). For these reasons, increasing tree canopy has become a high priority for city development (McGee et al., 2012).

Soil is the key to successful establishment of urban vegetation (Vegter, 2007). Furthermore, grading, compaction and the incorporation of building materials into soil can degrade soil physical, chemical, and biological properties. Studies have been conducted all over the world concerning soil pollution, particularly heavy metal contamination (e.g., Biasioli et al., 2006; Li et al., 2001; Manta et al., 2002; Möller et al., 2005). However, compacted soil is more commonly cited as a primary constraint to tree growth in urban areas (Bartens et al., 2008; Day et al., 1995; Gregory et al., 2006; Jim, 1998b; Unger and Kaspar, 1994). Reduced organic matter typically accompanies soil compaction, suggesting that C stores may be reduced in some urban soils. Studies about how to improve urban soil conditions to both benefit tree growth and optimize C stores are needed.

Urbanized land is characterized by the dominance of paved surfaces. Increasing tree canopy in urbanized areas has been identified as an effective way to achieve urban forest ecosystem services (Brack, 2002; McPherson and Simpson, 2003; Nowak et al., 2006). Local ordinances (Arlington County Chesapeake Bay Preservation Ordinance, Undated) and certification programs [e.g., (Sustainable Sites Initiative, 2009; U.S. Green Building Council, 2009)] (Sustainable Sites Initiative, 2009; Windhager et al., 2010) often stipulate tree canopy requirements in commercial and residential development and encourage shading of pavement to reduce the urban heat island effect. Ordinances alone have been shown to be ineffective at increasing canopy cover (Hill et al., 2010). Meanwhile, soil conditions are widely believed to exert considerable control on tree growth. In order to successfully implement tree ordinances and site development standards, it is essential to rehabilitate degraded urban soils to increase tree canopy growth.

1.2.9 Current approaches to rehabilitating degraded urban soils

It is challenging to restore degraded urban soils (Pavao-Zuckerman, 2008). Preplanning and design can reduce construction effects (Randrup and Dralle, 1997). A wide range of techniques have been developed to rehabilitate post-construction degraded urban soils: trenching, augering holes, mixing sand, mixing gravel and adding organic soil amendments (Day et al., 1995; Pittenger and Stamen, 1990; Spoor, 2006), in particular, deep tilling has been suggested as a tool to remediate subsoil compaction (Motavalli et al., 2003). However, most of these studies focused on remediation to improve plant growth. They did not quantify the effect on other ecosystem services, such as C sequestration and stormwater. Furthermore, it is well documented that plant roots can penetrate compacted soils in some situations and help improve soil condition (Bartens et al., 2008; Clark et al., 2003; Materechera et al., 1991). Thus, there is a need to develop a new urban soil rehabilitation approach that not only improves soil condition and plant growth but also provides compatible ecosystem services. In this study, we address urban soil rehabilitation with a focus on subsoil compaction. Our proposed urban rehabilitation focuses on subsoil loosening via compost amendment, subsoiling and tree planting.

1.2.10 Proposed urban soil rehabilitation technique

It is well documented that conservative tillage and crop residue management practices can help restore degraded soil and sequester C in agricultural soils (Lal, 1997). Urban wastes (e.g., yard waste) are a potential source of organic inputs and may have similar effects on recovering degraded soils. In a restoration study in Spain, soil organic amendment with the organic fraction of urban wastes (both fresh and composted) initially increased the levels of SOM, these effects decreased but always remained higher than those of the unamended soil 2 years after treatment which demonstrates the addition of urban organic waste can be beneficial for recovering degraded soils (Ros et al., 2003). However, different amendments have different influences on soil rehabilitation. Roldan et al. (1996) studied different organic amendment effects on aggregate stability and uncomposted urban refuse was the most effective in increasing soil stable aggregates among four types of organic amendments (sewage sludge, fresh uncomposted urban refuse, composted urban refuse, and horse manure). However, these studies focused on agricultural soils, while research on urban soils is limited. Whether these practices could be effective to address urban soil issues is unknown.

Organic soil amendments have been recommended for use in urban landscapes to improve disturbed, highly modified or degraded soils (Cogger, 2005; Larney and Angers, 2012; Sæbø and Ferrini, 2006; Sloan et al., 2012). For example, such amendments have been successfully used to improve turfgrass, grassland, urban gardens, and landscapes (González et al., 2010; Loschinkohl and Boehm, 2001; McIvor et al., 2012; Ohsowski et al., 2012). Organic soil amendments are favored because they can decrease soil bulk density and increase infiltration rates, aggregate stability, porosity and soil C storage

(Cogger, 2005; Khaleel et al., 1981; Loper et al., 2010; Persyn et al., 2004). Brown (2012) described a case study conducted in Tacoma, Washington and estimated that 6,000 Mg of organic amendment at various rates and frequencies across the city had resulted in an additional 460 Mg of C sequestration. Likewise, Loper et al. (2010) reported that composted dairy manure solids incorporated into the top 5 cm of soil (256 Mg ha^{-1}) increased SOM compared to an unamended control after 1 year in simulated new residential landscapes in Florida, US. Recently, Beesley (2012) demonstrated that a green waste compost amendment could help maintain and build soil C storage in newly created urban soils in the United Kingdom. Although soil C in new amended soils appeared to approximate that of old urban soils in this study, newly created and existing urban soils could not be objectively compared due to different regions and landuse histories. Amendments can be either surface-applied or incorporated. However, most studies focus on surface application. The limited literature comparing surface application vs. incorporation suggests that surface-application increases soil C storage compared to incorporated amendments. However, most of these studies were conducted in controlled laboratory environments over short time periods and do not address incorporation below the surface soil (Coppens et al., 2006; Nicolardot et al., 2007). Biederman and Whisenant (2011) studied the influence of amendment location in a severely disturbed landfill 3 years after application. However, the incorporated amendment application was only applied in the surface 6 cm of soil. Thus, it is still unknown how urban soil C storage is affected when organic amendments are incorporated into deeper soils, especially over the long term.

Unlike agricultural subsoils, urban subsoils are very likely to have been compacted during land development. Urban site development generally includes significant grading and compaction to both support structures and facilitate site drainage. However, subsoil quality affects environmental performance of urban sites because of its influence on water storage and transmission as well as the establishment and persistence of vegetation, especially trees which are a large part of urban above-ground C stores and provide a host of other ecosystem services (Johnson and Gerhold, 2003; Nowak, 1993; Nowak and Crane, 2002). Trees tend to have deeper maximum rooting depths than shrubs and herbaceous plants (Canadell et al., 1996). Roots of urban trees, however, are often impeded by dense subsoils, resulting in shallower root systems (Day et al., 2010). Thus, deeper soil regions that may be heavily compacted in urban areas are important to tree establishment in terms of supporting root penetration and development and consequently resilience to drought. Thus, improving the quality of lower soil regions is critical in terms of supporting plants and storing C in soils. Subsoil has been recognized recently as an important C sink to store more stable C (Lorenz et al., 2011; Salomé et al., 2010; Sanaullah et al., 2011). In forest soils near Coshocton, Ohio, total SOC in the A horizon was greater than at lower horizons, however, the proportion in the stabilized C pools (i.e., physically or chemically protected) increased with depth (Lorenz et al., 2011). These and other findings have led to increasing awareness that deep soil region is critical to global C storage (Helfrich et al., 2010; Lorenz et al., 2011; Rumpel et al., 2012; Sanaullah et al., 2011). Furthermore, in urban settings, surface soils are more likely to be influenced by human activities, e.g., topsoil removal or replacement during construction processes. In addition, different C pools have different levels of accessibility and stability since they have different (e.g., physical, chemical and biochemical) protection mechanisms. As a result, the comparatively more stable C pool in deeper soils becomes critical to store C for the long term. Based on current literature, Harrison et al. (2011) estimated that 27-77% of mineral C was found in the soil deeper than 20 cm when sampling depth was no less than 80 cm. Furthermore, it has also been proposed that topsoil and subsoil might have different C dynamic regulatory mechanisms (Salomé et al., 2010). Contact between microbes and other degraders and substrate is thought to be the main regulatory mechanism of C mineralization in the subsoil, while the supply of fresh organic matter controls C dynamics in the surface horizons. Introduction of organic amendments into deeper soil regions has the potential to increase contact between microbial communities and substrate, but it is unknown if this depletes existing C stocks (Dungait et al., 2012). Thus, studies are needed to test whether organic amendments can increase C storage in both surface and deep soil, especially in the stabilized C pools. The study presented here was designed to simulate typical urban land development practices and test effective urban soil rehabilitation practices to evaluate soil C storage changes after urbanization and rehabilitation.

Litterfall and fine root production are major pathways for C and nutrient cycling in forest ecosystems (Vogt et al., 1995). Tree root systems contribute to C sequestration through formation of long-lived belowground biomass and C in root necromass and woody debris (Maier and Johnsen, 2010). Living roots have a stimulatory effect on SOM decomposition due to the higher microbial activity induced by the roots (Cheng and Coleman, 1990). Dijkstra and Cheng (2007) studied interactions between soil and tree roots with Ponderosa pine (*Pinus ponderosa*) and Fremont cottonwood trees (*Populus* *fremontii*) in the greenhouse for 395 days and found that tree roots substantially accelerate SOC decomposition; SOC decomposition in planted treatments was significantly greater (up to 225%) than in soil incubations alone and thus increased C inputs from roots could still result in net soil C loss. But these rhizosphere effects can be positive or negative on C cycling depending on water availability and inter-specific plant interactions (Dijkstra et al., 2010). This is demonstrated in the study by Strickland et al. (2010); they found grass invasion of a hardwood forest decreased belowground C pools because this invader might accelerate C-cycling in forest soils and deplete C stocks. So there are no certain and exact explanations. Although the importance of soil-root interactions for SOC decomposition has increasingly been recognized, their long-term effect on SOC decomposition remains poorly understood. The information about urban trees under the conditions prevalent in urban environments is especially scarce.

1.2.11 Summary

Based on the discussion above, typical urban development practices often result in degraded urban soils, which are highly compacted and lack organic matter. Moreover, soils play an important role in mitigating global warming through their function as a C reservoir and provide other important ecosystem services that are of high value in urban settings, such as stormwater mitigation. Rehabilitation treatment practices that seek to improve soil structure have the potential to store more C and provide ecosystem services. This study will focus on urban soil rehabilitation and its influence on soil C storage, greenhouse gas emission and soil physical characteristics. The Soil Rehabilitation Experimental Site (SRES) plot was established in 2007 at Virginia Tech to explore the potential for improving damaged urban soils with subsoiling and organic amendment. Prior research examined rehabilitation treatment effects on tree growth and soil physical characteristics during tree establishment. In this study, it focuses on the effects of three different types of soils (pre-development undisturbed soils, typical-development degraded soils and post-development rehabilitated soils) resulting from urbanization process on soil C storage, GHG emissions and stormwater mitigation.

1.3 Research Questions and Objectives

1.3.1 Questions addressed in this study

1) Can urban soils be managed to expand their potential for C storage and mitigation of GHG emissions?
2) Can soil management practices that improve degraded soil structure help sequester more C in soils?
3) Can such management practices also enhance other ecosystem services, such as stormwater management through improvements to soil physical characteristics?

1.3.2 My research objectives

1) evaluate how urban land development processes followed by common post-development soil management protocols affect C pool size and microbial biomass carbon at three soil depths (0-5, 5-10, and 15-30 cm);
2) determine how a post-development soil rehabilitation practice directed at alleviating compaction and incorporating organic matter into subsoil can influence C pool sizes and microbial biomass carbon at these same depths;
3) examine aggregate-associated soil C concentration of typical urban land development and post-development rehabilitation;
4) compare GHG emissions after typical urban land development practices and post-development soil rehabilitation;

5) evaluate if urban soil rehabilitation improves soil physical characteristics including aggregate size distribution and bulk density over time; and
6) determine if urban soil rehabilitation will influence soil hydraulic conductivity and soil water content dynamics that might lead to mitigate urban stormwater runoff.

1.4 Research Site and Experimental Design

Figure 1.1. The Soil Rehabilitation Experimental Site (SRES) at Kentland Farm, Virginia Tech.

The Soil Rehabilitation Experimental Site (SRES) (Figure 1.1) was established at Kentland Farm (37°12' 1.1888" N, 80° 33’ 48.3768" W) near Blacksburg, VA in 2007 by graduate student Rachel Layman and others (Layman, 2010). Two related loamy soils, Shottower loam (fine, kaolinitic, mesic Typic Paleudults) and Slabtown loam (fine-loamy, mixed, mesic Aquic Paleudalfs), are present at the site. The plot site is oriented east to west on a slight east-facing slope (approximately 3-7°) and has historically been in agricultural use. The site was maintained in tall fescue until installation of the experiment. Between May 2007 and November 2007, 24 (6 replications × 4 soil treatments) 4.6 × 18.3 m plots were installed in a completely randomized experimental design. With the exception of the control plots, all plots were subjected to typical urban land development practices as a pre-treatment: The A horizon was removed, and then exposed soil surface was compacted via repeated trafficking with heavy equipment to a bulk density of approximately 1.98 Mg m^{-3}. The four installed soil treatments are (Figure 1.2): typical practice (TP) (topsoil replaced only), enhanced topsoil (ET) (topsoil replacement and rototilling), profile rebuilding (PR) (leaf compost applied to surface followed by subsoiling with a backhoe to 60-cm depth, topsoil replacement, and rototilling), and undisturbed (UN) (no grading, scraping or improvement methods applied). As pre-development agricultural soils, UN serves as a benchmark in this study. The typical practice represents typical urban land development soil conditions after vegetation clearing, topsoil removal, soil stockpiling, grading, compacting, and finally partial topsoil replacement. This typical practice is seen throughout the United States and elsewhere as land is developed and buildings constructed. As our urban soil rehabilitation treatment, PR is an aggressive post-development soil rehabilitation technique that includes compost amendment and deep tilling focusing on subsoil decompaction; while ET is intermediate and is often recommended in practice as an improvement over TP because it disrupts the interface between compacted subsoil and applied topsoil. We randomly planted five tree species at each plot: *Ulmus japonica* (Rehd.) Sarg. x *wilsoniana* Schneid. 'Morton', *Acer rubrum* L., *Quercus bicolor* Willd., *Prunus* 'First Lady' (*P.* ×*incam* Ingram ex R. Olsen & Whittemore 'Okamé' × *P. campanulata* Maximowicz), and *Quercus macrocarpa* Michx. (Figure 1.3).

Rachel Layman (2010) studied soil characteristics (bulk density, soil pH, C/N), tree growth (tree canopy, photosynthesis rate, etc.) during two years after installation of soil rehabilitation treatments. My study focuses primarily on soil C dynamics, water infiltration, and greenhouse gas emissions to continue the next phase of this long-term project.

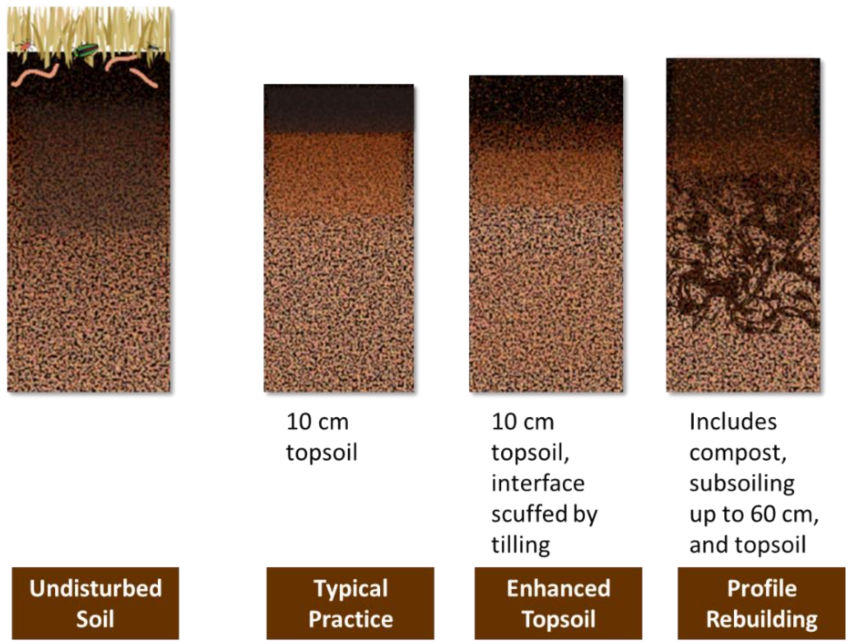

Figure 1.2. Four soil treatments: undisturbed soil (UN), typical practice (TP), enhanced topsoil (ET) and profile rebuilding (PR) (Artist: Sarah Gugercin).

Plot #	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24
Treatments	PR-I	UN-I	TP-I	ET-I	ET-II	UN-II	PR-II	UN-III	PR-III	ET-III	TP-II	UN-IV	UN-V	ET-IV	PR-IV	TP-III	TP-IV	PR-V	UN-VI	PR-VI	ET-V	ET-VI	TP-V	TP-VI
Tree	2	1	2	4	1	5	2	3	1	3	1	1	5	3	1	2	4	3	4	5	1	2	3	2
	5	3	3	5	5	1	4	5	2	5	4	5	3	1	2	1	2	2	1	2	4	4	5	1
	3	5	1	3	4	2	1	4	3	1	3	4	1	5	4	5	3	1	2	3	2	1	4	5
	4	2	4	2	2	4	3	1	5	4	2	3	4	2	3	3	5	4	5	4	5	3	1	4
	1	4	5	1	3	3	5	2	4	2	5	2	2	4	5	4	1	5	3	1	3	5	2	3

Figure 1.3. Plot map of the SRES site indicating soil treatments, plot number, and location of five tree species.

(Notes: 1- *Acer rubrum* L.; 2- *Quercus bicolor* Willd; 3-*Ulmus japonica* (Rehd.) Sarg. x *wilsoniana* Schneid. ‘Morton’”; 4- *Prunus* ‘First Lady’ (P. ×*incam* Ingram ex R. Olsen & Whittemore ‘Okamé’ × P. *campanulata* Maximowicz); 5- *Quercus* macrocarpa Michx.)

1.5 Overall Approach

The overall approach for this study is represented in Figure 1.4. Based on the established SRES plot, we study the effects of urban rehabilitation on soil C dynamics and soil physical characteristics. In terms of soil C dynamics, we examine C storage, C efflux, and microbial activity after soil rehabilitation by determining soil C pools, MBC, aggregate associated C and greenhouse gas emission. Furthermore, the effects of rehabilitation treatment on soil physical characteristics are estimated by measuring soil aggregate size distribution, soil moisture, hydraulic conductivity and bulk density. All measurements are carried out at several soil depths as is appropriate to the technique (Figure 1.5). Soil C pools, MBC, aggregate associated C and soil aggregate size distribution are measured in 0-5 cm, 5-10 cm and 15-30 cm depths; soil moisture at 0-10 cm, 0-25 cm and 0-40 cm; bulk density at 2.5-7.6 cm, 15.2-20.3 cm, 30.5-35.6 cm, 50.8-55.9 cm; and hydraulic conductivity at surface, 10-25 cm and 25-40 cm.

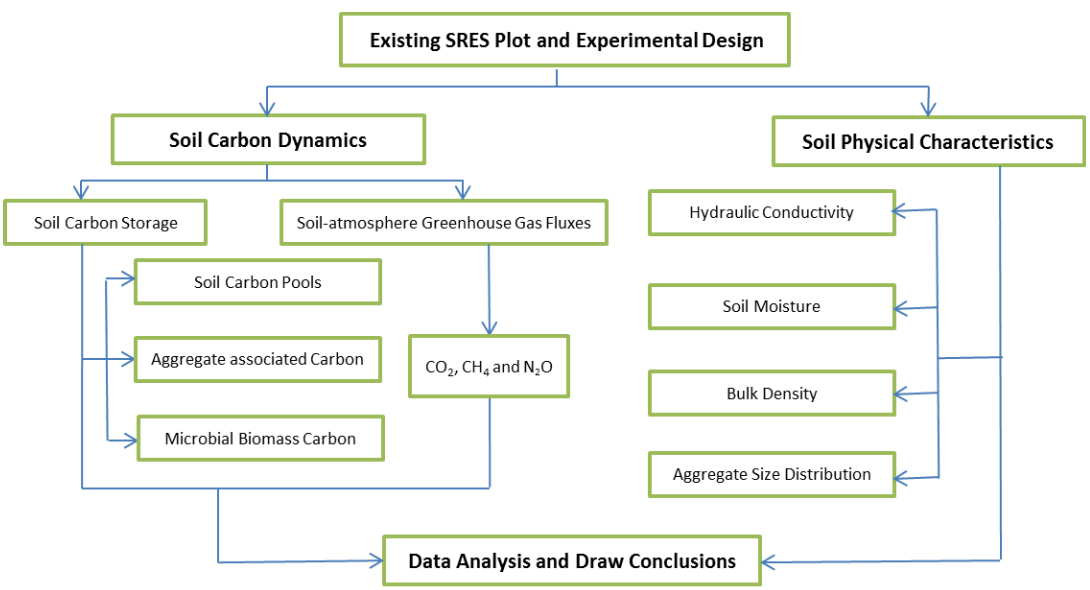

Figure 1.4. Flowchart indicating data collected in two schematic areas: soil carbon dynamics including greenhouse gas emissions and soil physical characteristics.

