Effects of Disturbance and Land Management on Water, Carbon, and Nitrogen Dynamics in the Terrestrial Ecosystems of the Southern United States

by

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Abstract

Although the climate in the Southern United States (SUS) is warm and wet, with mild winters and high humidity, the terrestrial ecosystems in this region have been greatly disturbed. These disturbances include hurricanes, storms, fires, insect and diseases, floods, extreme droughts and land use and land cover change. Meanwhile, to alleviate these effects, land management practices such as irrigation, fertilization, tillage, forest harvest and thinning have been increasing applied to terrestrial ecosystems. It is still unclear how land disturbance and management have changed the terrestrial ecosystem structure and function. In this dissertation, a systematic approach of integrating field observational data, regional inventory data, remote sensing, and a process-based global biogeochemical model was used to evaluate the impacts of disturbance and management on terrestrial ecosystem carbon, water and nitrogen fluxes in the SUS. Results indicated that although the intensity and duration of drought disturbance in the SUS was not significantly increased, drought events over short periods (a year to a few years) could significantly reduce net primary production (NPP) and C storage (up to 40%). Climate change in the SUS has resulted in a net release of 0.33 Pg C (1 Pg = 10^{15} g) into the atmosphere, while changes in precipitation and temperature patterns induced C emissions of 0.035 and 0.14 Pg, respectively during 1895-2007. The interactions between precipitation and air temperature induced a C emission of ~0.15 Pg, suggesting that changes in air temperature could significantly enhance drought impacts in the SUS. In total, C emission from drought impacts induced both by precipitation and temperature could be ~0.19 Pg. The western SUS (dry region) was found to act as a C sink, while the east (water-rich region) acted as a C source due to changing precipitation patterns. C sources in the east were significantly enhanced by the interactions between

precipitation and temperature changes. With changing climate, land use, and land management, both evapotranspiration (ET) and water yield were increased during 1895-2007, implying that available water in the terrestrial ecosystem of the SUS is decreased. N fertilization has greatly increased carbon storage by ~296 Tg (4.70 Tg yr^{-1}) in the SUS cropland during 1945-2007, while N_2O emissions were also significantly enhanced by 2.97 Tg N (0.047 Tg N yr^{-1}). The ratio of N_2O emission to fertilized N uses was 2.5% ± 0.2%, indicating that about 2.5% of fertilized N was emitted as N_2O. Combining the global warming potential (GWP) of these two gases, N fertilization was a net source that could enhance the GWP by 304.6 Tg CO_2 equivalents during this period. The GWP induced by N fertilization increased after mid 1970s and N fertilization showed a saturation effect for increasing C storage, suggesting that further increases in N fertilizer use would not significantly stimulate C sequestration. To decrease GWP and maintain high crop productivity in the future, crop N use efficiency needs to be increased rather than increasing N fertilizer amounts. Forest disturbance in Mississippi and Alabama has resulted in a 1.3% annual mortality of forest trees during 1984-2007, resulting in a net C source of 0.23 Pg C. Most of this C source is due to the loss of the vegetation C pool since forest biomass accumulation requires a longer recovery time. Although small disturbance events may not significantly change forest structure, the legacy effects of forest disturbance on C storage could last over 100 years. To improve estimation accuracy of US C budget, impacts from small but continuous disturbance events should be taken into account. Combining the impacts of disturbances (Drought, land use change, and forest mortality) and management (N fertilization, site preparation, and forest plantation management) on C, N and water dynamics, this research suggested that disturbances could reduce C storage, NPP and available water resources and increase N_2O emission in the SUS, while land management could increase C storage, NPP and

N_2O emission. Further research is needed to systematically explore the impacts of other major disturbance and management types on C, N and water dynamics in the SUS. The findings from this study could help policy makers and land managers to understand the potential consequences of various disturbance events and management practices, and thus taking precautions against these consequences through making appropriate policies.

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Chapter 1

Introduction

RESEARCH JUSTIFICATIONS

The government and scientists have paid much attention to land disturbance and management events in the Southern United States (SUS) during the past century and a few studies have explored how land management and disturbance influence ecological, economic and social benefits (e.g., Delcourt and Harris 1980; Turner et al. 1995; Birdsey and Lewis 2003; Woodbury et al. 2007); However, the mechanisms controlling of terrestrial ecosystem responses to changes in multiple environmental factors in the SUS remains unclear. Most of the previous work focused on the impacts of changes in one or a few environmental factors or ecological consequences. Many factors such as lack of computation capacity and sparse observational and experimental data limited the researchers' ability to explore these relationships at the landscape or regional levels. Due to increases in long-term observational and experimental data and great advances in technology such as remote sensing, computation capability, and laboratory instruments, it is possible to conduct new studies to explore comprehensive impacts of global change, land disturbance and management on multiple ecological processes (e.g., water, carbon and nitrogen cycles) in the terrestrial ecosystems in the SUS.

In the SUS, cropland has slightly decreased during the last century and accounts for about 20% of the current land area, while urban areas have been increasing and growing rapidly since 1970s. Forest area in the SUS has declined from ~66% in 1630 to ~40% at present (Smith et al. 2009). Historical (1700-present) land use change in the SUS could greatly change the C, water, and N cycles, thus increasing greenhouse gas emissions. Although several studies (e.g., Delcourt and Harris 1980; Houghton et al. 1999; Birdsey and Lewis 2003; Chen et al. 2006) have addressed the impacts of land use change on C cycles, these studies did not cover the entire land use history of the SUS, nor did they address impacts on other greenhouse gas emissions (i.e., N_2O and CH_4). Although the SUS has a warm and wet climate, significant natural and human caused disturbances have been occurred over time (Houghton et al. 1999; Woodbury et al. 2007; Karl et al. 2009; Seager et al. 2009; Pan et al. 2010; Hansen 2010). These disturbances (including harvesting, land use change, drought, hurricanes and storms, insect and diseases, prescribed fires, and wild fires) may have greatly altered the terrestrial ecosystem structure and function in the SUS. Few studies have addressed the comprehensive impacts from these disturbance events. The forest product industry has a major impact on economy in some of the SUS states. Historical forest harvest and management activities have resulted in very young forest age structure (most forests are less than 60 years old) in current forests (Smith et al. 2009; Pan et al. 2010). Although forest biomass can recover from forest harvest and management practices, the balance or stability of major ecological processes such as water, C and N cycles in these ecosystems could be greatly disturbed. In addition, genetically-improved, high-productivity tree species were planted in the SUS following the 1930s. In 1996, planted forests comprised 14.6 million ha (36 million acres), or ~17% of all forest land in the SUS (Smith et al. 2001). By 2006, planted forests comprised approximately 17.4 million ha, or ~20% of all forest land in the SUS (Smith et al. 2009). By 2040, the overall area of productive planted pine forests in the SUS is projected to increase to ~21.9 million ha (Wear and Greis 2002), with Georgia, Florida, and Alabama having the most acreage. Most of these forest plantations in the SUS are privately owned and are managed to improve forest productivity (Fox et al. 2007).

OBJECTIVES

In this study, we conducted a comprehensive analysis to determine the impacts of land management (forest harvest, afforestation/reforestation and nitrogen fertilization), and disturbance (land use change, and drought) on C, water, and N cycles (net primary productivity (NPP), biomass, C storage, evapotranspiration (ET), water yield, CO_2, and N_2O fluxes) in the SUS. The SUS is defined to include 13 states (Alabama, Arkansas, Florida, Georgia, Kentucky, Louisiana, Mississippi, North Carolina, Oklahoma, South Carolina, Tennessee, Texas, and Virginia) (Wear and Greis 2002; Smith et al. 2009). The time periods evaluated were as long as 308 years (1700-2007) and the periods varied according to different research objectives. Based on multiple sources of experimental and observational data and a sophisticated Dynamic Land Ecosystem Model (DLEM), the study objectives were to:

(1) Develop a comprehensive and effective modeling approach to simulate impacts of large-scale land management and disturbance on C, water and N cycles;
(2) Assess the impacts of historical drought events during 1895-2007 on NPP and C sequestration in the SUS;
(3) Assess changes in water cycles (ET and water yield) due to land use change, land management, and climate change in the terrestrial ecosystems of SUS;
(4) Estimate N fertilization impacts on C storage, N_2O emission, and global warming potential from croplands during 1945-2007;
(5) Assess impacts of forest disturbance on C storage in Alabama and Mississippi states; and
(6) Evaluate the impacts of forest plantation and management on C storage and N_2O emission.

HYPOTHESIS

Two hypotheses were put forward in this study according to the study questions and objectives: 1) impacts of land management and disturbance could result in significant changes in regional NPP, C sequestration, N_2O emission, and available water resource in the SUS; and 2) these changes could be modeled by process-based models within a reasonable confidence interval.

APPROACHES

Based on a widely applied Dynamic Land Ecosystem Model (DLEM), this dissertation developed a land disturbance and management modules which were integrated into the DLEM. Based on a series of newly developed or organized long-term spatial data, the impacts of land management and disturbance on C, water and N cycles were simulated with this improved DLEM. Combined with field observational and experimental data, regional inventory data, and remotely sensed data, this dissertation comprehensively assessed the impacts of multiple disturbance and management types on multiple ecological processes.

DISSERTATION STRUCTURE

Based on the objectives and approaches, this dissertation is organized as 7 chapters:

Chapter 1 justifies the research activities and described the objectives, and approaches used in this dissertation;

Chapter 2 provides a comprehensive literature review of the past and current progresses in assessing impacts of different land management and disturbance types on multiple ecological processes. This review points out the shortcomings of previous efforts, the need for this dissertation research, and future research directions;

Chapter 3 evaluates drought conditions during 1895-2007 in the SUS based on a widely used standard precipitation index (SPI), and then explores the impacts of drought on net primary productivity and C sequestration in the SUS. Results indicate that although drought intensity has not found to increased, extreme drought events have significantly reduced net primary productivity and carbon sequestration during this period.

Chapter 4 presents research regarding changes in ET and water yield in the SUS as influenced by land use change, land management, and climate change during 1895-2007. Results indicated that ET and water yield increased slightly during this period. Compared to change in land use and land management, climate variability had the largest impact and contributed the most to increases in both ET and water yield.

Chapter 5 comprehensively evaluates the effects of N fertilization of cropland on CO_2, and N_2O fluxes and greenhouse gas warming potential during 1945-2007. Results indicated that long-term N fertilizer application could both increase soil C storage and N_2O emissions in the SUS cropland. The global warming potential of combined CO_2 and N_2O fluxes was enhanced, indicating N fertilization was a C source during 1945-2007.

Chapter 6 assesses the effects of forest disturbance on C storage in Mississippi and Alabama. The forest mortality rate after disturbance (might be induced by multiple disturbance types) in MS and AL during 1984-2007 was quantified with Landsat TM/ETM+ images. Results indicated that forest disturbance significantly decreases C storage, especially for C storage in vegetation. The uncertainties underlying this study are also presented.

Chapter 7 evaluates the impacts of increased forest plantation areas and different management practices on changes in C storage and N_2O emissions.

Chapter 8 summarizes the research results and conclusions in this study, assesses the uncertainties, and proposes areas for future improvements.

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Chapter 2

Literature Review

The Southern United States (SUS) is composed of different plant functional types with forests accounting for about 50% of the land area. This region exhibits a vast spatial difference in climate characterized by an increasing precipitation gradient from west to east and a decreasing temperature gradient from south to north (Karl et al. 2009). Under the pressure of increasing population, industrialization, urban sprawl, increasing demands for food and resources, and global climate change (Chappelka and Samuelson 1998; Schimel et al. 2000; Felzer et al. 2004; Norby et al. 2005; Holland et al. 2005; Dentener et al. 2006; Chen et al. 2006a; Woodbury et al. 2007; Birdsey et al. 2006; IPCC 2007), the SUS has experienced significant changes in climate, atmospheric composition, and terrestrial ecosystem structure and function during 1700s-2000s. Changes in ecosystem function include water, carbon (C) and nitrogen (N) cycling, while changes in ecosystem structure include land use types, canopy structure, and plant community composition. Although many studies have focused on the potential impacts of climate change, land management, and disturbance on ecosystem water, C and N cycles in the past (Chappelka and Samuelson 1998; Schimel et al. 2000; Felzer et al. 2004; Norby et al. 2005; Holland et al. 2005; Dentener et al. 2006; Chen et al. 2006a; Birdsey et al. 2006; Woodbury et al. 2007), no comprehensive studies have included all of the above-mentioned factors at a relatively high spatial resolution over a long-term time period. In this research, the impacts of drought, land management (forest harvest, N fertilization, and site preparation), and disturbance (land use change and forest mortality) on ecosystem C, water and N pools and fluxes using a systematic approach of integrating field observational data, regional inventory data, remote sensing data and simulation results from an widely validated ecosystem model.

CLIMATE CHANGE, DROUGHT AND THEIR IMPACTS IN THE SUS

Drought actually means water deficiency at the land surface, while water capacity at land surface is mainly controlled by the balance between precipitation from atmosphere and evapotranspiration (ET) to the atmosphere. At the soil surface, precipitation increases soil water capacity, while ET has the opposite effect, and their combination decides the dry or wet status of soils. Therefore, it is important to consider the joint influence of precipitation and ET as they are related to drought issues. For example, in some areas, ET might increase more than that of precipitation, so the budget of land surface water is still negative and these areas may be still very dry even after increased precipitation. Only the amount of precipitation cannot represent the wetness/dryness, increases in air temperature due to global warming could also induce drought events (Houghton 2004). Global warming is predicted to cause massive drought events that will threaten the lives of millions and take over half the land surface on our planet in the next 100 years as predicted by the Hadley Centre for Climate Prediction and Research (IPCC 2007). Changes in temperature have raised concerns about impacts of changed drought frequency and intensity on terrestrial ecosystems throughout the USA (Houghton 2004; Sun et al. 2008; Karl et al. 2009; Seager et al. 2009). In addition, changes in climate (especially warmer and wetter winters and drier summers) may influence the hydrological cycle (Houghton 2004). As described by Houghton (2004), higher temperatures will cause precipitation events to become more extreme and less frequent. Furthermore, the higher temperatures will lead to increases in ET over land surface, thus reducing of the land surface moisture and adding to the drought conditions. Under most climate change scenarios, more frequent and/or intense drought is expected in the

Southern United States (SUS). In this region, the potential ET is predicted to increase and already exceeds summer precipitation (J. Sadler, USDA Cropping Systems and Water Quality Research Leader, Personal Communication).

The climate in the SUS is warm and wet, with mild winters and high humidity, compared to the remaining continental US (Karl et al. 2009); however, due to increased water use demand and climate variability, this water-rich region has experienced serious water stress conditions (McNulty et al. 2007). Precipitation in the SUS display a great spatial variability with an increasing gradient from the west to east and a great inter-annual variability alternating extreme high and low precipitation patterns. Although annual precipitation in the SUS has slightly increased during 1901-2008 (Table 1), 7.7% decrease in annual precipitation during 1970-2008 implies an intensified drought period. Furthermore, the largest decrease (29.2%) in precipitation in spring during 1970-2008 could mean a disastrous consequence for vegetation development and growth in the SUS. The US Climate Change report (Karl et al. 2009) noted that the average annual temperature of the SUS as a whole did not change significantly over the past century. Since 1970, however, the annual average temperature has increased ~0.9°C, with the greatest seasonal increase during the winter months (Table 1). The warmer winter in the SUS could indirectly cause higher ET and possibly causes drought events. In the near future (present to 2100), many climate change scenarios suggest that more frequent and/or intense drought episodes are expected across the SUS, and potential ET is predicted to exceed summer precipitation (Smith and Tirpak 1990; IPCC 2007; Karl et al. 2009; Seagers et al. 2009). Therefore, it is important to understand the effects of drought on forest health and productivity.

Table 1 Climate change during 1901-2008 in the SUS (Source: Karl et al. 2009)

	Temperature change in ^{o}C		Precipitation change in %	
	1901-2008	1970-2008	1901-2008	1970-2008
Spring	0.2	0.7	1.7	-29.2
Summer	0.2	0.9	-4.0	3.6
Autumn	0.1	0.6	27.4	0.1
Winter	0.1	1.5	1.2	-9.6
Annual	0.2	0.9	6.0	-7.7

While much attention has been paid to drought issues in the arid and semi-arid regions of the US such as the Great Plains, the interior West, and the Southwest (Seager et al. 2009), much less paid to drought impacts in the traditional high-precipitation regions such as the eastern US and SUS. Even though the SUS does not experience long-term extreme drought events, short periods of a year to a few years do occur when precipitation reductions stress water supplies, and thus impacts of drought events have often been reported and recorded. An obvious example is the recent drought event which started in winter of 2005 and extended to 2007. In addition, in recent history, the SUS has also experienced several severe droughts (e.g., 1954-1957, 1986-1989, 1998-2001, and 2005-2007) as indicated by the Palmer Drought Severity Index (PDSI) (Palmer 1965; http://www.drought.noaa.gov/index.html). Except for drought events induced by precipitation reductions and global warming, drought impacts may be intensified by increased water use demands (McNulty et al. 2007). The fast-growing population and urbanization caused by population migration and economic growth has increased water demands. For example, almost a quarter of total water use in Georgia is for public use (http://ga.water.usgs.gov/projects/projectwateruse.html). Thus, an increasing population, economic expansion and a decrease in precipitation in Georgia imposed enormous stresses on the available water resources (Sun et al. 2008). Due to this shrinking water resource, Georgia's Governor declared a drought emergency in 2007 (http://www.georgia.gov).

