



Multi-factor controls on terrestrial carbon dynamics in urbanized areas

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Abstract. As urban land expands rapidly across the globe, much concern has been raised that urbanization may alter the terrestrial carbon cycle. Urbanization involves complex changes in land structure and multiple environmental factors. Little is known about the relative contribution of these individual factors and their interactions to the terrestrial carbon dynamics, however, which is essential for assessing the effectiveness of carbon sequestration policies focusing on urban development. This study developed a comprehensive analysis framework for quantifying relative contribution of individual factors (and their interactions) to terrestrial carbon dynamics in urbanized areas. We identified 15 factors belonging to five categories, and we applied a newly developed factorial analysis scheme to the southern United States (SUS), a rapidly urbanizing region. In all, 24 numeric experiments were designed to systematically isolate and quantify the relative contribution of individual factors. We found that the impact of land conversion was far larger than other factors. Urban managements and the overall interactive effects among major factors, however, created a carbon sink that compensated for 42 % of the carbon loss in land conversion. Our findings provide valuable information for regional carbon management in the SUS: (1) it is preferable to preserve pre-urban carbon pools than to rely on the carbon sinks in urban ecosystems to compensate for the carbon loss in land conversion. (2) In forested areas, it is recommendable to improve landscape design (e.g., by arranging green spaces close to the city center) to maximize the urbanization-induced environmental change effect on carbon sequestration. Urbanization-induced

environmental change will be less effective in shrubland regions. (3) Urban carbon sequestration can be significantly improved through changes in management practices, such as increased irrigation and fertilizer and targeted use of vehicles and machinery with least-associated carbon emissions.

1 Introduction

Urbanization – the aggregation of population in cities and transformation of rural areas into urban/developed land use – has become a dominant demographic trend and important land transformation process in recent decades (Seto et al., 2010; Kuang et al., 2014; He et al., 2014). At present, about 3–5 % of global land area has been converted to urban and developed land use (hereafter referred to as urban) (Svirejeva-Hopkins and Schellnhuber, 2008; Seto et al., 2010), 13–17 % of which was intensively developed (Schneider et al., 2010; Kuang, 2013). Urban areas in the United States increased by about 130 % between 1960 and 2000 (www.census.gov, last access: July 2012). Global urban areas could increase by about 1 million square kilometers over the next 25 years (McDonald, 2008). The spatial prominence of urban areas and fast urban land conversion rate is reason enough to study its environmental impacts (Kuang, 2011; Zipperer and Pickett, 2012). A major finding of urban ecological research in the past decade is that urban ecosystems play an important role in both local and regional biogeochemical cycles (Imhoff et al., 2000; Pataki et al., 2003;

Kuang, 2012a). The urban ecosystems could account for a significant portion of terrestrial carbon (C) storage (Nowak and Crane, 2002; Pataki et al., 2006; Pouyat et al., 2006; Churkina et al., 2010; Davies et al., 2011; Hutyra et al., 2011; Edmondson et al., 2012). Zhang et al. (2012) estimated that urban and developed land accounts for about 6.7–7.6% of total ecosystem C storage within the southern United States (SUS), which is larger than the pool size of shrubland. The potential for C sequestration in urban vegetation (McPherson et al., 1997) and soil (Pouyat et al., 2008) has drawn attention from both ecologists and decision makers (Poudyal et al., 2010). Municipal interest in climate change mitigation through C offset trading has increased as many cities have established substantial programs, such as tree planting, to increase ecological services of urban ecosystems (Nowak, 2006; Tratalos et al., 2007; Young, 2010). A management strategy for urban and peri-urban land, as suggested by the Intergovernmental Panel on Climate Change (IPCC, 2000), including tree planting, improved waste management, and wood production, could lead to a C sink of $0.3 \text{ t C ha}^{-1} \text{ a}^{-1}$. Escobedo et al. (2010) indicated that urban forest management can create moderate carbon sink in the southeastern United States.

However, the ecological consequence of urbanization is highly complex (Pickett et al., 2011), not only because of the strong spatial heterogeneity of urban ecosystems (Kuang, 2012b), which is composed by land cover types with distinct biogeochemical characteristics (Cannell et al., 1999; Alberti, 2005; Buyantuyev et al., 2010), but also because urbanization usually results in significant changes in many interacting environmental factors that affect ecosystem C processes, such as land conversion from rural to urban land use (Schaldach and Alcamo, 2007), shifts in disturbance and management regimes (Kissling et al., 2009; Fissore et al., 2012), and urban-induced climate and atmospheric changes (Korner and Klopatek, 2002; Fenn et al., 2003; Kuttler, 2011; Li et al., 2011). Furthermore, the legacy effect of pre-urban land use changes (Ramalho and Hobbs, 2012) and influences from global climate changes (McCarthy et al., 2010) could also modify the ecosystem's responses to the urbanization-induced environmental changes. Analyzing the impacts of these changes and their interactive effects will help in our understanding of how regional C cycles are affected by urbanization, quantifying the impacts of various environmental stresses, and identifying the major factors that control C dynamics of developed areas. Such knowledge can be valuable for policy makers and managers to predict the long-term ecological consequences of urbanization, to elucidate where management efforts should focus, and to formulate meaningful guidelines and tailor strategies for urban C managements.

Despite its importance and complexity, urbanization is an often-missing component in global change studies (Kaye et al., 2005; Pouyat et al., 2006). There are several remote sensing analyses that addressed the urbanization effect on net primary productivity (NPP) (Imhoff et al., 2000; Milesi et al.,

2003; Lu et al., 2010). With an empirical inventory approach, Cannell et al. (1999) roughly estimated the effects of urbanization on the C budget of the United Kingdom. Only a few modeling studies have analyzed the responses of regional C dynamics to the environmental changes induced by urbanization. Many studies suggested that urban land conversion could have a strong negative impact on regional to global C storage (Schaldach and Alcamo, 2007; Svirejeva-Hopkins and Schellnhuber, 2008; Zhang et al., 2008; Eigenbrod et al., 2011). Trusilova and Churkina (2008) compared the impacts of different urban-induced environmental changes on the C cycle in Europe and found strong C sequestration due to urbanization-induced atmospheric changes. Milesi et al. (2005) assessed effects of different management practices on the C storage of United States urban lawn. Zhang et al. (2012) found that pre-urbanization vegetation type and time since land conversion were closely related to the extent of urbanization effects on C dynamics of the southern United States over the last 6 decades. Despite these efforts, a comprehensive study that investigates the dominant environmental changes and addresses their relative importance on regional C dynamics is still not available, although it has been repeatedly suggested that, due to the complex interactions among multiple involving factors, the ecological consequences of urbanization could not be fully understood without a full set of controlling drivers and their interactions being addressed (Hutyra et al., 2011; Pickett et al., 2011; Ramalho and Hobbs, 2012).

In this study, we first comprehensively analyzed the factors that may control the urbanization effect on the ecosystem C dynamic (Fig. 1), and we proposed a numeric experimental scheme: a scenario design to conduct factorial analysis on the effects of different factors in Sect. 2. Then, as a case study of the newly developed analysis scheme, the dynamic land ecosystem model (DLEM) (Tian et al., 2011a) was applied to quantify the urbanization effect on the C dynamics of the SUS from 1945 to 2007, and to analyze the relative contributions from each environmental factors and their interactive effects (Zhang et al., 2012). SUS was selected as the study area because it was identified as the region with the most rapid urbanization in the United States, where about one-third of the developed area had been added in the last 15 years of the 20th century (Alig et al., 2004). Our study only considered the C dynamics of ecosystem (i.e., vegetation and soil) in land. Fossil fuel emissions unrelated to urban managements were out of the scope of this study (Townsend-Small and Czimczik, 2010; Bartlett and James, 2011). The objectives of this study were to (1) identify the controlling factors on C dynamics during urbanization and organized them into a comprehensive analysis framework that shows the relationship among the factors; (2) to develop a set of numeric experimental scheme to isolate and quantify the relative contribution of each factor; (3) to provide useful information for regional C management in the SUS (as a case