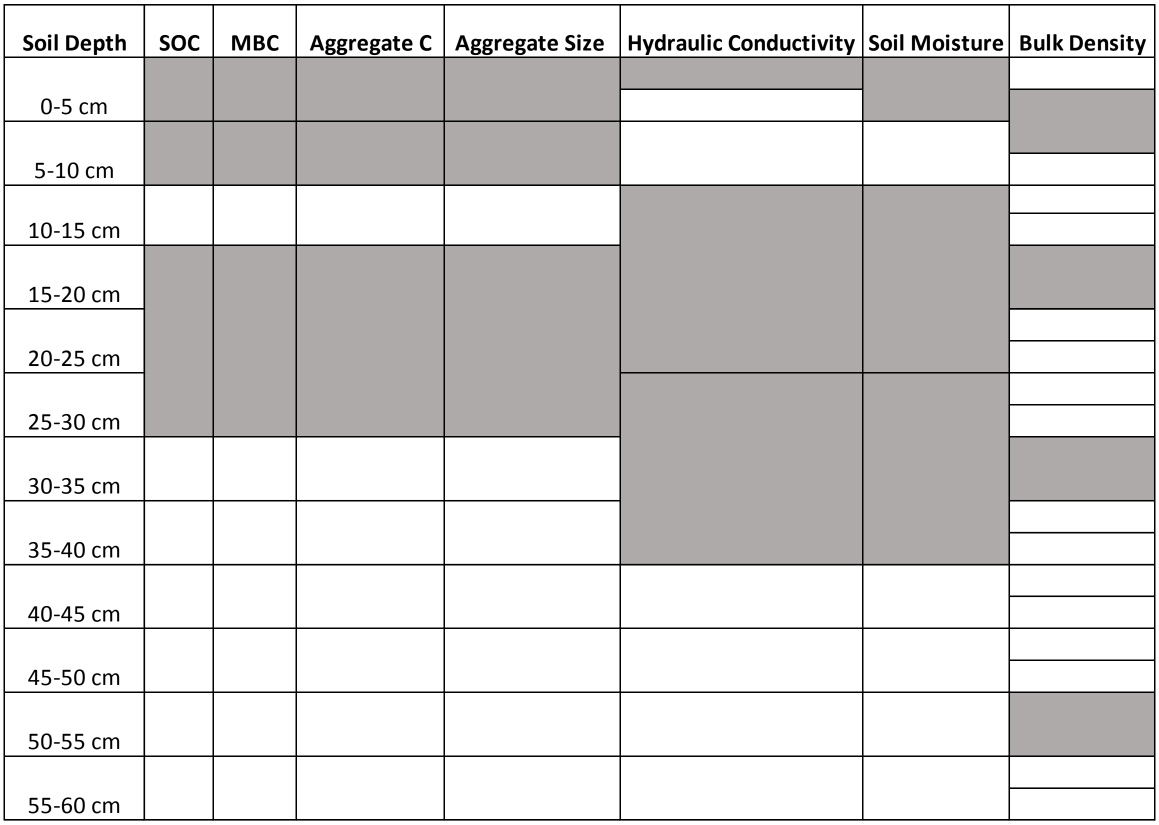

Figure 1.5. Sampling depths for soil-related measurements. Shaded areas indicate depth range of measurement (Notes: SOC-soil organic carbon; MBC-microbial biomass carbon).

Figure 1.6. Comparison of three different types of soils during urbanization process: pre-development undisturbed soils, typical development degraded soils and post-development rehabilitated soils.

One motivation of this study is to evaluate the soil disturbance resulting from typical urban land development practices compared to undisturbed soils. Another motivation of this study is to explore an effective approach to rehabilitate this disturbance and provide better ecosystem services. The proposed soil rehabilitation technique incorporates compost amendment, deep tilling and tree planting. In this study, we will assess three types of soil conditions of interest when we consider the urbanization process: pre-development undisturbed soils; typical urban land development practice soils, two post-development rehabilitated soils (common practice by surface tilling as an enhancement; our rehabilitation technique via a deep tillage, compost amendment and tree planting system named “soil profile rebuilding”). The goal of this study is to determine the change of soil physical characteristic (e.g., bulk density, aggregation), soil C storage, greenhouse gas emission, and other related ecosystem services (e.g., stormwater mitigation) during land conversion from rural or forested land to urban land. The ultimate goal of this study is to evaluate the effects of post-development soil management practices (e.g., soil rehabilitation) on these individual ecosystem elements and the whole ecosystem function. The results from this study can be applied to landscapes of different scales, ranging from a construction site to an urban ecosystem, even to the global scale, for different purposes. This study also can provide insight to explore effective policy and management practices to establish or maintain a sustainable environment.

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Chapter 2

Changes in Soil Carbon Pools and Microbial Biomass from Urban Land Development and Subsequent Post-development Soil Rehabilitation

Abstract

Urbanization degrades soil by surface soil removal and subsoil compaction. Subsoiling and organic matter incorporation offer potential to increase soil carbon (C) storage over time. We evaluated the effects of urban land development practices and post-development soil rehabilitation on soil C pools and microbial biomass carbon (MBC). In 2007, four treatments [typical development practice (A horizon removed, subsoil compacted, and A horizon soil partially replaced), enhanced topsoil (same as typical practice plus tillage), profile rebuilding (compost incorporation to 60-cm depth in subsoil; A horizon soil partially replaced plus tillage), and undisturbed soil] were applied to 24 plots in Virginia, USA. In 2011, soil C pools and MBC were measured at 0-5, 5-10 and 15-30 cm depths. In surface soils, undisturbed soils had greater total soil C (12.10 and 7.73 Mg C ha^{-1} at 0-5 and 5-10 cm depths, respectively) than the three practices where A horizon soils were disturbed. However, profile rebuilding had higher total soil C (6.12 Mg C ha^{-1}) and larger available and aggregate-protected C pools (1.59 and 2.04 Mg C ha^{-1}, respectively) at 15-30 cm depth than other treatments, including undisturbed. In surface soils, MBC ranged from 185.3 to 287.3 mg C kg^{-1} soil among all practices. At the15-30 cm soil depth, profile rebuilding had the greatest MBC (149.5 mg C kg^{-1} soil). Total soil C, available, aggregate-protected and mineral-bound C pools had positive

relationships with MBC at the 15-30 cm depth (r = 0.85, 0.71, 0.77, and 0.69, respectively). We found typical land development practices led to soil C loss during construction processes even when topsoil is replaced. Soil rehabilitation that includes compost additions and subsoiling has potential to increase soil C storage below the surface horizons.

Keywords: aggregate-protected carbon pool, compaction, compost, grading, iPOM, light fraction, mineral-bound carbon pool, urbanization

2.1 Introduction

Urban land is expected to triple during 2000-2030, resulting in a global urban land cover of at least 1.4% (Seto et al., 2012). With this rapid urbanization, the role of urban soils as a potential carbon (C) sink is of increasing interest. Urban land development practices including clearing, topsoil removal, surface grading, compacting and building construction lead to degraded urban soils with low vegetative cover, high bulk densities, low infiltration rates, and disturbed C cycles (Jim, 1998b; Kaye et al., 2006; Woltemade, 2010); however, there is potential to enhance the C storage potential of urban soils through post-development soil management strategies.

Soil rehabilitation practices of subsoiling (deep tillage) and organic matter incorporation can be used to enhance urban soil C reserves. In a case study in Tacoma, Washington Brown et al. (2012) demonstrated the potential for C sequestration in urban soils via compost amendment and calculated that these amendments could result in 0.22 Mg C ha^{-1} C sequestration per year for the pervious land area of the city. This estimate is derived from the available compost supplies and landuse characteristics for Tacoma and the portion of sequestered C resulting from single or multiple applications of biosolid or yardwaste compost determined in a series of studies over 7-15 years. Loper et al. (2010) reported that a single compost amendment tilled into the surface 15 cm of soil increased SOM compared to an unamended control after one year in simulated new residential landscapes in Florida, USA. However, the long-term effects of single applications of organic amendment have received little attention. The addition of organic soil amendments has been recommended for use in urban landscapes to improve disturbed, highly modified or degraded soils (Cogger, 2005; Larney and Angers, 2012; Sæbø and Ferrini, 2006; Sloan et al., 2012). Organic soil amendments decrease soil bulk density, while increasing infiltration rates, aggregate stability, porosity and soil C storage (Cogger, 2005; Khaleel et al., 1981; Loper et al., 2010; Persyn et al., 2004). Amendments can be either surface-applied or incorporated. However, most studies have focused on surface application.

Subsoil has been recognized recently as an important C sink to store more stable C (Helfrich et al., 2010; Lorenz et al., 2011; Rumpel et al., 2012; Salomé et al., 2010; Sanaullah et al., 2011). The proportion of C in stable pools (i.e., physically or chemically protected) is higher in deeper soils (Lorenz et al., 2011). Based on a review of data from ten profiles from a variety of soil types in forested land, Harrison et al. (2011) estimated that 27-77% of C in the mineral soil was found in the soil deeper than 20 cm when the sampling depth was no less than 80 cm, indicating the importance of including deeper soil regions in studies of soil C where possible. In the urban setting, although subsurface soils may be highly disturbed during initial urbanization, they are less likely than surface soils to be subjected to ongoing human influence. It has also been proposed that topsoil and subsoil might have different C dynamic regulatory mechanisms (Salomé et al., 2010). Contact between degraders and substrate is thought to be the main regulatory mechanism of C mineralization in the subsoil, while the supply of fresh organic matter controls C dynamics in surface horizons. Introduction of organic amendments into deeper soil zones has the potential to increase contact between degraders and substrate, but it is unknown if this depletes existing subsoil C stocks (Fontaine et al., 2007). Thus, studies are needed to test whether organic amendment can increase C storage in both surface and subsurface soils, especially in stabilized C pools.

Compaction, relocation, disruption of horizons, and various other physical disturbance activities have been recognized as characteristics of urban soils (Craul, 1992; Jim, 1993, 1998a, b; Short et al., 1986). The high level of human disturbance of these soils offers considerable opportunities for developing new management practices; however, there is little literature exploring the effects of urban land development practices on soil C pools. Pouyat et al. (2010) proposed several factors that could result in soil C losses during and after construction activities either because they remove soil or physically disrupt aggregates. These include: (1) site erosion; (2) grading processes and compaction; and (3) surface soil stockpiling.

Urban site development generally includes significant grading and compaction to support structures (e.g., buildings) and facilitate site drainage. However, it is unclear how this type of disturbance affects the soil C cycle (Pouyat et al., 2010). Soil compaction is pervasive on disturbed sites and has been well documented as a major limiting factor to urban vegetation establishment and growth (Day et al., 1995; Fleming et al., 2006; Jim,

1993) primarily because of root impedance and restricted air and water movement within compacted soil. Urban subsoils in particular are very likely to have been compacted during land development. Subsoil quality affects environmental performance of urban sites because of its influence on water storage and transmission as well as the establishment and persistence of vegetation. This is especially true for trees which are a large part of urban above-ground C stores and provide a host of other ecosystem services (Johnson and Gerhold, 2003; Nowak, 1993; Nowak and Crane, 2002). Soil compaction can thus affect the soil C cycle by directly influencing C input (from roots) and storage, and indirectly affecting soil physical properties such as bulk density, strength, porosity, and water relations which in turn can alter soil biological activity (Brevik et al., 2002; Deurer et al., 2012; Nawaz et al., 2012).

In addition to these effects on soil C, compaction can also affect the amount and distribution of MBC. The effect of compaction on MBC is highly variable and has been reported to have a positive, negative or no effect over a range of compaction levels (as review in Beylich et al., 2010). In a study conducted at a military training site in Fort Benning, Georgia, MBC decreased as the level of disturbance resulting from training activities increased (Silveira et al., 2010a). However, the most significant increases were associated with loss of vegetative cover that accompanied severe disturbance. On the other hand minor compaction in wheel-tracks didn't affect seasonal MBC in a study conducted at an orchard in New Zealand (Deurer et al., 2012), nor was microbial community size or activity affected by severe compaction from multiple passes of heavy machinery in a North American Long-Term Soil Productivity study conducted at North Carolina, Louisiana and California in loam forest mineral soils (Busse et al., 2006).

Compaction may also influence MBC distribution. In a study conducted in agriculture soils at Braunschweig, Germany, microaggregate MBC increased with depth in soils under high compaction resulting from wheel tracks whereas it decreased with depth in uncompacted areas (Santrucková et al., 1993). The variety of MBC responses to compaction is attributed to differences in SOC content, landuse, the severity and duration of compaction, site history, and management practices all of which may affect availability of food resource and environmental factors (Beylich et al., 2010; Busse et al., 2006; Deurer et al., 2012; Silveira et al., 2010a). Soil MBC is a critical component in the soil C cycle that regulates movement of C within different C pools (e.g., from active into more stable pools (Dalal, 1998). It will be helpful to understand the linkage between microbes and soil C pools by testing MBC change after soil rehabilitation.

Although topsoil removal, stockpiling and replacement are common processes during urban land development, few studies have examined the effects of these practices on soil C in this context. Effects on organic C content vary by the length of stockpiling period (Kundu and Ghose, 1997). However, unlike mining or other sites, topsoil stockpiled at urban construction sites that is later reapplied on site is typically for short durations (less than 1 year), although soil moved off site and later sold may be stockpiled for longer periods. Thus, there is a need to assess the effects of typical topsoil handling at urban development sites on C storage, in particular, on different C pools.

By documenting changes in each of these C pools under different soil management regimes, we can better understand how urban development and soil management practices affect soil C storage. This study was designed to simulate common urban land development practices and evaluate soil C storage changes after urbanization under three soil restoration regimes: (1) typical practice where topsoil is partially replaced, (2) a commonly practiced enhancement where this topsoil is tilled after placement, and (3) a soil rehabilitation technique where compost is incorporated into the subsoil before topsoil placement, tillage, and tree planting.

Our objectives were to:

(1) evaluate how urban land development processes followed by common post-development soil restoration protocols affect C pool size and MBC at three soil depths (0-5, 5-10, and 15-30 cm);

(2) determine how a post-development soil rehabilitation practice directed at alleviating compaction and incorporating organic matter into subsoil can influence C pool sizes and MBC at these same depths; and

(3) determine the relationship of MBC to C sequestration in these pools.

2.2 Materials and Methods

2.2.1 Site information

The study site was established in Montgomery County, Virginia (37°12' 1.1888" N, 80° 33’ 48.3768" W) between May and November in 2007. The site was in pasture before experiment installation and contained two closely related loamy soils, Shottower loam (fine, kaolinitic, mesic Typic Paleudults) and Slabtown loam (fine-loamy, mixed, mesic Aquic Paleudalfs). Twenty-four (6 replications × 4 soil treatments) 4.6 × 18.3 m plots were installed in a completely randomized experimental design. Treatments included an undisturbed control, and three post-development soil management practices. In June of 2007, with the exception of the undisturbed plots, all plots were subjected to pre-treatment that replicated the scraping and compaction typical of land development: the A horizon was removed and stockpiled adjacent to the study site, and the exposed subsoil surface was compacted via 8 passes of a 4,808 kg ride-on sheep's foot vibrating compactor Ingersoll Rand (Model SD45D) when the soil was at apparent field capacity to an average bulk density of 1.98 g cm^{-3}. This pre-treated land was then subjected to three soil management practices: typical practice (TP), 10-cm of stockpiled A horizon soil (topsoil) applied to the compacted subsoil; enhanced topsoil (ET), same as TP, except interface between compacted subsoil and applied topsoil was disrupted by tillage to 10-15 cm; and profile rebuilding (PR), a soil rehabilitation practice that focuses on subsoil loosening and deep organic matter amendment. The PR procedure includes: (1) 10 cm of leaf compost (C/N ratio 15.0) applied to the surface; (2) subsoiling with a backhoe to a 60-cm depth until no clumps of compacted soil larger than 30 cm diameter remain (inserting the backhoe rearbucket through the compost layer and into the subsoil to a depth of 60 cm and raising the bucket of soil at least 60 cm above the soil surface and allowing soil to fall); and (3) replacement of 10 cm topsoil, and tillage. For all treatments, any existing vegetation was killed with herbicide (glyphosate) and plots were maintained weed free for the duration of the study. The soil surface was covered with mesh/straw erosion mats for the first year after installation and maintained as bare soil thereafter. The UN treatment represents pre-urban-development soil and thus serves as a benchmark for soil rehabilitation. The TP treatment represents typical urban land development practice.

The PR treatment is the full soil rehabilitation treatment being evaluated in this study, and the ET treatment is intermediate practice sometimes specified by designers and contractors. Five species of trees (*Ulmus japonica ×wilsoniana* ‘Morton’, *Acer rubrum*, *Quercus bicolor*, *Prunus* spp. ‘First Lady’, and *Quercus macrocarpa*) were planted in each plot from February to April in 2008.

2.2.2 Soil sampling

In June 2011, soil samples for soil C pools and MBC measurements were collected at locations equidistant from the trunk of red maple (*Acer rubrum*) trees in each of the 24 plots for a total of six replication sampling sites for each treatment. Within each sampling site, four soil samples were extracted at 0-5 cm and 5-10 cm depths within 1 m from the trunk. Two of these four sampling locations were randomly selected to extract additional samples at 15-30 cm depth. All samples were immediately stored in a cooler after field collection. The four samples were composited by depths and the required amount of soil extracted from the pooled sample for C fractionation and MBC measurements. Extracted fresh samples for MBC analysis were covered with wet paper to maintain even soil moisture prior to sieving and passed through a 2‑mm sieve. Samples were stored at 4 °C and analyzed within one week. Soil samples for C fractionation analysis were dried at room temperature and passed through a 2 mm sieve. For bulk density measurements, soil cores (D=5 cm, H=5 cm) were collected at 3 depths (2.5-7.6 cm, 15.2-20.3 cm, 30.5-35.6 cm) in the middle of each of plot using a slide hammer. Bulk density was calculated for each sample after oven drying at 105°C. Soil C data was calculated based on bulk density at related depths.

2.2.3 Density floatation and carbon fractionation

Density C fractionation is a well-established procedure for separating different (available, aggregate-protected and mineral-bound) C pools based on the density and stability of organic matter (Six et al., 1998; Wick et al., 2009a; Wick et al., 2012; Wick et al., 2009b). Soil C pool analysis was conducted using a density fractionation procedure described by Six et al. (1998) and modified by Wick et al (2009b). One 10-g sample per plot (replication) was taken from the pooled field samples for each depth at that plot and dried at room temperature (6 replications × 4 treatments × 3 depths = 72 samples). The samples were suspended in 35 mL of 1.85 g cm^{-3} density sodium polytungstate (SPT) in a 50 ml centrifuge tube and shaken gently by hand to bring the sample into suspension (approximately 10 strokes). Material on the lid was washed into the cylinder using 10 mL of SPT. Samples were then placed under vacuum (138 kPa) for 10 min to remove air trapped within aggregates. Samples were centrifuged for 60 min at 2500 rpm and floating material [free light fraction (LF)] was aspirated through a 20 mm nylon filter and rinsed with deionized water. The material on the filter was transferred into a beaker and dried at 55 ºC overnight. The material remaining in the centrifuge tube [particulate organic matter occurring within aggregates (iPOM), sand, silt + clay] was rinsed twice with deionized water, flocculated with five drops of 0.25M $CaCl_2$ + 0.25M $MgCl_2$ solution and centrifuged for 15 min at 2500 rpm. Twelve 6-mm glass beads were added to each centrifuge tube, which were then placed on a reciprocal shaker for 18 hours. Samples were removed from the shaker and sieved with nested 250- and 53-mm sieves for macroaggregate samples and a 53-mm sieve for microaggregate samples. Material remaining on the sieve (iPOM + sand) and material washed through the sieve (silt + clay) were dried at 55 ºC overnight and weighed. These measurements were used to detect *Available C* [indicated by light fraction (LF); has a short turnover time (1-5 yr); and is the least stable soil C pool] *Aggregate-protected C* [indicated by intra-aggregate particulate organic matter (iPOM); and has an intermediate turnover time (20-40 yr)] and *Mineral-bound C* [indicated by organic matter directly bound to silt and clay (SC); has the longest turnover time (200-1500 yr); and is the most stable soil C] (Parton et al., 1987; Wick et al., 2012).

2.2.4 Microbial biomass carbon

Soil samples were extracted from pooled soil by depths at each plot (see *2.2.2 Soil Sampling*) to assay MBC in June 2011. Two pairs of 25-g subsamples were then collected from each sieved sample and kept in a constant field-moist condition by placing in a sealed desiccator chamber along with 250 mL of distilled water for 24 hours. Samples were then processed for MBC assay according to the chloroform fumigation extraction method of Horwath and Paul (1994) with one sample from each pair being fumigated and one non-fumigated, resulting in two MBC assays per replicate per depth (6 replications × 4 treatments × 3 depths × 2 subsamples= 144). Fumigation destroys membranes and cell walls, allowing subsequent extraction of cell constituents with K_2SO_4 (0.5 M). The extraction procedure was performed on both fumigated and non-fumigated samples and extracted constituents were shipped frozen to the Soil Analytical Service Laboratory at North Carolina State University (Raleigh, North Carolina USA) for total organic C analysis using a TOC-5050 analyzer fitted with an ASI-5000 autosampler (Shimadzu Corporation, Kyoto, Japan). Total MBC of the extractant was calculated as the difference between total organic C in fumigated and non-fumigated samples using an extraction efficiency factor of 0.35 (Voroney et al., 1991). Soil MBC was then calculated by adjusting extractant volume for soil mass and correcting for gravimetric moisture content at the time of sampling.

2.2.5 Statistical analysis

Differences in soil C pools and MBC among soil treatments at the same depth were determined using one-way analysis of variance by using Tukey's HSD at $\alpha=0.05$ using PROC GLM in SAS software (SAS Institute, Inc., Cary, North Carolina). Data to be analyzed by parametric techniques were tested for normality via Shapiro-Wilk in SigmaPlot 12.0 (Systat Software Inc., Point Richmond, CA). Spearman's rank correlation was used to identify the relationship of total soil C, different soil C pools and MBC using SigmaPlot 12.0.

2.3 Results

2.3.1 Total soil carbon

In the surface 10 cm, undisturbed soils had greater total soil C (12.10 and 7.73 Mg C ha^{-1} at 0-5 cm and 5-10 cm, respectively) than the soils subjected to urban development practices. The TP treatment decreased total soil C at 0-5 cm, and, although not significantly, at lower sampling depths. Profile rebuilding, the urban soil rehabilitation treatment, had higher total soil C in subsurface soils (6.12 Mg C ha^{-1} at 15-30cm) compared to TP, ET and UN. There was no evidence that total soil C was different between TP and ET at any depth. Total soil C distribution was similar across measured depths in PR. In contrast, total soil C in the other three treatments decreased with depth (Table 2.1).

Table 2.1.
Total soil carbon at 0-5 cm, 5-10 cm and 15-30 cm soil depths in typical urban land development practice (TP), two post-development soil rehabilitation practices (ET and PR), and undisturbed control soil (UN). Values in parentheses are standard error of the mean (n=6). Within each soil depth, superscript letters indicate significant differences across soil treatments at α = 0.05 using Tukey's HSD.

	Total soil carbon (Mg C ha^{-1})			
Depth	TP	ET	PR	UN
0-5 cm	7.08 (1.54)b	5.95 (1.28)b	6.42 (1.25)b	12.10 (1.95)a
5-10 cm	5.85 (1.12)ab	5.15 (1.45)b	6.96 (2.31)ab	7.73 (1.22)a
15-30 cm	2.51 (0.57)b	2.66 (1.93)b	6.12 (3.11)a	3.79 (0.92)ab

2.3.2 Available, aggregate-protected and mineral-bound carbon pools

Undisturbed soils had greater available, aggregate-protected, and mineral-bound C pools than the other three treatments at 0-10 cm (Figure 2.1). The TP treatment reduced the size of all soil C pools at this depth compared to UN, even the mineral-bound C pool, which is considered the most stable. In subsurface soils, urban soil rehabilitation via PR resulted in higher available and aggregate-protected C storage than the other three treatments, including undisturbed (1.59 and 2.04 Mg C ha^{-1}, respectively; Figure 2.1).