Carbon and water fluxes between the atmosphere and terrestrial ecosystems are interactively linked at various temporal and spatial scales (Farquhar et al. 1980; Running et al. 1988; Rodrigues-Iturbe 2000; Jackson et al. 2005). Drought directly influences photosynthesis of individual plants and thus the C cycle in an ecosystem (Reichstein et al. 2002; Rambal et al. 2003; Ciais et al. 2005; Barr et al. 2006; Grainer et al 2007; Ju et al. 2010; Tian et al. 1998, 2000, 2003, 2010a). Under most severe drought situations, plants will die due to the long-term low photosynthesis rates (Seagers et al. 2009; Klos et al. 2009). Drought could also influence canopy stomata conductance (through changing vapor pressure deficits at the canopy surface and canopy water potential), thus indirectly influence the gross primary productivity (GPP) (Farquhar et al. 1980; Grant et al. 2006a, b). In theory, canopy conductance (g_n) and photosynthetic rates are estimated as (Farquhar et al. 1980; Collatz et al. 1991; Sellers et al. 1992; Bonan 1996; Chen et al. 1999; Oleson et al. 2004):

$$g_n = \max(g_{\max} f(ppdf) f(T) f(VPD) f(W) f(CO2), g_{\min})$$

$$GPP = f(T, gn, CO2, ppf, K, N)$$

Where, g_{min} and g_{max} is the minimum and maximum canopy conductance, respectively, which are constant for specific plant functional types; *f(ppdf)* is the impact function of photosynthetic photo flux density (radiation); *f(VPD)* is the impact function of vapor pressure deficit at the canopy surface; *f(W)* is the impact function of soil moisture; $f(CO_2)$ is the impact function of atmospheric CO_2 concentration. *GPP* is a function of air temperature (*T*), enzyme (including Rubisco, light, and carbohydrate export-limited enzymes) kinetics (*K*), photosynthetic photo flux (*ppf*), stomata conductance (g_n), canopy surface CO_2 concentration (CO_2), and leaf nitrogen level (*N*). The function of soil moisture is estimated as:

$$f(W) = \sum_{i=1}^{n}(w_i r_i)$$

$$w_i = \begin{cases} 0 & vwc_i \le wp_i \\ \dfrac{vwc_i - wp_i}{fc_i - wp_i} & wp_i < vwc_i < fc_i \\ 1 & vwc_i \ge fc_i \end{cases}$$

Where vwc_i, wp_i, and fc_i are the soil volumetric water content, wilting point, and field capacity at soil layer *i*, respectively, and r_i is the root fraction in soil layer *i*.

In addition, drought will greatly decrease litter decomposition rate resulting in N limitation for plant growth through decreasing soil N mineralization:

$$Decom_i = k_i f(T) f(W) f(N)$$

where $Decom_i$ is the decomposition rate for litter pool *i*; k_i is decomposition rate of litter pool *i* at 25 °C air temperature and at optimum soil moisture; *f(W)* is the function of soil moisture; *f(N)* is the function of soil available N; *f(T)* is the function of temperature (for aboveground litter pools, it is the daily average air temperature; for belowground litter pools, it is the daily average soil temperature).

Drought impacts on plant productivity, plant mortality and ecosystem carbon sequestration have been extensively reported in the SUS. Based on PDSI data from NOAA and 1991-2005 Forest Heath and Monitoring (FHM) plot data from Alabama, Georgia, and Virginia, Klogs et al. (2009) examined drought impacts for three tree species groups (pines, oaks, and mesophytic species). They found that the pines and mesophytic species showed significant reductions in growth rate with increasing drought severity. However, no significant difference in growth rate was observed for oaks. Mean mortality rates within the no-drought class were significantly lower than those within the other drought classes. Mean mortality rates were not significantly different among drought classes for oaks. Their study also implied that older and denser stands are more susceptible to drought damage and stands with more species suffer less mortality. During drought periods in the 1950s and 1980s, many studies found that drought significantly affected tree mortality and growth (Buell et al. 1961; Small 1961; Elliott and Swank 1994; Olano and Palmer 2003).

LAND USE AND LAND COVER CHANGE AND ITS EFFECTS IN THE SUS

The SUS covers about 215.6 million ha of land in 13 states from the Mid-Atlantic coast west to Texas and Oklahoma. This represents ~24% of the land area, 60% of forest land, and 25% of agricultural land in the entire United States. Land use types in the SUS have changed dramatically over time. Four obvious eras from 1700s to present have occurred (Wear 2002; Wear and Greis 2002; Hansen et al. 2010). They were characterized as: agricultural expansion era (1700s-1880), industrial logging era (1880-1920), semi-regeneration era (1920-1970) and suburban encroachment era (1970 to present). During the agricultural expansion era, agricultural land expansion and forest area shrinkage is the major characteristic of land use change. Settlers in the SUS converted large areas of forest to agricultural fields and grazing lands. Between 1700 and 1880, an estimated 26.3 million ha of southern forest was cleared, primarily for agriculture. During the industrial logging era, large-scale industrial logging activity was accelerated due to the rapid expansion of railway networks. Much of the remaining primary or virgin forests of the SUS were cut, resulting in forests primarily composed of young-age trees. By the end of this era, the southern forest area declined to ~86.2 million ha by 1920 and southern forests reached their lowest acreage (Williams 1989). During the semi-regeneration era, due to the widespread use of electricity and transportation fuels, forests were not used as a major fuel sources in the SUS. In addition, the US government began to protect forest resources by strengthening forest management and encouraging reforestation. Forests began to recover in abandoned croplands, pasture areas, and regions that had been logged. The land use change was thus primarily characterized as forest regrowth and a slight increase in forest area. During the suburban encroachment era (1970s to present), suburban encroachment surpassed agriculture as the leading cause of land use change in the SUS. Urbanization became the most important land use change type during this period.

Land use change such as cropland establishment and cultivation, cropland abandonment, and subsequent forest regrowth is the primary mechanisms for transferring C between land and atmosphere (Houghton et al. 1999; Caspersen et al. 2000; Pacala et al. 2001; Tian et al. 2003; Birdsey and Lewis 2003). Based on the Forest Inventory and Analysis (FIA) biomass data from the late 1970s to early 1990s, Pacala et al. (2001) estimated a C sink between 0.30 and 0.58 Pg C yr^{-1} in the continental United States and about half of this sink was due to forest regrowth on abandoned cropland. Using FIA data of five eastern states, Caspersen et al. (2000) showed that land-use change is the dominant factor for C accumulation in the eastern US. Houghton et al. (1999) estimated an annual accumulation of 0.037 Pg C in the US because of land use change in the 1980s, and forest regrowth has taken up 0.28 Pg C each year during this same period based on a bookkeeping model. By using the terrestrial ecosystem model (TEM), Chen et al. (2006) estimated that land use change had resulted in a total of 9.4 Pg C or 0.065 Pg C yr^{-1} (1 Pg = 10^{15} g), including 6.5 Pg C in vegetation and 2.9 Pg C in soil, released to the atmosphere in the SUS during 1860–2003. The net C flux due to cropland expansion and forest regrowth on abandoned cropland was approximately zero in the entire SUS between 1980 and 2003. Tian et al. (2010b) found that land use change resulted in a net C source of 1.26 Pg C or 0.01 Pg C yr^{-1} during 1895-2007, with a small net C sink of 0.08 Pg C or 0.008 Pg C/yr during 1980s, which is greatly different from Houghton et al. (1999) estimates (0.037 Pg C yr^{-1}). The difference among these modeling efforts suggests that more robust and persuasive studies are required in the future.

There have been many studies on effects of land use change on C dynamics based on the use of FIA data and other land use inventory data sets (Houghton et al. 1983, 1999; Turner et al. 1995; Brown and Schroeder 1999; Caspersen et al. 2000; Houghton and Hackler 2000, 2000b; Pacala et al. 2001; Birdsey and Lewis 2003). The lack of continuous monitoring of land use dynamics and resultant legacy effects indicates that the statistical methods are not accurate enough to reflect the ecosystem-level C and water dynamics. Considering the large differences in modeling and statistical methods, it remains unclear how the spatial and temporal variability of C storage and fluxes resulting from land use change over a long time period. Spatial and temporal variability of C dynamics in terrestrial ecosystems could possibly be monitored by combining modeling and statistical methods.

FOREST HARVEST, REFORESTATION AND THEIR IMPACTS ON ECOSYSTEM C, N AND WATER CYCLES

Modern forests continue to be dramatically altered by two major anthropogenic disturbances: timber harvesting (Kittredge et al. 2003) and permanent conversion due to land-use change (Riitters et al. 2002). Since harvesting is common across the SUS (Birdsey and Lewis 2003; Woodbury et al. 2006), it is important to understand the implications of both land conversion and harvesting patterns for many ecological processes such as C, water and N cycles. Along with the changes of land use, forest harvest has been commonly used in the SUS. From the early 19th to early 20th century, much of the forest was cleared for agriculture and these areas were frequently burned (Riitters et al. 2002; Wear and Greis 2002). Afterwards, large forest area was cut for wood products and converted to short-rotation pine forests. Also, a portion of natural or naturally-regenerated forests were converted to pine plantations since the middle of the 20th century. Until 2007, over 22% of forests were pine plantations in the SUS, over 26% of this being located in the coastal region (Smith et al. 2009). These plantations are routinely harvested and replanted, which results in relatively evenly-distributed age groups that are less than 60 years old (Smith et al. 2000, 2004, 2009; Pan et al. 2010; Hansen et al. 2010). Natural forests are also harvested frequently and clear-cutting is used in most situations. Therefore, most forests in the SUS are dominated by young- and even-aged trees and have shorter rotation ages of approximately 80–100 yr, with few as old as 180-200 years (Pan et al. 2010; Smith et al. 2009; Figure 1).

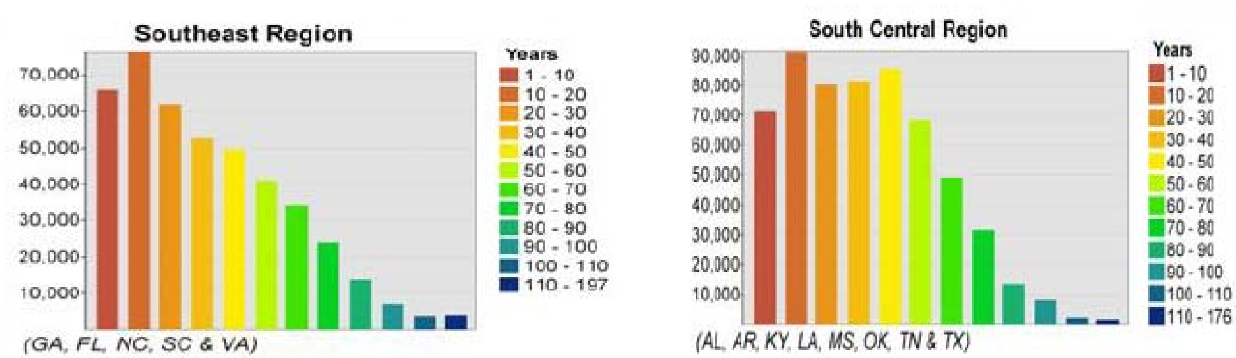

Figure 1 Current forest age structure in the SUS (South Central + Southeast Region, source: Pan et al. 2010). Note: unit for y-axis is thousand ha.

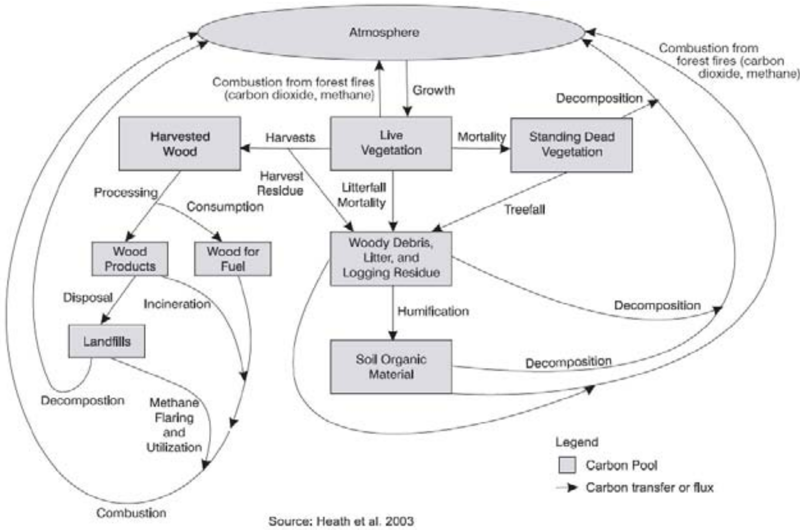

Figure 2 Schematic diagram of forest harvest and its impacts on C cycle (Source: Heath et al. 2003).

Forest harvest and subsequent reforestation could have a strong legacy effect on the forest ecosystem C and N cycles. The impacts of forest harvest on C pools are described in detail in Figures 1 and 2. Forests are the largest terrestrial pool for atmospheric C, which remove CO_2 from the atmosphere and store it in the soil organic matter and standing biomass (Alexandrov 2007). The current C stock in tree biomass comprises half of the atmospheric storage and is reported to continue increasing despite deforestation which decreases C storage in forest (Watson et al. 2000). The amount of C stored in forest stands depends on its age and productivity. The terrestrial C sink, inferred from changes in the atmospheric gas concentrations and its isotopic composition, is primarily attributed to the increase in productivity (Chambers et al. 2001). Much less attention has been paid to changes in forest age, another important characteristic of this C reservoir (Alexandrov 2007). Since biomass increases with stand age, delaying harvest to the age of biological maturity may result in the formation of a larger C sink (Alexandrov and Yamagata 2002; DOE 2007). In the Technical Guidelines Voluntary Reporting of Greenhouse Gases Program (DOE 2007) listed the relationships between age and biomass for different tree species at high-productivity sites and suggested that young forest biomass is significantly lower than that at biological maturity (Figure 3). For example, biomass could be doubled from 15 yr old (6.74 kg C m^{-2}) to 30 yr old (13.4 kg C m^{-2}) loblolly pine (*Pinus taeda L.*) forest. In addition to the impacts on forest biomass and productivity, forest harvest could greatly influence organic C storage in the forest soils and wood products. Forest harvest for agriculture, forest management, and use of wood has a significant effect on terrestrial C stocks in the US (Birdsey and Heath 1995; Houghton 1999; Birdsey and Lewis 2003; Birdsey et al. 2006). Globally, soils contain more carbon than any other terrestrial C pool (Schlesinger 1977; Jobaggy and Jackson 2000), and the forest floor is the most dynamic part of soil organic matter. Estimates of the effect of forest harvest on soil C storage are critical to predictions of carbon exchange with the atmosphere in the SUS. Forest harvest may have a significant effect on forest floor structure and function through mechanical disturbance, inputs of logging slash, alterations in litter production, and leaching of dissolved organic matter, as well as the alteration of temperature and moisture regimes (Birdsey et al. 2006; Figure 1, 2). One of the most influential chronosequence studies in recent history is from Covington (1981), who created the “Covington’s curve”. Covington’s curve described differences in organic matter storage in the forest floors of northern hardwood forests that had been harvested at different time periods. This study concluded that forest floor mass declined sharply following harvest, with 50% of forest floor organic matter lost in the first 20 years. The apparent losses of organic matter were attributed to increases in decomposition rates and decreases in litter inputs as the ecosystem reorganized (Covington 1981; Yanai et al. 2003). Yanai et al. (2003) recommended that the mechanisms underlying Covington’s curve are still not well understood and fortunately, the impressive magnitude of the reported C and N loss inspired a multitude of follow-up studies. By collecting a large amount of field experimental data, Johnson et al. (1992, 2001) did a meta-data analysis to indentify the relationships between forest harvest and soil carbon storage. They concluded that soil C and N could decrease or increase after forest harvest, but this depends on harvest and fire regimes, and tree species. Yanai et al. (2003) revisited Covington's curve and the relationships reported by Johnson et al. (2001) and concluded that more research is still needed to understand forest floor dynamics following disturbance. They also reported that forest harvest has a much smaller effect on forest floor and soil C pools than was predicted from early interpretations of Covington's curve.