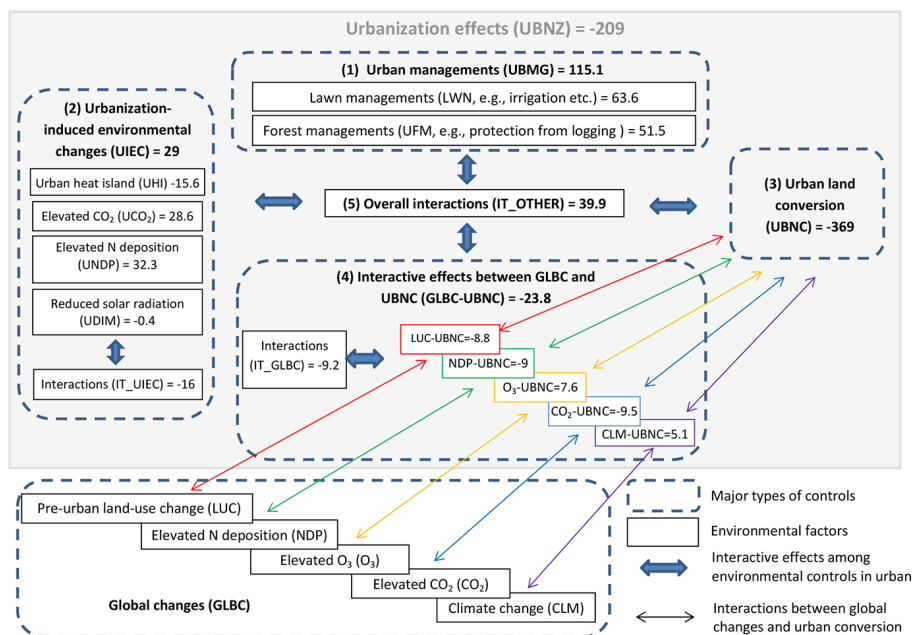


Figure 1. The urbanization (UBNZ) effects on regional carbon dynamics are controlled by four major types of environmental changes, including (1) urban land conversion (UBNC) during which rural land use type is converted to urban and developed land use composed by impervious surfaces, managed urban lawn, and other urban vegetation (e.g., urban forest); (2) urban management (UBMG), including lawn (LWN) management, such as irrigation and fertilization, and urban forest management, such as protection from logging and fire disturbances; (3) urbanization-induced environmental changes (UIECs), including effects of urban heat island (UHI), elevated CO_2 (UCO_2) and N deposition (UNDP), reduced solar radiation due to air pollution (UDIM), and interactions among these UIEC factors (IT_UIEC); (4) the interactive effects between UBNC and multiple global environmental changes (GLBC–UBNC), including changes in climate (CLM–UBNC), CO_2 (CO_2 –UBNC), N deposition (NDP–UBNC), ozone exposure (O_3 –UBNC), pre-urban land use change history (LUC–UBNC), such as cropland conversion and abandonment, and the interactions among all GLBC–UBNC factors. IT_OTHER represents the overall interactive effects among the four major controls (i.e., UBNC, UBMG, UIEC, and GLBC–UBNC). The numbers in the figure show the carbon flux in response to each factor from 1945 to 2007 in the southern United States. Unit: TgC.

study) by identifying the dominant controls on the C dynamics and elucidating where management efforts should focus.

2 Factors controlling urbanization effects

To study the effects of urbanization on regional C balance, Zhang et al. (2012) compared the model simulation results of the urbanization scenario (or the “business as usual” scenario) against the results from a non-urbanization scenario in which the urbanization process was controlled and all lands remained in pre-urban land types. They found that the urbanization from 1945 to 2007 resulted in a regional C loss of 0.21 Pg C in the SUS. The study, like others (McCarthy et al., 2010), also indicated that urbanization is not a simple C release process, but involves complex changes in land structure and multiple environment factors, whose effects should not be treated independently. Whenever an ecosystem component is modified by one environmental stress, the ecosystem’s responses to other factors could also be altered due to the non-linear interactions among the coupled ecosystem components and processes (Wu, 1999). For example, elevated

CO_2 in urban areas could be particularly important in relieving water stress induced by urban heat island effect (Groffman et al., 2006). Therefore, it is important to consider all the major environmental factors and their interactive effects on C processes when studying the urbanization effects on regional C balance.

In Fig. 1, we generalize the factors that may control the urbanization (UBNZ) effects (descriptions for the abbreviations are found in Fig. 1): (1) urban vegetation is intensively managed. Irrigation, fertilization, and weed disease controls improve lawn productivity (Milesi et al., 2005). Remnant ecosystems in urban areas are generally protected from intensive disturbances, such as agricultural soil tillage, wild fire, and commercial logging (Raciti et al., 2011). All of these urban managements (UBMGs) could result in high C density in urban ecosystems, as observed in former studies (Nowak and Crane, 2002; Hutyrá et al., 2011; Edmondson et al., 2012). (2) Urbanization-induced environmental changes (UIECs), such as urban heat island (UHI), elevated CO_2 (UCO_2) and N deposition (UNDP), and reduced solar radiation due to aerosol pollutions (UDIM) could affect plant growth, succession, and soil respiration in urban areas

(Lovett et al., 2000; Awal et al., 2010; Zipperer, 2011). According to Shen et al. (2008), the interactive effects among these UIEC factors (IT_UIEC) should not be ignored. (3) Urban land conversion (UBNC) alters the landscape structure, where pre-urban land covers are replaced by impervious surfaces and artificial green spaces, such as urban lawns. During the process, vegetation biomass is removed, soils are disturbed, and a large amount of C are released from the ecosystem (Schaldach and Alcamo, 2007; Zhang et al., 2008). (4) Global changes (GLBCs) in climate (CLM), land use (LUC), and atmosphere – e.g., CO₂, N deposition (NDP), and O₃ – have different effects on different vegetation/land cover types. In this study, LUC only refers to pre-urban land use change. Because UBNC alters the vegetation/land cover type, it also indirectly affects the ecosystem's responses to global changes. For example, the legacy effects of pre-urban land use history could explain the spatial pattern and temporal dynamic of ecosystem C pools in urban and developed areas (Golubiewski, 2006; Jenerette et al., 2006). Therefore, the interactive effects between GLBC and UBNC (GLBC-UBNC) should not be ignored when investigating urbanization effects. Furthermore, the interactive effects among global changes (IT_GLBC) could have important ecological impacts (McMurtrie et al., 2008). (5) Finally, the interactive effects among the above four major type of urban controls (IT_OTHER in Fig. 1) should not be overlooked (Wu, 1999).

Numeric experiments and factorial analyses can be conducted to quantify the effects of each of the above factors on carbon balance. For this purpose, a model scenario scheme is presented in Table 1. Based on these scenario outputs, factorial analyses can be conducted to isolate the effect of individual factors and their interactive effects. According to Fig. 1, we have

$$\begin{aligned} \text{UBNZ} &= \text{UBNC} + \text{UBMG} + \text{UIEC} + \text{GLB-UBNC} \quad (1) \\ &+ \text{IT_OTHER} = S_{\text{UBNZ}} - S_{\text{GLBC}} \\ &\rightarrow \text{IT_OTHER} = (S_{\text{UBNZ}} - S_{\text{GLBC}}) \\ &- (\text{UBNC} + \text{UBMG} + \text{UIEC} + \text{GLB-UBNC}), \end{aligned}$$

where S_{UBNZ} is the urbanization scenario (or the business as usual scenario), and S_{GLBC} is the control scenario in which no urbanization takes place (Table 1). The difference indicates the overall urbanization effect on C balance (Zhang et al., 2012). UBNC is estimated with the S_{UBNC} scenario in which only urban land conversion occurs.

$$\begin{aligned} \text{UBMG} &= \text{LWN} + \text{UFM} \quad (2) \\ \text{LWN} &= S_{\text{LWN} \& \text{UBNC}} - S_{\text{UBNC}} \\ \text{UFM} &= S_{\text{UFM} \& \text{UBNC}} - S_{\text{UBNC}} \end{aligned}$$

$S_{\text{LWN} \& \text{UBNC}}$ and $S_{\text{UFM} \& \text{UBNC}}$ simulate the C balance in managed grass (lawn) and urban forests in (converted) urban areas, respectively. It should be noted that it is impossible to simulate urban land management without also simulating the urban land conversion. Their results are compared against the

UBNC to isolate the effects of lawn (LWN) and urban forest management (UFM).

$$\begin{aligned} \text{UIEC} &= \text{UHI} + \text{UCO}_2 + \text{UNDP} \quad (3) \\ &+ \text{UDIM} + \text{IT_UIEC} \\ &= S_{\text{UIEC} \& \text{UBNC}} - S_{\text{UBNC}} \\ &\rightarrow \text{IT_UIEC} = (S_{\text{UIEC} \& \text{UBNC}} - S_{\text{UBNC}}) \\ &- (\text{UHI} + \text{UCO}_2 + \text{UNDP} + \text{UDIM}) \end{aligned}$$