Across all treatments, available and aggregate-protected C pools constituted 8.6% and 13.5% respectively of the total soil C while the mineral-bound C pool contributed 77.9%. Profile rebuilding increased the proportion of available and aggregate-protected C pools to total soil C, especially the aggregate-protected C pool (20.1%, 23.5% and 33.3% from topsoil to subsoil).

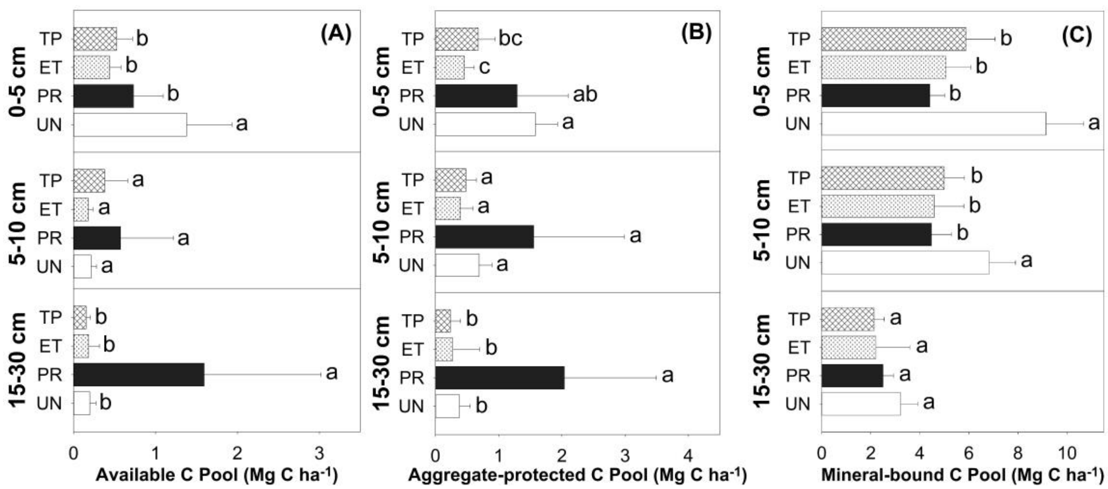

Figure 2.1. Available (A), aggregate-protected (B) and mineral-bound (B) carbon pools at three soil depths in typical urban land development practice (TP), two post- development soil rehabilitation practices (ET and PR) and undisturbed control soil (UN) four years after rehabilitation. Error bars depict standard error of the mean (n=6). Within each carbon pool in the same year at the same soil depth, letters denote differences among treatments using Tukey's HSD at α=0.05.

2.3.3 Microbial biomass carbon

Table 2.2.
Microbial biomass carbon at 0-5 cm, 5-10 cm, and 15-30 cm soil depths in typical urban land development practice (TP), two post-development soil rehabilitation practices (ET and PR) and undisturbed control soil (UN). Values in parentheses are standard error of the mean (n=6). Within each soil depth, superscript letters indicate significant differences across soil treatments at α = 0.05 using Tukey's HSD.

	Microbial biomass carbon (mg C kg^{-1} soil)			
Depth	TP	ET	PR	UN
0-5 cm	255.8 (96.3)a	249.5 (113.7)a	209.9 (118.8)a	287.3 (93.3)a
5-10 cm	196.0 (30.4)a	185.3 (47.8)a	256.4 (97.4)a	238.4 (55.8)a
15-30 cm	40.4 (20.2)b	45.1 (31.5)b	149.8 (89.8)a	90.5 (25.2)ab

Microbial biomass carbon (ranging from 185.3 to 287.3 mg C kg^{-1} soil) was not affected by soil treatment at 0-5 cm and 5-10 cm (overall p-values were 0.69 and 0.19, respectively). These values are in line with other regional data in that they are slightly higher than surface soil MBC measured at minimally disturbed upland forested sites in Georgia (Silveira et al., 2010a) and similar to surface soils at reclaimed mining sites in

Virginia 16-42 years after reclamation (Clayton et al., 2009). At 15-30 cm soil depth, however, PR had the greatest MBC (149.5 mg C kg^{-1} soil) while TP and ET had the least (40.4 and 45.1 mg C kg^{-1} soil, respectively). Undisturbed soils had an intermediate MBC (90.5 mg C kg^{-1} soil) (Table 2.2).

2.3.4 Relation of soil carbon pools to microbial biomass carbon

Table 2.3.
Correlation coefficients between microbial biomass carbon (MBC) and total soil carbon (C), available, aggregate-protected, and mineral-bound C pool at 0-5, 5-10 and 15-30 cm soil depths (n=24) using Spearman's rank correlation (*: $0.001<p<0.05$; **: $0.0001<p<0.001$; ***: $p<0.0001$).

Correlation coefficient (r)	MBC		
	0-5 cm	5-10 cm	15-30 cm
Total soil C	0.38	0.84***	0.85***
Available C pool	0.22	0.29	0.71***
Aggregate-protected C pool	0.18	0.80***	0.77***
Mineral-bound C pool	0.34	0.62 *	0.69**

Total soil C had a positive relationship with MBC at 5-10 cm and 15-30 cm depths ($p<0.0001$; r=0.68 and 0.85, respectively). The available soil C pool had a positive relationship with MBC only at the 15-30 cm depth ($p<0.0001$; r=0.71). The aggregate-protected C pool had a positive relationship with MBC at both 5-10 cm and 15-30 cm depths ($p<0.0001$; r=0.80 and 0.77, respectively). The mineral-bound C pool had a positive relationship with MBC at both 5-10 cm and 15-30 cm depths ($p<0.05$ and $p<0.001$, r=0.62 and 0.69, respectively). At 0-5 cm depth, there was no relationship between MBC and different soil C pools. However, at 15-30 cm soil depth, MBC was positively related to total soil C, available C, aggregate-protected C and mineral-bound C pools (Table 2.3).

2.4 Discussion

In our study, soil management practices typical of urban land development, as represented by TP, resulted in losses of soil C in surface soils when compared to undisturbed sites. This is consistent with studies conducted in street tree pits (Jim, 1998a, b), but in contrast with other studies (Golubiewski, 2006; Pouyat, 2002; Pouyat et al., 2009; Pouyat et al., 2007; Qian and Follett, 2002; Raciti et al., 2011) which found soil C content was higher in urban areas than rural areas. However, this second group of studies were either conducted in remnant urban forests or parks that may have not experienced the disturbance typical of urban development, or on highly maintained home lawns that have received significant fertilizer and C additions over a number of years from root turnover or grass clippings. Research on residential landscapes in Idaho and Washington State (Scharenbroch et al., 2005) and in school yards in Ohio (Park et al., 2010) both with silt loam soils found that soil C storage typically increases with age since development. Based on a study estimating C dynamics of urbanized/developed lands in the Southern United States from 1945 to 2007 using a process-based model (Dynamic Land Ecosystem Model), Zhang et al. (2012) proposed that soil C storage rapidly decreases at the beginning of land conversion to urban uses, and then gradually increases. This suggests that there may be potential for changes in soil C management soon after development to accelerate C sequestration over the long term.

Our study indicates that simply short time stockpiling and replacing surface soils with no additional disturbance can result in significant C losses after 4 years. In fact, the surface soil removal and replacement represented by TP reduced the mineral-bound C pool as well as less stable pools in surface soils. As the most stable C pool, the mineral-bound C was expected to be unaffected by short-term topsoil handling such as topsoil removal, stockpiling and replacement. Studies have observed soil C losses due to disturbance, but multiple contributing factors make it difficult to determine the driver of the observed C losses (Brevik et al., 2002; DeBusk et al., 2005; Silveira et al., 2009; Whitecotton et al., 2000). For example, at a military training site at Fort Benning, Georgia, a series of studies evaluated the effects of military training related disturbance (e.g., foot traffic, tank training exercises) on soil C storage. Silveira et al. (2010b) found the high level of surface disturbance due to foot traffic and tank training exercises reduced total soil C compared to low-impact sites. However, unlike our study, disturbance occurred repeatedly and resulted in significant loss of vegetation compared with the low-impact sites. In a companion study, Debusk et al. (2005) observed the reduction in SOM content of surface horizons with increasing levels of disturbance and proposed that chronic physical disturbance of topsoil might accelerate decomposition of the more recalcitrant soil organic matter fractions. Furthermore, decreased MBC suggested that labile C also decreased with increasing level of disturbance (Silveira et al., 2010b). However, we did not find any differences in MBC due to treatment in the surface 10 cm of soil and conditions contributing to C loss in these studies (loss of topsoil, vegetation and forest floor removal, increased soil erosion and sedimentation) did not occur in our study. Instead, the only soil manipulation was physical: scraping, stockpiling, and replacement. Previous reports of loss of soil C during urbanization (Beesley, 2012; Jim, 1998a, b) did not differentiate among C pools. Our study, however, found C losses in all pools, including the most stable, mineral-bound C. Our findings suggest that even the most stable mineral-bound soil C may be released by the soil disturbance that commonly results from urban development. Soil disturbance during urbanization can be significantly more severe where deep cuts or fills are required or soil is compacted in lifts, suggesting that C losses may be greater in many cases.

Wick et al. (2009c) observed changes in aggregate size distribution after topsoil stockpiling and replacement on a coal mining site. Macroaggregate proportions increased after three years of storage, but decreased following redistribution on the site, while microaggregate proportions had the opposite trend, suggesting that any physical soil movement destroys some soil aggregates even if tilling is not involved. Furthermore, disturbance might lead to more free microaggregates resulting in reduction of C stabilization within microaggregates (Six et al., 2000). Thus, the proportion of stable macroaggregates and the enrichment of C concentration within microaggregates were suggested as two main protection mechanisms of maintaining soil C (Plaza-Bonilla et al., 2013). However, prior to our findings, no studies to our knowledge have documented mineral-bound C pool reduction after disturbance. We hypothesize the loss of mineral-bound C was associated with breaking down of aggregates during the construction processes which disrupted organic matter-mineral complexes and released C. However, whether C was released during stockpiling, after redistribution, or throughout the study period is not known.

As we expected, the PR treatment increased total soil C and MBC at 15-30 cm depth. Furthermore, PR increased both available and aggregate-protected C pools four years after organic amendment incorporation. Carbon inputs can add to the available C pool immediately, but it takes time for C to be integrated into the more stable C pools such as the aggregate-protected C pool. In our study, we observed that organic matter was transferred to the aggregate-protected C pool in the PR treatment at 15-30 cm over 4 years. Most previous studies concluded that the available C pool is the most sensitive pool to management practices such as tillage (Bremer et al., 1994; Haynes, 2005). However, we planted trees with compost application at our site. Thus, the tree root-soil-compost complex may have contributed to the rapid increase in the aggregate-protected C pool. Tree roots can accelerate soil aggregation, input C, and increase biological activity (Day et al., 2010). Moreover, higher MBC was found at the lower depth in rehabilitated soil. The observed increases in available and aggregate-protected C pools may be a result of the incorporation of compost amendments at depth which facilitated more microbial biomass by supplying more food resources, and a subsequent microbial contribution to available and aggregate-protected C pools. These observations suggest that urban soil management practices can affect deeper soil layers as well as surface layers and have potential to alter soil C sequestration through different mechanisms. Compost amendment can provide C and energy sources to soil microbes, and thus can increase soil MBC (Bastida et al., 2012; Pascual et al., 1999; Wiseman et al., 2012). A meta-analysis (Kallenbach and Grandy, 2011) indicated that even small amounts of organic amendments can rapidly increase microbial biomass in agricultural systems. Our findings suggest that urban soil rehabilitation by compost amendment and soil loosening via deep tillage could increase MBC in urban soils, potentially mediating movement of C into aggregate-protected pools.

In our study, soil C loss resulting from the TP treatment (compared to undisturbed soil) was 6.9 Mg C ha^{-1} in the surface soil (0-10 cm). Thus the TP treatment resulted in a 35%

loss of C in surface soils (0-10 cm), which increased to 44% with the additional tilling of the ET treatment. Carbon was lost from all three pools, yet notably these practices resulted in 32% (TP) and 40% (ET) losses in the mineral-bound C, the largest and most stable pool. Typical development practices thus have the potential to lead to significant C loss as urban land area is predicted to triple between 2000 and 2030 (Seto et al., 2012). Surface soils in rehabilitated soil experienced a similar total C loss (34%), but distribution among C pools was significantly altered. Rehabilitated soils experienced an increase of 25% in the aggregate-protected pool that was offset by a 44% loss of mineral-bound C, while changes in available C were minimal (18% loss). The total C loss could be significantly greater if soils below 10 cm depth are accounted for. Rehabilitated urban soil via profile rebuilding can increase soil C by 3.61 Mg C ha^{-1} in lower soil regions (15-30 cm). Hence, urban soil rehabilitation could sequester C in subsurface soils, which could potentially mitigate half the C loss resulting from surface soil handling. Moreover, the potential of urban soil rehabilitation to increase soil C sequestration could be greater than observed in this study. First of all, compost was incorporated up to 60 cm, but we only measured soil C changes within the upper 30 cm depth. In addition, urban land development often results in more severe subsoil grading and compaction than in our study, especially when significant changes in elevation are required to support buildings and other structures. Thus potential C losses in subsoils would be magnified resulting in greater potential absolute increases from subsurface soil rehabilitation. Our results suggest that C loss occurs during urban land development, but can be mitigated by soil rehabilitation practices such as profile rebuilding that includes subsoiling and deep compost incorporation. Additional studies of greenhouse gas emission in developed urban land and rehabilitated urban soils would help more fully evaluate rehabilitation effects on C sequestration and atmosphere-soil C efflux mitigation.

2.5 Conclusions

Urban land development practices with a wide range of topsoil removal, handling, grading, compacting, and building result in losses of C in every soil C pool, even the most stable mineral-bound C pool in surface soil. This study demonstrated how urban construction activities can affect soil C storage at the beginning of the urbanization process. This effect is surprisingly strong because it appears to reduce the stable soil C pool as well as more labile pools. Profile rebuilding as a post-development soil rehabilitation practice offers potential to store more C in subsoil by increasing available and aggregate-protected C pool at greater soil depths. There are several possible factors responsible for these changes: soil physical properties (aggregates) affect C storage in surface soils; while the soil microbe-tree root-organic matter complex may be the driving force in the soil C cycle at greater depths. Profile rebuilding can offset more than half of the C loss resulting from topsoil handling. However, our study site was in a humid temperate region previously in agricultural use, thus, the results may be most relevant to sites with similar climate and land use history. In conclusion, typical urban land development practices degrade soil and lead to C loss, even from the most stable soil C pools, however, profile rebuilding can rehabilitate degraded urban soil to offset C loss.

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Chapter 3

Greenhouse Gas Efflux from Undisturbed, Compacted, and Rehabilitated Soil

Abstract

Urban land development and subsequent soil management can affect efflux of the major greenhouse gases (GHG) including carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4). Rehabilitation of degraded urban soils through deep tillage and organic amendment may improve the soil's ability to support vegetation, but its effect on soil GHG emissions is unknown. In 2007, twenty-four plots were either left undisturbed or subjected to topsoil stripping, grading and compacting to mimic typical land development followed by one of three treatments:10 cm topsoil replaced (typical practice); typical practice plus tilling (enhanced practice); compost incorporation to 60-cm depth, followed by typical practice plus tilling (rehabilitation). Soil-atmosphere CO_2, N_2O, and CH_4 efflux was measured seasonally from fall 2011 to summer 2013. Post-development soil rehabilitation resulted in greater CO_2 fluxes (ranging from 0.8 to 12.0 g C m^{-2} d^{-1}) than other treatments (ranging from 0.2 to 5.5 g C m^{-2} d^{-1}) in all seasons except winter 2012. All soils were CH_4 sinks (ranging from -0.8 to -0.2 mg C m^{-2} d^{-1}) but no treatment-related differences were evident. Likewise, N_2O fluxes were largely consistent across treatments. Overall, rehabilitated soils had greater total GHG emissions (CO_2eq ranging from 0.7 to 12.1 g C m^{-2} d^{-1}) than both undisturbed soils and those subjected to typical development practices (CO_2eq ranging from 0.1 to 5.6 g C m^{-2} d^{-1}), which did not significantly differ.

Increased GHG emissions from rehabilitated soils may be offset by increased plant biomass production on these sites, but our study only assessed soil-atmosphere fluxes.

Keywords: carbon dioxide, methane, nitrous oxide, urbanization, climate change, landuse change

3.1 Introduction

Over the last three decades, greenhouse gas (GHG) emissions have increased by an average of 1.6% per year (IPCC, 2007). Global carbon dioxide (CO_2) emissions, the greatest overall contributor to global warming, have increased by 3.4 Pg C y^{-1} from 1990 to 2011 with average annual growth rates of 1.9% per year in the 1980s, 1.0% per year in the 1990s, and 3.1% per year since 2000 (Peters et al., 2012). By 2030, GHG emissions are projected to have increased by 25-90% compared to 2000 (IPCC, 2007). Thus, there are compelling reasons to attempt to mitigate climate change by offsetting GHG emissions across a broad range of human activities. Modern climate change is dominated by human influences and land use change resulting from urbanization is one important driving force (Karl and Trenberth, 2003). A significant element of land use change is the soil disruption that typically accompanies urban land development, including topsoil removal and replacement, grading, compacting and building. Beyond initial urbanization, urban areas are dominant anthropogenic sources of GHG emissions, with large and increasing emissions resulting from the construction and operation of cities (Grimmond, 2007). Some of these emissions likely relate to soil movement. Since 70% of the world's population is expected to live in urban areas by 2050 (Seto and Shepherd, 2009), maximum average annual per capita GHG emission could be more than 15 tons CO_2

equivalent (CO_2eq) (Hoornweg et al., 2011). Thus, reducing soil-movement-related GHG emissions during urban development or re-development is a potential strategy to mitigate regional GHG emissions.

Some authors have suggested that urban soils have potential to sequester large amounts of soil organic carbon (SOC) and consequently mitigate GHG emissions (Lal, 2003; Lorenz and Lal, 2009; Pouyat et al., 2006). Along with carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4) are major GHGs contributing to global climate change. North America is a net atmospheric CO_2 source even though net primary production offsets about 35% of the fossil-fuel-based CO_2 emissions by transferring atmospheric C to vegetation, soils, and wood products (King et al., 2012). The higher urban emissions lead to atmospheric CO_2 levels in urban areas that are much higher than in nonurban areas. For example, in Phoenix, Arizona, the CO_2 levels in the metropolitan region were 50% greater than the surrounding non-urban areas. Eighty percent was from anthropogenic sources (Koerner and Klopatek, 2002) and only 16% from soils. However, it is still unclear whether urban soils are a net CO_2 source or sink and to what degree this is influenced by management practices. In Fort Collins, Colorado, CO_2 emission from urban lawns was more than five times that from native grasslands. In Chicago, Grimmond et al. (2002) found the atmospheric CO_2 concentration was elevated in the city. However, they also observed negative surface-atmosphere CO_2 fluxes in daylight during the growing season over a suburban forested measurement site, indicating the significant effect of urban vegetation on GHG emission and C sequestration through photosynthesis. Nowak (1994) demonstrated that urban trees in the city of Chicago can store about 855,000 Mg of C. Urban vegetation can affect the C cycle via above ground canopy photosynthesis and below ground root turnover (Jha and Mohapatra, 2010; Nowak and Crane, 2002).

Differences in GHG emissions between urban and rural areas may also vary from region to region and with management practices. For example, in Phoenix, Arizona, CO_2 fluxes were lower in urban sites compared to rural sites along an urban-rural gradient in a desert landscape (Koerner and Klopatek, 2010). This variability in emissions is likely related to climate and moisture as well as the wide variety of land management practices found in urban areas. Urban lawns have been studied intensively recently and may play a significant role in regional soil-atmosphere gas exchange budgets even given the small portion of the landscape represented by lawns (Kaye et al., 2004). However, CO_2 fluxes from urban grasses likely differ from those of urban forests (Groffman et al., 2009). Nonetheless, most studies to date primarily compare the differences in CO_2 fluxes between urban and nonurban systems; while few have explored whether these differences can be diminished via soil or vegetation management practices.

Nitrous oxide is a more powerful greenhouse gas than CO_2 (its global warming potential is 298 times of CO_2 in a 100 year horizon), although it is a much smaller contributor to the global warming effect (Dalal et al., 2003). During 1990-2010, biogenic N_2O emissions ranged from 0.802 to 1.016 Pg CO_2eq yr^{-1}. Projected N_2O emissions will increase 157-227% by the end of this century (Tian et al., 2012). Previous studies have documented that urbanization increases N_2O emissions compared to natural lands. In a study conducted by Kaye et al. (2004) in Fort Collins, Colorado, they found that urban lawns had more than 10 times the N_2O emissions of natural grasslands. In Phoenix, Arizona, urban lawns also had higher N_2O emissions compared to natural landscapes (Hall et al., 2008). In southern California, however, N_2O emissions from urban landscapes were approximately equal to or only slightly greater than agricultural emissions (Townsend-Small et al., 2011). Fertilizer, soil moisture, soil temperature, and the frequency of disturbance, and soil amendments have been reported as factors associated with increased N_2O emissions in urban soils (Bijoor et al., 2008; Groffman et al., 2009; Huang et al., 2004; Maggiotto et al., 2000). However, the heterogeneity of urban soils leads to uncertainties concerning the significance of these factors and their interactions.