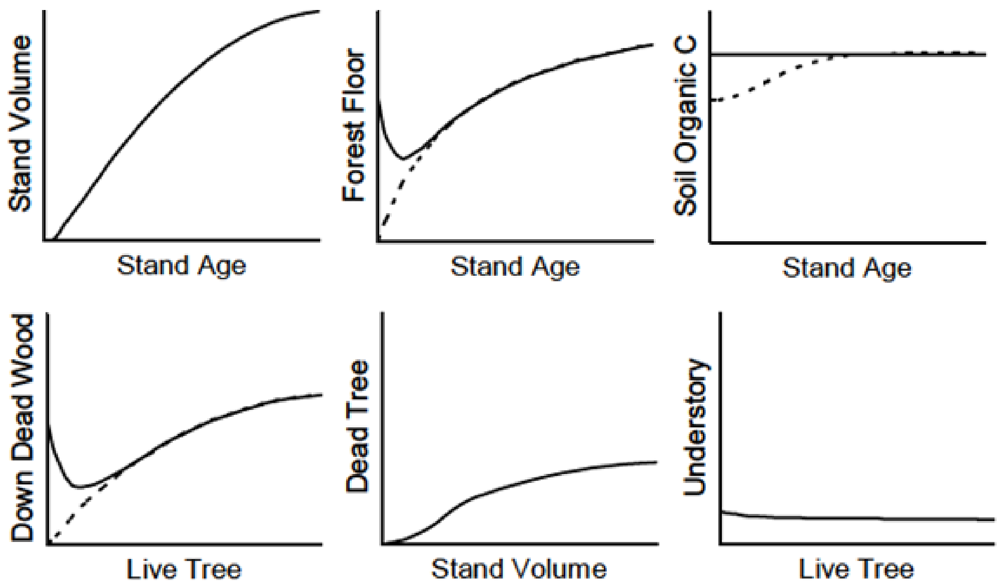

Figure 3 The basic relationships among the components of the forest ecosystem C pools. Figures are not drawn to scale; dashed lines qualitatively represent the difference between afforestation and reforestation. (Source: DOE 2007).

Wood products are another important C pool induced by forest harvest (Figure 3). When forest trees are harvested, C remains stored in resulting value-added wood products. The categories of harvested wood products commonly include: 2) C in wood products in use; 2) C in landfills; 3) emissions from wood burned to produce energy; and 4) emissions from wood either decaying or burned without producing energy. Wood products in use can be then divided into three types in terms of their life span. Short-term wood products such as pulp paper has an average life span of 1-5 years, middle-term wood products such as panels and wood containers could have an average life span of 20-30 years, while long-term wood products could have an average life span of 60-100 years (Birdsey et al. 2006; DOE 2007; Howard et al. 2009). Since forest biomass and soil organic C after harvest could be lower than that before harvest in the long term, C storage for in-use wood products and their life span become one of the most important factors to determine whether this forest ecosystem is a C sink or source (Birdsey et al. 2006; Skog and Nicholson 2000). Combustion of wood residues is a CO_2-neutral operation; hence, wood products could be used as energy to replace fossil fuel and thus could be one of the most important solutions to potentially mitigate global climate change (Birdsey et al. 2006; Malmsheimer et al. 2008). One cubic meter of wood can reduce CO_2 emission from fossil fuels by ~1.1 tonnes. Wood products require less energy for manufacturing than alternative raw materials, and hence contribute even more to the reduction of fossil fuel consumption. By using the full potential of wood in buildings, Europe could reduce emissions of CO_2 by 300 million tons or 15 to 20% (source: http://ec.europa.eu/enterprise/sectors/wood-paper-printing/files/ccmreport.pdf). FAO (2004) estimated that stocks of sequestered C in products in use and in landfills were increasing at a rate of 139 million tons of C in 1990. This annual increase in sequestered harvested wood product C represents an equivalent net removal of C from the atmosphere that is substantially more than the annual direct greenhouse gas emissions from the global forest products industry and perhaps more than twice the overall industry's emissions.

Similar to the C cycle, harvest could greatly change the N cycles in the forest ecosystems. Different from C flux and storage, soil available and total N may greatly increase in a short term after harvest; however, they could decrease rapidly and even lower than contents before harvest (Johnson and Curtis 2001). The increase in available N following harvest could be related to four reasons: more litter N input, reduced N uptake by vegetation, faster litter N mineralization rates and more N-fixing vegetation establishment (Johnson and Curtis 2001; Johnson et al. 1995; Aber et al. 1978). The quick decrease in available N after a short term (3-5 years) could result from: 1) more N leaching due to higher available soil N; 2) higher N immobilization in the soil organic matter or litter; 3) increasing uptake by fast-growing vegetation; and 4) more N loss from nitrification or denitrification. N gas emissions due to higher available N are similar to the effects of N fertilization, which will be discussed in the next paragraph.

The relationships between timber harvest and runoff (or water yield) has long been of interest (e.g., Bates and Henry 1928). Worldwide studies showed that water yield usually increases immediately after timber harvest (Bosch and Hewlett 1982; Douglass 1983; Harr 1983; Hibbert 1983; Troendle 1983; Stednick 1996; Sun et al. 2005). The relative amount of increase depends on climate regime and forest type and tends to decrease as forests regenerate (Bosch and Hewlett 1982; Whitehead and Robinson 1993; Bari et al. 1996; Lesch and Scott 1997). Decreases in ET are the primary causes for resulting increases in water yield following forest harvests (Bosch and Hewlett 1982; Trimble and Weirich 1987; Sun et al. 2005). Because leaf area index (LAI) decreases after harvest, rainfall or snow intercepted by forest canopy could be greatly reduced, and thus canopy evaporation or sublimation losses are reduced during precipitation events (Troendle and King 1987; Sun et al. 2005). Increases in water yield also depend on the percentage of land area that is harvested, and harvest regimes emplyed (Trimble and Weirich 1987). In general, increases in water yield are assumed to be positively correlated to the percentage of area harvested. For partial cuts or canopy thinning, removal of vegetation may result in smaller increases in water yield than predicted by area alone because of increased use of available moisture by remaining vegetation and surrounding uncut vegetation (Hibbert 1966). Bosch and Hewlett (1982) reviewed 94 catchment experiments to determine the effects of vegetation changes on water yield and ET. They found that annual streamflow increases with decreases in vegetation cover because of decreasing ET. Many studies have been evaluated the effects of various treatments of vegetation (Williamson et al. 1987; Maimer 1992; Rowe and Pearce 1994), seasonal effects of vegetation change (Bren and Papworth 1991), and differences in land use practices including forest clearing, reforestation, and forest thinning (Trimble and Weirich 1987; Ruprecht and Stoneman 1993; Sun et al. 2005). Although several regional studies have specifically addressed the impacts of reforestation and deforestation on water yield (or runoff or streamflow) and evapotranspiration in the SUS (e.g. Trimble and Weirich 1987, Trimble et al. 1987, and Sun et al. 2005; Jackson et al. 2005), no work has studied the impacts of long-term (e.g. longer than 100 years) land use change on water cycles in the entire SUS. This is primarily due to lack of spatial data sets and sophisticated process-based models.

PRESCRIBED FIRE

Fire has played an important role in the structure of natural ecosystems throughout North America and represents a major disturbance that could result in a large quantity of C being emitted from the land to the atmosphere (Malmsheimer et al. 2008). Due to intentional suppression of wildfire and increases in forest plantation area, prescribed fire is widely used to manage forest ecosystems. Prescribed fire is an important forest management tool in the SUS, and could result in beneficial ecological, economic and social impacts. The history of prescribed burning can be dated back to the 1950s when intensive forest management emerged (Malmsheimer et al. 2008; Wiedinmyer et al. 2010). During the 20th century, prescribed burning was also used for site preparation to clean the forest floor before planting seedlings or sowing seeds, especially in forest plantations. Slash burning has been used to remove logging slash to uncover the soil in preparation for planting or seeding to secure regeneration, reduce competition of non-dominant species, improve wildlife habitat, and minimize catastrophic wildfire hazards due to accumulating limbs and unmerchantable trees. Due to more plantation area in the SUS, land area subjected to prescribed burns has greatly increased (http://www.firescience.gov/). Approximately 0.91 million ha of forests have received prescribed fire each year for the entire IS and ~ 0.43 million ha (47.8%) of this occurs in the SUS (Source: http://www.fire.uni-freiburg.de/iffn/country/us/us_9.htm). The USDA Forest Service has set a goal of burning 1.2 million hectares per year for the entire United States by the year 2010 (Bell et al. 1995). Management-ignited prescribed fires accounted for the most at 62.2 percent of the system total in the US; followed by slash reduction (25.3%), brush and rangeland (8.3%), and prescribed natural fire (4.2%). Overall, the national forests conducted an average of 6,763 burns per year, of which 75% were for slash reduction and 20% were management-ignited burns of natural fuels. Changes in prescribed burning size and intensity could greatly influence the C and N cycles in forest ecosystems.

Prescribed burning is either reported as an important C sink or a C source, depending on fire intensity and regimes. If the prescribed fire is too intense, most of the litter including the coarse woody debris on forest floors could be burnt and could result in a larger C emission than uptake by enhanced forest productivity and biomass. In contrast, prescribed fire could reduce C emission by reducing wildfire size and severity (Malmsheimer et al. 2008; Wiedinmyer and Hurteau 2010). Wiedinmyer and Hurteau (2010) found that wide-scale prescribed fire application can reduce CO_2 fire emissions by 18-25% in the western U.S., and by as much as 60% in specific forest systems. Based on a model and statistical data, Vose et al. (1996) found that ~ 0.33 million ha forests were burned due to site preparation in 1982 in the six southeastern states (Alabama, Georgia, North Carolina, South Carolina, Tennessee, and Virginia).

Similar to wildfires, prescribed fire could result in large amounts of trace greenhouse gas emissions such as N_2O, CH_4, and CO. Vose et al. (1996) found that 11.8 tonnes CO_2, 202.0 kg CO, 36.1 kg CH_4, and 6.7 kg N_2O were emitted into the atmosphere due to wildfire burning in 1982 for the 6 states in the SUS, while 13.5 tonnes CO_2, 391.5 kg CO, 43.1 kg CH_4, and 13.6 kg N_2O were released to the atmosphere due to prescribed fire. Although slash burning is widely used, few studies (e.g. Vose et al. 1996) have evaluated the regional impacts of slash burning on C sequestration and trace greenhouse emissions in the SUS.

NITROGEN DEPOSITION AND FERTILIZATION

Human activities have greatly accelerated emissions of both CO_2 and biologically reactive N to the atmosphere. The amount of reactive N deposited on land has doubled globally and has become at least five-times higher in Europe, Eastern United States, and South East Asia since 1860, mostly attributed to increases in fertilizer production and fossil fuel burning (Galloway et al. 2004). Due to rapid industrialization and population growth in the SUS during the 1960s-1970s, N deposition increased rapidly during this period (Dentener 2006). With the successful execution of the Clean Air Act in the 1970s, the N deposition rate in the US remained relatively constant or even decreased to the level at the early of 1990s as indicated by the observational data from 270 NADP sites (http://nadp.sws.uiuc.edu) and 87 CASTNET sites (www.epa.gov/castnet) in the US and regional reanalysis data from Dentener (2006) and Holland et al. (2005). Although N deposition rate in the SUS is relatively low compared to the northeastern US, the long-term accumulation of deposited N could augment the effects of N deposition.

Nitrogen fertilization is one of the major contributors to the enhanced crop productivity worldwide. The annual removal of crop products and residues from these ecosystems could greatly reduce the soil N content. This continuous removal can decrease cropland nitrogen, thereby decreasing crop productivity. In addition, due to continuous disturbance to the cropland soil (e.g. tillage), soil N leaching could be much higher in the cropland than other natural ecosystems. Although N fixation and deposition from atmosphere and residue return could alleviate the N limitation to a certain degree, N losses still dominate the N fluxes in cropland. Many cropland areas were reported to abandon after several years of cultivation due to significant decreases in productivity induced by N limitation around the world. After the first emergence of synthetic N fertilizers during the early 20th century, N fertilizer amounts in the cropland have greatly increased in the SUS. N fertilizer use has greatly increased from about 0.4 Tg N in 1945 to about 3.0 Tg N in the early of 2000s in the SUS based on the county-level statistical N fertilization data (Alexander and Smith 1990; Ruddy et al. 2006; Tian et al. 2010a). The increased N fertilizer on cropland has been reported to greatly increase crop productivity and soil C storage in the SUS.

Since forests are long-lived (if undisturbed), N limitations could be gradually alleviated during its life cycle through N fixation and atmospheric deposition. However, disturbances such as forest harvest could greatly change the N status in forest ecosystems. Disturbances could not only result in a reduction in total N content, but can also change N demand and leaching rate.

Forest regrowth after land use and land cover change or harvest was identified as a major driver of elevated C uptake in the Northern Hemisphere in 1980-1999 (Fox et al. 2007). The SUS produces more industrial timbers than any other regions of the world and now comprises almost one-half of the world's industrial forest plantations (Prestemon and Abt 2002; Fox et al. 2007). Currently, there are 13 million ha of pine plantations in the SUS (Wear and Greis 2002), predominantly comprised of loblolly pine. In 1990, about 0.08 million ha of pine plantations were fertilized whereas over 0.49 million ha were fertilized in 2004 (Fox et al. 2007; Albaugh et al. 2007). From 1969 to 1991, fertilized forests increased from 0.01 to 0.08 million ha per year and a total of ~6.47 million ha forested areas were fertilized during this period in the SUS. However, fertilized forest rapidly increased from 1991 to 2004 and peaked in 1999 (0.6 million ha per year). Based on county-level statistical data from Alexander and Smith (1990) and Ruddy et al. (2006), non-farm (managed grassland and forestland) N fertilizer amounts have increased from 48.5 to 93.9 thousand tons per year in the SUS. Among which most are put in the forests. This enormous anthropogenic N input could greatly change the nitrogen cycle in managed ecosystems of the SUS.

Many studies have found that N fertilization effects are partly responsible for the "missing C sink" (Townsend et al. 1996; Holland et al. 1997, 2005; Reich et al. 2006; Magnani et al. 2007). Especially for N-limited ecosystems such as temperate forests, anthropogenic N deposition plays a critical role in stimulating plant growth, altering soil respiration, and determining the C allocation pattern and subsequent C storage patterns (Vitousek and Howarth 1991). Because N is a primary limiting nutrient throughout terrestrial ecosystems of mid and high latitudes and an important limiting nutrient for plant growth throughout subtropical and tropical ecosystems (Vitosek et al. 1998), increased N deposition could have an attenuating effect on rising atmospheric CO_2 by stimulating the vegetation productivity and accumulation of C in biomass (Churkina et al. 2007). One third of the global N inputs entered the land ecosystems. Due to the high C:N ratios and long lifetimes of C in wood, the impacts of N deposition on C storage in forests could be very large (Churkina et al. 2007). Studies have agreed on the location of major response (i.e., the temperate forests located between 25° and 55° north), but no consensus on the response magnitude (Churkina et al. 2007). Based on results from ^{15}N-tracer experiments, Nadelhoffer et al. (1999) argued that increased N inputs from atmosphere made a minor contribution to terrestrial C sink. They suggested that temperate forests sequestered only 0.25 Pg C per year by increasing N deposition. In contrast, model based estimates (e.g., Townsend et al. 1996; Holland et al. 1997, 2005) showed significant increases in C uptake. Townsend et al. (1996) estimated an additional C uptake in the order of 0.3-1.3 Pg C per year in global terrestrial ecosystems by using an ecosystem model and spatially explicit N deposition data. Using the same ecosystem model and different nitrogen deposition data, Holland et al. (1997) showed an even higher C uptake of 1.5-2.0 Pg per year. Fenn et al. (2003) found the low levels of N deposition in the western United States could lead to a significant increase in plant productivity. The potentially detrimental effects of excessive N have also been found in N-saturated ecosystems (Aber et al. 1998; Matson et al. 1999; Asner et al. 2001; Fenn et al. 2003). Many studies have shown that N_2O emissions increase substantially after N additions from N deposition and fertilization (e.g., Neff et al. 1994; Matson et al. 1992; Papen et al. 2001; Butterbach-Bahl et al. 2002). These studies indicated a key role of N addition in the control of N_2O emissions in N limited ecosystems.