$S_{\text{UIEC} \& \text{UBNC}}$ simulates the combination effects of multiple urban induced environmental changes and urban land conversion. We cannot simulate urban induced environmental changes without also simulating urban land conversion (land use change). Therefore, the effects of UHI, UCO₂, UNDP, and UDIM are calculated similarly to the LWN and UFM:

$$\begin{aligned} \text{UHI} &= S_{\text{UHI} \& \text{UBNC}} - S_{\text{UBNC}} \quad (4) \\ \text{UCO}_2 &= S_{\text{UCO}_2 \& \text{UBNC}} - S_{\text{UBNC}} \\ \text{UNDP} &= S_{\text{UNDP} \& \text{UBNC}} - S_{\text{UBNC}} \\ \text{UDIM} &= S_{\text{UDIM} \& \text{UBNC}} - S_{\text{UBNC}}. \end{aligned}$$

Finally, the interactive effects between global changes and urban land conversion can be derived as

$$\begin{aligned} \text{GLB-UBNC} &= \text{LUC-UBNC} + \text{NDP-UBNC} \quad (5) \\ &+ \text{O}_3\text{-UBNC} + \text{CO}_2\text{-UBNC} \\ &+ \text{CLM-UBNC} + \text{IT_GLBC} \\ &= S_{\text{GLBC} \& \text{UBNC}} - (S_{\text{GLBC}} + S_{\text{UBNC}}) \\ &\rightarrow \text{IT_GLBC} = S_{\text{GLBC} \& \text{UBNC}} - (S_{\text{GLBC}} + S_{\text{UBNC}}) \\ &- (\text{LUC-UBNC} + \text{NDP-UBNC} \\ &+ \text{O}_3\text{-UBNC} + \text{CO}_2\text{-UBNC} + \text{CLM-UBNC}), \end{aligned}$$

where S_{GLBC} simulates the global change effects, and $S_{\text{GLBC} \& \text{UBNC}}$ simulates the combined effects of global change and urban land conversion. The difference between the result from the combined scenario and the sum of the GLBC and UBNC scenarios (i.e., S_{GLBC} and S_{UBNC}) shows the interactive effects between the two factors. Similarly,

$$\text{LUC-UBNC} = S_{\text{LUC} \& \text{UBNC}} - (S_{\text{LUC}} + S_{\text{UBNC}}) \quad (6)$$

$$\text{NDP-UBNC} = S_{\text{NDP} \& \text{UBNC}} - (S_{\text{NDP}} + S_{\text{UBNC}}) \quad (7)$$

$$\text{O}_3\text{-UBNC} = S_{\text{O}_3 \& \text{UBNC}} - (S_{\text{O}_3} + S_{\text{UBNC}}) \quad (8)$$

$$\text{CO}_2\text{-UBNC} = S_{\text{CO}_2 \& \text{UBNC}} - (S_{\text{CO}_2} + S_{\text{UBNC}}) \quad (9)$$

$$\text{CLM-UBNC} = S_{\text{CLM} \& \text{UBNC}} - (S_{\text{CLM}} + S_{\text{UBNC}}). \quad (10)$$

Detailed information about scenario design can be found in Table 1. Based on the work reported by Zhang et al. (2012), we conducted two additional scenarios to simulate urban C storage under extreme conditions (S_{CMAX} and S_{CMIN}) to assess the uncertainties related to model parameters. For S_{CMAX} , parameters were selected to maximize the C sequestration capacity of the urban ecosystem, while, for S_{CMIN} , parameters were selected to provide a conservative estimation.

Table 1. Scenario design for numeric experiments and factorial analysis.

Description	Scenarios	Factors												
		Urban land conversion	Global environmental changes					Urbanization-induced environmental changes					Urban managements and disturbance regimes	
			Climate	CO ₂	Nitrogen deposition	Ozone exposure	Pre-urban LUC	Urban heat island	Elevated urban CO ₂	Elevated urban N deposition	Aerosol dimming effect	Urban lawn management	Urban forest management	
All combined	S ^U BNZ	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
	S ^U BNZ_Cmin	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
	S ^U BNZ_Cmax	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Control (global changes only)	S ^G LBC	-	✓	✓	✓	✓	✓	-	-	-	-	-	-	-
	S ^C LM	-	✓	-	-	-	-	-	-	-	-	-	-	-
	S ^C O ₂	-	-	✓	-	-	-	-	-	-	-	-	-	-
	S ^N DP	-	-	-	✓	-	-	-	-	-	-	-	-	-
	S ^O 3	-	-	-	-	✓	-	-	-	-	-	-	-	-
	S ^L UC	-	-	-	-	-	✓	-	-	-	-	-	-	-
Urban land conversion only	S ^U BNC	✓	-	-	-	-	-	-	-	-	-	-	-	-
Global changes with urban land conversion	S ^G LBC & UBNC	✓	✓	✓	✓	✓	✓	-	-	-	-	-	-	-
	S ^C LM & UBNC	✓	✓	-	-	-	-	-	-	-	-	-	-	-
	S ^C O ₂ & UBNC	✓	-	✓	-	-	-	-	-	-	-	-	-	-
	S ^N DP & UBNC	✓	-	-	✓	-	-	-	-	-	-	-	-	-
	S ^O 3 & UBNC	✓	-	-	-	✓	-	-	-	-	-	-	-	-
	S ^L UC & UBNC	✓	-	-	-	-	✓	-	-	-	-	-	-	-
Urbanization-induced environmental changes	S ^U IEC & UBNC	✓	-	-	-	-	-	✓	✓	✓	✓	-	-	-
	S ^U HI & UBNC	✓	-	-	-	-	-	✓	-	-	-	-	-	-
	S ^U CO ₂ & UBNC	✓	-	-	-	-	-	-	✓	-	-	-	-	-
	S ^U NDP & UBNC	✓	-	-	-	-	-	-	-	✓	-	-	-	-
	S ^U DIM & UBNC	✓	-	-	-	-	-	-	-	-	✓	-	-	-
Urban managements	S ^U BMG & UBNC	✓	-	-	-	-	-	-	-	-	-	-	✓	✓
	S ^U LWN & UBNC	✓	-	-	-	-	-	-	-	-	-	-	✓	-
	S ^U FM & UBNC	✓	-	-	-	-	-	-	-	-	-	-	-	✓

Note: “✓” means changes in the environmental factor were considered, while “-” means the factor was unchanged in the simulation.
 * Following Zhang et al. (2012), UBNZ_Cmin and UBNZ_Cmax were designed to examine the effect of uncertainties in model parameters on the estimated urbanization effects. Parameters of UBNZ_Cmin were set so that carbon sequestration was minimized, while carbon loss was maximized during urbanization; UBNZ_Cmax was the contrary (Table 2).

3 Materials and methods in the case study

The DLEM is a process-based model that integrates the biophysical, biogeochemical, and hydrological processes to simulate impacts of environmental changes on water, C, and N cycles (see Fig. S1a in the Supplement). The model has been parameterized and validated against intensively studied natural sites, and it has been applied in multiple regional C dynamic studies (Tian et al., 2011a, b, 2012). Zhang et al. (2012) have developed an urbanization module for the DLEM to assess the impacts of urbanization on long-term C dynamics in the SUS. Their study, however, only focused on the overall effects of urbanization without investigating the relative contribution from individual factors. In this current study, by conducting factorial analysis, we examined the relative contribution of different environmental controls and their interactive effects on regional C dynamics during urbanization. Here, we briefly introduce the study area, model structure, and the development of model inputs, including the background of global-change data sets (Table S1 in the Supplement), as well as the parameters for human-induced changes in urban areas. More detailed descriptions are found in Zhang et al. (2012).

3.1 Study area

Because an ecological understanding of urban effect must include the suburban areas and settled villages, as well as city cores (Pickett et al., 2011; Liu et al., 2014), the “urban” areas

refer to all the urban and developed regions in the SUS in this study. This study focuses on the 1.2×10^5 km² urban lands in the SUS (red areas in Fig. S2 in the Supplement). Following Zhang et al. (2012), this study focuses on the impacts of urbanization from 1945 to 2007 on regional net carbon exchange (NCE). NCE quantifies the C balance (with positive values indicating C sequestration) of the ecosystems in response to environmental change in a certain period (Tian et al., 2003, 2011b).