Methane (CH_4), also a powerful GHG, is produced in anaerobic soil environments. Soils represent the most important source of atmospheric CH_4 (Schütz et al., 1990). During 1990-2010, biogenic CH_4 emissions ranged from 0.16 to 0.50 Pg of CO_2eq per year. Projected CH_4 emissions will increase by 137-151% by the end of this century (Tian et al., 2012). On the other hand, many soils serve as a CH_4 sink, playing a significant role in the global atmospheric methane budget (Dutaur and Verchot, 2007). Soils can take up about 40 Tg CH_4 from the atmosphere annually. Atmospheric methane consumption is especially sensitive to anthropogenic disturbances, which typically decrease methane consumption (King, 1997). Methane uptake capacity varies by land use, climate, latitude, rainfall, temperature, and soil texture (Dutaur and Verchot, 2007; Goldman et al., 1995; Kaye et al., 2004). Temperate forests with coarse soil texture tend to have large CH_4 consumption (Dutaur and Verchot, 2007). It has been documented that urbanization reduces soil CH_4 uptake capacity. Goldman et al. (1995) observed lower CH_4 consumption rates in urban forests than in rural forests along an urban to rural land-use gradient. In Fort Collins, Colorado, CH_4 uptake was reduced by about half in urban lawns compared to natural grasslands (Kaye et al., 2004). Moreover, in Baltimore, Maryland, CH_4 uptake capacity was almost completely eliminated in urban lawns compared to rural forests (Groffman and Pouyat, 2009). Whether soil management practices could restore eliminated CH_4 uptake capacity is uncertain.

Several techniques for mitigating GHG emissions through soil management, such as biochar application, deep soil C sequestration, and genetic modification of plants have been studied (as reviewed in Post et al., 2012). In our study, we address whether rehabilitating degraded urban soils via a soil preparation practice that is desirable in urban areas for vegetation management can contribute to GHG mitigation. Typical urban land development practices including clearing, topsoil removal, surface grading, compacting and building construction lead to degraded urban soils which could influence GHG emissions. Soil profile rebuilding (SPR) is a technique developed to rehabilitate degraded urban soils to better support vegetation. This additional soil treatment mitigates the subsurface soil compaction that results from typical urban land development practices via deep tillage and compost amendment. Previous work with SPR indicates post-development rehabilitated soils can sequester more C, improve tree growth than nonrehabilitated soils and perform on a par with pre-development soils (Chen et al., 2013; Layman, 2010). However the effect of urban land development with or without post-development rehabilitation on GHG emissions is not known. Our objectives were to: (1) compare GHG emissions of soils subjected to typical urban land development practices with undisturbed soils; and (2) assess the GHG emission effect of post-development soil rehabilitation.

3.2 Materials and Methods

3.2.1 Site information

The study site in Montgomery County, Virginia (37°12' 1.1888" N, 80° 33’ 48.3768" W) contains two closely related loamy soils, Shottower loam (fine, kaolinitic, mesic Typic Paleudults) and Slabtown loam (fine-loamy, mixed, mesic Aquic Paleudalfs) and was previously under pasture landuse. Between May and November in 2007, 24 (6 replications × 4 soil treatments) 4.6 × 18.3 m plots were installed in a completely randomized experimental design. Treatments included an undisturbed control typical urban land development practice and two post-development soil management practices. With the exception of the control plots, all plots were subjected to pre-treatments that replicated the scraping and compaction, typical of land development: the A horizon was removed and stockpiled adjacent to the study site, and the exposed subsoil surface was compacted via 8 passes of a 4,808 kilogram ride-on sheep’s foot vibrating compactor Ingersoll Rand (Model SD45D) to an average bulk density of 1.98 g cm^{-3}. In addition to the undisturbed control (UN), treatments included: typical practice (TP), 10-cm of stockpiled topsoil applied to the compacted subsoil; enhanced topsoil (ET), same as TP, except tilled to 10-15 cm depth; and profile rebuilding (PR), a soil rehabilitation technique that focuses on subsoil loosening and deep compost amendment. The PR treatments includes: 10 cm of leaf compost (C/N ratio 15.0) applied to the surface; deep tillage with a backhoe to a 60-cm depth; and replacement of 10 cm topsoil. The control (UN), represents pre-urban development soil and thus serves as a benchmark for soil rehabilitation. The TP treatment is similar to typical urban land development practices employed in the United States. The PR treatment is a recommended soil rehabilitation practice, while ET is a modification of TP sometimes specified by site designers. Five species of trees (*Ulmus japonica* (Rehd.) Sarg. x *wilsoniana* Schneid. ‘Morton’, *Acer rubrum* L., *Quercus bicolor* Willd., *Prunus* ‘First Lady’ (*P.* ×*incam* Ingram ex R. Olsen & Whittemore ‘Okamé’ × *P. campanulata* Maximowicz), and *Quercus macrocarpa* Michx.) were planted in each plot from February to April in 2008.

3.2.2 Gas sampling and lab analysis

Soil-atmosphere CO_2, CH_4 and N_2O fluxes were measured by vented static chambers (Hutchinson and Livingston, 2001) every three months (seasonally) over two years from fall 2011 to summer 2013. In July 2011, 24 polyvinyl chloride (PVC) chambers (height=17.8 cm and diameter=25.4 cm) were installed 1.5 m away (northeast) from the *Quercus bicolor* tree in each plot. PVC chambers were inserted approximately 7.5 cm into the soil and leaving 10 cm above ground. They were installed more than 1 month before the first measurement to mitigate the effect of placement disturbance on GHG emissions. The chambers were left open in the field to mimic the natural environment, but covered with PVC end caps to close the chamber during gas sampling. Each cap was equipped with a vent and a sampling port with a rubber septum. At 0, 30, 60 and 90 min, 10 ml gas samples were collected from the sampling port with a syringe after covering with the lid. The zero-minute samples were taken immediately after each chamber was covered. Samples were then transferred to sealed pre-vacuumed vials (10 mL) and stored at room temperature until analysis. The gas samples were analyzed by gas chromatography (Shimadzu GC-14A). The concentration of CO_2 was detected by a thermal conductivity detector (TCD) at 100°C and the concentration of CH_4 was determined by a flame ionization detector (FID) at 150°C. The concentration of N_2O was analyzed after modifying the gas chromatograph configuration, which was fitted with a precolumn and an analytical column. Fluxes were calculated from the linear rate of change in gas concentration, the chamber internal volume, and soil surface area.

3.2.3 Carbon dioxide equivalents calculation

Negative fluxes of GHG indicate the uptake of a given gas by soil and positive fluxes indicate net emissions from soil. The global warming potential (GWP) of CH_4 and N_2O are 25 and 298 times that of CO_2 in an 100-year timeframe (IPCC, 2007). Therefore, CO_2eq was estimated by multiplying CO_2, CH_4, and N_2O effluxes by 1, 25, and 298, respectively; and adding the three gases up for the total (Eq.1).

$$\text{GWP } (CO_2\text{eq, g } CO_2\text{-C m}^{-2}\text{ d}^{-1}) = CO_2\text{-C (g m}^{-2}\text{ d}^{-1}) + CH_4\text{-C (g m}^{-2}\text{ d}^{-1}) \times 25 + N_2O\text{-N (g m}^{-2}\text{ d}^{-1}) \times 298 \qquad \text{(Eq.1)}$$

3.2.4 Precipitation, air temperature, soil moisture and soil temperature measurements

Volumetric soil moisture and soil temperature were measured immediately after gas sampling at each sampling time. Volumetric soil moisture was measured by time-domain reflectometry (Trase System I, Soilmoisture Equipment Corp, Santa Barbara, California) at 0-10cm soil depth. Soil temperature was measured by Traceable® Workhorse™ thermometer from Control Company, Friendswood, TX at 20 cm depth. Daily precipitation and air temperature (maximum and minimum) were monitored by the College of Agriculture and Life Sciences Kentland Farm's Dynamet weather station (assembled by Dynamax, Inc., Houston, TX using Campbell Scientific, Logan, UT, hardware and a model CR10X data logger).

3.2.5 Statistical analysis

Differences of CO_2, CH_4, N_2O fluxes and total CO_2eq between soil treatments in the same season were determined using one-way analysis of variance using PROC GLM in SAS software (SAS Institute, Inc., Cary, North Carolina). Linear regression was used to identify the relationship of CO_2, CH_4, N_2O fluxes, total CO_2eq and soil moisture, soil temperature using SigmaPlot 12.0 (Systat Software Inc., Point Richmond, CA).

3.3 Results and Discussion

3.3.1 Environmental factors

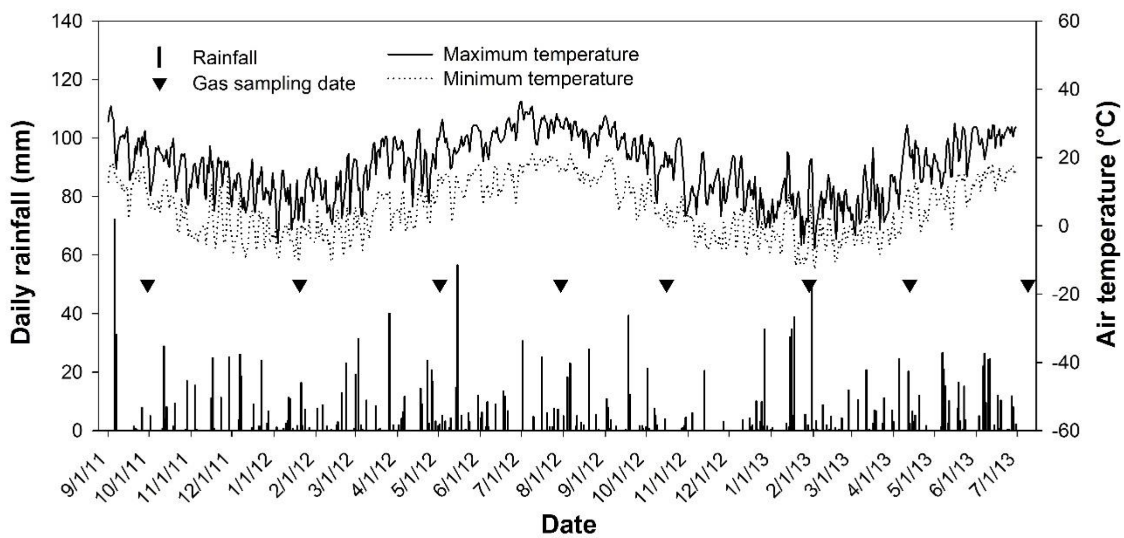

Figure 3.1. Daily rainfall (mm) and maximum and minimum air temperature (°C) during the experimental period from September 2011 to July 2013.

In 2012, the annual rainfall was 945 mm. The mean maximum and minimum temperatures were 19.1 °C and 6.1 °C, respectively (Fig. 3.1). Soil temperature ranged from 5.1 °C to 29.8 °C during the study period. Volumetric soil moisture at 0-10 cm depth within PVC chambers ranged from 6.9% to 38.3% from fall 2011 to summer 2013 (Fig. 3.2 A).

3.3.2 CO_2 flux

The post-urbanization rehabilitation treatment PR via compost incorporation and deep tilling had greater CO_2 fluxes (ranging from 0.8 to 12.0 g C m^{-2} d^{-1}) than the other three treatments (ranging from 0.2 to 5.5 g C m^{-2} d^{-1}) in all seasons from fall 2011 to summer 2013. There are few CO_2 studies of compost application in urban areas. One study conducted by Livesley et al. (2010) in Victoria, Australia reported that surface organic mulch application had no effect on CO_2 emissions from lawn and mulched garden beds. The lack of effect from organic amendment observed in their study might be due to difference in methods of organic amendment application or climate variation. The Australia study site received only 681 mm of precipitation per year, considerably drier than our site. Moreover, the C:N ratio of any incorporated amendment may influence CO_2 emissions (Huang et al., 2004). In our study, there was no evidence to show that CO_2 fluxes were affected by the other two soil treatments compared to undisturbed soil. This demonstrated that newly created urban soils have similar CO_2 emissions compared to natural soils. This finding is different from previous studies (Byrne et al., 2008; Kaye et al., 2005; Koerner and Klopatek, 2010) that reported that urban landscapes had greater CO_2 emissions compared to natural landscapes. Among four seasons, CO_2 had the least flux (ranging from 0.2 to 0.8 g C m^{-2} d^{-1}) in winter than during the other three seasons (ranging from 0.8 to 12.0 g C m^{-2} d^{-1}) (Fig. 3.2).

3.3.3 CH_4 uptake

All soil treatments were CH_4 sinks (ranging from -0.8 to -0.2 mg C m^{-2} d^{-1}). However, there was no evidence to show that CH_4 fluxes were affected by different soil treatments in any season (Fig. 3.2). Methane consumption capacity did not change between pre- and post-development soils as represented by the UN and TP treatments. This is contrary to findings by others that urbanized land had greatly reduced CH_4 uptake capacity (Groffman and Pouyat, 2009; Kaye et al., 2004). Two possible reasons for this difference are: (1) there was no irrigation in our study, reducing the likelihood of anaerobic conditions and (2) soils in other studies may have been more highly disturbed and thus more poorly drained than those in our study (Goldman et al., 1995; Kaye et al., 2004).

3.3.4 N_2O flux

In spring 2012, PR had greater N_2O fluxes (0.9 mg N m^{-2} d^{-1}) than the other three treatments (ranging from 0.1 to 0.3 mg N m^{-2} d^{-1}). The compost incorporation that was a part of the PR treatment may have increased biological activity in the soil. Soil microorganisms are controllers of GHG efflux (Conrad, 1996) and they are more active in spring. Our finding is consistent with other urban N_2O studies which reported urban soils have higher or equal N_2O emission than rural soils (Kaye et al., 2004; Townsend-Small et al., 2011). There was no evidence to show that N_2O fluxes were affected by different soil treatments in other seasons (ranging from -0.01 to 0.4 mg N m^{-2} d^{-1}). In our study, N_2O had higher flux (ranging from 0.03 to 0.9 g N m^{-2} d^{-1}) in spring than other three seasons (ranging from -0.01 to 0.4 mg C m^{-2} d^{-1}) among the four seasons (Fig. 3.2).

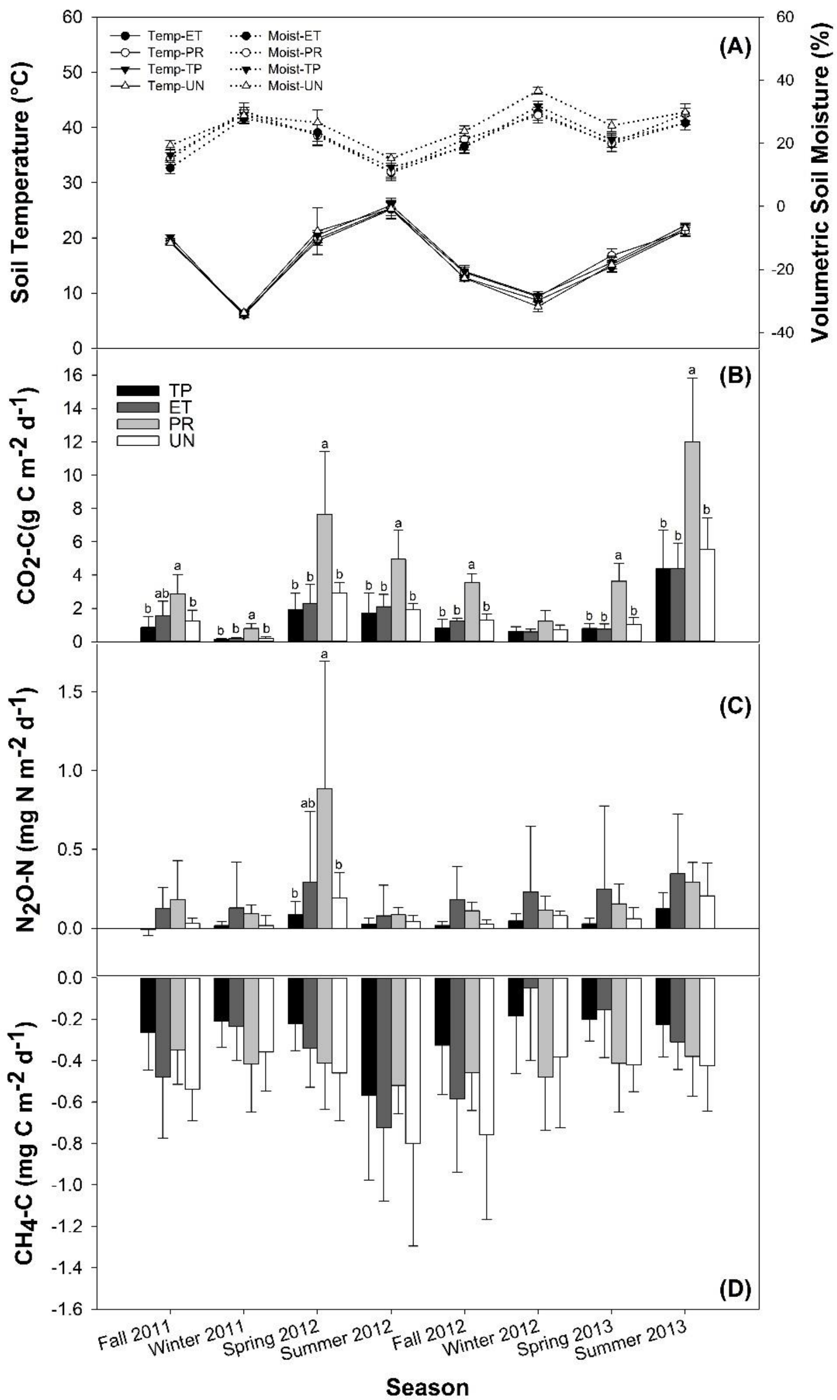

Figure 3.2. Mean soil temperature (°C) at 20 cm soil depth and volumetric soil moisture (%) at 0-10 cm soil depth of different treatments (n=6) on each sampling date (A). Seasonal CO_2 (B), N_2O (C), and CH_4 (D) fluxes from typical urban land development practice (TP), two post- development soil rehabilitation practices (ET and PR) and undisturbed control (UN) soils to atmosphere from fall 2011 to summer 2013. Error bars depict the standard error of the mean (n=6). In each season, letters denote differences among treatments using Tukey's HSD at α=0.05.

3.3.5 CO_2 equivalents

Table 3.1. Carbon dioxide equivalents (g C m^{-2} d^{-1}) of typical urban land development practice (TP), two post- development soil rehabilitation practices (ET and PR) and an undisturbed control soil (UN) from fall 2011 to summer 2013.

	Carbon dioxide equivalents (g C m^{-2} d^{-1}) †							
	Seasons							
Treatments	Fall 2011	Winter 2011	Spring 2012	Summer 2012	Fall 2012	Winter 2012	Spring 2013	Summer 2013
TP	0.8 (0.6)b‡	0.2 (0.1)b	1.9 (1.0)b	1.7 (1.2)b	0.8 (0.6)b	0.6 (0.3)	0.8 (0.3)b	4.4 (2.3)b
ET	1.6(0.9)ab	0.2 (0.1)b	2.4 (1.1)b	2.1 (0.8)b	1.3 (0.2)b	0.7 (0.2)	0.8 (0.4)b	4.5(1.5)b
PR	2.9 (1.2)a	0.8 (0.3)a	7.9 (4.0)a	5.0 (1.7)a	3.6 (0.6)a	1.2 (0.7)	3.7 (1.1)a	12.1(3.8)a
UN	1.3 (0.7)b	0.3 (0.1)b	3.0 (0.6)b	1.9 (0.4)b	1.3 (0.4)b	0.7 (0.3)	1.0 (0.4)b	5.6 (1.9)b

†n=6. Values in parentheses represent standard errors of the means.

‡Within each season, letters denote differences among treatments using Tukey's HSD at α=0.05.

Profile rebuilding had greater total GHG emissions (0.8 to 12.1 g CO_2-C equivalent m^{-2} d^{-1}) than other treatments (0.2 to 5.6 g CO_2-C equivalent m^{-2} d^{-1}) through all seasons except winter 2012. ET and TP had no effects on CO_2eq compared to UN (Table 3.1). Carbon dioxide is the primary contributor to net GHG emission (ranging from 96.9%-99.6%). Methane alone could offset only a very small portion of GHG emission (from -0.2% to -0.7%). Profile rebuilding had a lower CH_4 consumption rate (0.2%) than other treatments (0.5-0.7%). This lower consumption might due to higher CO_2 emission and related higher total GHG emissions in PR (Table 3.2).

Table 3.2. Mean daily GHG emissions (CO_2 equivalents) of typical urban land development practice (TP), two post- development soil rehabilitation practices (ET and PR) and an undisturbed control soil (UN) (n=192) four/five years after experiment installation. Data is based on repeated measurements over two years. The percent contribution to global warming potential (GWP) is the contribution of CO_2, N_2O and CH_4 to the total GHG emissions.

Treatments	CO2 equivalents (g C m^{-2} d^{-1})	Percent contribution to GWP† (%)		
		CO_2	N_2O	CH_4
TP	1.4	99.6	0.9	-0.5
ET	1.7	96.9	3.6	-0.5
PR	4.6	98.7	1.5	-0.2
UN	1.9	99.4	1.3	-0.7

† GWP (CO_2eq) = CO_2-C ×1 + N_2O-N ×25 + CH_4-C ×298

3.3.6 Relations of GHGs to soil temperature and soil moisture

Overall, total GHG emissions were positively related to soil temperature ($R^2 = 0.21$, $p<0.001$); there was little evidence showing total GHG emissions was associated with volumetric soil moisture (Figs. 3.3 and 3.4). In our study, soil temperature was more consistent across treatments than soil moisture. Furthermore, soil moisture did not change greatly throughout the year. Carbon dioxide was the dominant gas and had similar relationships with soil temperature and moisture as the total GHG emissions. In addition, CH_4 uptake was weakly related to soil temperature and soil moisture (p-values were <0.01 and < 0.001, respectively). However, there was little evidence showing N_2O emission was association with soil temperature and soil moisture (p-values were 0.06 and 0.92, respectively) (Figs. 3.3 and 3.4). Soil water content has been recognized as one of main controlling factors for methane oxidation compared to soil temperature (Jang et al., 2006). Soil moisture could affect the direction of the soil CH_4 flux from sink to source (Castro et al., 2000). Moreover, it can also influence the net CH_4 flux rate. In the dry tropical region of India, Singh et al.(1997) observed that the mean uptake rate was higher in dry seasons than in the rainy season. Furthermore, they found the CH_4 uptake rate was positively related to soil moisture in dry summers but negatively related to soil moisture in other wetter seasons, suggesting there is an optimal soil moisture level for methane uptake.