A wide range of short-term N addition experiments have been conducted to examine changes in plant growth, soil respiration, N retention and loss in response to chronic N loads or removal in various ecosystems (Wright and Rasmussen 1998; Magill et al. 2004; Mo et al. 2004; Bowden et al. 2004; Niu et al. 2009). These studies provide insightful points on the effects of altered N availability even though the reported ecosystem responses have been divergent or even controversial. However, Högberg et al. (2006) pointed out high doses of N addition over short periods could not truly reflect the long-term effects at lower rates. Little is known about the consequences of increased N inputs into ecosystems from a long-term perspective. In addition, it's difficult to study the interactions between N deposition and changes in climate, atmospheric composition, and land use and land cover using field experiments. Modeling simulations were widely used as an alternative tool for studying the impacts of long-term N addition (Nadelhoffer et al. 1999; Asner et al. 2001; Lu et al. 2010). Currently, forest area accounts for ~50% of the SUS land area. Although most forest areas in this region have not received N fertilization, the long-term N deposition could contribute much to tree growth. In addition, disturbed young-aged forests dominate the SUS. These areas display higher productivity than old-aged forest and have thus a higher N demand. Many previous studies (e.g., Townsend et al. 1996; Holland et al. 1997, 2005) assumed the forest is mature and other factors could have not significant interaction with N fertilization or deposition effect. This might underestimate impacts of N depositionon on forests in a long term.

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Chapter 3

Drought in the Southern United States over the Last Century: Variability and its Impacts on Terrestrial Ecosystem Productivity and Carbon Storage

ABSTRACT

Droughts are one of the most devastating natural hazards faced by the Southern United States (SUS) today. Precipitation in the SUS as a whole has been found to increase during 1895-2007, drought events and their adverse impacts on the economy, society and environment have been extensively reported. In this study, our objective is to use the standard precipitation index (SPI) to characterize drought intensity and duration and explore the relationship between drought and the terrestrial ecosystem function (i.e., net primary productivity (NPP) and net carbon exchange (NCE) in the SUS using a process-based Dynamic Land Ecosystem Model (DLEM). Results indicated that drought varied spatially and temporally throughout the SUS. Although annual and growing-season SPI in the SUS has slightly increased, both percent drought area and drought duration exhibited no significant change. Combining the overall information of growing-season SPI, drought area, and duration, we concluded there was no obvious evidence of a significant change in drought condition in the SUS as a whole over 1895-2007. However, area of the SUS experiencing excessive rainfall appeared to be increasing. NPP varied enormously among years due to changing precipitation patterns, but there was no significant ($P < 0.05$) change trend during this time period. NPP was noted to decrease up to 40% in some areas during extreme droughts. Climate change in the SUS has resulted in a net release of 0.33 Pg C (1 Pg = 10^{15} g) into the atmosphere, while changes in precipitation and temperature patterns induced C emissions of 0.035 and 0.14 Pg, respectively during 1895-2007. The interaction between precipitation and air temperature induced a C emission of ~0.15 Pg, suggesting that changes in air temperature could significantly enhance drought impacts in the SUS. In total, C emission from drought impacts induced both by precipitation and temperature could be ~0.19 Pg. The western SUS (dry region) was found to act as a C sink, while the east (water-rich region) acted as a C source due to changing precipitation patterns. C source in the east was significantly enhanced by the interactions between precipitation and temperature changes. These findings suggested that climate change could greatly increase C sequestration in the drier regions of the SUS. Both NPP and NCE significantly increased along a gradient of declining drought intensity. Changes in air temperature could significantly enhance or reduce drought impacts on NPP and NCE for different vegetation types. Changes in precipitation induced a C source in forest and wetland ecosystems, while a C sink in shrubland, grassland, and cropland ecosystems. Drought impacts on C sources for forest were enhanced by the interactive effects between precipitation and temperature changes, while reduced for wetland. However, this interaction enhanced C sinks in shrubland and reduced in grassland and cropland. More experimental evidence is also needed for the further improvement of ecosystem models to adequately simulate complex interactive processes among air temperature, precipitation, and other environmental factors.

Key words: Southern United States; climate change; drought; standard precipitation index; net primary productivity (NPP); net carbon exchange (NCE)

INTRODUCTION

A widespread increase in annual precipitation is projected by most models over the majority of the North American continent, except the Southern and Southwestern United States and

Mexico (IPCC 2001, 2007; CENR 2008; Karl et al. 2009), and it is likely that water distribution among different terrestrial ecosystems will be more variable (Saxe et al. 2001; Salinger 2005). Drought actually means water deficiency of the land surface, while water availability at land surface is mainly controlled by the balance between precipitation and evapotranspiration (ET), and their combination governs the dry or wet status of land surfaces. Although droughts are generally associated with decreased precipitation, increased rainfall does not necessarily mean less intense droughts. If the increased magnitude of ET is larger than that of precipitation, the soil water budget could be negative. Therefore, precipitation amount alone could not accurately represent the wetness/dryness of the land surface. Other factors such as increases in air temperature could also induce drought events. The higher temperatures could increase ET over the land surface, thus reducing soil moisture and contributing to drought. In addition, higher temperatures will cause precipitation events to become more extreme and less frequent. Global warming is predicted to cause massive droughts that will threaten the lives of millions and cover half the land surface in the next 100 years (IPCC 2007). Likely increases in temperature in the future have raised concerns about drought frequency and intensity on terrestrial ecosystems in the USA (Houghton 2004).

To effectively characterize drought conditions, several indices have been introduced during past decades, such as Palmer Drought Severity Index (PDSI, Palmer 1965), Standard Precipitation Index (SPI, McKee et al. 1993), and PHDI (Palmer Hydrologic Drought Index, Karl and Knight 1985). SPI has been widely used to evaluate drought events worldwide (McKee et al. 1993, 1995; Wilhite et al. 2000). It was first created by McKee et al. (1993, 1995) to address some of the limitations that exist in PDSI. Like PDSI, this index is negative for dry, and positive for wet conditions. But the SPI is a probability index that considers only precipitation, while PDSI is water balance index that considers water supply, demand, and loss. To identify the relationships between drought and ecosystem functions, the SPI which is independent from the ecosystem functions could be better than PDSI. In addition, SPI is relatively simple, spatially consistent and temporally flexible, thus making this index more reliable for detecting emerging drought, and became an increasingly important tool for assessing moisture condition and initiating mitigation and response actions (Guttman 1998; Wilhite et al. 2000; Ji and Peters 2003).

Carbon and water fluxes between the atmosphere and terrestrial ecosystems are interactively linked at various temporal and spatial scales (Farquhar et al. 1980; Running et al. 1988; Rodrigues-Iturbe 2000; Jackson et al. 2005). Drought directly influences photosynthesis of individual plants and thus the carbon cycle in an ecosystem (Hanson and Weltzin 2000; Barr et al. 2006; Ciais et al. 2005; Grainer et al 2007; Tian et al. 2003, 2010a, d). Drought impacts on ecosystem functions such as mortality, gross/net primary productivity (GPP/NPP), and carbon fluxes have been reported worldwide (e.g., Hanson and Weltzin 2000; Ciais et al. 2005; Zeng and Qian 2005; Zhao and Running 2010; Asner and Alencar 2010). In the southern United States (SUS), few studies have been conducted to address drought impacts on plant productivity, plant mortality and ecosystem C sequestration (e.g. Elliott and Swank 1994; Olano and Palmer 2003; Karl et al. 2009; Klogs et al. 2009). Based on PDSI data from NOAA and 1991-2005 Forest Heath and Monitoring (FHM) plot data from Alabama, Georgia, and Virginia, Klogs et al. (2009) examined drought impacts on three tree species groups (pines, oaks and mesophytic species). They found that the pines and mesophytic species showed significant reductions in growth rate with increasing drought severity. During drought periods in the 1950s and 1980s, many studies found that drought significantly affected tree mortality and growth (Buell et al. 1961; Small

1961; Elliott and Swank 1994; Olano and Palmer 2003). Reports have also indicated that drought could change ecosystem plant composition and structure resulting in dominance or increasing of plant species that tolerate water stress (Elliott and Swank 1994). Drought can also indirectly affect plant communities by predisposing some plant species to damage from other abiotic (e.g., fire) or biotic (e.g., disease and insects) factors (Olano and Palmer 2003). Drought inducing a decrease in ground litter moisture is primary prerequisite factors for the occurrence of wildfire.

The climate in the SUS is warm and wet, with mild winters and high humidity, compared to the rest of the continental United States (Karl et al. 2009); however, due to increased water use demand and climate variability, this water-rich region has experienced serious water stress (McNulty et al. 2007). Although climate (precipitation and air temperature) was not reported to change significantly in the SUS during 1895-2007 (Karl et al. 2009), drought events and their impacts have been frequently reported. Precipitation patterns in the SUS exhibited a great spatial variability with a gradient that increases from the west to the east and inter-annual variability of alternate extreme high and low precipitation. The report of climate change in the United States (Karl et al. 2009) noted that the average annual temperature of the SUS did not change significantly over the past century as a whole. Since 1970, however, annual average temperature has raised ~1.6°F, with the greatest seasonal increase during the winter months. The warmer winter in the SUS could indirectly cause more water evapotranspirated to the atmosphere and possibly results in drought. In the near future (present to 2100), many climate change scenarios suggest that more frequent and/or intense drought episodes are expected across the SUS, and potential ET is predicted to exceed summer precipitation (Smith and Tirpak 1990; IPCC 2007; Karl et al. 2009; Seagers et al. 2010). Therefore, it is important to understand the effect of drought on ecosystem health and function. Although drought conditions in the continental U.S.

have been addressed in many studies (e.g., Karl 1983; Soule 1990; Andreadis and Lettenmaier 2006; Cook et al. 2007; Luo et al. 2008; Seagers et al. 2010), it is rather limited for specifically studying how drought has influenced ecosystem long-term productivity and C sequestration in the SUS. Based on the Standard Precipitation Index and a process-based Dynamic Land Ecosystem Model (DLEM) which fully couples carbon, nitrogen and water cycles in terrestrial ecosystems, our objectives in this study are to: 1) quantify spatial and temporal patterns of drought intensity and duration in the SUS; and 2) explore drought impacts on ecosystem NPP and C sequestration.

METHODS

Study region

The SUS defined in this study includes 13 states: Alabama, Arkansas, Florida, Georgia, Kentucky, Louisiana, Mississippi, North Carolina, Oklahoma, South Carolina, Virginia, Tennessee, and Texas (Tian et al. 2010a, c). Elevations within the region range from near sea level along the Gulf and Atlantic coasts to more than 1,800 m in the Appalachian Mountains. Overall, the climate is temperate, becoming largely subtropical near the coast. The dominant land cover types in this region are temperate evergreen needleleaf forest and temperate deciduous broadleaf forests. Cropland area accounts for ~20% of the total land area in 2000 (Tian et al. 2010a; Chen et al. 2006).

Data description

Climate and other model input data

In this study we reconstructed a 8 km × 8 km resolution daily climate dataset (including precipitation, daily maximum, minimum and average air temperature, and dew point) for the entire SUS from 1895 to 2007 by integrating the daily climate patterns of the North American Regional Reanalysis (NARR) dataset (http://wwwt.emc.ncep.noaa.gov /mmb/rreanl/) that covers the period of 1979 to 2005 into the monthly climate dataset (1895 to 2007) developed by PRISM (Parameter-elevation Regressions on Independent Slopes Model) Group at Oregon State University (http://prism.oregonstate.edu/). Furthermore, we generated a 30-year detrended (i.e., no obvious change trend) climate dataset from the daily climate data for the period 1895-1924 for model spin-up run after equilibrium state which could avoid sudden vibrations in model results due to simulation mode changes from equilibrium to transient mode.

Temperature exhibited an increasing trend from 1895 to the mid-1950s (0.12^{o}C per decade) and then showed a sudden decrease over several years (1955-1960) (Figure 1a). After the 1960s, air temperature increased again with time (0.12^{o}C per decade). Therefore, during the entire study period, no obvious trend for air temperature was observed for the entire SUS. From 1895 to 2007, precipitation showed an increasing trend at a rate about 10 mm per decade (Figure 1b). Annual mean precipitation for the entire region increased ~ 40 mm from 1895 – 1950 to 1951 – 2007. Precipitation varied from 797 mm in 1954 to 1316 mm in 1973 with huge inter-annual variability. Annual mean precipitation in the SUS decreased along an east-west gradient with a semiarid climate in the west and a humid climate in the east. Although precipitation showed a general increasing trend for the entire SUS, it decreased in some areas in the eastern part of the SUS from 1895-1950 to 1951-2007 (Tian et al. 2010a, c), which suggests that spatial variability should be considered when address impacts of changing precipitation pattern on ecosystem functions.

The generation of other model input data including air humidity (or dew point), soil texture, and topography data has been described in detail by Tian et al. (2010a, c) and Zhang (2008).

Standard precipitation index

To characterize the drought intensity and duration for the SUS and to compare the spatial difference in ecosystem function response to drought, SPI was used. SPI has been widely used to evaluate the drought events around the world. The monthly precipitation data collected from the PRISM (http://prism.oregonstate.edu/) were used to calculate SPI. We used the same methods of McKee et al. (1993, 1995) to reconstruct historical SPI. 1-, 2-, 3-, 4-, 5-, 6-, 9-, 12- and 24-month SPI have been used to represent short-term, middle-term and long-term drought conditions and proved to be useful in monitoring drought conditions. In this study, we used 1-, 3-, 6- and 12- month SPI to represent monthly (short-term), seasonal, growing season and annual drought conditions. Seven categories of SPI were defined by McKee et al. (1993, 1995): extremely wet (> 2.0), very wet (1.5 to 1.99), moderately wet (1.0 to 1.49), near normal (-0.99 to 0.99), moderately dry (-1.49 to -1.0), severely dry (-1.99 to -1.5), and extremely dry (< -2.0). We redefined dry areas as SPI < -1.0 in this study.

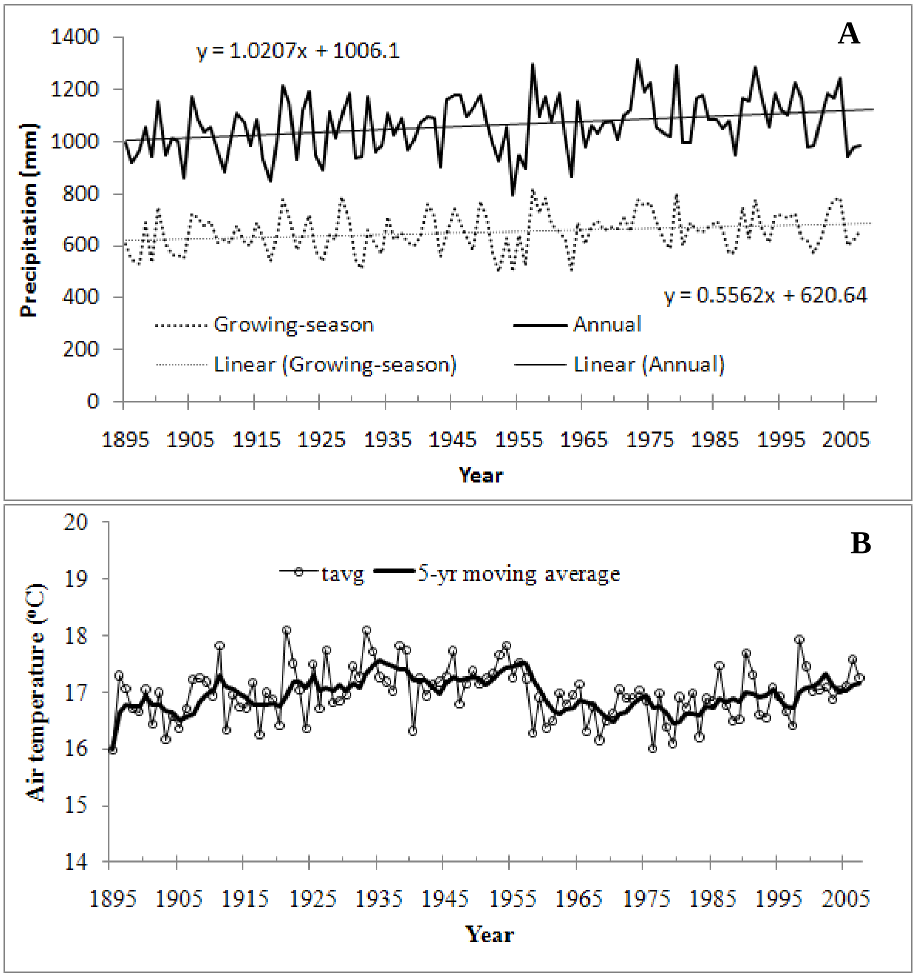


Figure 1 Annual and growing-season (May-October) precipitation (A), and mean air temperature (B) during 1895-2007.