3.2 Model description

Urban landscapes are composed by two major land functional types: urban impervious surface (UIS) and urban vegetation. Stearns (1971) identified three urban vegetation types: ruderal, residual, and managed. For simplification, ruderal and residual are merged into the dominant/potential local vegetation type in urban (UVG), and the managed vegetation is represented by urban lawn (ULW); an important characteristic of urban land use conversion with respect to the C cycle (Kaye et al., 2005; Golubiewski, 2006). Therefore, an urban landscape is treated as a mosaic of UIS, UVG, and ULW in the DLEM. The development of UIS and ULW land typically includes the clearing of existing vegetation, and massive movements of soil. The DLEM not only models the disturbances on vegetation and soil during land clearing, but also tracks the fate of removed biomasses, following the study of Houghton (1999) and Nowak and Crane (2002). Converting agricultural land to UVG will result in cropland abandonment

and regeneration of potential vegetation (Dwyer et al., 2000). Otherwise, UVG conversion will not directly disturb the pre-urban ecosystem. The disturbance regimes in UVG land, however, change after urbanization in the DLEM. Urban forest and other residual ecosystems are protected from wild-fire and commercial logging (Campbell et al., 2007; Defosse et al., 2011), disturbances that are responsible for the low biomass density in the SUS forest (Birdsey, 1992). Taking the disturbances' effect into account, the overall mortality rate of rural forest (Tian et al., 2012) is about 10 % higher than that of urban forest.

3.3 Model inputs

In model simulation, the background climate, atmosphere, and land use drivers were modified by urbanization-induced environmental changes, the values of which were estimated based on literature reviews (Table 2). The background environmental drivers provided global change information because they were transient data sets that changed annually or daily from 1945 to 2007. To control a certain global change driver, we fixed its value to the year 1945. For example, in the climate-only scenario (S_{CLM} in Table 1), only climate data changed from 1945 to 2007, the values of other drivers (CO_2 , N deposition, O_3 , and land use) being fixed to the value of 1945. If a certain urbanization-induced environmental change factor is considered in the simulation, the corresponding background value will be modified by its parameter from Table 2.

3.3.1 The background climate, atmosphere, and land use data set

We reconstructed an 8 km resolution daily climate data set of the entire SUS from 1895 to 2005 (Figs. 2a, b, c) by integrating the daily climate pattern of the North American regional reanalysis (NARR) (32 km resolution) data set (Mesinger et al., 2006) into the monthly PRISM (Parameter-elevation Regressions on Independent Slopes Model; 4 km resolution; 1895–present) climate data (Daly et al., 2008; prism.oregonstate.edu). A detailed description of the method is found in Zhang (2008). PRISM is a knowledge-based system to interpolate climate elements under the assumption that, for a localized region, elevation is the most important factor in the distribution of temperature and precipitation. To make predictions, PRISM dynamically calculates a linear climate–elevation relationship for each DEM grid cell using a moving window, a procedure that smooths out signals of urbanization-induced climate changes (Daly et al., 2008). Like most reanalysis data, the surface temperatures of NARR were estimated from the atmospheric values by regional climate modeling, and thus were not sensitive to changes in land surface (Kanamitsu et al., 2002). Therefore, the reanalysis data sets were used to provide information of background climate change in former UHI studies (Si et al., 2012). The cli-

mate change data from 1895 to 2005 that were reconstructed based on the PRISM and NARR data sets were used to address the background global climate change (Fig. 1; GLBC-CLM) in this study.

Ozone AOT40 data (Fig. 2d) were retrieved from a global data set developed by Felzer et al. (2004). EDGAR-HYDE 1.3 nitrogen emission data (Van Aardenne et al., 2001) were used to interpolate three maps from Dentener (2006) to generate a time-varying annual nitrogen deposition data set (Fig. 2e). Both data sources had coarse spatial resolutions ($0.5\text{--}1^\circ$) and were downscaled to $8 \times 8 \text{ km}^2$ using bilinear interpolation. Due to their coarse resolutions, and because the atmospheric models that were used to generate these data sets did not consider the local urbanization effects (Felzer et al., 2005; Dentener, 2006), the AOT40 and nitrogen deposition data sets represented the background global atmospheric changes in this study. The background global annual CO_2 concentration was obtained from the National Oceanic and Atmospheric Administration (NOAA) (www.esrl.noaa.gov).

To simulate the land use changes, the DLEM requires annual urban and cropland maps (1 represents urban or croplands; 0 represents natural vegetation). Distribution maps for cropland and urban/developed lands from 1895 to 2007 (Fig. 2f) were reconstructed by combining the contemporary land use map that was derived from NLCD2001 (Homer et al., 2007) with historical census data set for cropland, urban land, and population (Waisanen and Bliss, 2002). A detailed description can be found in Zhang et al. (2012).

Sources of other inputs, including the base maps (potential vegetation, soil properties, and topographic characteristics, etc.) and cropland management (irrigation and fertilization) data sets can be found in Table S1. A detailed description of the data development methodologies is found in Zhang (2008).

3.3.2 Urban-induced environmental changes

The DLEM further models the effect of urban-induced environmental changes (i.e., UHI, aerosol pollutions, and increased CO_2 and N deposition) to the urban ecosystem, which (except for the aerosol pollutions) generally enhance the growth and biomass accumulation rate of urban vegetation (Ziska et al., 2004). Based on literature reviews, we estimated the parameters controlling urban-induced environmental changes (Table 2). To evaluate the effects of parameterization uncertainties on the model simulations, we designed two additional scenarios to simulate urban C storage under extreme conditions: UBNZ_Cmin and UBNZ_Cmax (Table 1). Parameters of UBNZ_Cmin were set so that carbon sequestration was minimized, while carbon loss was maximized during urbanization; UBNZ_Cmax was the contrary (Table 2).

Table 2. The parameters of urban managements and urban-induced environmental changes.

Scenarios *	Lawn managements				Urban forest managements		Urbanization-induced environmental changes		
	Irrigation (Y/N)	Litter remove (Y/N)	Clipping interval (days)	Nitrogen fertilization (g N m ⁻² a ⁻¹)	% of biomass pruned	% of litter removed to land fill	Elevated CO ₂ (ppmv)	Reduced solar radiation due to aerosol (%)	Elevated NO ₂ (ppmv)
S _{UBNZ} _C _{min}	Y	Y	5	6.4	10%	25%	5	-10%	0.03
S _{UBNZ}	Y	Y	10	6.8	5%	10%	10	-5%	0.045
S _{UBNZ} _C _{max}	Y	N	15	7.3	0%	0%	20	0%	0.06

* S_{UBNZ} represented the normal condition (i.e., business as usual scenario). The S_{UBNZ}_C_{min} scenario provided a conservative estimation of the urban carbon storage, while the S_{UBNZ}_C_{max} scenario simulated the maximum carbon storage of urban/developed area.

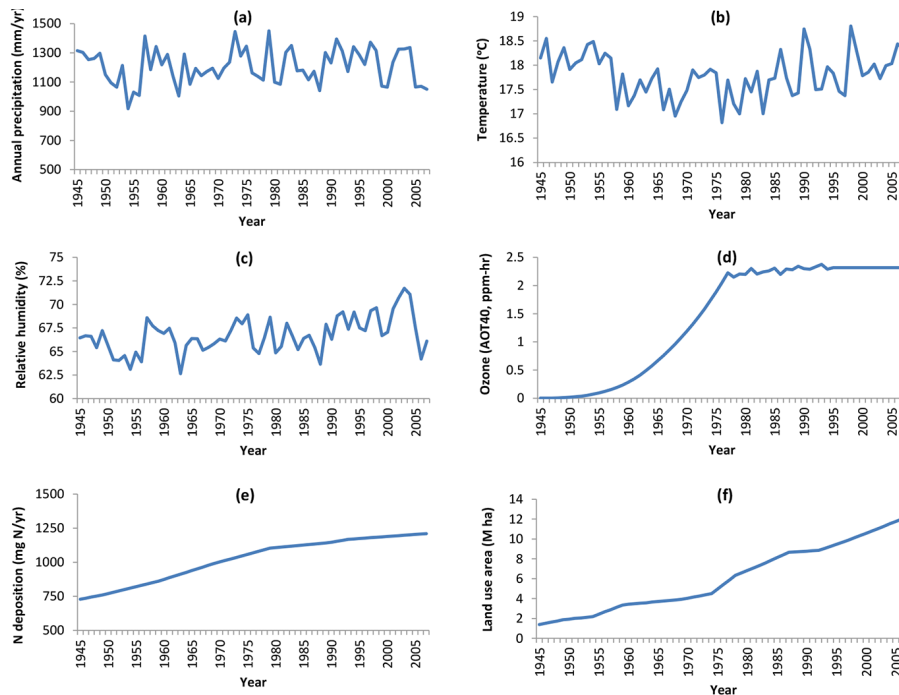


Figure 2. Temporal patterns of major global change factors in the study region from 1945 to 2007. (a) Annual precipitation; (b) temperature; (c) relative humidity; (d) ambient ozone exposure; AOT40 is the accumulated dose over a threshold of 40 ppb during daylight hours in a month (Felzer et al., 2004); (e) annual nitrogen deposition rate; (f) urban land changes.