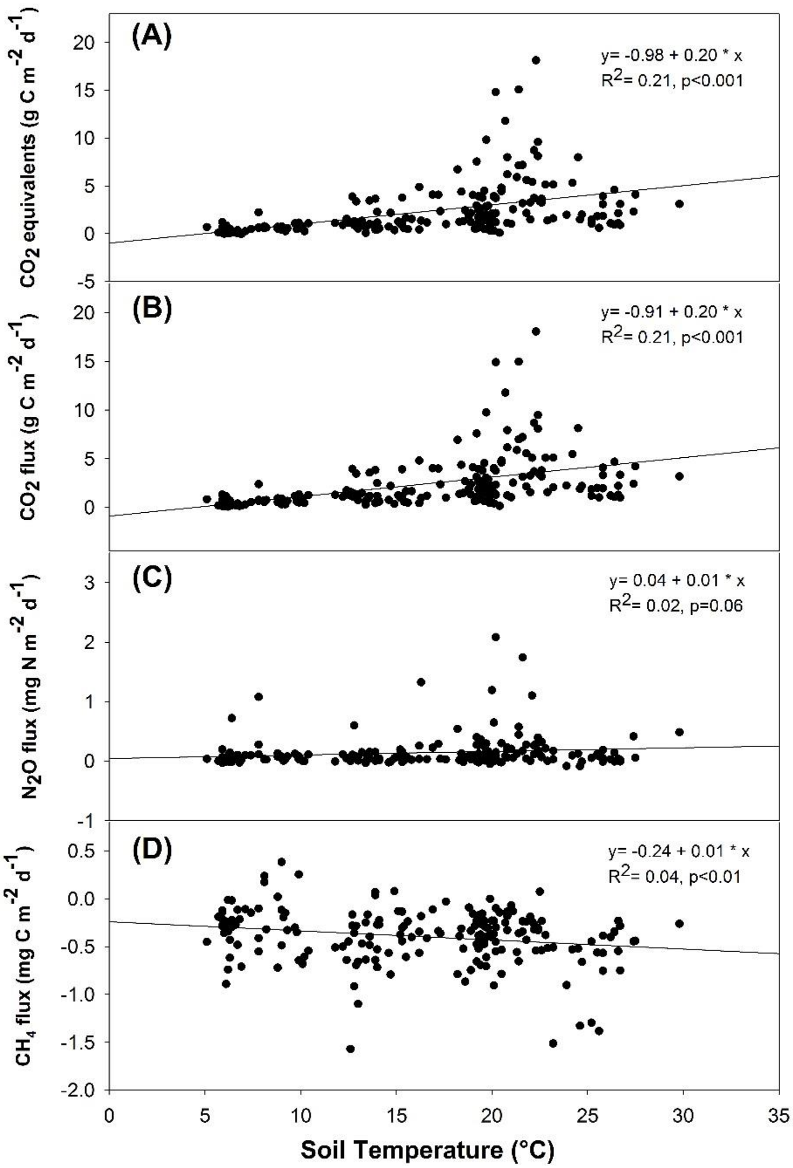

Figure 3.3. Relationship between CO_2 equivalents (A), CO_2 (B), N_2O (C), and CH_4 (D) fluxes from soil to atmosphere and soil temperature (20 cm soil depth) (n=192).

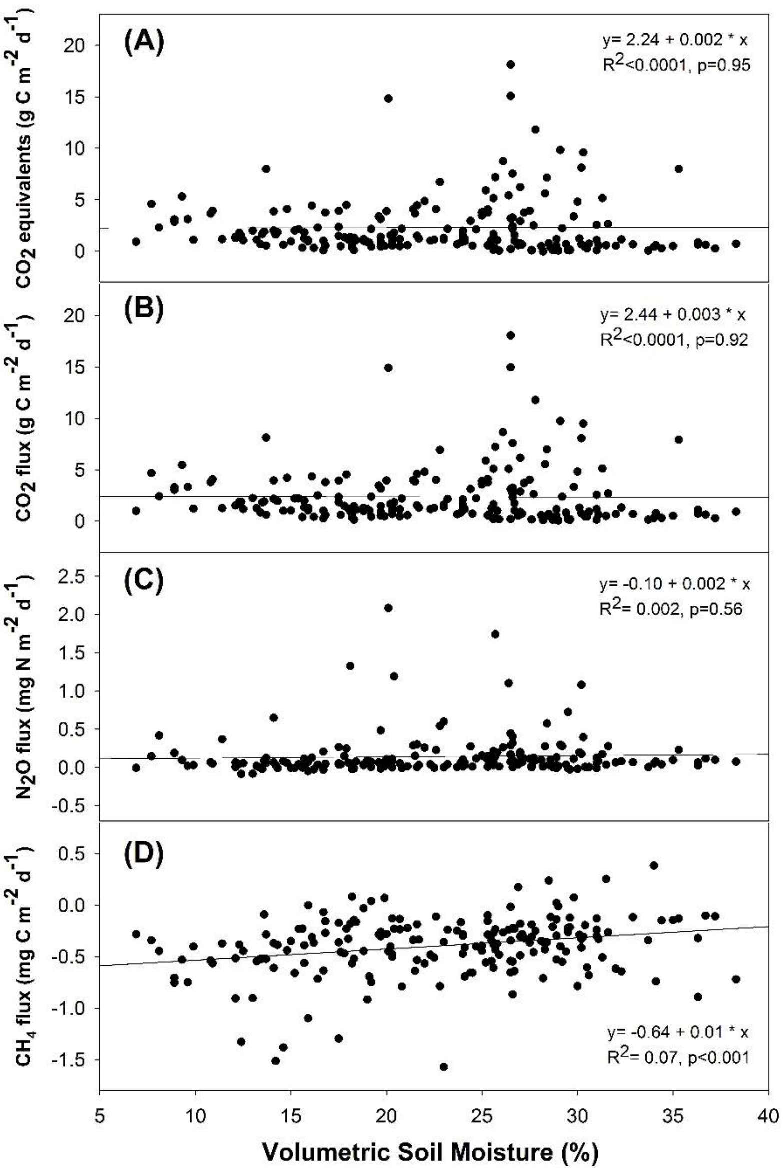

Figure 3.4. Relationship between CO_2 equivalents (A), CO_2 (B), N_2O (C) and CH_4 (D) fluxes from soil to atmosphere and volumetric soil moisture (0-10 cm soil depth) (n=192).

3.4 Conclusions

Typical urban land development practices alone did not alter GHG emissions compared to undisturbed soils. However, urban soil rehabilitation via incorporating compost and subsoiling increased GHG emissions, especially CO_2. Carbon dioxide was the primary contributor to GHG emissions in our study. Methane uptake could only offset a very small portion of GHG emissions. Soil CH_4 uptake capacity did not change after soil manipulations. Carbon dioxide flux and related total GHG emission were positively related to soil temperature. However, our study site was in a humid temperate region previously in agricultural use, and there was no irrigation or fertilization during the study period, thus, the results may be most relevant to sites with similar climate, land use and management practice history.

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Chapter 4

Changes in Aggregate Size Distribution and Aggregate-associated Carbon and Nitrogen due to Urban Land Development and Post-development Soil Rehabilitation

Abstract

Soil aggregates physically protect soil organic matter. However, disturbance from urban land development may disrupt aggregation and alter aggregate-associated carbon (C) and nitrogen (N). We evaluated the effects of urban land development and post-development soil rehabilitation on aggregate size distribution and aggregate-associated C and N. In 2007, four treatments [typical development (A horizon removed, subsoil compacted, and A horizon soil partially replaced), enhanced topsoil (same as typical practice plus tillage), profile rebuilding (soil rehabilitation consisting of compost incorporation to 60-cm depth in subsoil; A horizon soil partially replaced plus tillage), and undisturbed] were applied to 24 plots in Virginia, USA. In 2011 and 2012, soil aggregate size distribution and aggregate-associated C and N concentration were measured at three depths. In surface soils (0-10 cm of A horizon), undisturbed soils had 19 to 78% greater proportion of macroaggregates (250-2000 μm) than when soils were disturbed regardless of whether they were simply stockpiled and replaced, or tilled. At depth (15-30 cm), however, treatment did not affect aggregate size distribution. At 0-5 cm, undisturbed soils always had higher concentrations of microaggregate (53-250 μm) C and N and larger macroaggregate C and N pools. However, at 15-30 cm, rehabilitated soils had both the highest macroaggregate C and N concentrations and the largest microaggregate C and N pools. Typical development disrupted macroaggregates resulting in smaller aggregates in surface soils. After four years of soil redevelopment, soil rehabilitation increased the concentration of C in both macro- and microaggregates in subsurface soils, resulting in larger aggregate-associated C pools. Although soil rehabilitation can increase soil carbon stores, reaggregation of the soil did not occur, suggesting that increases in soil C observed were not due to physical protection by aggregates.

Keywords: compaction, compost, deep tillage, fine fraction, soil organic carbon, urbanization

4.1 Introduction

Rapid urbanization is occurring globally. By 2030, urban land cover will increase by 1.2 million km^2 which will nearly triple the global urban land area extant in 2000 (Seto et al., 2012). During the urbanization process, urban land development practices including clearing, topsoil removal, grading, compacting and building lead to degraded urban soils with low vegetative cover (Jim, 1993), high bulk densities (Jim, 1998), low infiltration rates (Gregory et al., 2006), disrupted aggregation (Craul, 1992), and disturbed carbon (C) cycles (Kaye et al., 2006; Pouyat, 2002), ultimately leading to soil C loss. For example, urbanization resulted in an estimated 0.21 Pg C loss during 1945-2007 in the Southern United States (Zhang et al., 2012). Furthermore, the C loss may be greater than estimated if impervious surfaces lose more C than open space (Raciti et al., 2012). Most studies have quantified the difference in soil C between urban and non-urban landscapes (Golubiewski, 2006; Pouyat, 2002; Pouyat et al., 2007; Qian and Follett, 2002; Raciti et al., 2011). Relatively few studies have explored rehabilitation of degraded urban soil and whether soil C storage levels can be increased by management practices (Beesley and Dickinson, 2010a; Cogger, 2005).

Typical urban land development practices not only affect the C cycle, but also modify physical characteristics by disrupting soil structure, resulting in increased bulk density, altered drainage, and disturbed soil aggregation (Craul, 1992), although urban studies exploring these phenomena are limited. In highly disturbed roadside soils in Hong Kong, the proportion of water stable aggregates was very low (Jim, 1998), which may impair conservation of soil C. Aggregates physically protect organic C (Tisdall and Oades, 1982), and indirectly affect C by regulating microbes, water, oxygen, and nutrients in the soils, which in turn influence soil C dynamics (Six et al., 2004). But different size aggregates have different effects on soil C dynamics. Microaggregates (53-250 μm) protect SOM in the long term; while macroaggregate (250-2000 μm) turnover is a crucial process influencing the stabilization of SOM immediately after disturbance (Six et al., 2004). Because soil aggregates are sensitive to management practices (Six et al., 1998), urban land development practices may lead to aggregates breaking apart and ultimately soil C loss. Thus, it is desirable to understand the degree of aggregate disruption that takes place during urban land development and whether subsequent management practices, such as soil rehabilitation, can accelerate aggregate formation in the near term.

In a previous study, we found that urban land development resulted in significant C losses from soil (Chen et al., 2013). In addition to the influence of C sequestration on climate regulation, associated increases in soil aggregation could also lead to improved provision of other ecosystem services including greater stormwater capture and plant growth. In our previous work, soil rehabilitation via deep tillage and compost application resulted in greater C sequestration including increases to the aggregate-protected C pool, especially in subsurface soils. This implies that rehabilitation affected the aggregate-organic matter complex, but it is unclear if this is because of improved aggregation, increased aggregate-associated C concentration, or a combination. Consequently, our objectives in this study were to:

(1) quantify changes in soil aggregation after typical urban land development and post-development rehabilitation;
(2) determine aggregate-associated soil C and N concentration and pool size of typical urban land development and post-development rehabilitation compared to undisturbed soils; and
(3) assess the mechanisms of C loss due to typical land development practices and C storage resulting from soil rehabilitation.

4.2 Materials and Methods

4.2.1 Site information

The study site was established in Montgomery County, Virginia (37°12' 1.1888" N, 80° 33’ 48.3768" W) between May and November in 2007. The site was in pasture before experiment installation and contains two closely related loamy soils, Shottower loam (fine, kaolinitic, mesic Typic Paleudults) and Slabtown loam (fine-loamy, mixed, mesic Aquic Paleudalfs). Twenty-four (6 replications × 4 soil treatments) 4.6 × 18.3 m plots were installed in a completely randomized experimental design. Treatments included an undisturbed control and three post-development soil management practices. With the exception of the control plots, all plots were subjected to pre-treatment to replicate the scraping and compaction typical of urban land development: the A horizon was removed and stockpiled adjacent to the study site and the exposed subsoil surface was compacted via 8 passes of a 4,808 kilogram ride-on sheep's foot vibrating compactor Ingersoll Rand (Model SD45D) to an average bulk density of 1.98 g cm^{-3}. In addition to the undisturbed control, treatments included: typical practice (TP), 10-cm of stockpiled topsoil applied to the compacted subsoil; enhanced topsoil (ET), same as TP, except rototilled to 12-15 cm depth; and profile rebuilding (PR) 10 cm of leaf compost (C/N ratio 15.0) applied to surface followed by subsoiling with a backhoe to a 60-cm depth, replacement of 10 cm topsoil, and rototilling. Any existing vegetation was killed with herbicide (glyphosate). UN represents pre-urban development soil and thus serves as a benchmark for soil rehabilitation. Typical urban land development practice is represented by TP. Profile rebuilding is the full soil rehabilitation treatment being evaluated in this study, and ET is an intermediate practice often specified by landscape architects. Five species of trees (*Ulmus japonica* (Rehd.) Sarg. x *wilsoniana* Schneid. 'Morton', *Acer rubrum* L., *Quercus bicolor* Willd., *Prunus* 'First Lady' (*P.* ×*incam* Ingram ex R. Olsen & Whittemore 'Okamé' × *P. campanulata* Maximowicz), and *Quercus macrocarpa* Michx.) were planted in each plot from February to April in 2008.

4.2.2 Soil sampling

In June of 2011 and 2012, soil samples were collected at locations equidistant from the trunk of *Acer rubrum* trees in each plot. Four soil samples were extracted at 0-5 cm and 5-10 cm depths 1 m from the trunk. Two of these four sampling locations were randomly selected and additional samples extracted at 15-30 cm depth. Soil samples were composited by depth and the required amount of soil extracted from the pooled sample for aggregate analysis and dried at room temperature.

4.2.3 Aggregate size distribution

We used the wet sieving protocol described by Six et al. (1998) and modified by Wick et al. (2009a; 2009b) to determine the water stable aggregate size distribution. A 50 ± 0.02 g soil sample was air dried and then submerged in deionized water for 5 min at room temperature on a 2000 µm sieve. Water stable soil aggregates (>2000 µm) were separated from the whole soils by moving the sieve up and down 50 times in 2 min. The remainder of the material (soil and water) were then passed through first a 250 µm sieve and then a 53 µm sieve to separate water stable soil macroaggregates (250-2000 µm) and microaggregates (53-250 µm) using the same procedure. The material collected from each sieve (2000, 250 and 53µm) was dried at 55 °C until a constant weight was achieved. The fine fraction (<53 µm) was determined by subtracting the three aggregate weights from the whole soil weight (50±0.2 g).

Because sand may be separated out in the sieving procedure and sand content varies with different aggregate size classes (Elliott et al., 1991), samples were corrected for sand content according to Denef et al. (2001) and (Wick et al., 2009a) using 5 g of each aggregate sample. We dispersed these samples with 0.5% sodium hexametaphosphate on a shaker for 18 h. Dispersed samples were then sieved with 250 and 53 µm nested sieves for macroaggregates and a 53 µm sieve for microaggregates. Collected sand was dried and weighed and sand-corrected aggregate weights determined according to Eq. (1).

Sand corrected weight = aggregate weight - [(sand weight/5g) × aggregate weight] (1)

4.2.4 Aggregate-associated carbon and nitrogen

Macroaggregate and microaggregate samples were analyzed for total C and N using dry combustion on an Elementar Variomacro CN Analyzer (Hanau, Germany). In order to compare C and N concentrations across sites, we determined sand-free C and N by the following formulas for sand correction [Eq. (2) and (3)] for each size class (Denef et al., 2001; Wick et al., 2009a):

Sand free C =C × [g aggregate / 1- sand] (2)

Sand free N =N × [g aggregate / 1- sand] (3)

4.2.5 Statistical analysis

Differences in aggregate size distribution and aggregate-associated C, N and C/N ratio among soil treatments at the same depth were determined using one-way analysis of variance using PROC GLM in SAS software (SAS Institute, Inc., Cary, North Carolina) and Tukey's HSD at α=0.05 or 0.01 as indicated.

4.3 Results and Discussion

4.3.1 Aggregate size distribution

In 2011, four years after simulated land urbanization and soil rehabilitation, the surface 0-5 cm of soil in UN plots had a greater proportion of macroaggregates, but smaller proportions of microaggregates and fine fraction than the three post-development treatments (Table 4.1). At 5-10 cm depth, UN had a greater proportion of macroaggregates (0.44 g sand-free aggregate g^{-1} soil) and a lower proportion of microaggregates (0.12 g sand-free aggregate g^{-1} soil) compared to the other three treatments, but there was no evidence to show differences among treatments in the fine fraction (overall p-value= 0.12). Likewise, at 15-30 cm soil depth, there was little evidence of differences in aggregate size distribution among all treatments (p-values = 0.06, 0.20 and 0.10, for macroaggregates, microaggregates, and fine fraction respectively) (Table 4.1).

Physical disturbance from urban land development and post-development practices appear to have fractured macroaggregates leading to increased microaggregate and fine fractions in surface soils. This finding is consistent with previous studies that mostly looked at the effect of tillage disturbance on aggregate size distribution in agricultural systems (Lawal et al., 2009; Six et al., 2004; Six et al., 2000; Wick et al., 2009c). Soil disturbance from tillage has been recognized as a major cause of reduction in the number and stability of soil aggregates when land changes from natural to agricultural use (Six et al., 2000). However, in urban areas, there are a wide range of human induced disturbances. For example, urban land development processes may involve vegetation clearance, topsoil removal, stockpiling, compaction, building, or soil replacement. Collectively, these activities could influence soil aggregation even more dramatically than the tillage associated with conversion to agricultural uses. In our study we applied these urban land development practices (with the exception of building/excavating) to study plots. Although the soil was scraped and stockpiled, the site did not include significant cutting or filling nor was it compacted in lifts as sometimes occurs on land development sites. Our findings demonstrate that the minimal level of disturbance likely to result from urban land development reduced the proportion of macroaggregates in the 0-5 and 5-10 cm depths by about 1/3 and 1/6, respectively (Table 4.1).

In 2012, soil aggregate size distribution exhibited similar trends among treatments as in the previous year, although overall there was a slightly greater proportion of aggregates in all size classes, especially macroaggregates, which implies that continued soil aggregation was underway during this period of time (Table 4.1). In the top 10 cm of soil, treatments subjected to urban land development followed by post-development practices (i.e., ET, TP, and PR) had lower macroaggregate proportions but greater microaggregate and fine fraction proportions compared to undisturbed soils. However, as in 2011, different soil treatments had no effect on any aggregate size classes at 15-30 cm soil depth.

Based on the results above, surface soil aggregate size distribution was more likely to be affected by typical urban land development practices than subsurface soils. Of interest is that after surface soil scraping, stockpiling, and replacement, additional disturbance via tillage (as in the ET treatment) did not further degrade aggregates. In our study, only surface soils were subjected to removal and replacement and conventional tillage (the PR treatment received deep tillage with a backhoe). Thus, aggregates were more strongly affected by these practices rather than compaction alone, as is indicated by the similar aggregate size distribution at 15-30 cm soil depth. Interestingly, the PR treatment also had similar aggregate size distributions at that depth, suggesting that either (A) the backhoe-style tillage (that separates soil into large clumps from approximately 5-30 cm in diameter) combined with compost addition, did not disrupt aggregates as much as might be expected, or (B) that aggregates had reformed within the study time period. As might be expected, others have also found surface soils to be more vulnerable to disturbance during manipulation processes (Six et al., 1998). Thus, efforts directed at redeveloping subsurface soils have potential to persist over time, even if surface soils are subjected to additional disturbances such as might occur during urban land use. Thus, restoring degraded subsoils may provide opportunities for additional strategies for improving soil functions for the long term. We did not find evidence, however, that PR improved aggregation 5 years after installation compared to other treatments. Our results differ from other studies that observed indications of rapid changes to aggregation after compost application (Tejada et al., 2009; Whalen et al., 2003). However, these studies focused on aggregate stability (Tejada et al., 2009) or observed changes in very large aggregates (> 4 mm) (Whalen et al., 2003) rather than macro or micro aggregates as defined in the present study.

Table 4.1. Macroaggregates (250-2000 µm), microaggregates (53-250 µm), and fine fraction (<53 µm) of soils from typical urban land development practice (TP), two post-development soil rehabilitation practices (ET and PR), and undisturbed control soil (UN) at 0-5 cm, 5-10 cm, and 15-30 cm soil depths four and five years after treatment.

			Treatments			
	Soil		TP	ET	PR	UN
Year	depth	Aggregates	g sand-free aggregate per g soil†			
2011	0-5 cm	Macroaggregate	0.23 (0.02)b‡	0.23 (0.03)b	0.22 (0.02)b	0.34 (0.02)a
		Microaggregate	0.18(0.02)a	0.20 (0.04)a	0.19 (0.02)a	0.13 (0.01)b
		Fine fraction	0.18 (0.02)a	0.16 (0.02)a	0.18 (0.02) a	0.12 (0.01)b
	5-10 cm	Macroaggregate	0.35 (0.04)b	0.37 (0.06)ab	0.37 (0.04)b	0.44 (0.02)a
		Microaggregate	0.18 (0.04)a	0.18 (0.05)a	0.16 (0.03)ab	0.12 (0.02)b
		Fine fraction	0.10 (0.03)	0.10 (0.02)	0.10 (0.01)	0.08 (0.02)
	15-30 cm	Macroaggregate	0.22 (0.04)	0.22 (0.04)	0.20 (0.02)	0.26(0.04)
		Microaggregate	0.30 (0.04)	0.30 (0.04)	0.29 (0.03)	0.26 (0.03)
		Fine fraction	0.12 (0.03)	0.11 (0.02)	0.13 (0.02)	0.11(0.02)
2012	0-5 cm	Macroaggregate	0.34 (0.04)b	0.29 (0.07)b	0.30 (0.04)b	0.43 (0.02)a
		Microaggregate	0.16 (0.03)ab	0.18 (0.04)a	0.19 (0.02)a	0.13 (0.01)b
		Fine fraction	0.21 (0.02)a	0.19 (0.04)ab	0.20 (0.02)ab	0.16 (0.02)b
	5-10 cm	Macroaggregate	0.27 (0.04)c	0.35 (0.06)b	0.35 (0.04)b	0.48 (0.03)a
		Microaggregate	0.18 (0.03)ab	0.21 (0.04)a	0.21 (0.03)a	0.14 (0.02)b
		Fine fraction	0.14 (0.02)ab	0.16 (0.02)a	0.15 (0.03)a	0.11 (0.02)b
	15-30 cm	Macroaggregate	0.34 (0.08)	0.33 (0.08)	0.29 (0.03)	0.38 (0.09)
		Microaggregate	0.27 (0.05)	0.27 (0.06)	0.26 (0.02)	0.21 (0.05)
		Fine fraction	0.14 (0.03)	0.13 (0.04)	0.17 (0.02)	0.12 (0.04)

†Values in parentheses represent standard errors of the means (n=6).
‡Within each aggregate size, soil depth, and year, letters denote differences among treatments using Tukey's HSD at α=0.01.