Table 1 Photosynthesis and C allocation parameters and their values for DLEM simulations

DBF	ENF	SHR	GRA	CRO*	Units	Description
0.015	0.02	0.025	-	-	% yr^{-1}	Annual natural mortality
0.1	0.1	0.1	-	-	yr^{-1}	Turnover rate for sapwood
15	25	15	25	15	kg C kg N^{-1}	Minimum leaf C:N ratio
35	50	40	40	35	kg C kg N^{-1}	Maximum leaf C:N ratio
300	350	300	-	-	kg C kg N^{-1}	C:N ratio of dead wood
55	55	55	40	40	kg C kg N^{-1}	C:N ratio of fine root
150	200	150	60	60	kg C kg N^{-1}	C:N ratio of coarse root
150	200	150	75	75	kg C kg N^{-1}	C:N ratio of sapwood
0.5	0.5	0.5	0.5	0.5	dim	Canopy light extinction coefficient
2.5	2	1.6	2.2	3	mm	Canopy water interception coefficient
7	7	3.5	4	7	m^2 m^{-2}	Max. leaf area
7	15	15	12	20	m^2 kg C^{-1}	Canopy average specific leaf area
2	2	2	2		dim	Ratio of SLA in shaded to sunlit canopy
2	2.6	2	2	2	dim	Ratio of all-sided to projected leaf area
0.005	0.002	0.004	0.005	0.005	m s^{-1}	Max. stomatal conductance
630	630	630	630	630	Pa	Vapor pressure deficit at start of stomatal conductance reduction
4100	4100	4100	4100	4100	Pa	Vapor pressure deficit at complete stomatal conductance reduction
45	33	25	33	55	umol CO_2 m^{-2} s^{-1}	Max. rate of carboxylation at 25C

*Different crop types have different parameter values, here given the mean values.

Abbreviations: DBF, deciduous broadleaf forest; ENF, evergreen needleleaf forest; SHR, shrubland; GRA, grassland; CRO, cropland.

Model description

General description

The Dynamic Land Ecosystem Model (DLEM) is used to simulate the influences of climate change on GPP, NPP, and net C exchange (NCE, positive indicate a C sink in this study). The DLEM is designed to simulate variations in biogeography, hydrological cycle, plant physiological processes, and soil biogeochemical cycle in land ecosystems driven by natural and anthropogenic forces such as climate variability and change, atmospheric CO_2, tropospheric ozone, land-use change, nitrogen deposition, and disturbances (e.g., fire, harvest, hurricanes) on terrestrial C, water and N cycles. The hydrological cycle of current DLEM couples the groundwater submodel (multiple soil layers, Niu et al. 2007), river routing system, and a set of sophisticated algorithms to relate water stress to ecosystem functions (Figure 2). DLEM has been widely used in China, Asia, US, and North America (e.g., Chen et al. 2006a, b; Ren et al. 2007, 2010; Liu et al. 2008; Tian et al., 2008, 2010a, b, c, d; Xu et al. 2010; Schwalm et al. 2010).

The modeling mechanisms of gross primary production (GPP) and NPP have been described in Tian et al. (2010a). Here, we listed the major parameters and their values for modeling GPP and NPP (Table 1), and described the algorithms for simulating drought impacts on NPP, GPP, and C storage below.

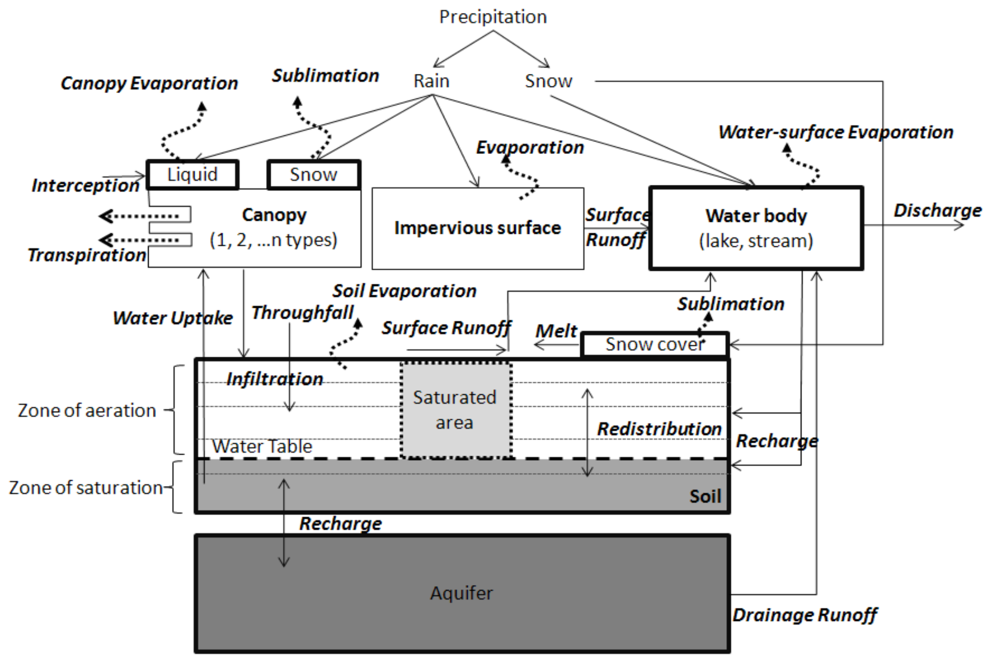


Figure 2 The schematic diagram of major hydrological processes in the Dynamic Land Ecosystem Model (DLEM). Description: Snow and rain are separated from precipitation. When precipitation enters into the ecosystem, canopy intercepts a portion of it into canopy snow and rain pools, respectively. The intercepted precipitation eventually evaporates or sublimates back to the air. The remaining precipitation (snow or rain) enters into the ground surface (soil, impervious surface and water body) as throughfall. Throughfall water arrives at the first soil layer and a portion will be transported out of the ecosystem through surface runoff and the remains will infiltrate into the second soil layer and then to deeper soil layers. The excess water from the deepest soil layer (drainage runoff) will enter into the zone of saturation (groundwater) to form surface runoff. An aquifer layer is added to account for water recharge in the zone of saturation, water body and the aquifer. Vertical soil water transport is governed by infiltration, surface, and sub-surface runoff, gradient diffusion, gravity, evaporation, and root extraction through canopy transpiration.

Hydrological cycle and drought related algorithms in DLEM

In this study, we specifically improved DLEM by integrating part of the water cycle processes in TOPMODEL (Beven and Kirkby 1979; Niu et al. 2005) and Community Land Model (CLM, Oleson et al. 2007) models. The new water cycle is illustrated in Figure 2. Compared to the last version in DLEM (e.g. Liu et al. 2008; Tian et al. 2005, 2010a, d), the new water cycle features as multiple soil layers (10 layers), multiple driving forces for soil water movement, shorter time step (bihourly step) for water movement and balance, and inclusion of water transport between land and water bodies. To better address the impacts of drought on ecosystem functions, the accuracy in simulating water cycles is critical. A set of sophisticated algorithms are included in DLEM to relate water stress to ecosystem function. We describe some of these algorithms here.

Drought will influence canopy stomatal conductance via changing vapor pressure deficit at the canopy surface and canopy water potential and thus indirectly influence plant GPP (Farquhar et al. 1980; Grant et al. 2006a, b). In theory, the canopy conductance (g_n) and photosynthesis rate are estimated as (Farquhar et al. 1980; Collatz et al. 1991; Sellers et al. 1992; Bonan 1996; Chen et al. 1999; Oleson et al., 2004):

$$g_n = \max(g_{\max} f(ppdf) f(T) f(VPD) f(W) f(CO_2), g_{\min})$$

$$GPP = f(T, g_n, CO_2, ppf, K, N)$$

Where, g_{min} and g_{max} are the minimum and maximum canopy conductance, respectively (both are constants for a specific plant functional type); *f(ppdf)* is the impact function of photosynthetic photo flux density (radiation); *f(VPD)* is the impact function of vapor pressure deficit at the canopy surface; *f(W)* is the impact function of soil moisture; $f(CO_2)$ is the impact function of atmospheric CO_2 concentration. GPP is a function of air temperature (T), enzyme (including Rubisco, light, and carbohydrate export-limited enzymes) kinetics (K), photosynthetic photo flux (ppf), stomata conductance (g_n), canopy surface CO_2 concentration (CO_2), and leaf nitrogen level (N). The function of soil moisture is estimated as:

$$w_i = \begin{cases} 0 & vwc_i \le wp_i \\ \dfrac{vwc_i - wp_i}{fc_i - wp_i} & wp_i < vwc_i < fc_i \\ 1 & vwc_i \ge fc_i \end{cases}$$

$$f(W) = \sum_{i=1}^{n}(w_i r_i)$$

Where w_i, vwc_i, wp_i, and fc_i are the relative water content, soil volumetric water content ($m^3\ m^{-3}$), wilting point, and field capacity at soil layer i, respectively. r_i is the root fraction in soil layer i. Wilting point and field capacity for a specific soil type are estimated according to soil texture (fraction of silt, sand and clay) within each soil layer.

Drought will also greatly decrease litter decomposition rate and thus result in nitrogen limitation for plant growth via decreased soil nitrogen mineralization:

$$Decom_i = k_i f(T) f(W) f(N)$$

where $Decom_i$ is the decomposition rate for litter pool i (three litter pools in DLEM: very labile litter; labile litter; resistant litter); k_i is decomposition rate of litter pool i at 25 °C air temperature and at optimum soil moisture; $f(W)$ is the function of soil moisture at the 0-20 cm soil depth; $f(N)$ is the function for available soil nitrogen; $f(T)$ is the function of temperature (for aboveground litter pools, it is daily average air temperature; for belowground litter pools, it is daily average soil temperature). $f(W)$is estimated with the CENTURY-RWC approach, which was proved to be one of the most effective approach to reflect moisture effects on decomposition (Keryn I. Paul, *CSIRO* Forestry and Forest Products, pers. comm., 2001):

$$f(W) = \frac{1}{(1 + 4 \times e^{-6 \times wi})}$$

Where w_i is the relative water content at the 0-20 cm soil layer.

In addition, drought will directly influence plant nitrogen uptake from soil. In DLEM, potential nitrogen uptake ($N_{pot,\ nup}$) from soil is estimated as:

$$N_{pot,nup} = K_{maxnup} \times f(T_{soil}) \times f_{nup}(W) \times f(N)$$

Where, K_{maxnup} is the daily maximum uptake of nitrogen; $f\ (T_{soil})$, $f_{nup}\ (W)$ *and* $f(N)$ is the impact factors of soil temperature (T_{soil}) at 0-20 cm soil layer, soil moisture at the 0-20 cm soil layer, and soil available nitrogen content, respectively. The moisture impact factor is estimated as:

$$f_{nup\ (W)} = 0.9?\,(\frac{vwc}{fc})^3 + 0.1$$

Where, *vwc* and *fc* are volumetric water content ($m^3\ m^{-3}$) and field capacity at the 0-20 cm soil layer.

Model parameterization, initialization and simulation experiments

Before running DLEM, it has been parameterized against field measurement data in the SUS. The sites and processes for DLEM parameterization have been described in detail by Tian et al. (2010a). DLEM was first run to an equilibrium state using the mean (1895-2007) climate dataset to develop the simulation baseline for C, N, and water pools. Then a 90 year spin-up simulation was conducted using the detrended climate data to stabilize unusual fluctuations caused by simulation mode shifts from equilibrium to transient modes. The data-detrending approach subtracts the best-fit line from transient climate dataset, and only retains the fluctuations about the trend. Four simulation experiments were designed to achieve the objectives: Climate change only (CLM, changes in all climate factors including precipitation, air temperature and air humidity), precipitation change only (PREC, other factors keep constant while precipitation changes with time), and air temperature change only (TEMP). The interactive effect between precipitation and temperature was calculated as: CLM – PREC – TEMP.

Since drought effects could be induced both by precipitation and temperature changes, we added the interactive effects between precipitation and temperature scenarios to the drought effects from precipitation change to reflect drought effects induced by both precipitation and temperature changes. This meant that drought effect = PREC + interactive effect.

Model evaluation and performance

The DLEM's performance to simulation climate and drought impacts on carbon storage and net primary productivity has been extensively evaluated against field observation data and regional inventory data in SUS (Tian et al. 2010a, c), China (Tian et al. 2010d; Liu et al. 2008; Ren et al. 2010), and North America (Tian et al. 2010b; Schwalm et al. 2010). Specifically, in the SUS, DLEM has been validated to successfully simulate gross primary productivity (GPP), ET, carbon storage, and net carbon exchange rate under climate change in Tian et al. (2010a, c) and Schwalm et al. (2010). DLEM's performance has also been proved to be effective in simulating impacts of drought in the North American Carbon Program (NACP) site synthesis project (Schwalm et al. 2010). In this study, we further evaluated DLEM's performance in simulating runoff (or streamflow), and NPP under the impacts of drought in the SUS. The long-term observation data for streamflow was compared with DLEM-simulated streamflow in Coweeta Basin, North Carolina, USA (35.05^{o} N, 82.42 ^{o} W) (Figure 2). We performed a t-test and results indicated that both annual and monthly patterns of observed data had no significant difference ($P > 0.4$), which implied that DLEM could capture the inter-annual and monthly patterns of runoff under climate change. DLEM-simulated aboveground NPP was also compared to the site observation data in the SUS. We selected 138 measurements from the multi-biome NPP data set published by the Oak Ridge National Laboratory (ORNL) Distributed Active Archive Center (Zheng et al. 2003). These observation data include different plant functional types including forest, cropland, grassland, and shrubland. We extracted simulated NPP from our regional simulation outputs (8 km × 8 km per pixel) to match the geographic information of these 138 sites. We found a good agreement between the simulated and measured NPP (Figure 4, slope = 1.09 and R^2 = 0.82).

RESULTS AND ANALYSIS

Drought conditions in the SUS during 1895-2007

SPI was used to characterize drought conditions in the SUS. From an annual perspective, we found that there were only 6 dry years (1904, 1910, 1917, 1954, 1956 and 1963) for the entire SUS (Figure 5A) based on the 12-month SPI. The 12-month SPI displayed an increasing trend during 1895-2007. Most years after 1970s were normal or wet years, indicating a wetter trend for the entire SUS. For growing season (May-October), we found 3 dry years (1952, 1954, and 1956) during 1895-2007 for the entire SUS (Figure 5B). A consistent dry period was found during 1951-1956. Both from annual and growing season perspectives, the drought conditions were alleviated from 1895 to 2007.

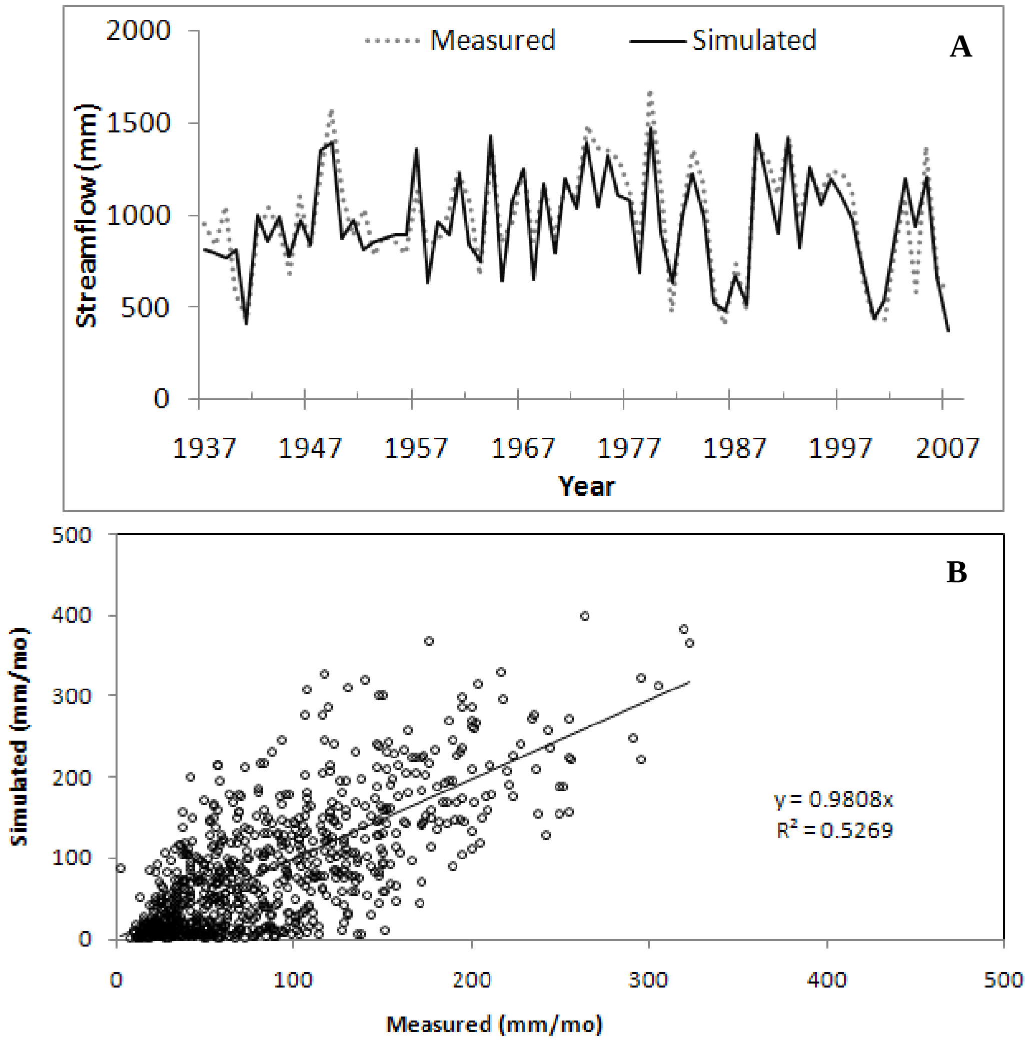


Figure 3 Comparison of DLEM-simulated streamflow with long-term observation data for watershed 18 at the Coweeta Basin, North Carolina, USA (35.05° N, 82.42 ° W). A: Long-term annual pattern; B: monthly pattern (averaged over 1937-2007).