The urban heat island effect

The DLEM estimates the elevated temperature in urban areas (i.e., UHI, unit: °C) with the regression model developed by Karl et al. (1988):

$$\text{UHI} = \alpha \times (p)^{0.45}, \quad (11)$$

where α is a regression coefficient that varies with seasons and size of urban population (p). Based on the climate records from 1219 stations in the United States, Karl et al. (1988) determined the values of α for the maximum, minimum, and average temperature for each season in three different urban sizes ($p < 10\,000$; $p \in [10\,000, 100\,000]$; $p > 100\,000$). To develop an 8 km resolution urban population data set from 1945 to 2007, county-level urban population data developed by Goldewijk (2005) was divided by the area of urban/developed land to calculate the mean urban population density of each county. Then, the

urban population map for each year was developed by multiplying the area of each urban region/patch in the NLCD 2001 land use map with the urban population density of the local county.

The elevated atmospheric CO₂ concentration in urban/developed lands

Rural-urban CO₂ gradient is highly variable depending on time, location, wind direction, and distance from traffic, etc. (Idso et al., 1998, 2001, 2002; Vogt et al., 2006). The reported daily urban-rural CO₂ gradient ranged from 5 (Berry and Colls, 1990) to 66 ppmv (George et al., 2007). However, the daytime CO₂ gradient that determines the CO₂ fertilization effect on urban ecosystem is usually much smaller than the daily average due to solar-induced convective mixing (Idso et al., 2002) and the C uptake by plants (Kordowski and Kuttler, 2010). Day et al. (2002) reported that the daytime

CO₂ concentration of the vegetated area in the center of Phoenix, AZ was only 8 ppmv higher than the background value. According to the measurements by Clark-Thorne and Yapp (2003), the daytime CO₂ concentration of the urban interior was averagely 5–7 ppmv higher than the rural CO₂ in Dallas, TX. Garcia et al. (2012) found that the daytime suburban CO₂ concentrations were 6–16 ppmv higher than rural levels in northern Spain. Based on these and other reports (Berry and Colls, 1990; Li et al., 2010; Rice and Bostrom, 2011), we assumed that the atmospheric CO₂ concentration of an urban vegetated area is 10 ppmv higher than the background value.

Urbanization-induced air pollution

Urban atmospheres have higher concentrations of nitrogen and aerosols than those in rural regions (Lovett et al., 2000; Azimi et al., 2005). In general, urban boundary layer pollutants are believed to reduce solar irradiance by 0–10% in North American cities (Oke, 1979, 1982; Peterson and Stoffel, 1980; Estournel et al., 1983). The DLEM assumed that aerosol pollutions reduce the urban solar radiation by 5%. Urban air pollution generates high concentrations of both ozone precursors and ozone scavengers. Gregg et al. (2003) found the detrimental effects of tropospheric ozone were lower in urban than in suburban areas. Due to the uncertainties in urban ozone (Trusilova and Churkina, 2008), we did not consider the urbanization-induced ozone change in this study. Like atmospheric CO₂, the temporal and spatial patterns of urban nitrogen deposition are highly variable. Previous studies indicate that the daytime atmospheric NO₂ concentration of urban and developed land usually ranges from 0.03 to 0.06 ppmv; an order higher than the value measured in rural ecosystem (Hanson et al., 1989). In this study, we used the mean value of 0.045 ppmv as the elevated urban NO₂ concentration.

It should be noted that these UIECs were not static but changed through time. In this study, we assumed that the rise of CO₂ and air pollutants in urban areas were positively correlated to the historical per capita fossil fuel emissions:

$$PL_i = PL_{\text{mean}} \times f_{\text{EMS}}, \quad (12)$$

where PL_i is the urbanization-induced atmospheric change (i.e., elevated NO₂, CO₂, and aerosol) in year i ; PL_{mean} refers to the mean urbanization-induced air pollution according to recent (since 1980) studies (parameters in Table 2); f_{EMS_i} is the normalized fossil fuel emission factor for year i .

$$f_{\text{EMS}_i} = \frac{\text{EMS}_i}{\text{EMS}_{1980_2000}}, \quad (13)$$

where EMS_i is the per capita annual fossil fuel emission of the United States in year i ; EMS_{1980_2000} denotes the mean value between 1980 and 2000. To calculate the annual per capita fossil fuel emissions, we obtained the annual

national fossil fuel emission data compiled by Marland et al. (2008) and the historical United States population data from the United States Census Bureau (<http://www.census.gov/population/www/popclockus.html>).

3.3.3 Urban managements

Lawn managements

Urban lawns are irrigated, fertilized, and clipped. In the DLEM, urban lawns are irrigated whenever the soil water content is lower than 50% of the field capacity. Since many of the United States lawns are irrigated excessively (Milesi et al., 2005), we may have underestimated the water use in irrigation. This uncertainty in irrigated water did not significantly affect the predicted C dynamics in urban lawn.

Based on the values provided by several reports in the literature, e.g., 8–9 g N m⁻² yr⁻¹ (Rockwell, 1929), 5–10 g N m⁻² yr⁻¹ (Thompson, 1961), 10 g N m⁻² yr⁻¹ (Qian et al., 2003), 2.4–15 g N m⁻² yr⁻¹ (Osmond and Hardy, 2004), 9 g N m⁻² yr⁻¹ (Law et al., 2004), and 9.7 g N m⁻² yr⁻¹ (Zhou et al., 2008), the DLEM assumes that 10 g N m⁻² yr⁻¹ will be the N fertilization rate for the professionally managed lawns. This value is close to the 10.9 g N m⁻² yr⁻¹ fertilization rate for the managed lawns in the United States as estimated by Zirkle et al. (2011). In reality, however, the rates of N fertilization to lawns varied significantly from household to household in the United States (Augustin, 2007). The professionally managed lawn only accounts for half of the United States lawn area (Grounds Maintenance, 1996). Only half of the home lawns are fertilized in a given year (Augustin, 2007). Only 25% of the fertilized home lawns are professionally managed. The remaining 75% are managed by the home owners with fertilization rates ranging from 4 to 9 g N m⁻² yr⁻¹. Combining all of this information, we deduced that the annual N fertilization rates to United States lawns vary from 6.4 g N m⁻² yr⁻¹ (when 4 g N m⁻² yr⁻¹ is applied by home owners) to 7.3 g N m⁻² yr⁻¹ (when 9 g N m⁻² yr⁻¹ is applied by home owners). In the simulation, the DLEM used the average value of 6.8 g N m⁻² yr⁻¹ for the urban lawn.

Urban lawns are usually clipped every 0.5 to 2 weeks in the United States (Milesi et al., 2005; Kaye et al., 2005). In this study, we assumed an averaged mowing cycle of 10 days in SUS. This estimation agrees with a 900-person survey in Illinois, which reported an average mowing rate of 30 per year (Zirkle et al., 2011). Following Milesi et al. (2005), a lawn will only be mowed if its leaf area index exceeds the threshold value of 1.5. After mowing, 20% of the vegetation biomass will be removed. The belowground biomass will enter the soil litter pool, while the aboveground portion will enter the product pool and decay in 1 year. All clipped biomass will enter the product pool and decay in 1 year.

Mortality and management of urban forest

We assumed that urban trees were protected from commercial logging, and thus could grow very old. Large uncertainties exist in the mortality rate of urban trees. Field measurements revealed that street trees could have various mortality rates depending on their size: -2.1 to 3.0% for trees whose DBH < 77 cm; and 5.4% for larger trees (Nowak, 1986). Nowak (1994) assumed an annual mortality rate of 2.6% in their urban forest modeling study. In the DLEM, the annual background mortality rate of urban trees ranged from 2.2 to 3.5% , positively correlated with tree size (Nowak, 1986). Following Sitch et al. (2003), the background mortality is modified by light competition at the stand level. In the DLEM, forest die back will take place to maintain the foliage-projected coverage under 95% . Urban forests have a relatively open canopy compared to rural forests, providing it an advantage to suppress light competition and support bigger trees. The DLEM calculates the foliage-projected coverage of urban forest based on the total area of the urban land to simulate the open canopy effect.

For simplification, we assumed rural and urban trees had the same background mortality. Unlike urban forests, rural forests in the SUS were frequently disturbed by wildfire and commercial logging (Campbell et al., 2007; Defosse et al., 2011), which were responsible for the low biomass density in the SUS forest (Birdsey, 1992). Taking the disturbances' effect into account, the overall mortality rate of rural forest is about 10% higher than that of protected forest (Tian et al., 2012). Following Chen (2010), we implicitly modeled the disturbances' effect on the rural forest in the SUS by increasing their background mortality by 10% .