4.3.2 Aggregate-associated carbon

In 2011, at 0-5 cm, UN had the greatest macroaggregate and microaggregate C concentration and C pool size; other than greater microaggregate C concentration these effects were no longer evident at 5-10 cm. In contrast, at 15-30 cm, PR had the greatest macroaggregate and microaggregate C concentrations and C pool sizes among all treatments (Figure 4.1 and Table 4.2). Our finding of C loss in surface soils is consistent with other studies (Mikha and Rice, 2004; Six et al., 1998; Wick et al., 2009c). The increase in microaggregates and fine fraction at the expense of macroaggregates suggest this C loss is due to disturbance breaking aggregates apart, leading to organic matter exposure and thus easier access to heterotrophic soil microbes (Rovira and Greacen, 1957).

Table 4.2. Carbon pool size of macroaggregates (250-2000 µm) and microaggregates (53-250 µm) of typical urban land development practice (TP), two post-development soil rehabilitation practices (ET and PR) and undisturbed control soil (UN) at 0-5 cm, 5-10 cm, and 15-30 cm soil depths four and five years after treatment.

			Treatments			
	Soil		TP	ET	PR	UN
Year	depth	Aggregates	Carbon pool size (g C kg^{-1} whole soil)†			
2011	0-5 cm	Macroaggregate	4.82 (1.43)b‡	3.82 (1.47)b	4.91 (1.23)b	11.10 (1.72)a
		Microaggregate	4.78 (0.75)ab	4.47 (1.06)b	4.96 (0.60)ab	6.24 (1.18)a
	5-10 cm	Macroaggregate	5.23 (1.64)	5.25 (1.72)	7.46 (2.99)	7.64 (1.11)
		Microaggregate	3.27 (0.46)	3.21 (0.63)	3.72 (0.72)	3.17 (0.63)
	15-30 cm	Macroaggregate	1.26 (0.41)b	1.50 (1.38)b	4.80 (2.65)a	2.59 (1.30)ab
		Microaggregate	1.80 (0.50)b	1.87 (0.86)b	4.12 (1.96)a	2.88 (0.68)ab
2012	0-5 cm	Macroaggregate	3.77 (1.46)b	2.56 (1.38)b	3.98 (1.33)ab	5.83 (0.68)a
		Microaggregate	2.06 (0.31)	2.00 (0.37)	2.66 (0.59)	2.39 (0.34)
	5-10 cm	Macroaggregate	3.03 (1.16)b	2.99 (1.29)b	4.81 (1.57)ab	5.39 (0.85)a
		Microaggregate	2.18 (0.42)ab	2.12 (0.26)b	2.88 (0.61)a	2.00 (0.44)b
	15-30 cm	Macroaggregate	1.19 (0.57)	1.26 (1.29)	2.94 (1.61)	2.59 (1.20)
		Microaggregate	1.03 (0.18)	1.09 (0.27)	2.34 (1.67)	1.42 (0.32)

† Values in parentheses represent standard errors of the means (n=6).
‡Within each aggregate size, soil depth, and year, letters denote differences among treatments using Tukey's HSD at α=0.05.

In 2012, in the top 10 cm, aggregate C concentration was not affected by treatment except microaggregate C concentration at 0-5 cm. In contrast, UN had greater aggregate C pool size than the other three treatments except the microaggregate C pool at 0-5 cm. At 15-30 cm depth, PR increased macroaggregate C concentration, but had no effect on aggregate C pool size (Figure 4.1 and Table 4.2). Profile rebuilding as urban soil rehabilitation increased aggregate-associated C in subsurface soils four years after rehabilitation installation, and macroaggregate-associated C continued to accumulate in the subsequent year.

Previous studies demonstrated that compost could increase degraded urban soil C storage (Beesley and Dickinson, 2010b; Beesley et al., 2010; Cogger, 2005; Sæbø and Ferrini, 2006). Our findings indicate that compost amendment with subsoiling could not only increase macroaggregate-associated C concentration and pool size, but also lead to more stable microaggregate-associated C four years after installation. It implies that C inputs can increase macroaggregate-associated C concentration rapidly, but it takes longer to repair disrupted aggregation, which is important for enhanced long-term C storage. Aggregate C concentration increased from 2011 to 2012, especially macroaggregate associated C. However, aggregate C pool size exhibited the opposite trend.

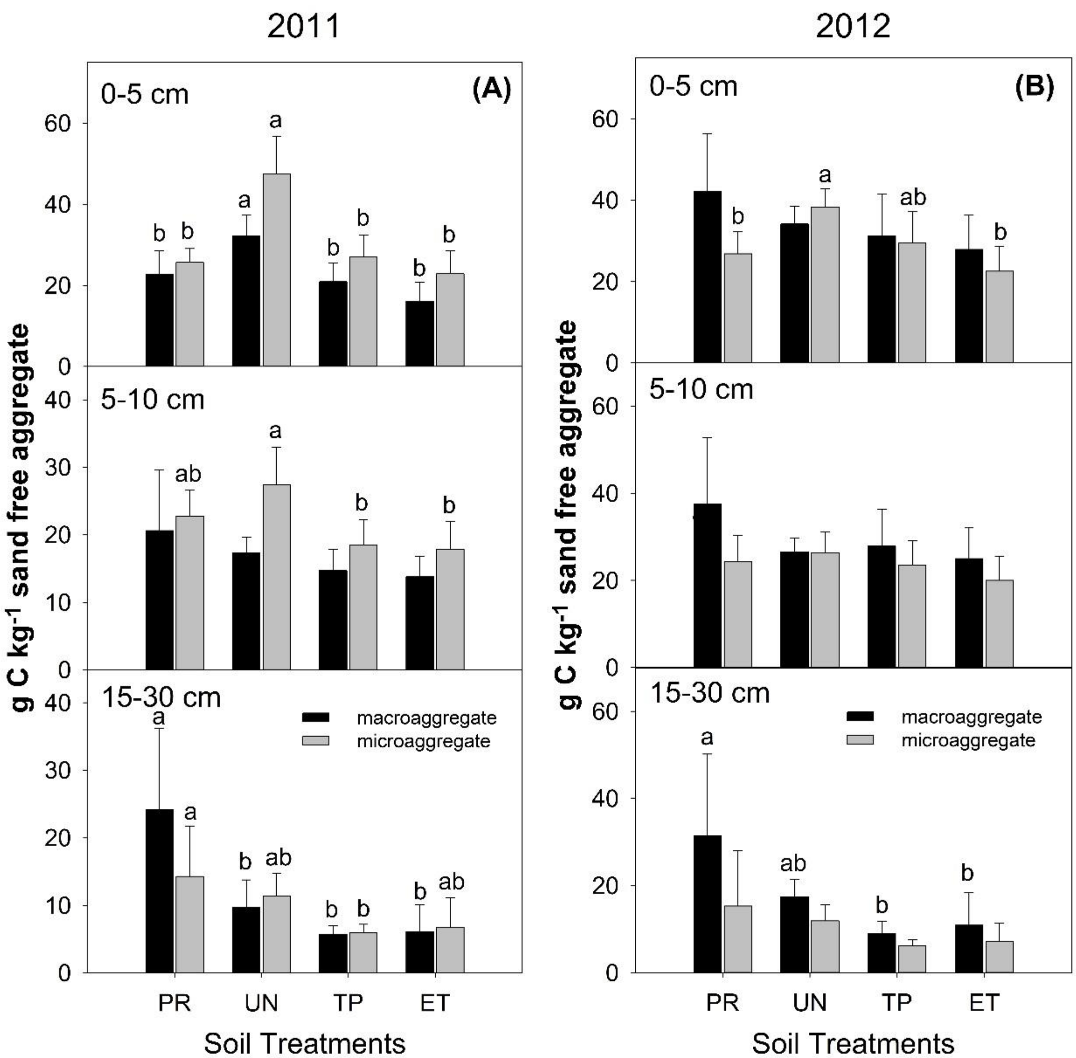

Figure 4.1. Carbon concentration in macroaggregate (250-2000µm) and microaggregate (53-250 µm) at three soil depths of typical urban land development practice (TP), two post-development soil rehabilitation practices (ET and PR) and undisturbed control soil (UN) in June 2011(A) and in June 2012 (B). Error bars depict standard errors of the means (n=6). Within each size aggregate at the same soil depth, letters denote differences among treatments using Tukey's HSD at α=0.05.

4.3.3 Aggregate-associated nitrogen

In 2011, at 0-5 cm and 5-10 cm, UN had the greatest macro and microaggregate N concentrations and pool sizes; however, at 15-30 cm, PR had the greatest macroaggregate N concentration and macro- and microaggregate N pool sizes (Figure 4.2 and Table 4.3). In 2012, in the top 10 cm soils, UN had the greatest macroaggregate N pool. At 15-30 cm depth, PR had greatest macroaggregate N concentration; however, microaggregate N concentration and aggregate N pools were similar among all treatments (Figure 4.2 and Table 4.3). Profile rebuilding as urban soil rehabilitation increased macroaggregate-associated N concentrations and pool sizes in subsurface soils four years after rehabilitation, and macroaggregate-associated N concentrations continued to accumulate, but the effects on macroaggregate-associated N pool size were reduced in the subsequent year. The aggregate-associated N concentration displayed similar trends as the aggregate associated C content.

Table 4.3. Nitrogen pool size of macroaggregates (250-2000 µm), microaggregates (53-250 µm) of typical urban land development practice (TP), two post- development soil rehabilitation practices (ET and PR) and undisturbed control soil (UN) at 0-5 cm, 5-10 cm and 15-30 cm soil depths four and five years after treatment.

			Treatments			
			TP	ET	PR	UN
Year	Soil depth	Aggregates	Nitrogen pool size (g N kg^{-1} whole soil)†			
2011	0-5 cm	Macroaggregate	0.48 (0.15)b‡	0.40 (0.15)b	0.43 (0.10)b	1.09 (0.17)a
		Microaggregate	0.48 (0.07)b	0.47 (0.10)b	0.48 (0.04)b	0.63 (0.11)a
	5-10 cm	Macroaggregate	0.54 (0.16)b	0.58 (0.15)ab	0.70 (0.21)ab	0.84 (0.16)a
		Microaggregate	0.35 (0.04)	0.35 (0.06)	0.37 (0.06)	0.34 (0.06)
	15-30 cm	Macroaggregate	0.15 (0.05)b	0.16 (0.12)ab	0.35 (0.17)a	0.25 (0.12)ab
		Microaggregate	0.25 (0.06)b	0.26 (0.07)b	0.42 (0.14)a	0.35 (0.07)ab
2012	0-5 cm	Macroaggregate	0.37 (0.14)b	0.26 (0.12)b	0.34 (0.10)b	0.60 (0.08)a
		Microaggregate	0.21 (0.03)	0.21 (0.04)	0.24 (0.04)	0.25 (0.04)
	5-10 cm	Macroaggregate	0.32 (0.11)b	0.31 (0.13)b	0.40 (0.09)ab	0.57 (0.09)a
		Microaggregate	0.25 (0.06)	0.24 (0.03)	0.28 (0.05)	0.22 (0.05)
	15-30 cm	Macroaggregate	0.14 (0.06)	0.15 (0.13)	0.23 (0.09)	0.27 (0.12)
		Microaggregate	0.14 (0.02)	0.14 (0.02)	0.23 (0.12)	0.16 (0.03)

†n=6. Values in parentheses represent standard errors of the means.
‡Within each aggregate size in the same year, letters denote differences among treatments using Tukey's HSD at α=0.05.

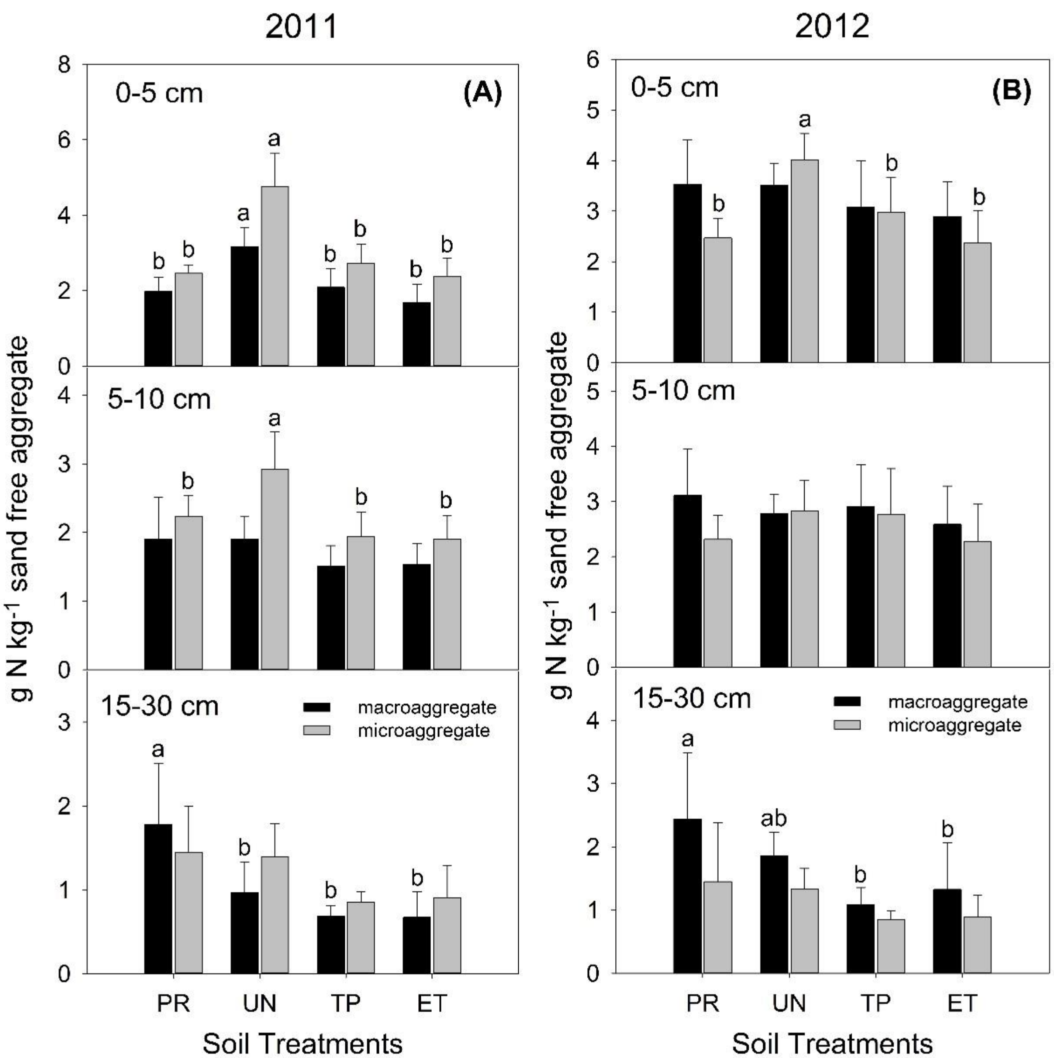

Figure 4.2. Nitrogen concentration in macroaggregate (250-2000μm) and microaggregate (53-250 μm) at three soil depths of typical urban land development practice (TP), two post-development soil rehabilitation practices (ET and PR) and undisturbed control soil (UN) in June 2011 (A) and in June 2012 (B). Error bars depict standard errors of the mean (n=6). Within each size aggregate at the same soil depth, letters denote differences among treatments using Tukey's HSD at α=0.05.

4.3.4 Aggregate-associated C/N ratio

In 2011, at 0-5 cm and 15-30 cm depths, PR had the highest C/N ratio in macroaggregates and microaggregates. At 5-10 cm, C/N ratio of these two aggregate classes was not affected by treatment. In 2012, PR had highest C/N ratio in macroaggregates and microaggregates at all 3 soil depths (Table 4.4). The higher C/N in the PR treatment might due to higher C concentrations introduced by incorporated compost in subsurface soils. In surface soils, C input from tree roots and microbes might have contributed to increased C and higher C/N ratios, but further study would be needed to test these hypotheses.

Table 4.4. Carbon to nitrogen ratio of macroaggregates (250-2000 µm) and microaggregates (53-250 µm) of typical urban land development practice (TP), two post- development soil rehabilitation practices (ET and PR) and undisturbed control soil (UN) at 0-5 cm, 5-10 cm, and 15-30 cm soil depths four and five years after treatment.

			Treatments			
			TP	ET	PR	UN
Year	Soil depth	Aggregates	C/N Ratio†			
2011	0-5 cm	Macroaggregate	10.0 (0.5)b‡	9.5 (0.5)b	11.5 (1.3)a	10.2 (0.3)b
		Microaggregate	9.8 (0.3)ab	9.5 (0.5)b	10.4 (0.6)a	10.0 (0.2)ab
	5-10 cm	Macroaggregate	9.7 (0.6)	9.0 (1.4)	10.6 (1.4)	9.2 (0.7)
		Microaggregate	9.5 (0.3)	9.3 (0.5)	10.1 (0.9)	9.4 (0.2)
	15-30 cm	Macroaggregate	8.4 (1.0)b	8.7 (1.5)b	13.1 (1.9)a	9.0 (0.8)b
		Microaggregate	7.0 (0.4)b	7.0 (1.5)b	9.5 (1.2)a	8.2 (0.6)ab
2012	0-5 cm	Macroaggregate	10.1 (0.4)b	9.5 (0.8)b	11.8 (1.6)a	9.8 (0.4)b
		Microaggregate	9.8 (0.4)b	9.5 (0.5)b	10.8 (0.7)a	9.6 (0.3)b
	5-10 cm	Macroaggregate	9.5 (0.7)b	9.6 (0.4)b	11.8 (1.4)a	9.5 (0.3)b
		Microaggregate	8.6 (0.6)b	8.8 (0.4)b	10.4 (1.0)a	9.3 (0.2)b
	15-30 cm	Macroaggregate	8.3 (0.9)b	8.0 (1.2)b	12.0 (2.6)a	9.4 (0.8)b
		Microaggregate	7.2 (0.6)b	7.7 (1.3)ab	9.8 (2.0)a	8.8 (0.9)ab

†Values in parentheses represent standard errors of the means (n=6).
‡Within each aggregate size, soil depth, and year, letters denote differences among treatments using Tukey's HSD at α=0.05.

4.4 Conclusions

During urbanization, typical urban land development practices disrupted soil aggregation, reducing the proportion of macroaggregates and leading to macroaggregate-associated C loss in surface soils. This C loss is mainly due to reduced proportions of macroaggregates and the decreased microaggregate C concentration that likely results from loss of macroaggregate protection. Post-development rehabilitation increased macroaggregate-associated C concentration via introduction of soil organic amendment in subsurface soils and increased the microaggregate-associated C pool. In our study, aggregate size distribution in the subsoil was not significantly altered by the compaction applied while the surface soils were removed. This implies that subsoil is more resistant to disturbance resulting from typical urban land development practices. It may also indicate that soil itself can recover four to five years after disturbance. Thus, there is no evidence showing the effect of profile rebuilding on subsoil aggregate size distribution. In contrast, simply scraping, stockpiling, and replacing surface soils resulted in dramatic decreases in macroaggregates. Subsequent tillage, however, did not exacerbate this loss of macroaggregates. These findings imply that C input by compost can increase C storage in a short period but it is uncertain how or when aggregates will reform in degraded soils. Thus, the protection of soils at the early stage of urban development is critical to achieving healthy urban soil systems.

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Chapter 5

Soil Hydraulic Conductivity Change from Urban Land Development and Response to Post-development Soil Rehabilitation

Abstract

Typical urban development practices result in degraded urban soils that are often highly compacted, slowing stormwater movement into soils. We evaluated the effects of typical urban land development practices and post-development rehabilitation on soil permeability. In 2007, four treatments [typical land development practice (A horizon removed, subsoil compacted, and A horizon soil partially replaced), enhanced topsoil (same as typical practice plus tillage), post-development rehabilitated soils (soil rehabilitation consisting of compost incorporation to 60-cm depth in subsoil; A horizon soil partially replaced plus tillage), and pre-development (undisturbed) soils] were applied to 24 plots in Virginia, USA. All plots were planted with five tree species but otherwise kept free of vegetation and were not mulched. In 2012, surface unsaturated hydraulic conductivity (K_h) and saturated hydraulic conductivity (K_{sat}) at 10-25 cm and 25-40 cm soil depths were measured. Surface K_h was not affected by different soil treatments (ranging from 0.44 to 2.28 cm hr^{-1}) and was the most limiting layer, due in part to surface sealing. At the 10-25 cm and 25-40 cm depths, K_{sat} was higher in rehabilitated soil than undisturbed soils which, in turn, was higher than K_h in typical post-development soils. We found post-development rehabilitated soils (14.8 and 6.3 cm hr^{-1} at 10-25 cm and 25-40 cm, respectively) had about twice the K_{sat} of undisturbed soils and

approximately 6-11 times that of soils subjected to typical land development practices. The rehabilitation system studied has strong potential as a tool for urban stormwater mitigation. However, techniques for maintaining rapid surface infiltration require further study. We also observed lower soil water contents in rehabilitated soils. This could be due to greater tree water uptake, more rapid movement of water into deeper soil regions, reduced water holding capacity, or a combination of these.