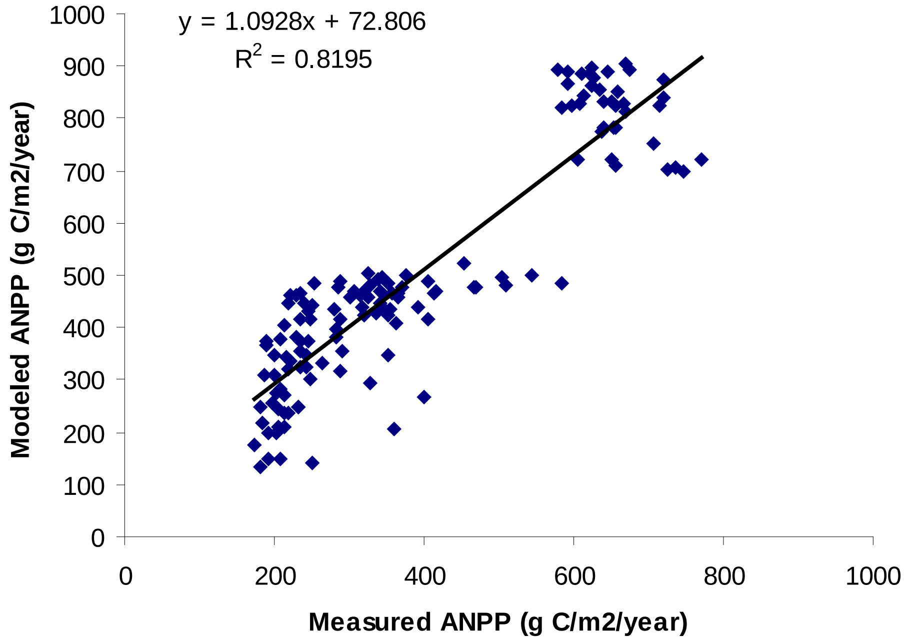


Figure 4 Comparisons of modeled annual aboveground NPP against 138 field measurements in the SUS selected from Zheng et al. (2003). Note: To match the geographic information of 138 sites, we extracted simulated NPP from our regional simulation outputs (8 km × 8 km per pixel); therefore, some uncertainties and inconsistence might exist in matching model input data with site measurements.

The regional mean annual and growing-season SPI might not fully reflect the spatial information for drought conditions in the SUS. Therefore, we used drought area and duration (which conveys the spatial drought information) to further represent drought conditions. We found that drought area slightly (not significant at $P < 0.05$) decreased in the SUS (Figure 6A). The highest percent of drought area was ~33% in 1954. This year was reported as the driest year during 1895-2007 (Seagers et al. 2010). The other high drought area percent occurred in 1917 (27%), 1925 (30%), and 1963 (27%). Drought duration also showed a slight decreasing trend

(Figure 6B), indicating that drought periods from 1895 to 2007 have not extended. A consistent dry period was also found during 1951-1956 and the longest drought duration was ~4 months in 1954 for the entire SUS. We found that the longer drought durations occurred in the same years with the higher percent drought area. Combining the trends for drought area and drought duration, we concluded that drought intensity has not been increased during 1895-2007 for the SUS as a whole. The annual and growing-season mean SPI (Figure 3A, B) indicated a decreasing trend of drought but drought area and duration did not reflect this pattern. Although there appears to be no long-term trends in drought, the area of the SUS experiencing excessive wetness appears to be increasing, particularly since the 1970s, which means flooding frequency could have increased during this study period. This pattern of climate change has also been reported for the continental US by Karl et al. (1996) and Easterling et al. (2000).

The regional average SPI, drought area and duration could not reflect the true drought conditions at spatial scales. Large spatial variation trend for SPI from 1895 to 2007 was found (Figure 7). The SPI showed an increasing tendency in most areas of the SUS, but did decrease in some areas in the east, suggesting that most areas in the SUS became wetter while some areas in the east became drier. The wetter trend in the traditionally dry region (such as Texas) and drier trend in the traditionally wet region (such as South and North Carolina) could induce tremendous changes in ecosystem functions such as NPP and C sequestration. We selected the 4 driest years (1917, 1925, 1954, and 1963) to further illustrate spatial drought distribution patterns. The longest drought duration in these 4 years was distributed across different locations (Figure 8). The longest drought duration occurred in the southern and southwestern Texas, the central east of SUS, the northwestern Texas, and the northeast of the SUS for drought events in 1917, 1925, 1954 and 1963, respectively.

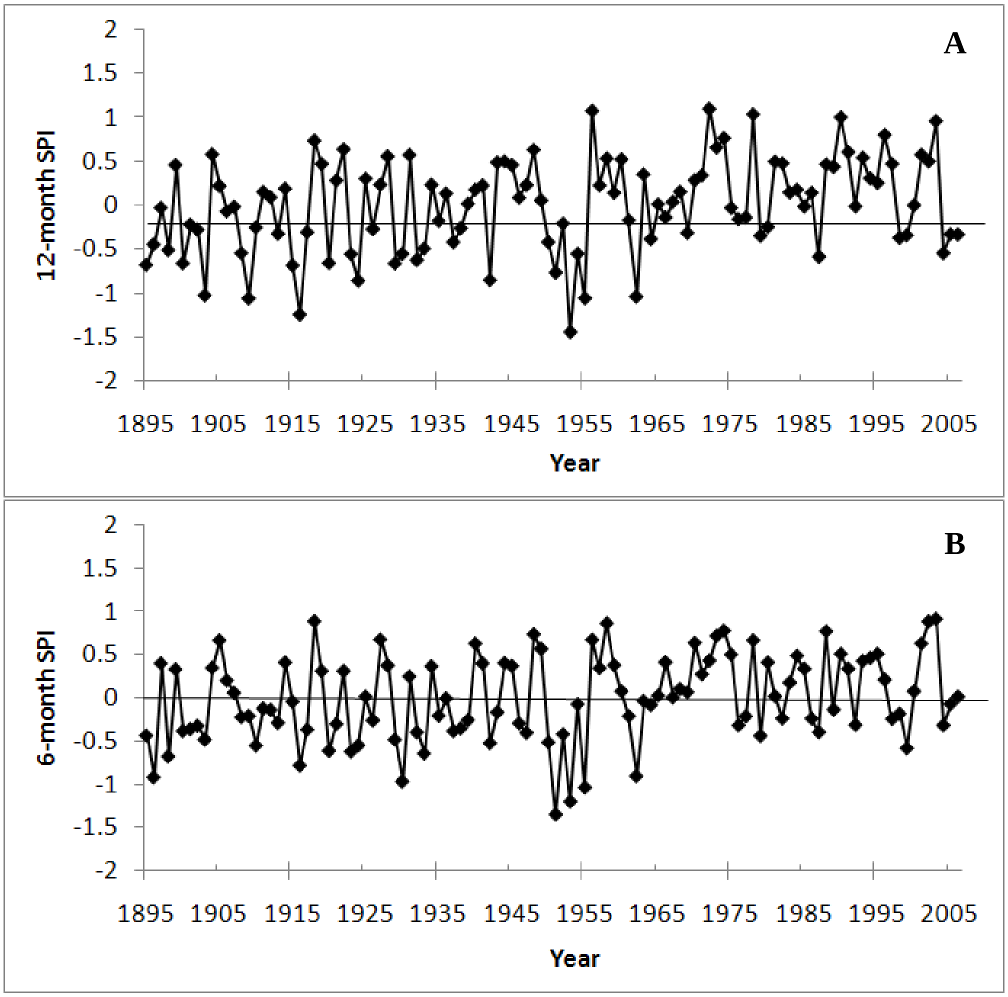


Figure 5 Regional mean annual (A) and growing-season (May-October, B) SPI (based on 12- and 6- month SPI), respectively for the entire SUS. R squares are less than 0.05, indicating no significant changing trends.

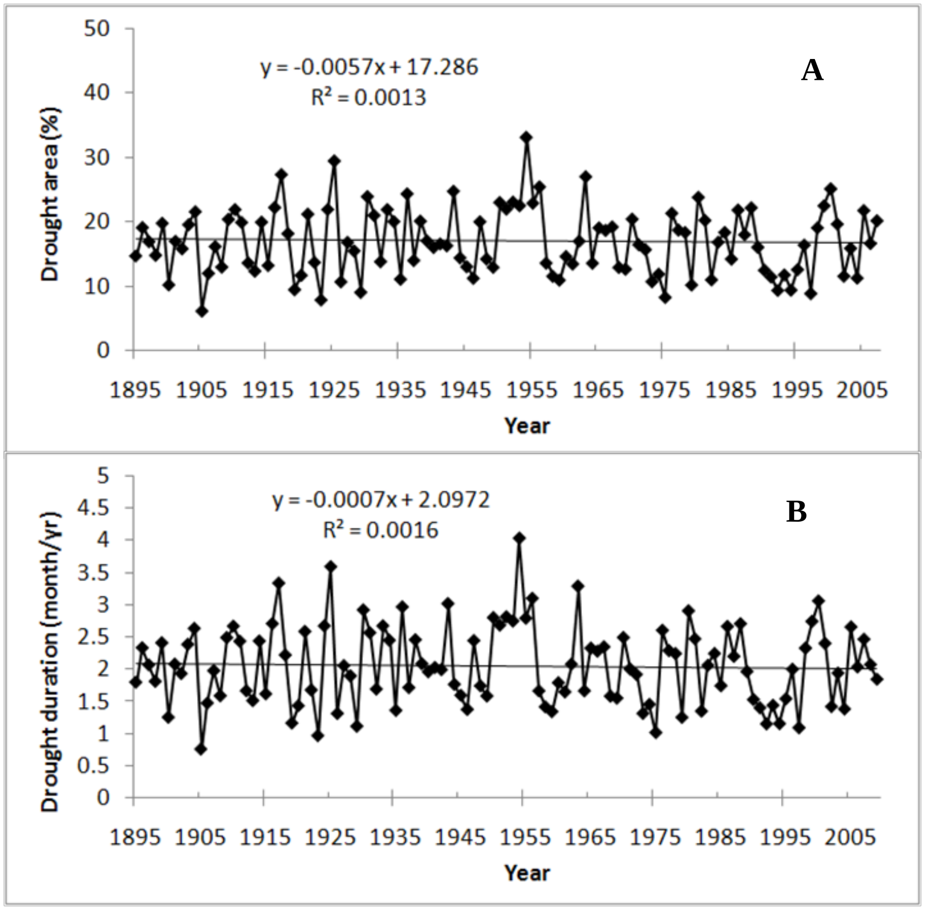


Figure 6 Percent of drought area (%, when 1-month SPI < -1.0, A) and mean drought duration (months, when 1-month SPI <- 1.0, B).

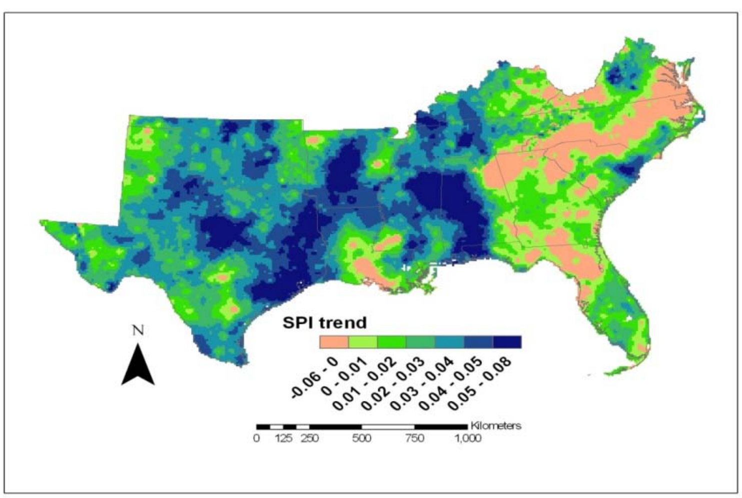


Figure 7 Changing trends for 12-month SPI during 1895-2007 (per decade).

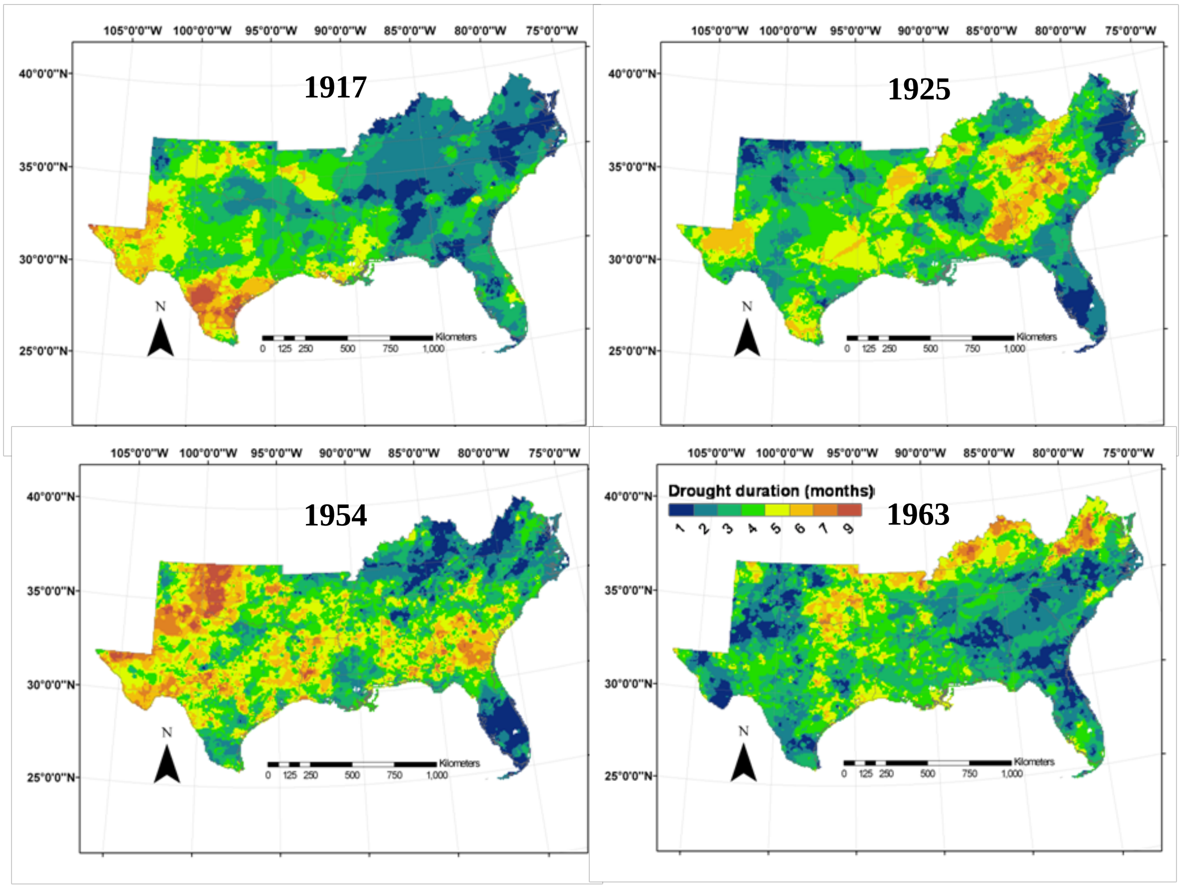


Figure 8 Spatial distribution of drought duration (months) calculated based on 1-month SPI in the 4 driest years (1917, 1925, 1954 and 1963).