Like lawns, urban forests may be managed by pruning and litter raking. It was found that intensive pruning might reduce the biomass of urban trees by as much as 25% (Nowak, 1994; Nowak et al., 2002; Escobedo et al., 2010). Unlike lawn, however, intensively managed trees, such as street trees that account for about 62% of the managed urban forest in the United States (Kielbaso, 2008), only contribute to a small fraction (e.g., $2-4\%$ in Oakland, CA, and Chicago) of urban forests (Dwyer et al., 2000). Furthermore, a national survey revealed that more than 60% of United States cities do not have urban forest management programs (Kielbaso, 2008). Even if all cities in the SUS have a forest management program, and 10% of urban forest is street tree that accounts for 50% of the managed forests, managed trees will only account for 20% of urban forests. Under this assumption, about $20\% \times 25\% = 5\%$ of the forest biomass was removed by pruning (Nowak, 1994; Nowak et al., 2002) (Table 2).

In some managed urban forests, a fraction of the litter (such as the litter from the pruned trees) will be removed and disposed of in a landfill. Nowak et al. (2002) assumed that only 3.7% of the removed carbon would be released during the first 5 years, and the remaining would be permanently locked up in a landfill. Accordingly, the DLEM sim-

ulated the process of litter removal by allocating 1.85 , 1.85 , and 96.3% of the removed litter to 1-, 10-, 100-year product pools that have turnover rates of 1, 10, and 100 years, respectively. No information about the patterns of litter management is currently available for the urban/developed land in the SUS. Since the fraction of intensively managed urban forest is quite low (Dwyer et al., 2000), we assumed that only 10% of litter will be removed and disposed of in a landfill (Table 2).

3.4 Model evaluation

Urban ecosystem modeling is bound to large uncertainties (Churkina, 2008). Consistency between model results and field measurements is essential to establish the credibility of simulated C dynamics. Previously, we validated the DLEM simulated C and water fluxes, nitrogen cycle, soil processes, and trace gas emissions against intensively studied ecological research sites (Tian et al., 2011a, b). Because urbanization does not change genetic characteristics of plants or fundamental mechanisms of ecological processes (Niemela, 1999), former validation results indicated that the DLEM can correctly simulate the ecosystem's responses to multiple environmental stresses in urbanized areas. To evaluate the DLEM's performance for simulating C processes in urban ecosystems, we further compared model predictions with 16 field observations – including vegetation carbon (VEGC), soil organic carbon (SOC), and NPP – from 12 studies that were located in or close to the SUS (Table 4). For those studies with sample variance, all of our model predictions fall into the range of 1 standard error.

Urbanization has complex effects on local climate, atmosphere, and disturbance regimes. Urban environment conditions and land management vary from place to place and from time to time (Alberti, 2005). Because the ecophysical and socioeconomic mechanisms underlying these urban-induced environmental changes are largely unclear (Pickett et al., 2011), we have to rely on an empirical parameterization approach to address the multiple controls on urban C dynamics in this study. As described in Sect. 3.3, based on an extensive literature review and academic reasoning, we derived the model parameters to approximate the average urban-induced environmental changes in the study region (Table 2). To evaluate the effects of parameterization uncertainties on the simulation results, we designed two additional scenarios to simulate urban C storage under extreme conditions: UBNZ_Cmin and UBNZ_Cmax (Table 1). The simulation results indicated that uncertainties related to parameterization of urban-induced environmental changes amounted to -2 to 3% of the urban-induced C dynamics (Fig. 3).

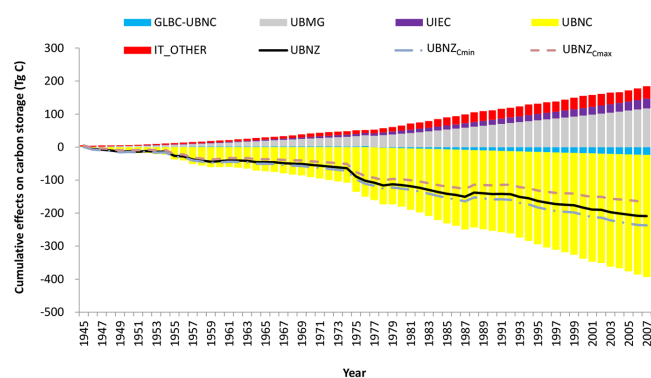


Figure 3. Cumulative effects of urbanization (UBNZ) to the carbon dynamic of the southern United States from 1945 to 2007 and the contributions of multiple environmental drivers. UBNC is the effect of urban land conversion; GLBC–UBNC is the interactive effect between global environmental changes (GLBC, including changes in climate, CO₂, N deposition, ozone, and landuse) and UBNC; UBMG is the management effect on the carbon dynamic of urban vegetation (such as lawn and urban forest); UIEC is the effects due to urbanization-induced environmental changes (such as urban heat island, CO₂ dome effect, and elevated N deposition in urban); the overall interactive effects among UBNC, GLBC–UBNC, UBMG, and UIEC are represented by IT_OTHER. Following Zhang et al. (2012), UBNZ_{Cmin} and UBNZ_{Cmax} represent the minimum and maximum urbanization effects, respectively, as influenced by uncertainties in model parameterization (Table 2).

4 Case study results

The temporal pattern of carbon dynamic during urbanization was controlled by the UBNC, which was estimated to result in about 0.37 Pg C loss from 1945 to 2007 (Fig. 3). In contrast, the UBMGs and UIEC enhanced C storage by about 0.12 and 0.03 Pg, respectively. Factorial analysis based on numeric experiments indicated that the interactive effects between global changes and urban land conversion has a negative effect on C storage, causing the study area to lose about 0.02 Pg C from 1945 to 2007. The complex interactive effects (i.e., IT_OTHER) among the four major types of environmental changes, urban land conversion, urban managements, UIECs, and GLBC–UBNC, resulted in a C sequestration of 0.04 Pg, comparable to the effects of UIEC and GLBC–UBNC.

The effects of UIEC, urban managements, and GLBC–UBNC can be further broken down to reflect the effect of individual factors (Fig. 1). From 1945 to 2007, urban LWN management enhanced C storage by 489.9 g m⁻² (the SUS subgroup in Table 3) or 63.6 Tg in the SUS (Fig. 1), having the strongest C sequestration effect among all factors. UFM, including direct management (Table 2) and indirect effects from altered disturbance regimes (e.g., protection from commercial logging and wildfire), also resulted in a large C sequestration of 396.3 g m⁻² or 51.5 Tg. Other factors that have significant positive effects on C sequestration

included the increased N deposition (248.9 g m⁻² and 32.3 Tg in the SUS) and CO₂ (220.5 g m⁻² and 28.6 Tg in the SUS) in urban areas. In comparison, UHI and interactive effects among UIEC factors caused 15.6 Tg (120.3 g m⁻²) and 16.0 Tg (123.2 g m⁻²) C loss from the SUS, respectively. The interactive effect between UBNC and global change factors were smaller than other controls. While its interactions with global O₃ and climate change may enhance C sequestration, interactions between UBNC and other global changes (pre-urban land use change, atmospheric CO₂ and N deposition change, and the interactive effects among the global change factors) caused C loss (Fig. 1).

Because the juxtaposition of land use and ecotypes strongly influences regional patterns of urban ecosystem functions (Nowak et al., 1996), we further analyze the impacts of urbanization on ecosystem C density based on the dominant/potential local vegetation type (i.e., UVG; Table 3). The results indicated that urbanization had a strong negative effect on C density (−2084 g m⁻²) in forest areas, only a slight negative effect on C density (−95 g m⁻²) in grasslands, and a positive effect on C density (390 g m⁻²) in shrubland/desert (Table 3). The C sequestration effects of UIECs and forest managements were strongest in forest areas, followed by grassland and shrubland/desert areas. The interactive effects between global change and urban land conversion had a negative effect (−276 g m⁻²) on C density in forest areas and a positive effect (168 g m⁻²) on C density in grassland areas. Because of the large forest areas in the SUS and because of the relatively strong responses of forest C dynamics to land conversion and urban induced changes, forest areas determined the pattern of regional C dynamics in response to urbanization from 1945 to 2007 (Fig. 1; Table 3).