Keywords: bulk density, compost, deep tillage, hydraulic conductivity, landuse change, urbanization

5.1 Introduction

Urbanization can increase short-term peak discharge and total stormwater runoff, and degrade water quality (Leopold, 1968). The high proportion of impervious surfaces found in urbanized areas is associated with increased stormwater runoff leading to flooding, streams with rapidly fluctuating water levels, and degraded surface water quality (Paul and Meyer, 2001). The U.S Environmental Protection Agency (EPA, 2003) reported that impervious surfaces can generate more than five times as much runoff as a woodland area. In urbanized settings, even open soil areas (i.e. without pavement or buildings) designated for landscaping, tree lawns, or parks, may be effectively impervious or nearly so due to grading, scraping, and soil compaction during and after urbanization (Gregory et al., 2006). Urban soils are typically highly compacted with high bulk densities (Jim, 1993, 1998a, b; Short et al., 1986) and compacted urban soils severely impede water infiltration (Gregory et al., 2006; Pitt et al., 2008). Pitt et al. (2008) demonstrated that compaction reduced soil infiltration rate significantly in a variety of texture soils (sandy, clay and loam), especially in sandy soils. They also found steady-state infiltration rates decreased as the level of compaction increased. In sandy soils in North Central Florida, average soil infiltration rates were 37.7- 63.4 cm hr^{-1} in undeveloped non-compacted natural forest soil and 0.8-17.5 cm hr^{-1} in post-development compacted yard soils (Gregory et al., 2006). The soil infiltration decreases were clearly due to home construction activities even with high variation among sites. Similarly, Hamilton and Waddington (1999) reported that the average infiltration rate of home lawns in central Pennsylvania was between 0.4 and 10.0 cm hr^{-1}. They also concluded that construction processes, e.g., excavation procedures and lawn establishment methods, affected infiltration rates. Furthermore, the time since development of a residential site of residential site can affect infiltration. In another Pennsylvania study, pre-2000 constructed residential sites had higher mean infiltration rates (9.0 cm hr^{-1}) than those constructed post-2000 (2.8 cm hr^{-1}) (Woltemade, 2010). Whether this is due to changes in construction practices or increases in infiltration that occur over time due to soil development and aggregation is unknown.

There is increasing interest in environmentally sensitive stormwater management practices that take into account the influence of site and soil variables on water movement (Pitt and Clark, 2008). Low impact development (LID), in particular, has been presented as alternative stormwater management and site design approach to minimize construction impacts on water flow (Williams and Wise, 2006). A wide range of best management practices (BMPs) have been studied (e.g., Bartens et al., 2008; Collins et al., 2010; Xiao and McPherson, 2011). However, all these practices were mainly focused on creating a facility to collect and store stormwater although LID practices tend to collect water from relatively small areas, compared with traditional detention basins. A completely integrated stormwater mitigation technique is needed to utilize existing landscapes in conjunction with these practices to optimize stormwater interception and mitigation as well as enhance other ecosystem services, such as net primary production.

Tillage is commonly used to loosen compacted agricultural soils, but is relatively rare in urban settings. One study, however, found deep tillage (up to 60 cm) alone slightly improved soil infiltration at three parks in the Minneapolis Metropolitan area (Olson et al., 2013). Compost amendment has also been reported as an effective practice to increase infiltration rates in disturbed urban soils (Pitt et al., 1999). A surface compost amendment of 7 cm, for example, was found to increase soil infiltration (saturated hydraulic conductivity of compost treatment was 2.7-5.7 times that of the control) (Olson et al., 2013). However, the degree to which deep tillage accompanied by compost amendment improves infiltration rates is uncertain. In addition, urban vegetation can help reduce peak discharge and stormwater runoff (Sanders, 1986). It is well documented that the tree canopy can intercept rainfall reducing peak discharge (Xiao et al., 2000). In a recent study conducted in Manchester, UK, the role of urban trees in reducing surface water runoff has been highlighted (Armson et al., 2013). In this study, trees and their associated tree pits (the cutouts in pavement designed for tree planting in sidewalks and other paved areas) reduced runoff from asphalt by 62%. Furthermore, tree roots can play important role in stormwater mitigation by increases in permeability due to root penetration. Bartens et al.(2008) demonstrated that, in a compacted soil, tree roots increased average K_{sat} 27-fold compared to unplanted controls. Thus, an urban stormwater mitigation system is needed to integrate these potential components.

In our study, we address whether rehabilitating degraded urban soils via deep tillage and compost incorporation plus tree planting that is desirable in urban areas for vegetation management can also improve soil infiltration rates permeability and potentially mitigate urban stormwater runoff. Typical urban land development practices including clearing, topsoil removal, surface grading, compacting and building construction may lead to degraded urban soils and decreased soil infiltration. Soil profile rebuilding (SPR) is a technique to rehabilitate post-development degraded urban soils to better support vegetation. This technique is an integrated system including deep tillage, deep compost amendment, and tree planting. Previous work with SPR indicates post-development rehabilitated soils have improved soil physical characteristics, such as soil bulk density (Layman, 2010). However, the effect of post-development rehabilitation on stormwater mitigation is not known. Our objectives were to: (1) compare permeability of pre-development undisturbed soils, soils subjected to typical land development practices, and post-development rehabilitated soils; (2) determine bulk density and soil water content among these soils and their relationship to soil permeability.

5.2 Methods

5.2.1 Site information

The study site in Montgomery County, Virginia (37°12' 1.1888" N, 80° 33’ 48.3768" W) contains two closely related loamy soils, Shottower loam (fine, kaolinitic, mesic Typic Paleudults) and Slabtown loam (fine-loamy, mixed, mesic Aquic Paleudalfs) and was previously in pasture. Four soil treatments were installed in a completely randomized experimental design between May and November in 2007 (6 replications × 4 soil treatments) in plots measuring 4.6 × 18.3 m. Treatments included an undisturbed control, and three post-development soil management practices. With the exception of the control plots, all plots were subjected to pre-treatment replicating the scraping and compaction typical of land development: the A horizon was removed and stockpiled nearby, and the exposed subsoil surface was compacted with a 4,808 kilogram ride-on sheep's foot vibrating compactor Ingersoll Rand (Model SD45D) in 8 passes to an average bulk density of 1.98 Mg m^{-3}. In addition to the undisturbed control (UN), treatments included: typical practice (TP), stockpiled topsoil applied to the compacted subsoil to a depth of 10 cm; enhanced topsoil (ET), same as TP, except after soil application, the site was rototilled to a 12-15 cm depth; and profile rebuilding (PR) with 10 cm of leaf compost (C/N ratio 15.0) applied to surface followed by subsoiling with a backhoe to a 60-cm depth to both break up the compacted subsoil and incorporate veins of compost deep into the soil, replacement of 10 cm topsoil, and finally rototilling. Existing vegetation on all plots was killed with herbicide (glyphosate) and maintained weed free throughout the study period. The control (UN), represents pre-urban development soil and thus serves as a benchmark for soil rehabilitation. The TP treatment is similar to typical urban land development practices employed in the United States. The PR treatment is a soil rehabilitation practice intended to loosen compacted soil and create opportunities for soil structure to build over time, while ET is a modification of TP sometimes specified by site designers to disrupt the interface between compacted subgrades and reapplied topsoil. Five species of trees (*Ulmus japonica* (Rehd.) Sarg. x *wilsoniana* Schneid. 'Morton', *Acer rubrum* L., *Quercus bicolor* Willd., *Prunus* 'First Lady' (*P.* ×*incam* Ingram ex R.

Olsen & Whittemore 'Okamé' × *P. campanulata* Maximowicz), and *Quercus macrocarpa* Michx.) were planted in each plot from February to April in 2008.

5.2.2 Hydraulic conductivity measurement

Surface unsaturated hydraulic conductivity (K_h) was measured using a mini disk tension infiltrometer (Decagon Devices, Inc., Pullman, WA) with h = -2 cm tension in each plot from June to July in 2012. Measurement locations were randomly selected within the central portion of each plot to avoid edge effects. Three minidisk measurements were conducted at the selected spot simultaneously. Then, a 6-cm diameter cylindrical hole is bored to 25 cm at the same measurement locations as the mini disk infiltrometer measurements. A constant 15-cm head of water was maintained at the bottom of the hole. Saturated hydraulic conductivity (K_{sat}) was measured by the Compact Constant Head Permeameter (Ksat, Inc., Raleigh, NC) from 10 to 25 cm. Similarly, a 6-cm diameter cylindrical hole is bored to 40 cm at the same location to measure K_{sat} from 25 to 40 cm with a 15-cm constant head of water during July and August 2012. One measurement was conducted in each plot for a total of six replications per treatment.

5.2.3 Soil moisture and rainfall

Volumetric soil moisture was measured monthly by time-domain reflectometry (Trase System I, Soil moisture Equipment Corp, Santa Barbara, California) at 0-10 cm, 0-25 cm and 0-40 cm soil depths in the middle of each plot from August 2010 to July 2011 (except December 2010). Daily precipitation was monitored by the College of Agriculture and Life Sciences Kentland Farm's Dynamet weather station (assembled by

Dynamax, Inc., Houston, TX using Campbell Scientific, Logan, UT, hardware and a model CR10X data logger) located approximately 50 m from the study site.

5.2.4 Soil bulk density

For bulk density measurements, soil cores (diameter=5 cm, height=5 cm) were collected at 4 depths (2.5-7.6 cm, 15.2-20.3 cm, 30.5-35.6 cm and 50.8-55.9 cm) in the middle of each of plot using a slide hammer in July 2012. Bulk density was calculated for each sample after oven drying at 105°C.

5.2.5 Statistical analysis

Differences in unsaturated hydraulic conductivity, saturated hydraulic conductivity and soil bulk density among soil treatments at the same depth were determined using one-way analysis of variance using PROC GLM in SAS software (SAS Institute, Inc., Cary, North Carolina) and Tukey's HSD at α=0.05. Difference of volumetric soil moisture among soil treatments was analyzed by repeated measurement analysis using PROC GLIMMIX (LS mean separation at α=0.05) in SAS 9.2 software (SAS Institute, Inc., Cary, North Carolina). Spearman's rank correlation was used to identify the relationship of soil saturated hydraulic conductivity at 25 cm and soil bulk density at 15.2-20.3 cm using SigmaPlot 12.0 (Systat Software Inc., Point Richmond, CA).

5.3 Results

5.3.1 Soil hydraulic conductivity

Unsaturated hydraulic conductivity at the surface was not affected by different soil treatments (ranging from 0.4 to 2.3 cm hr^{-1}) (Fig. 5.1). At 10-25 cm depth, profile rebuilding as our urban soil rehabilitation practice had the highest mean saturated hydraulic conductivity among all treatments (14.8 cm hr^{-1}) followed by undisturbed soils (6.7 cm hr^{-1}) while typical practice and enhanced topsoil had lower saturated hydraulic conductivity (1.3 and 1.7cm hr^{-1}, respectively). At the 25-40 cm depth, saturated hydraulic conductivity followed a similar trend as at 10-25 cm depth. Profile rebuilding had the highest mean saturated hydraulic conductivity among all treatments (6.3 cm hr^{-1}) followed by undisturbed soils and enhanced topsoil (3.0 and 3.7 cm hr^{-1}, respectively) while typical practice had the lowest saturated hydraulic conductivity (1.03 cm hr^{-1}) (Fig. 5.1).

5.3.2 Soil moisture and rainfall

In general, undisturbed soil had higher soil moisture contents while profile rebuilding had the lowest soil moisture, especially in 25-40 cm soils. Typical practice and enhanced topsoil maintained intermediate soil moisture contents. Soil moisture was highly varied at 0-10 cm compared to other soil depths. At the three different soil depths across all treatments, soil moisture was greatest at 25-40 cm, and lowest at 10-25 cm, while the 0-10 cm depth was intermediate. This implies there was a dry layer in the middle and deep soils can store more water. During winter, soil moisture was affected by different treatments at 10-25 cm. Throughout the rest of the year, soil moisture was affected by different treatments in the top layer and in deeper soils (Figs. 5.2, 5.3 and 5.4).

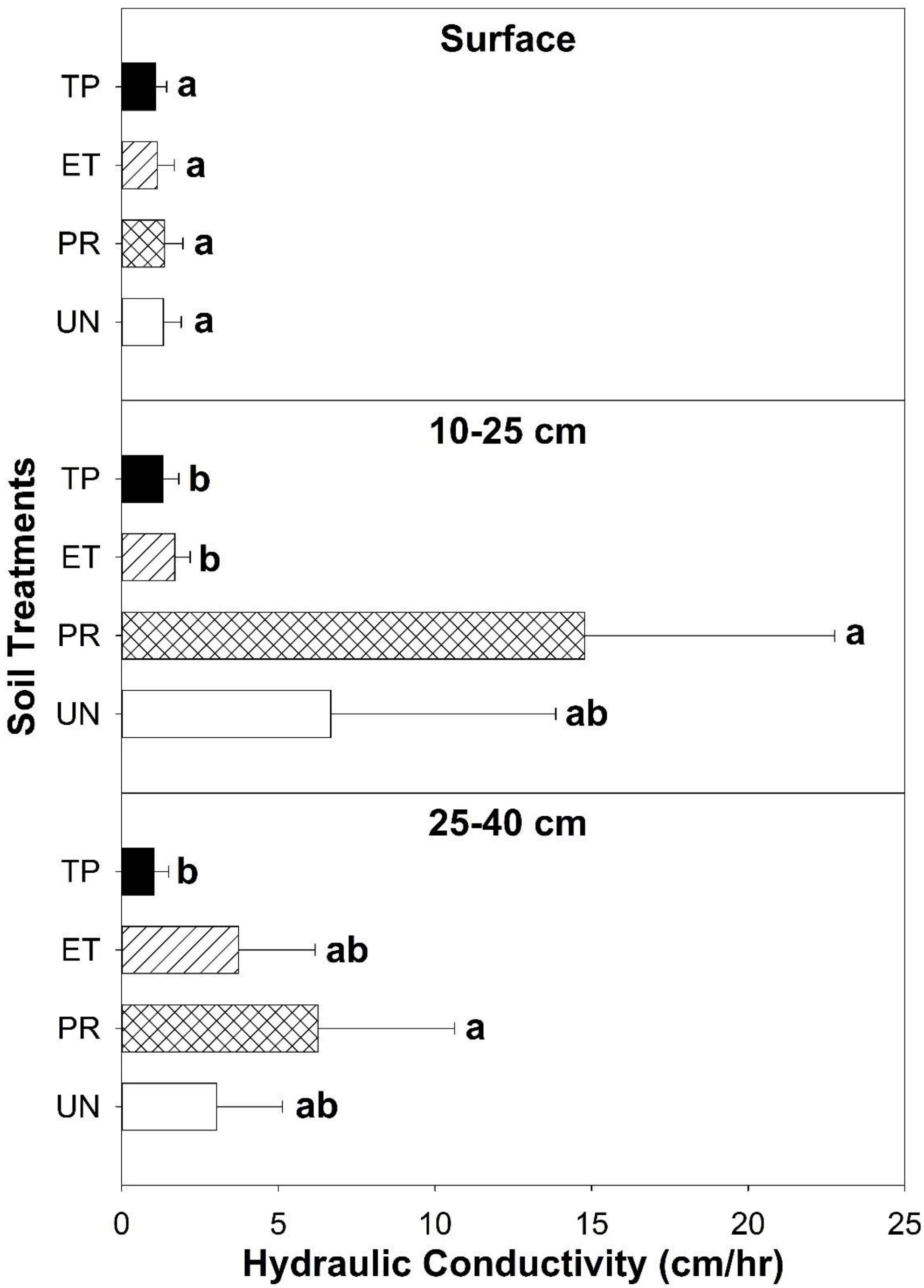

Figure 5.1. Mean unsaturated hydraulic conductivity (K_h) at soil surface when pressure head h= -2.0 cm (n=18) and saturated hydraulic conductivity (K_{sat}) at 10-25 cm and 25-40 cm soil depths (n=6) of typical urban land development practice (TP), two post-development soil rehabilitation practices (ET and PR), and undisturbed control soil (UN) five years after treatment installation. Bars depict standard error of the mean. At each soil depth, letters indicate significant differences across soil treatments at $\alpha = 0.05$ using Tukey's HSD.

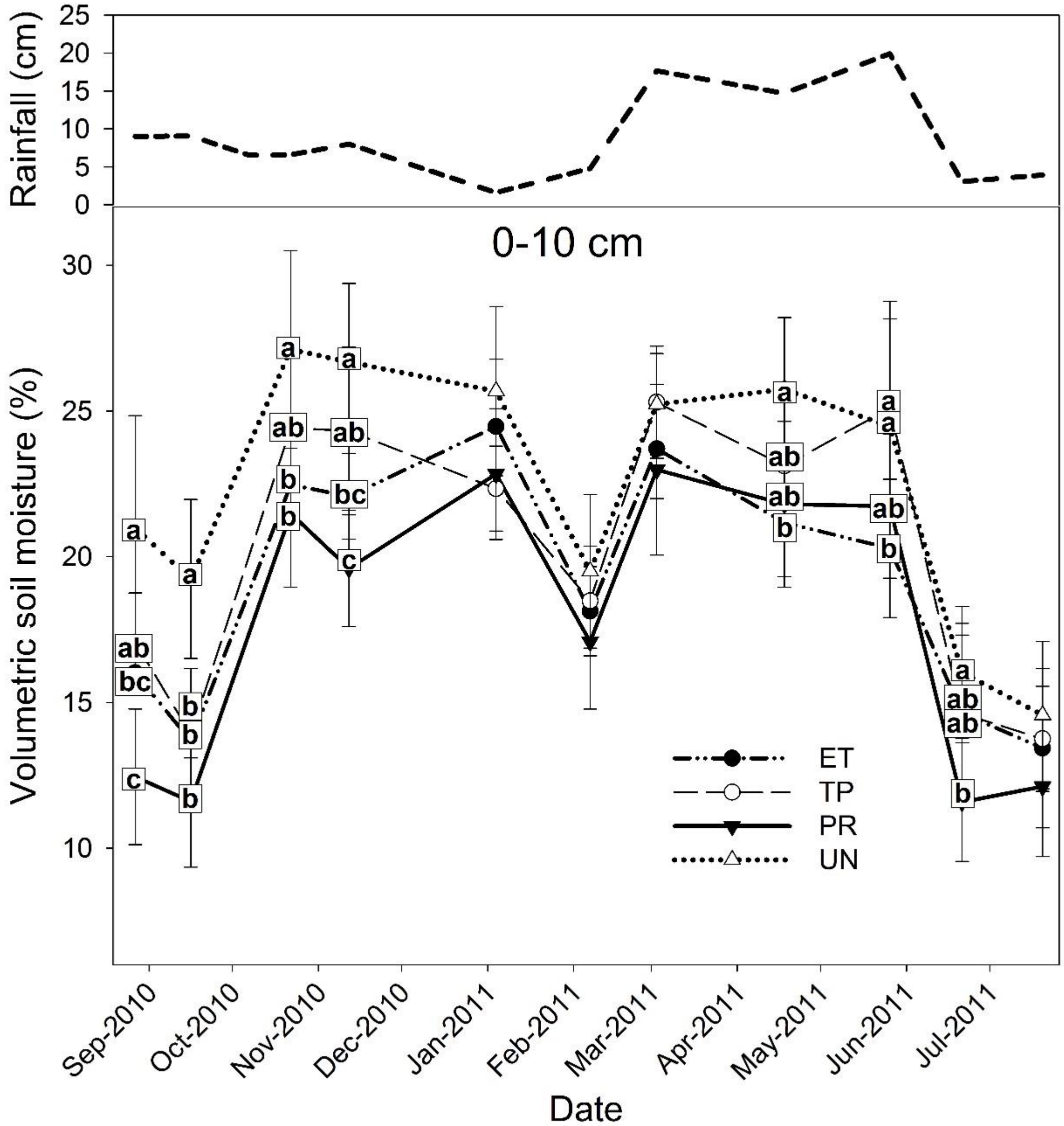

Figure 5.2. The monthly rainfall (cm) at site and monthly mean volumetric soil moisture (%) of typical urban land development practice (TP), two post-development soil rehabilitation practices (ET and PR) and undisturbed control soil (UN) in 0-10 cm soils from August 2010 to July 2011 (no data for December 2010). Error bars depict ± standard error of mean (n=6). Data were analyzed by repeated-measures. In each month, letters denote differences among treatments using Tukey's HSD at α=0.05.

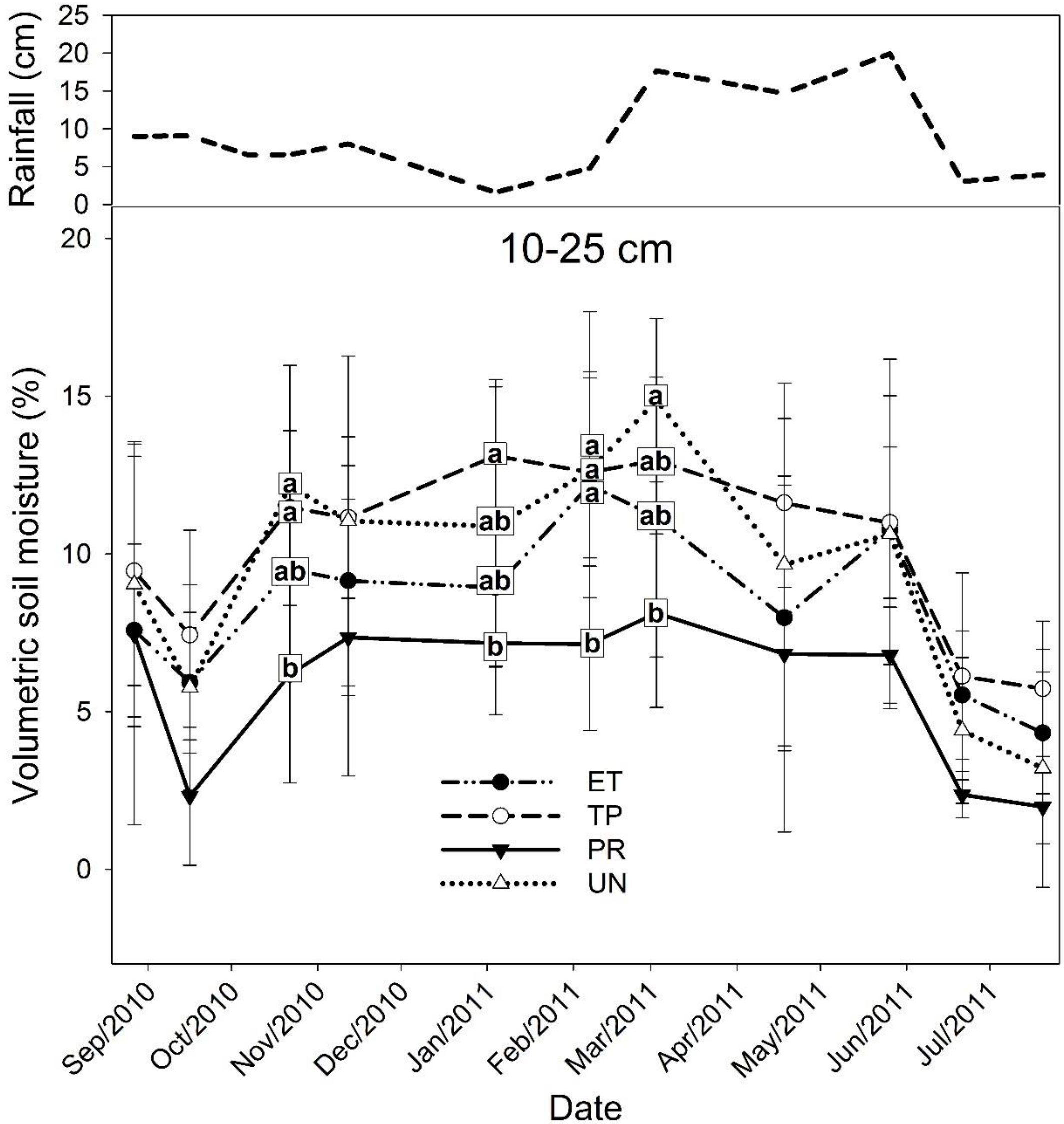

Figure 5.3. The monthly rainfall (cm) at site and monthly mean volumetric soil moisture (%) of typical urban land development practice (TP), two post-development soil rehabilitation practices (ET and PR) and undisturbed control soil (UN) in 10-25 cm soils from August 2010 to July 2011 (no data for December 2010). Error bars depict ± standard error of mean (n=6). Data were analyzed by repeated-measures. In each month, letters denote differences among treatments using Tukey's HSD at α=0.05.