Drought impacts on net primary productivity

NPP did not exhibit significant change trend but varied significantly among years during 1895-2007 with changes in climate, precipitation only, and temperature only (Figure 9). NPP ranged from 0.84 Pg C in 1980 to 1.30 Pg C in 1957 under the climate change scenario, while it ranged from 0.94 Pg C in 1917 to 1.34 Pg C in 1977 under the precipitation change scenario. Although change in air temperature has a larger influence on the magnitude of NPP, the inter-annual variation was primarily influenced by precipitation pattern. During the dry years, the effects of temperature and precipitation changes were additively enhanced to lower down NPP under the combined scenario. This resulted in significantly reductions in NPP in very dry years; but in very wet years, NPP under the combined scenario was primarily influenced by changing precipitation. By selecting 10 most dry and wet years during 1895-2007 based on drought duration and area, we found that annual total NPP in the extreme dry year (1.03 Pg C) was significantly ($P < 0.05$) lower than that in both normal (1.10 Pg C) and extreme wet (1.18) years (Table 2). Due to large spatial variation, however, the regional total NPP might underestimate the actual impacts of different drought intensities on NPP. During the four driest years, we found that NPP could be reduced by up to 40% in some areas of the SUS, while NPP could also increase over 40% in the wet areas in these years (Figure 10), which could offset most of the reduced NPP induced by drought. Due to the difference in spatial distribution of different drought events (Figure 86), the highest NPP reduction varied across space; however, reductions generally occurred either in the western or in the eastern part of the SUS.

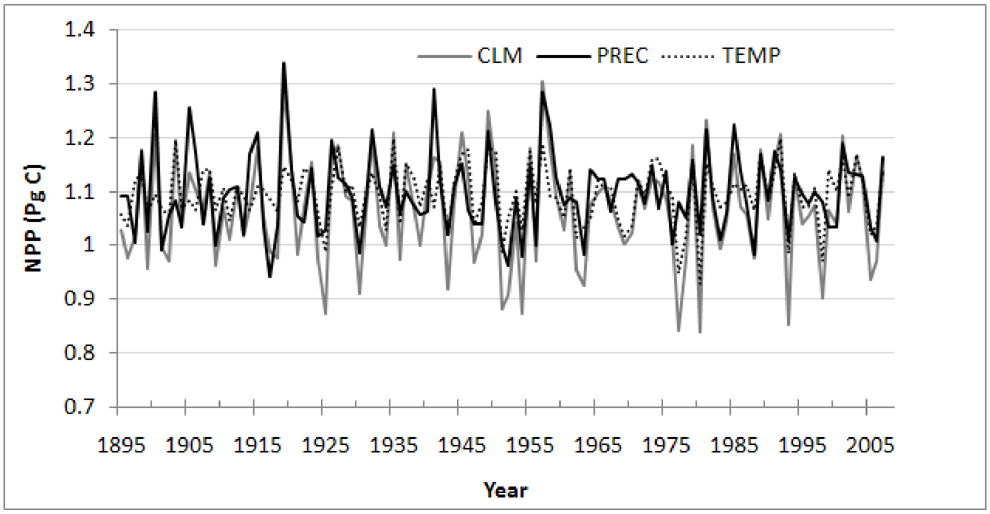


Figure 9 Inter-annual variations in NPP under changing climate (CLM), precipitation only (PREC) and temperature only (TEMP) scenarios.

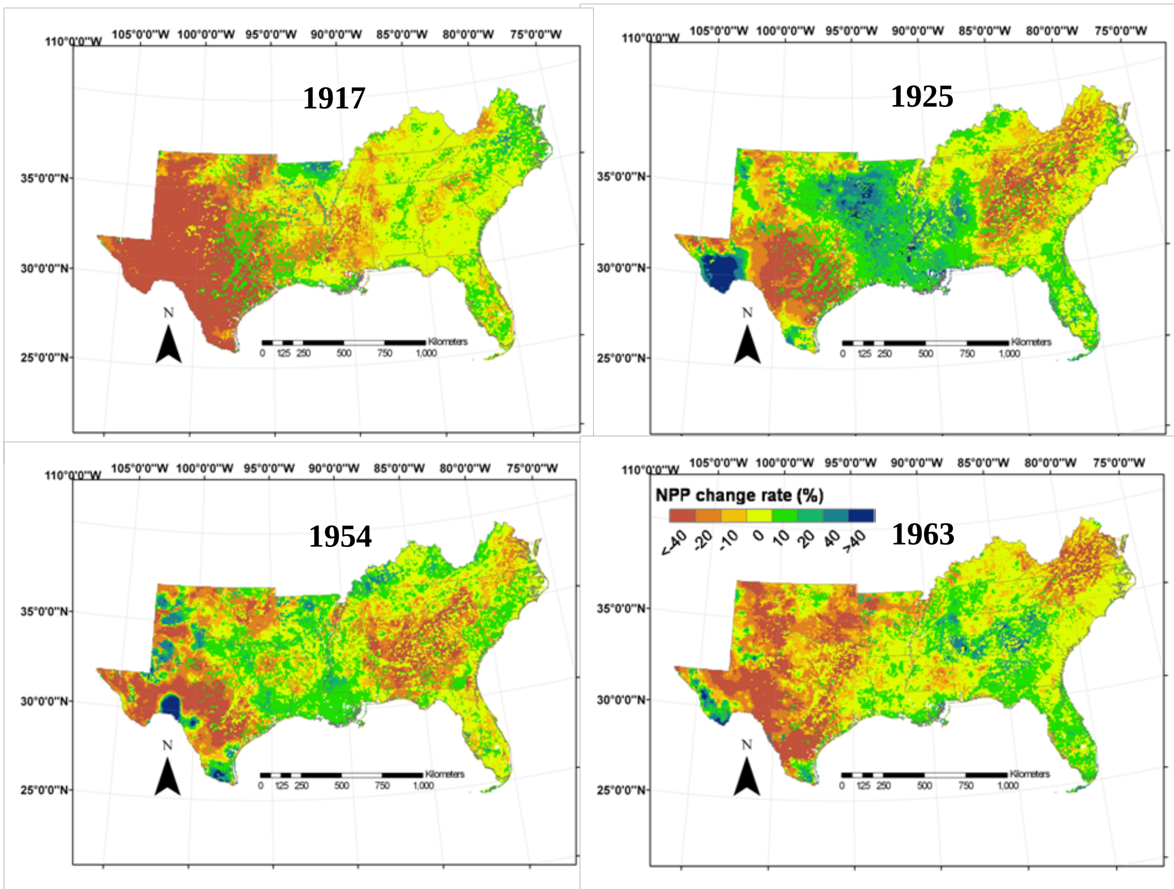


Figure 10 Changes of NPP relative to the mean NPP during 1895-2007 in the 4 driest years (1917, 1925, 1954 and 1963).

Table 2 Annual mean NPP and NCE under precipitation only scenario for selecting 10 driest, 10 wettest and normal years based on drought area and drought percent data.

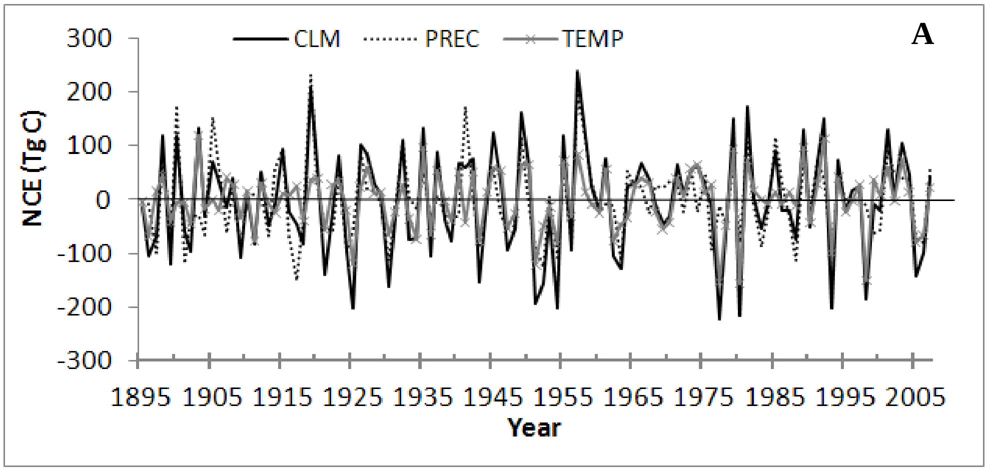
Variables	Extreme dry	Extreme wet	Normal
NPP (Pg C yr^{-1})	1.03 (0.06) a*	1.18 (0.07) b	1.10 (0.07) c
NCE (Tg C yr^{-1})	-2.30 (0.06) a	2.71 (0.07) b	-0.06 (0.07) c

*Different letters within rows indicated significant difference at $P < 0.05$. Values in the parenthesis are standard deviations.

Drought impacts on carbon storage in the SUS

NCE varied significantly among years under climate, precipitation only and temperature only scenarios (Figure 11A). NCE ranged from -149 Tg C yr^{-1} in 1917 (one of the driest years) to 233 Tg C yr^{-1} in 1919 (the wettest year) under the changing precipitation scenario. A continuous C source was found during the period of 1951-1956 (a period with continuous dry years). Changes in climate, precipitation, and air temperature have collectively resulted in about 0.33, 0.036, and 0.14 Pg C being released into the atmosphere during 1895-2007 (Figure 11B). The interactions among precipitation and air temperature, have induced an emission of 0.15 Pg C, implying a significant enhanced drought impact induced by temperature changes. The drought impacts induced both by precipitation and temperature could be ~0.19 Pg C. Although extreme drought events were reported during 1970-2007, precipitation changes in this period resulted in an important C sink (0.018 Pg C). Regional mean precipitation has an increasing trend; however, the SUS was still a C source, which might due to the large spatial difference in precipitation distribution.

Although precipitation change resulted in a relatively smaller C emission compared to climate change (i.e., combined changes in precipitation and temperature), large spatial variations were found across the SUS. A large part of the eastern SUS was a C source, while most areas in the west are C sinks (Figure 12A). The highest C sinks and sources were ~0.079 and -0.27 Tg C per grid cell (8×8 km^2), respectively. The western SUS is traditionally a semi-arid region with annual precipitation ranging from 200 to 800 mm. Precipitation is the major limiting factor for NPP and C sequestration. Climate change has resulted in a large increase and slight decrease in precipitation in the western and eastern SUS (Tian et al. 2010a). The combined changes in precipitation and temperature could significantly enhance C emissions in the eastern SUS and expand the distribution extent of C sources (Figure 12B). This implied that temperature change induced drought was significantly augmented in the eastern SUS. Most of the central SUS were also C sources as impacted by climate change.



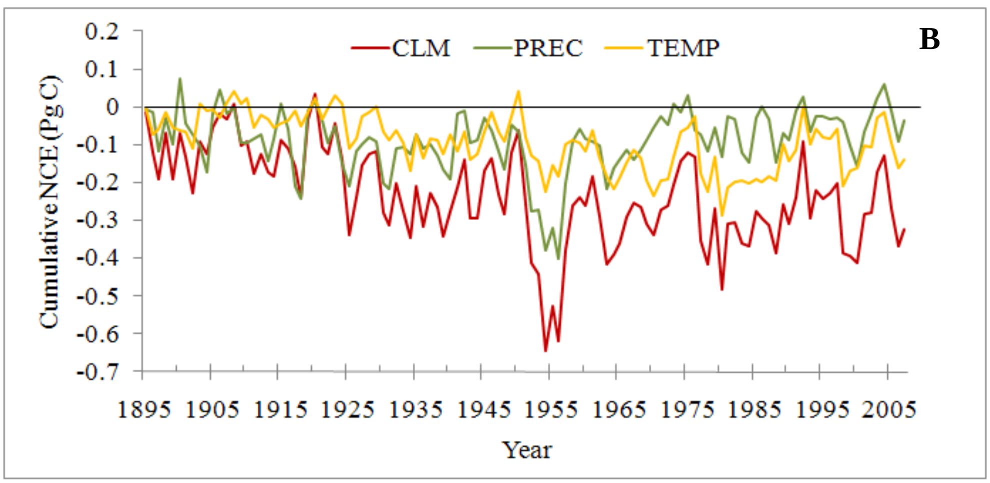


Figure 11 Inter-annual variation (A) and accumulative (B) net carbon exchange (NCE) resulting from changes in climate (CLM), precipitation (PREC), and temperature (TEMP).

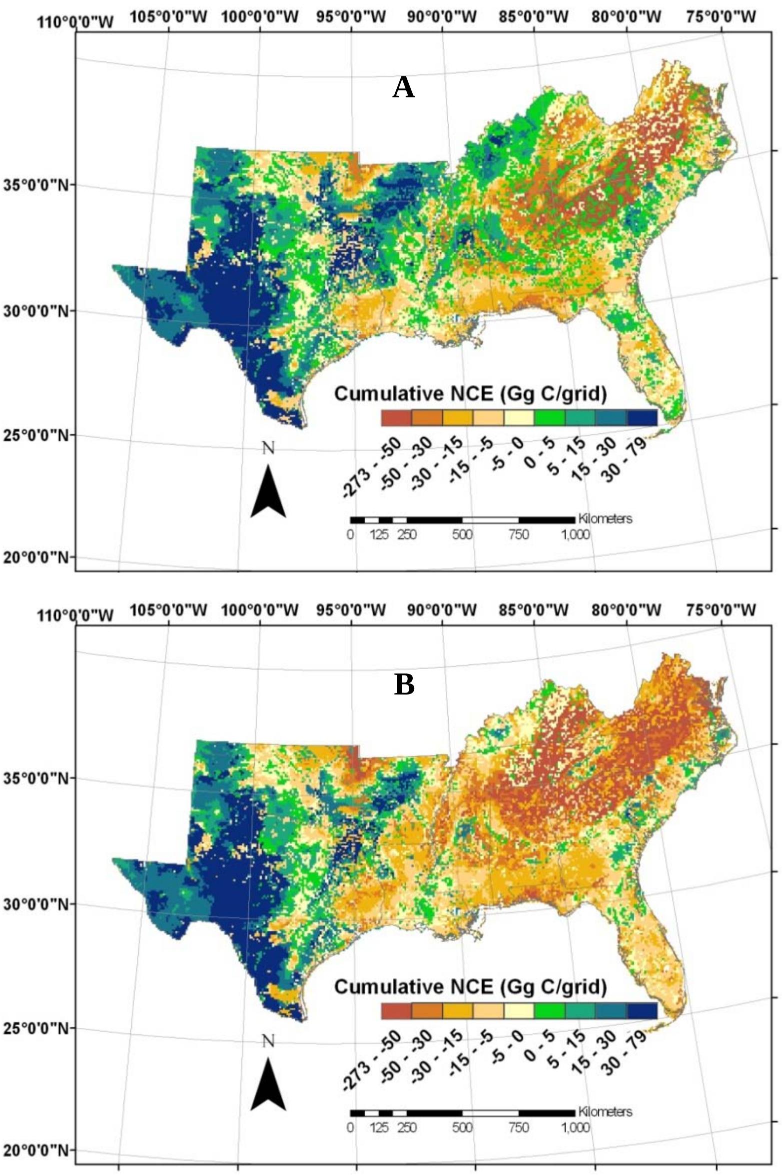


Figure 12 Spatial distribution of cumulative NCE (Gg C grid^{-1}; 1 Gg = 10^9 g; grid size: 8 × 8 km^2) during 1895-2007 under changes in precipitation (PREC, A) and climate (CLM, B) scenarios.

Table 3 Accumulated changes in carbon storage (Tg C) for different vegetation types as influenced by drought and climate change during 1895-2007.

Scenario	Upland forest	Shrubland	Grassland	Wetland	Cropland
CLM	-414.47	201.91	3.84	-34.80	-1.76
PREC	-198.47	187.55	8.13	-32.21	1.75
TEMP	-101.35	-20.16	-2.7	-11.70	-2.19
Interactive effect*	-114.66	34.52	-1.58	9.11	-1.32
Drought effect®	-313.13	222.07	6.55	-23.10	0.43

* Interactive effect was calculated as CLM-PREC-TEMP, indicating the interactive impacts of changes in temperature and precipitation on carbon storage; ®Drought effect was calculated as PREC + interactive effect, indicating drought effects from both precipitation and temperature changes.