5 Discussion

5.1 Relative importance of the controlling factors and the implications for urban management

Although many of the factors in our urban C analysis framework (Fig. 1) have been individually investigated in previous studies, their relative importance has rarely been compared. Our SUS case study showed that urban land conversion was by far the most important control on the regional C dynamics from 1945 to 2007, followed by urban managements, overall interactive effects among major factors (i.e., IT_OTHER), urbanization-induced environmental changes, and the GLBC–UBNC interactive effect, in descending order of importance (Fig. 1). Our findings provide valuable information for regional C management. First, we found that the C loss (−2845 g C m⁻²) caused by urban land conversion dominated the C sink that was induced by all other factors. Cannel (1999) estimated that Britain and Northern Ireland could lose as much as 8000 ± 4000 g C m⁻² due to urban land conversion. Their study, however, might overestimate the impact

Table 3. Contributions of multiple environmental controls to urbanization (UBNZ) effect on carbon (C) dynamic of the forest (including needleleaf, broadleaf, mixture, and wetland forests), grass (including C3 and C4 grasslands, and grassy wetland), and arid shrubs (including shrubland and desert) ecosystems in the southern United States (SUS) from 1945 to 2007. Unit: g C m^{-2} .

		Forested areas	Grassland areas	Shrubland areas	Southern United States
Urban land conversion (UBNC)		−2655	−1078	−303	−2845
Urbanization (UBNZ) effects	UBNC interact with climate change	43	33	21	39
	UBNC interact with global CO_2 change	−89	−25	−28	−73
	UBNC interact with N deposition change	−86	−19	−10	−69
	UBNC interact with global O_3 change	70	28	27	58
	UBNC interact with pre-urban land use change	−96	12	41	−68
	Interactive effects among the global change factors	−117	141	−68	−71
	Overall effect of GLBC–UBNC	−276	168	−18	−183
	Urbanization-induced environmental changes (UIECs)				
	Urban heat island effect	−136	−98	−33	−120
	Urban CO_2 dome effect	252	155	88	221
Effect from elevated N deposition in urban areas	270	245	92	249	
Interactive effects among the UIECs	−130	−131	−64	−123	
Overall effect of UIECs	256	171	82	226	
Urban managements (UBMG)					
Urban lawn management	455	639	694	490	
Urban forest management	525			396	
Overall effect of UBMG	980	639	694	886	
Overall interactive effects among UBNC, UIECs, UBMG, and GLBC–UBNC (IT_OTHER)	411	4	−6.7	307	
Overall effect of UBNZ	−2084	−95	390	−1609	

by assuming zero SOC in built-up areas. We estimated the SOC loss due to land conversion to be 944 g C m^{-2} in the SUS, close to the estimated SOC loss (820 g C m^{-2}) in central Germany (Schaldach and Alcamo, 2007). Because the effect of land conversion was more than twice the combination effects from all other factors (Table 3), it is preferable to preserve pre-urban C pools during land development, probably by reducing soil disturbances or reserving large areas of remnant green space, rather than relying on the carbon sink in urban ecosystems to compensate for the C loss during land conversion. Like us, Escobedo et al. (2010) also suggested that preserving large trees had a larger C benefit than planting young trees in the urbanized areas of the SUS. This is

especially important for the forested regions, which, when converted to urban lands, could release nine times more C than the shrubland (Table 3). Our analysis shows that about 77 % of urban and developed areas in the SUS were converted from forest, becoming a primary threat to the C sequestration in the forested area of the SUS (Wear, 2002). Currently, many cities of the United States have allocated a large amount of resources in tree planting, which was seen as an effective way to improve ecosystem service, such as C sequestration (Nowak, 2006; Young, 2010). We suggest that an equal amount of (if not more) resources should be allocated in preserving remnant forest in developed areas for effective C management.

Table 4. Comparison of model predictions against observed carbon pools and fluxes of urban ecosystems.

City	Long/lat (dd)	PFT ^a	Age	NPP (g C m ⁻² /yr) + SE ^b		VEGC (kg C m ⁻²) + SE ^b		SOC1m (kg C m ⁻²) + SE ^b		Sources of observations
				Observation	Prediction	Observation	Prediction	Observation	Prediction	
Atlanta, GA	-84.4/33.65	CF	NA			9.7 ± 0.7	9.3			Nowak and Crane (2002)
Baltimore, MD	-76.6/39.28	BF	NA			10.0 ± 1.3	11.2	12.1 ± 1.8	11.0	Nowak and Crane (2002) Pouyat et al. (2008)
Baltimore, MD	-76.6/39.28	Lawn	>50					12.2 ± 1.1	11.9	Pouyat et al. (2008)
Baltimore, MD	-76.6/39.28	Lawn	30					10.7	9.6	Pouyat et al. (2008)
Baltimore, MD	-76.6/39.28	Lawn	20					8.1	8.9	Pouyat et al. (2008)
Boston, MA	-71.03/42.37	BF	NA			9.1 ± 1.1	9.9			Nowak and Crane (2002)
Fort Collins, CO ^d	-105.1/40.6	Lawn	>60					13.1	13.5	Kaye et al. (2005)
Front Range, CO ^c	-105/40	Lawn	NA	762 ± 92	731	1.55 ± 0.16	1.59	11.6	12.0	Golubiewski (2006)
Miami-Dade, FL	-80.20/25.77	BF	NA			7.47	6.83			Escobedo et al. (2010)
Philadelphia, PA ^d	-75.17/39.95	BF	NA			9.0 ± 0.9	10.3			Nowak and Crane (2002)
Syracuse, NY ^c	-76.12/43.12	BF	NA			9.4 ± 1.0	9.7			Nowak and Crane (2002)
Washington DC	-76.5/30.88	Lawn	NA	737	715	1.49	1.54			Falk (1976)

^a Plant functional type (PFT); CF denotes coniferous forest; BF denotes deciduous forests.

^b NPP: net primary productivity; VEGC: vegetation carbon; SOC1m: soil organic carbon (0–1 m). Table 1 values (root NPP:NPP, root C:VEGC365, C:Biomass) were used to convert the aboveground NPP and biomass to total NPP and VEGC. By assuming that 53 % of SOC of grassland is located in the upper 30 cm (Jobbagy and Jackson, 2000), we calculated the SOC of lawns to 1 m for those studies only measuring top 30 cm.

^c Intensively managed sites. 23 samples. Two tailed Student's *t* test indicated differences between model predictions and observations were not significant (90 % level).

Intensively managed lawn: annual N fertilization rate = 11 g N m⁻² yr⁻¹; clipped litter left on site.

^d The species composition of the northern forest is not the same as that of the southern forest. Before conducting simulations in NY and PA, we estimated/calibrated the physiological parameters of the BF functional type against the intensively studied Harvard Forest LTER site. For other cities and the regional simulations in the SUS, we used the parameters, which were developed based on the studies in the Duke Forest.

Secondly, our study showed that the urban management effects have compensated for nearly one-third of the C loss due to land conversion from 1945 to 2007 in the SUS. Therefore, urban C sequestration can be enhanced through improved management practices (e.g., irrigation and fertilization) and targeted use of vehicles and machinery with least associated C emissions. According to Milesi et al. (2005), the urban lawns in the United States had a mean C sequestration rate of 37–104 g C m⁻² a⁻¹. Similarly, Townsend-Small and Czimczik (2010) found the C sink of managed lawns in Irvine, CA to be 140 g C m⁻² a⁻¹. Zirkle et al. (2011) estimated that lawn management increased the SOC of the United States home lawn by 78.5–79.5 g C m⁻² a⁻¹. These reports agreed well with the C sequestration effect (125 g C m⁻² a⁻¹) of lawn management in the SUS, as estimated by this study. In urban ecosystem management, it is important to quantify and reduce the hidden C cost from fossil fuel CO₂ emitted during maintenance (Townsend-Small and Czimczik, 2010). According to Zirkle et al. (2011), the fossil fuel C emission in maintenance currently cost 30–56 % of the C sink induced by N fertilization and irrigation in the United States home lawn (Zirkle et al., 2011). Previous studies also indicated that, if carbon-based maintenance is performed, urban forest will eventually become a C source (Nowak et al., 2002) or a weak sink (Escobedo et al., 2010). However, intensively managed trees only account for a small fraction of the urban forest in the United States (see Sect. 3.3.3 (2)). Therefore, hidden C cost from tree maintenance should be relatively small at regional scale. It is worth noting that there is substantial scope in reducing management-related CO₂ emissions, because different equipment and maintenance techniques may have distinct C emission rates (Reid et al., 2010). For exam-

ple, it was estimated that half of the lawnmowers used by the United States homeowners belong to riding mower (Quigley, 2001), which has far larger C emissions than walk-behind mowers (Zirkle et al., 2011). By improving the efficiency of riding mowers or choosing the walk-behind mower, the maintenance C emission of urban vegetation could be significantly reduced. Another possibility is to collect and utilize the 164 Tg dry biomass of lawn clippings and pruned tree twigs/limbs produced annually in the managed urban ecosystem in the United States for bioenergy production (Springer, 2012). Finally, well-managed urban vegetation can also indirectly reduce the C emission with its shading and cooling effects (Akbari et al., 1992).