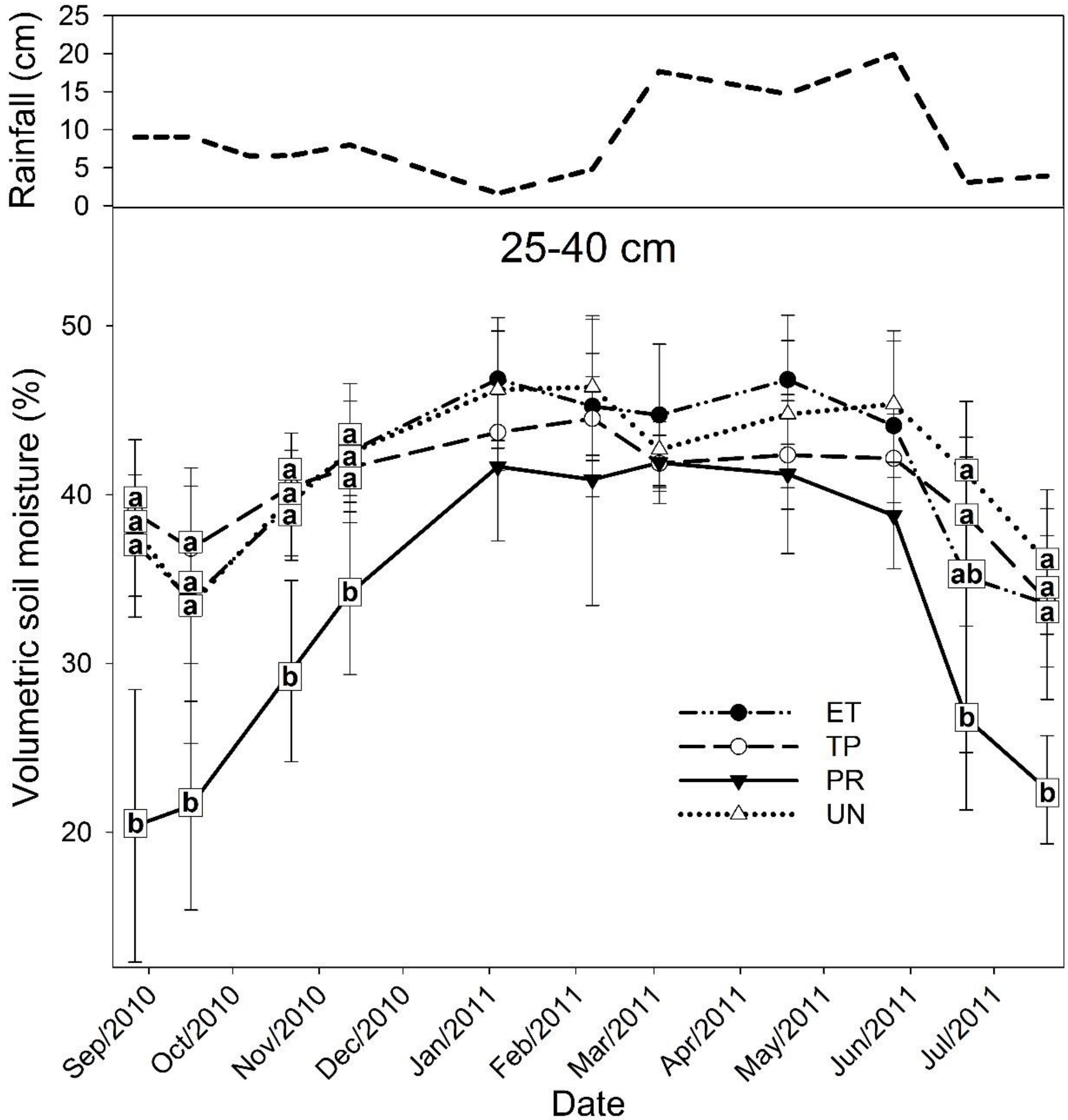

Figure 5.4. The monthly rainfall (cm) at site and monthly mean volumetric soil moisture (%) of typical urban land development practice (TP), two post-development soil rehabilitation practices (ET and PR) and undisturbed control soil (UN) in 25-40 cm soils from August 2010 to July 2011 (no data for December 2010). Error bars depict ± standard error of mean (n=6). Data were analyzed by repeated-measures. In each month, letters denote differences among treatments using Tukey's HSD at α=0.05.

5.3.3 Soil bulk density

Soil bulk density was not affected by different soil treatments at 2.5-7.6 cm and 20.5-35.5 cm depths. Profile rebuilding had lower soil bulk density (1.49 g cm^{-3}) at 15.2-20.3 cm depth but a slightly higher soil bulk density (1.76 g cm^{-3}) at 50.8-55.9 cm depth than the other three soil treatments (ranging from 1.61-1.76 g cm^{-3} and 1.67 to 1.70 g cm^{-3} at 15.2-20.3 cm and 50.8-55.9 cm, respectively) (Table 5.1).

Table 5.1. Mean soil bulk density (g cm^{-3}) of typical urban land development practice (TP), two post-development soil rehabilitation practices (ET and PR) and undisturbed control soil (UN) at four soil depths five years after treatment (n=6). Values in parentheses are standard error of the mean. At each soil depth, letters denote differences among treatments using Tukey's HSD at α=0.05.

	Bulk Density (g cm^{-3})			
	Soil depths			
Treatments	2.5-7.6 cm	15.2-20.3 cm	30.5-35.5 cm	50.8-55.9 cm
TP	1.38 (0.16)	1.76 (0.11)a	1.76 (0.06)	1.67 (0.05)b
ET	1.44 (0.06)	1.68 (0.09)ab	1.72 (0.04)	1.68 (0.03)b
PR	1.33 (0.04)	1.49 (0.22)b	1.79 (0.08)	1.76 (0.04)a
UN	1.43 (0.05)	1.61 (0.04)ab	1.72 (0.04)	1.70 (0.04)ab

5.4 Discussion

5.4.1 Hydraulic conductivity change during typical urban land development process

In our study, there was no evidence showing differences in surface unsaturated hydraulic conductivity among treatments (Fig. 5.1). Moreover, these values were low (0.44 to 2.28 cm hr^{-1}). This is likely because there was no surface treatment (e.g., grass, mulch) applied to any of the treatments. Surface measurements were made under slight tension to reduce any macropore effects in the measurements that might be source of high variability. Further study is needed to address the effects vegetation coverage or mulching treatment on surface permeability. At 10-25 cm and 25-40 cm soil depths, K_{sat} was greatest in rehabilitated soil, followed by undisturbed soils, while compacted soil was the lowest (Fig. 5.1). In most urban infiltration studies, soil infiltration rates decreased after urbanization (Gregory et al., 2006; Pitt et al., 2008; Yang and Zhang, 2011). In our study, the mean surface K_h of compacted soils was 1.1 cm hr^{-1}, which is close to the lower bound values reported in other similar studies (Hamilton and Waddington, 1999; Woltemade, 2010). In our study, post-development rehabilitated soils had about twice K_{sat} of undisturbed soils while as much as 11 times that of compacted urban soil (Fig. 5.1). The improvement resulting from urban soil rehabilitation via deep tillage-compost amendment-tree planting in our study is greater than surface compost amendment alone in another study that found K_{sat} of compost treatment was only 2.7-5.7 times that of the control (Olson et al., 2013). This effect might also be due to the role of trees in profile rebuilding because tree roots can improve infiltration significantly by deep root penetration and soil structure improvement.

5.4.2 The role of soil compaction and water content in soil infiltration change

We observed soil bulk density difference among treatments only at certain soil depths. At 15.2-20.3 cm soil depth, relative soil bulk density value was lowest in rehabilitated urban soil, and highest in typical land development practice soil, while undisturbed natural soil was intermediate. In contrast, at 50.8-55.9 cm soil depth, soil bulk density value was highest in rehabilitated urban soil, following by its in undisturbed soils, while typical land development practice soil was the lowest (Table 5.1). Soil saturated hydraulic conductivity at 25 cm was negatively correlated with soil bulk density measured at a slightly shallower location (15.2-20.3 cm) (r= -0.50), supporting previous findings of others (Pitt et al., 2004; Yang and Zhang, 2011).

We also found an interesting soil water content trend which rehabilitated urban soils had lower soil water content than undisturbed natural soils and typical land development practice soils was intermediate in our study (Figs. 5.2, 5.3, and 5.4). Pre-development natural soils had the highest soil water content compared to other treatments. This indicates that in the context of this experiment, undisturbed soils have optimal water holding capacity while disturbance resulting from typical development or post-development rehabilitation practice decreased this capacity. Perhaps most surprising was that rehabilitated soil had lower water contents than soils subjected to typical development practices. This finding is similar to a study conducted at Manchester, UK with Amsterdam soil (Rahman et al., 2011). They found lower soil moisture content at 20 cm depth in Amsterdam soil and hypothesized this might be due to higher infiltration rate and greater plant uptake. However, as we only measured the top 40 cm of the soils, it is unclear if rehabilitated soils have high soil water contents at greater depths.

5.5 Conclusions

Compacted urban soils resulting from typical urban land development practices decreased soil saturated hydraulic conductivity compared to pre-development undisturbed soils. However, post-development soil rehabilitation via a deep tillage-compost amendment-tree planting system improved soil infiltration significantly but not at the surface. The improvement is better than tillage or surface compost application alone indicating the important role of below-surface soil physical properties in stormwater mitigation.

However, disturbance resulting from typical practice and post-development rehabilitation reduced soil water holding capacity compared to undisturbed soils. Further study is needed to explore this effect on deeper soils. Our study site was in a humid temperate region previously in agricultural use, the results may be most relevant to sites with similar climate, land uses and management practices.

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Chapter 6

Summary and Conclusions

6.1 Summary and Study Implications

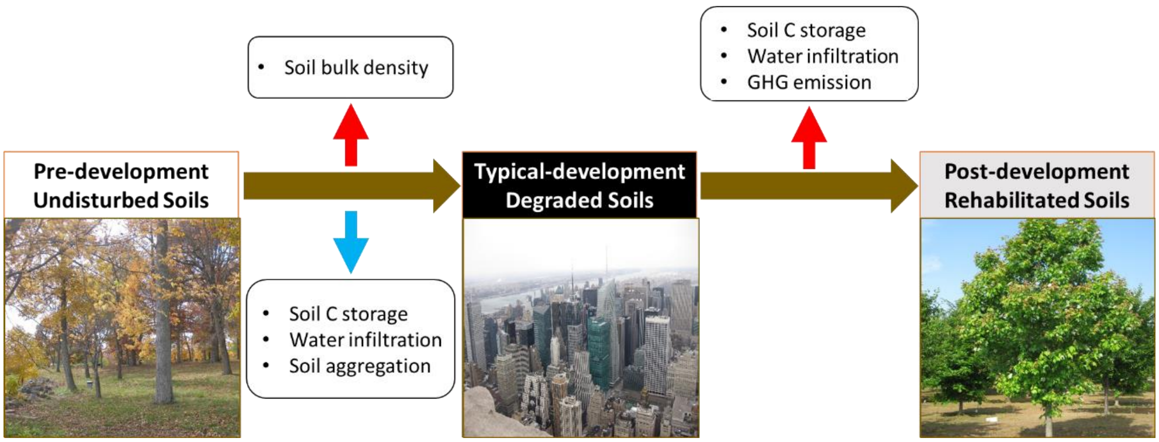

Figure 6.1. The overall comparison diagram of three different types of lands during urbanization: the changes of soil bulk density, soil aggregation, water infiltration, soil carbon (C) storage and soil-atmosphere greenhouse gas (GHG) emissions from pre-development undisturbed soils to typical-development degraded soils and from typical-development degraded soils to post-development rehabilitated soils.

This study assessed the effects of typical urban land development practices including topsoil removal and replacement, soil stockpiling and compacting and post-development soil rehabilitation via compost amendment, deep tilling and tree planting on soil physical properties, soil C dynamics, greenhouse gas emission and stormwater mitigation. We found that typical urban land development practices can alter soil physical properties dramatically and lead to significant soil C loss and reduced soil permeability, compared to pre-developed undisturbed soils. However, our findings demonstrate that urban soil rehabilitation is an effective approach to improve degraded urban soils in terms of providing compatible ecosystem services, although greenhouse gas emissions may increase due to increased biological activity and the disturbance associated with rehabilitation (Figure 6.1).

6.1.1 Soil physical characteristics

Urban soils are typically highly compacted with correspondingly high bulk density. However, urbanization not only increases soil bulk density, but also can lead to degraded soil structure. In our study, we found typical urban land development practices, including vegetation clearing, topsoil removal, stockpiling, compaction, and grading changed soil physical characteristics dramatically. Typical practices broke macroaggregates into microaggregates or fine fraction, especially in surface soils. Profile rebuilding did not appear to repair this damage within five years. Thus, soil disturbance resulting from topsoil handling during construction processes can dramatically damage soil structure, which may take a very long time to recover.

6.1.2 Soil carbon dynamics

Urban soil has the potential to be a C sink due to its increased and continuously expanding area and the capacity created by C loss during urbanization. Our study demonstrated that significant C loss occurs during construction processes, e.g., topsoil removal, stockpiling, replacement, and compaction, especially in surface soils. Topsoil handing not only reduced available and aggregate-protected C pools, but also decreased the most stable mineral-bound C pool. Effective management approaches are needed to minimize C loss during construction processes while rehabilitation practices are needed post-development to gain C in soils. Profile rebuilding as our urban soil rehabilitation practice increased C in both available and aggregate-protected pools in subsurface soils which could partially offset C loss in surface soils.

To further understand C changes tied to different management practices, we studied aggregate size distribution and aggregate associated C. We found that breaking macroaggregates into smaller aggregates might contribute to C loss during construction processes. Profile rebuilding increased C storage in subsurface soil because it increased aggregate associated C concentration. Long term study is needed to fully assess rehabilitation effects on soil C dynamics.

6.1.3 Greenhouse gas emission

To fully understand typical urban land development practice and post-development rehabilitation practice effects on the atmosphere-soil system, we studied GHG emission. We found soils under typical urban land development practices have similar GHG emissions compared to undisturbed soils. However, urban soil rehabilitation via incorporating compost and subsoiling increased GHG emissions, especially CO_2. We did not evaluate the C sequestration role of trees, however, thus our results represent only the soil component rather than the GHG emissions of the entire system. Future studies could integrate vegetation into the GHG emission evaluation of different soil management practices to give a more holistic view.

6.1.4 Stormwater mitigation

In our study, due to surface sealing, surface infiltration rates did not differ among different treatments. However, at deeper soil depths, we found post-development rehabilitated soils had about twice the permeability of pre-development undisturbed soils and approximately 6 -11 times that of typical development degraded soils. This indicates that the proposed rehabilitation system (compost amendment-deep tilling-tree planting) has the potential to mitigate urban stormwater. The influence of particular land management practices on stormwater runoff generation in urban areas is of great interest to policy makers as they seek to protect water quality by reducing nonpoint source pollution. Further study is needed to explore surface management practices (e.g., mulching, turf grass) that could enhance this mitigation effect.

6.1.5 Comparison of three different types of soils during the urbanization process

Integrating individual ecosystem elements presented above, we found a decrease in soil C storage, soil permeability, and aggregation, but an increase in soil bulk density when the land converted from pre-developed undisturbed land to land subject to typical land development practices. Typical urban land development practices dramatically disturbed soils. This disturbance not only physically damaged soil structure, but also influenced soil function and capacities, such as soil C storage and water movement through the profile. In contrast, post-development rehabilitation effectively increased soil C storage and soil infiltration rates five years after rehabilitation treatment. Thus, rehabilitation is an effective approach to repair degraded urban soils. We also found that rehabilitated soils had higher soil-atmosphere GHG emission. However, we did not account for CO_2 uptake by vegetation, thus, it is unclear if rehabilitated soils create a net GHG source or sink when the whole system is considered. We replicated compacted urban soils with high soil bulk density in this study and found the compaction level decreased in all treatments after five years. Thus, soils have the ability to recover some important soil characteristics after a certain period of time (e.g. 5 years). However, other degraded soil properties (e.g., aggregation) cannot be recovered within this time frame even with additional rehabilitation effort.

Land use change and soil management practice not only affect soil properties, but also can influence regional environment and global climate. We discussed the changes in soil C storage and GHG emission associated with these three different types of soils. The loss or gain of C and related GHG emissions can shape the regional C budget and global C cycle, especially as urban land is increasing all over the world. We can gain insight that we do have opportunities to act effectively to achieve a sustainable environment.

6.2. Methodology Evaluation

In the previous studies, heterogeneity has been documented as the biggest issue for urban ecosystem studies due to high variation of land uses, human activities, management practices, and development histories. In our study, we initiated a controlled experiment site to replicate different types of potential lands during urbanization processes including pre-development undisturbed, typical development practice and post-development rehabilitated lands. This study approach reduces the high variation in the actual urban settings, allowing the effects of management practices to be more easily discerned. On the other hand, a controlled experiment, such as ours, can eliminate some influential factors, such as human activities and reduce inference space. The combination of this type of controlled experiment and actual urban study will be helpful to better understand the urban ecosystem.

In our study, we evaluated the effects of typical practices and soil rehabilitation techniques five years after treatment. However, these effects could change over time. Future long term studies can be considered to have repeated measurements at different land development stages over time, e.g., initial status of pre-developed, right after typical development, a short time period after post-development and a long term after post-development. This can provide dynamic information for understanding the influence of management practices.

In our study, we determined three different soil C pools among different treatments. This provided valuable information to learn the effects of our rehabilitation techniques and the mechanism of C transformation in the soils. However, we have a sampling gap between 10 cm to 15 cm. Thus, there was a limitation to our estimates of the total soil C storage across different soil depths. There is a growing interest to study subsoil C. However, the soil C content in subsoil is lower than topsoils, especially in deeper horizons. Moreover, small amounts of soil C were not detectable by the soil density C fractionation technique we used in our study. Thus, an alternative technique is needed if we want to study subsoil C in future studies. We also measured soil-atmosphere GHG emissions. However, a full carbon budget for this type of land management would be valuable. Such an analysis would include information regarding the C input from compost and the C storage in the trees to fully understand the C cycle.

In our stormwater study, we measured soil hydraulic conductivity at the surface, and at 10-25 cm and 25-40 cm depths. The results provide information on the water movements from the surface into the soil, and from surface soils to subsurface soils. However, the information below 40 cm would improve understanding of the dynamics in deeper soils. Also, we did not account for the role of the trees in stormwater mitigation, e.g., tree canopy interception, tree root penetration. Future study can incorporate deep soil depths and vegetation.

6.3 Recommendations for Future Research

We replicated urban soils in our control study and, the next step for further study will be to explore the effects of these practices in actual urban situations. Our results indicate that it might take longer than we expected to rehabilitate degraded soils, thus, long term study is needed to assess rehabilitation effects. Further study might also include quantifying C input and output during the rehabilitation process. In addition, in this study, we did not study the role of vegetation, e.g., C sequestration, even though we presented the indirect effect of trees on stormwater mitigation. In summary, this study demonstrated that urban soil rehabilitation has potential to sequester more C and improve soil hydraulic conductivity in subsurface soils. Although not the topic of this dissertation, further benefits may also be realized through improved tree growth and the ecosystem services provided by this dynamic tree/soil system in urban settings.

Appendix A: Initial Soil and Compost Data from Layman (2010)

Table A.1. The total soil carbon (C), total soil nitrogen (N), and carbon to nitrogen ratio (C/N ratio) in initial soil profile before treatment installation in 2007.

Sample #	Horizon	% N	% C	C/N Ratio
1	Ap	0.13	1.36	10.09
	AB	0.04	0.39	10.34
	Bt1	0.02	0.15	6.44
	Bt2	0.12	0.14	5.59
2	Ap	0.13	1.33	10.56
	AB	0.03	0.36	11.22
	Bt1	0.02	0.16	8.45
	Bt2	0.03	0.23	8.44
	Bt3	0.02	0.14	8.26

Table A.2. The total carbon (C) and nitrogen (N) content of leaf compost

Sample #	Weight (g)	% N	% C
1	302.5	0.83	15.0
2	295.1	0.77	14.5
3	304.1	0.79	15.6

Table A.3. Mean soil bulk density (g cm^{-3}) of typical urban land development practice (TP), two post-development soil rehabilitation practices (ET and PR) and undisturbed control soil (UN) at four soil depths in 2008 (n=12). Values in parentheses are standard error of the mean. At each soil depth, letters denote differences among treatments using Tukey's HSD at α=0.05.

	Soil depth			
Treatments	2.5-7.6 cm	15.2-20.3 cm	30.5-35.5 cm	50.8-55.9 cm
TP	1.39 (0.13)	1.76 (0.17) a	1.76 (0.02)	1.77 (0.03)
ET	1.39 (0.06)	1.84 (0.04) a	1.69 (0.03)	1.76 (0.01)
PR	1.28 (0.03)	1.34 (0.05) b	1.68 (0.06)	1.75 (0.01)
UN	1.52 (0.04)	1.51 (0.14) ab	1.55 (0.15)	1.62 (0.08)