Thirdly, we found that an altered disturbance regime might explain the observed C sink in urban forest. Urban forest was reported to have higher C density and growth rate than the average rural forest (Nowak and Crane, 2002; Golubiewski, 2006). This phenomenon has been attributed to urbanization-induced environmental changes in previous studies (Nowak, 1994). We propose another possible mechanism: the altered disturbance regime after urbanization may enhance C sink in urban forest. Most city trees are protected from frequent tillage, wildfire, and commercial logging, leading to suppressed soil disturbances and increased tree age (lower stand turnover and higher biomass) (Hutyra et al., 2011). In comparison, the influence of more intensive management of plantations and natural forests resulted in a mosaic of different age classes and an averaged low biomass density (Birdsey, 1992). The risk of C emissions through catastrophic wildfire burning is also considerably reduced by a shifted fire regime (Guilden et al., 1990; Pickett et al., 2011) or fire control/management in urban areas (Campbell et al.,

2007; Defosse et al., 2011). Based on a literature review (see Sects. 3.2 and 3.3.3), we estimated that the overall mortality rate (considering both the background mortality and the disturbance effect) of rural forest is about 10 % higher than that of urban forest (assuming zero disturbance from fire or commercial logging in urban areas) in SUS. Our simulation showed that the effect of management and altered disturbance regimes together resulted in a C sink of 51.5 Tg in the SUS urban forest, stronger than the effect from urbanization-induced environmental changes. Because of direct management, such as pruning negatively affected C storage in our model simulation, the C sink can be attributed to the altered disturbance regime; a potentially important mechanism that should be further investigated in future study.

Fourthly, our study, as well as others (Ziska et al., 2004; Shen et al., 2008; Trusilova and Churkina, 2008), indicated that the UIECs had complex impacts on C dynamics, with an overall effect that promoted NPP and C sequestration. We found that urban heat island could induce C emissions, while the elevated urban nitrogen deposition and CO₂ concentration could enhance C sequestration in the SUS (Fig. 1). Our findings agreed well with the reported pattern in a Europe urbanization study (Trusilova and Churkina, 2008). Both studies found that the urban nitrogen deposition effect was stronger than the urban CO₂ effect, and that the urban CO₂ effect was much (100 to 200 %) stronger than the UHI effect. Because these UIEC factors generally have a “dome” pattern that peaks at the city center and gradually levels off along urban–rural gradient (Idso et al., 1998), it is advisable to arrange green spaces close to the city center to maximize their C sequestration capacity. The distinct responses from different vegetation types to urbanization should also be taken into consideration. Shen et al. (2008) suggested grass to be more sensitive to the urban CO₂ dome effect than desert shrub. We found that the carbon sink effect of UEIC decreased, while the carbon sink effect of lawn managements increased, in the sequence of forest, grass, and shrub areas (Table 3). Therefore, in the forested areas, it is recommendable to improve landscape design (such as arranging green spaces close to the city center) to maximize the UIEC effect, while, in the arid shrubland areas, the focus should be put on improving urban managements to enhance C sequestration.

5.2 Complex interactive effects among factors

One of the major uncertainties in urban C dynamic is from the interactive effects among environmental factors, which could have strong ecological impacts, sometimes even determining the direction of the overall ecosystem C balance (Shen et al., 2008; Tian et al., 2011b). However, many former urban studies overlooked the interactive effects and assumed the effects of multiple factors to be additive (e.g., Zirkle et al., 2011). Our case study found that the overall interactive effects of the major control factors could increase C sequestration in the SUS by about 39.9 Tg, larger than the effect of

urbanization-induced environmental changes (29 Tg, Fig. 1). This C sink is mainly located in the forested areas, which, on average, gained 411 gC m⁻² due to the overall interactive effects of urbanization from 1945 to 2007 (Table 3). Compared to the pre-urban forests, urban trees in general had higher biomass and productivity, because they were protected (by human managements) from disturbances (such as commercial logging) that caused the high overall mortality rates (considering both the background mortality and the disturbance effect) and low biomass of the rural forest in SUS (Birdsey, 1992). The strong C sink due to interactions among urban land conversion, managements, and UIECs indicated that these larger urban trees are more responsive to urbanization-induced environmental changes and can fix more C, a phenomenon confirmed by recent observations from Escobedo et al. (2010) and Stephenson et al. (2014). The underlying mechanism is related to the relatively large total leaf area of big trees. According to the Pipe model (Shinozaki et al., 1964) that controls the photosynthate allocation in woody plants, total tree leaf mass increases as the square of trunk diameter. A typical tree that experiences a 10-fold increase in diameter will therefore undergo roughly a 100-fold increase in total leaf mass. Larger leaf mass means the tree has higher growth potential if not limited by water and nutrient availability. Therefore, bigger trees are more sensitive to elevated CO₂ and N deposition in urban areas. In rural forest stand, the high C sequestration rate of large, old trees could be offset by the intensified mortality related to light competition. The urban forest, however, has a relatively open canopy, and is able to support large trees (see Sect. 3.3.3). Therefore, when a rural forest became a remnant forest in urban areas, its trees could grow bigger, faster, and were more sensitive to the increased urban CO₂ and N deposition because of the urban management effect that suppressed disturbances (commercial logging) and light competition (Table 3).

We also found a strong interactive effect (−16 Tg) among the UIEC factors (UHI, CO₂ dome, and elevated N deposition), comparable to the negative effects of UHI (−15.6 Tg) (Fig. 1). Unlike Trusilova and Churkina (2008), who found that the UIEC interactive effect increased C sequestration in Europe, we found that it suppresses the urban C sink in the SUS (Fig. 1). This is mainly because the two regions experienced different urbanization-induced climate changes. In our simulation, urbanization will increase local surface temperature in the SUS, but the data of Trusilova and Churkina (2008) indicated significant reduction of temperature by 0.73–1.26 °C, followed by the urbanization in Europe. We found that the UHI effect increased potential evapotranspiration and exacerbated the water stress in the warm temperate ecosystems of the urban areas in the SUS. Like Shen et al. (2008), our simulation indicated that increasing water stress suppressed elevated CO₂ and N deposition effects on the ecosystem C sequestration. The data of Trusilova and Churkina (2008), in contrast, indicated reduced temperature and increased precipitation in urbanized areas in Europe;

both climate changes improved water availability and magnified elevated CO₂ and N deposition effects.

Shen et al. (2008) suggested that the effect of urbanization-induced changes are difficult to predict due to the influence of other factors, such as global climate change. Guided by the factorial analysis scheme developed in this study, for the first time, we found a way to separate the global change effects (i.e., GLBC) from the urban land conversion (i.e., UBNC) and quantify their interactions (i.e., GLBC–UBNC) (Fig. 1). We found that GLBC–UBNC had negative effects on regional C storage (–24 Tg), almost offsetting the C sink induced by UEIC (29 Tg, Fig. 1). Such an important mechanism, however, had been overlooked in previous studies. The interaction between UBNC and different global change factors had different effects on C dynamics. In general, GLBC–UBNC would have a negative impact on C storage if the global change factor enhanced the ecosystem C sequestration. This is because the lands converted to impervious surface are no more responsive to global change. For example, elevated CO₂ and N deposition in atmosphere stimulate C sequestration. After a pre-urban ecosystem is converted to an impervious surface, the related C sinks (in response to CO₂ fertilization) disappear. Therefore, the interactive effects between urban land conversion and changes in global CO₂ and N deposition seem to have a negative effect on C sequestration (Fig. 1).

6 Conclusions

Urbanization involves complex changes in land structure and multiple environment factors, whose effects should not be treated independently. As urban land cover and human population continue increasing rapidly across the globe, it is important to investigate the individual effects of and complex interactions among multiple factors on the ecosystem structure and processes in urbanized lands. Our case study revealed how the C dynamics in the 1.2×10^5 km² urbanized areas of the SUS were influenced by multiple environmental factors during the period 1945–2007. And the numeric experimental design and the factorial analysis schemes proposed in this study could be applied in other regions. Such efforts as the one reported here not only improve our understanding of the complex effects of urbanization on regional C dynamics, but also provide a quantitative approach for assessing the effectiveness of landscape design in urbanized areas and urban development strategies.

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