Responses of different vegetation types to drought

Different vegetation types differ in their responses to drought and climate change due to differences in drought tolerance and environmental conditions. Precipitation change (PREC) induced a C source by 198.47 and 32.21 Tg C in forest and wetland (including grass and forest wetlands) ecosystems, respectively, while a C sink by 187.55, 8.13 and 1.75 Tg C in shrubland, grassland, and cropland ecosystems, respectively (Table 3). The grassland and shrubland are located in the western SUS, where drought intensity and duration were greatly decreased during 1895-2007 (Figure 7). Forest and wetland are primarily located in the eastern SUS or the coastal regions, where drought intensity and duration were found to increase (Figure 7). Drought impacts on C sources for forest were enhanced by the interactive effects between precipitation and temperature changes, while reduced for wetland. The interactions between precipitation and temperature changes enhanced C sinks in shrubland, while reduced in grassland and cropland. These findings suggested that there were significantly different responses of various vegetation types to global changes. Further studies are needed to specifically identify the mechanisms controlling these differences.

DISCUSSION

Drought conditions in the SUS

Even though the SUS does not experience multiple years of extreme droughts, short periods (a year to a few years) do occur when precipitation reductions induce serious stresses on water supply, and thus drought events and adverse impacts have often been reported and recorded. However, we did not find an obvious trend in drought duration and intensity during 1895-2007 based on the SPI. In contrast, we found a slight decreasing trend in drought intensity. Although reports from IPCC (IPCC 2007) and the United States climate report (Karl et al. 2009) indicated that it is likely that drought intensity, frequency, and duration will increase in the future for the SUS, we did not find this trend in the historical data. The IPCC (2007) and US climate report (Karl et al. 2009) predicted a fast increase in air temperature, which would result in a higher ET thereby reducing available water. However, we did not find an obvious increase in air temperature in the SUS as a whole during 1895-2007 (Figure 1B). Although temperature increased from 1895 to the mid-1950s and from mid-1950s to 2007, there was a sharp decrease in air temperature in the middle of 1950s, which resulted in no significant change in air temperature for the whole time period.

Although drought intensity and duration due to global climate change were not found to increase, drought impacts may be intensified by increased water use demands for human and land use changes from natural vegetation to irrigated cropland (McNulty et al. 2007). We found that irrigated and fertilized croplands have a higher ET and runoff compared to those unmanaged croplands and most of the natural ecosystems (see Chapter 4), resulting in less water storage in the terrestrial ecosystems. In addition, the fast-growing population, industrialization, and urbanization caused by population migration and economic growth could significantly increase water demands. For example, almost a quarter of total water use in Georgia is for public use (http://ga.water.usgs.gov/projects/projectwateruse.html). Thus, under the stresses of significant increases in water demands, a decrease in precipitation in 2007 caused a serious water shortage and thus a drought emergency in Georgia (http://www.georgia.gov), even though precipitation in 2007 for Georgia was not extremely low (Slightly lower than the long-term mean precipitation; Figures 1 and 6; Tian et al. 2010a). Further studies are needed to explore how human activities have influenced the drought intensity and duration in the SUS.

Relationships between drought and ecosystem function

Drought has been reported to significantly reduce NPP and C storage worldwide (e.g., Elliott and Swank 1994; Ciais et al. 2005; Zhao and Running 2010; Asner and Alencar 2010; Zeng and Qian 2005; Hanson and Weltzin 2000; Klogs et al. 2009). Due to the difficulties in characterizing and monitoring drought at larger scales, the assessment of drought effects on NPP and C storage generally relies on modeling. Therefore, performance of the process-based models directly influences the estimation results. DLEM model comprehensively includes most of the hydrological processes and a series of sophisticated algorithms required to address the impacts of water stress on plant physiological, bio- chemical and physical characteristics. The model has been extensively evaluated based on observational, experimental, and regional inventory data. Our simulation results indicated that drought could decrease NPP up to 40% resulting in a significant reduction in C storage in the SUS. Although drought intensity and duration have not been found to increase during 1895-2007, extreme drought events could significantly reduce regional NPP and C storage. Through collection of observational data, Klos et al. (2009) found that the mean annual growth rate of pines decreased significantly with increasing drought intensity, while it only decreased slightly for oaks and mesophytes in the SUS. Further, the growth rate could be reduced by ~ 20% for pines during the extreme drought years. Our simulation results also indicated a reduction in NPP of 10-30% for pine forests in the southeastern part of SUS (e.g., extreme drought in 1954, Figure 10) and 10-20% for broadleaf forests (e.g. extreme drought in 1925 and 1963) in the northeastern part. Although our simulation results have been evaluated against field experimental, observational, and inventory data, many uncertainties might still exist due to lack of observation data to directly validate the extent of drought impacts on C storage and NPP in a large region like the SUS.

Results from this study indicated that the SUS acted as a large C source in the extreme dry years and a small C source in normal years, while a C sink in extreme wet years. This suggested that water is still one of the most important limiting factors to C sequestration capacity in the SUS even though this region has a relatively wet climate. Although the SUS is characterized as a random and occasional drought region (Hanson and Weltzin 2000; Karl et al. 2009), the accumulative drought impacts on C storage could be very large. We found that drought could induce a C emission as high as 0.27 Pg C in some regions in the northeastern SUS during 1895-2007 (Figure 12A). It is notable that about half of terrestrial ecosystems in the eastern SUS were C sources under the impacts of changing precipitation and air temperature, while most parts of the western SUS were C sinks. The eastern SUS is traditionally a water-rich region (McNulty et al. 2007; Sun et al. 2008). Most of the vegetation types in this region have adapted to high precipitation and soil moisture conditions. Although no significant changing trend in precipitation during 1895-2007 was found in this region (Karl et al. 2009; Seagers et al. 2009; Tian et al. 2010a), the alternating high and low precipitation patterns could disrupt the stability of the plant ecosystems, thus causing large amounts of C emissions. In contrast, the western SUS is traditionally a low precipitation region and shrubland and grassland are the major land cover types. These vegetation types have adapted to dry conditions, causing a significant increase in plant growth and large C sinks in this region.

Interactions of drought with other factors

Drought could interact with other factors (e.g., atmospheric CO_2 concentration, air temperature, fires, and pests) to augment or attenuate its impacts on plant productivity and C sequestration (Rogers and Vint 1987; Norby et al. 1999; Olano and Palmer 2003; Hanson and Weltzin 2000; Houghton 2004; Luo et al. 2008). Results from this study suggested that drought impacts were significantly augmented by the changes in air temperature. The interactive effects between air temperature and precipitation could induce a C emission of 0.15 Pg during 1895-2007 in the SUS. This number is greater than the individual effect from changing precipitation (-0.036 Pg C) and air temperature (-0.14 Pg C), and about half of the effects from combined changes in precipitation and air temperature (-0.33 Pg C). This suggested that global warming could impose a significant influence on drought intensity and frequency. In addition, we also found that these interactive effects on C storage were different for various vegetation types.

Drought impacts on C sources for forest were enhanced by the interactive effects between precipitation and temperature changes, while reduced for wetlands. However, this interaction enhanced C sinks in shrubland and reduced in grassland and cropland. Further studies are needed to explore the underlying mechanisms controlling the interactive impacts among drought, global warming, and other environmental factors such as atmospheric CO_2 concentrations, N deposition and land use change (Norby et al. 1999; Hanson and Weltzin 2000; Luo et al. 2008; Bell et al. 2010).

CONCLUSION

Based on the SPI, we characterized drought intensity and duration in the SUS during 1895-2007. No significant changes in drought intensity and duration were found for this time period. However, we found that the area of the SUS experiencing extreme high rainfall events appeared to be increasing, which might imply an increased flooding frequency. NPP was greatly reduced during dry years and could induce more than 40% decrease in some areas. Although no significant trend in drought intensity and duration was found during 1895-2007, changes in precipitation pattern resulted in a C emission of 0.036 Pg and showed great spatial variability. Changes in precipitation induced C sinks in most of the western SUS and C sources in most of the eastern SUS. Drought impacts on NPP and C storage could be enhanced by changes in other climatic factors. Therefore, the future studies on drought impacts should also consider the interactive effects of drought events and other climatic factors such as solar radiation and air temperature.

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Chapter 4

Changes of Water Resources as Influenced by Changes in Climate, Land Use, and Management in the Southern United States

ABSTRACT

Climate and land use have changed greatly in the Southern United States (SUS) during the 20th century, which could significantly change the water cycles in this region. Changes of water yield (i.e., amount of water leaving an ecosystem) and evapotranspiration (ET) in response to multiple global changes have been observed on both site- and watershed- scales in this region. Due to lack of direct and accurate measurements at appropriate scales for predicting impacts of individual factors, our understanding of the regional to global ET is therefore generally dependent on modeling. In this study, we explored the impacts of climate, land use, and land management changes on ET and water yield during 1895-2007 using the widely applied global biogeochemical model (Dynamic Land Ecosystem Model, DLEM). The modeled water yield and ET were evaluated against field observational data and shown to agree well with observations. The simulation results indicated that ET and water yield increased by 3.5 and 6.7 mm per decade, respectively during this period. Compared with changes in land use and land management, climate variability had the largest contributions to the increase in ET and water yield. Land use change caused a decrease in ET and an increase in water yield. In contrast, land management caused an increase in ET and a decrease in water yield. Although the overall increase in ET and water yield were not significantly large, the much larger spatial and inter-annual variation could

result in significant impacts on ecosystem structure and function especially in the dry regions of the SUS.

Key words: Evapotranspiration (ET); water yield; climate change; land use change; land management; Dynamic Land Ecosystem Model (DLEM); Southern United States

INTRODUCTION

Evapotranspiration (ET) is one of the most important components of water balance which indicates water efflux that returns to the atmosphere from land surfaces, and water yield indicates the water amount leaving an ecosystem or watershed, which represents the potential water availability to living organisms in an ecosystem. Therefore, qualifying ET and water yield is essential to the research on the global water cycle. ET accounts for more than 60 percent of annual precipitation at the global scale (Vörösmarty et al. 1998; Sun et al. 2002). Environmental factors such as climate, soil texture, land use types, land management, and atmospheric CO_2 concentration could greatly affect the ET and the associated water-vapour flux partitioning at the land surface, and all freshwater flows and discharges to the sea (Shibuo et al. 2007). Climate change and land use change are two of the most direct influencing factors which could alter both magnitude and spatial distribution patterns of ET and water yield (Dow and DeWalle 2000; Sun et al. 2002). Impacts of climate change on ET have been studied extensively through both field experiments and empirical models. Climate change was suggested as the major factor determining the general spatial distribution pattern of ET and water yield (Dow and DeWalle 2000; Mu et al. 2007). Land use change was also found to be one of the most important factors that controlled water resources on both local and global scales during the last century (Hutjes et al. 1998; Vörösmarty et al. 2000; Costa et al. 2003; Mu et al. 2007). Some research suggests that the consequences might outweigh those from climate change (Sala et al. 2000; Vorosmarty et al. 2000; DeFries and Eshleman 2004). Climate and land use change will also interact to affect ET and water yield through their interactive effects on gross primary productivity and vegetation redistribution. Results from climate models indicated that land use change could affect regional climate through biogeochemical and biogeophysical processes (Feddema et al. 2005; Gordon et al. 2005; Pielke et al. 2007), which then could influence regional water cycles. In the meanwhile, climate change could alter the regional distribution of land use types through gradually changing vegetation species and their biogeographical boundary (Scheffer et al. 2001; Kelly and Goulden 2008).

Our current understanding of land use change effects on hydrology is primarily derived from controlled experimental manipulations of the land surface and climate condition, coupled with pre- and post-manipulation of hydrological processes (Bosch and Hewlett 1982; Andréassian 2004; Sun et al. 2008). The effects of land use change have also been studied using empirical models in the past decades (e.g., Lu et al. 2003; Andréassian 2004; Ice and Stednick 2004; Sun et al. 2005). The lack of direct and accurate measurements at appropriate scales has limited our efforts to understand ET and water yield processes over large scales. Our understanding of regional to global ET is therefore generally dependent on model simulations which are subjected to much uncertainty (Destouni et al. 2008). Modeling capabilities for evaluating and predicting hydrological consequences of climate and land use change at multiple scales have progressed at a rapid rate in recent years largely due to technological improvements in data collection and computing capabilities (DeFries and Eshleman 2004). However, there still are many factors limiting the modeling accuracy and applicability on a large scale as noted by Sun et al. (2005).

The climate in the Southern United States (SUS) is warm and wet, with mild winters and high humidity, compared with the rest of the continental United States (Karl et al. 2009); However, due to increased water use demands and climate variability, this water-rich region has experienced severe water stress (McNulty et al. 2007), for example, severe droughts in 2002 and 2007. The precipitation in the SUS displays a great spatial variability with an increasing gradient of precipitation from west to east and great inter-annual variability alternating extreme high and low precipitation patterns. Land use types have also changed greatly during the last century due to rapid economic development and increasing population (Chen et al. 2006a; Wear 2002). Associated with land use change, land management (including cropland irrigation, fertilization, and forest harvest) in this region is also experiencing enormous changes. Impacts of land use, land management, and climate change on ET and water yield at the field or watershed scales have been intensively studied or monitored in this region (e.g., Swank and Crossley 1988; Ice and Stednick 2004; Lu et al. 2003; Sun et al. 2002, 2005). Compared to land use and climate change, less attention has been paid to exploring how land management affects water yield and ET.

In our previous work (Tian et al. 2010a), we reported spatial and temporal patterns of ET in the SUS as influenced by combined multiple environmental factors. In this study, we will further examine the effects of climate, land use change, and land management on both ET and water yield through incorporating previous studies and new simulation results using the process-based global biogeochemical and hydrological model (Dynamic Land Ecosystem Model, DLEM. Chen et al. 2006b; Ren et al. 2007a; Zhang et al. 2007; Liu et al. 2008; Tian et al. 2010a, b, c).

Specifically, our objectives are to: 1) estimate spatial and temporal ET and water yield patterns in the SUS under the impacts of land use change, and land management, and climate change; and 2) identify the contributions of these factors to ET and water yield.

METHODS

Data description

Model input data in this study include annual land use maps, daily climate data, annual N fertilizer inputs to cropland, irrigation maps, and non-changed soil property and topography data. The sources, development, and changing patterns of these data were described in detail by Tian et al. (2010a, c). Geospatial data were scaled to the same spatial scale (8 km × 8 km) to drive the DLEM model.

The model calibration data were collected from various sources as described in Tian et al. (2010a). The data for validation were collected from AmeriFlux sites (http://public.ornl.gov/ameriflux/) and the Long Term Ecological Research (LTER) network (http://www.lternet.edu/).

Model description

DLEM is a highly-integrated process-based terrestrial ecosystem model that simulates daily C, water and N cycles driven by the changes in atmospheric chemistry (ozone and N deposition), climate, atmospheric CO_2 concentration, land-use and land-cover types, and disturbances (e.g., fire, hurricane, and harvest). DLEM is well-documented and evaluated and has been extensively used to evaluate terrestrial C, water and N cycles over China, Monsoon Asia, the continental United States, and North America (e.g., Tian et al. 2005, 2008, 2010a, b, c, d; Chen et al. 2006b; Ren et al. 2007a, b; Liu et al. 2007, 2008; Zhang et al. 2007). The DLEM model structure and major processes were described in these publications. In this study, we describe the algorithms for simulating ET and water yield in DLEM. The major processes for simulating water cycles in DLEM are listed in Figure 1.

Partition of precipitation: The allocation of precipitation (P) into rain or snow is based on air temperature (Wigmosta et al. 1994):

$$
\begin{aligned}
P_s &= P && T_a \le T_{\min} \\
P_s &= \frac{T_{\max} - T_a}{T_{\max} - T_{\min}} && T_{\min} < T_a < T_{\max} \\
P_s &= 0 && T_a \ge T_{\max} \\
P_r &= P - P_s
\end{aligned}
$$

Where, P_r and P_s are the water equivalent depths of rain and snow, respectively; T_{min} is a threshold temperature below which all P is in the form of snow, and T_{max} is a threshold temperature above which all P is rain; Between T_{min} and T_{max}, P is assumed to be a mix of rain and snow; T_a is the daily average temperature. -1.1°C and 3.3°C are used as T_{min} and T_{max} (Wigmosta et al. 1994).

Canopy interception of rain and snow: Precipitation is either intercepted by the canopy or falls to the ground as throughfall and stemflow. The minimum interception rate is set as 20% of precipitation and should be less than the maximum canopy water holding capacity (Bonan 1996).

$$
\begin{aligned}
P_{can,I} &= \min(0.2 \times P, \quad W_{can,\max} - W_{can}) \\
W_{can,\max} &= p_{\mathrm{int}\,c} \times LAI
\end{aligned}
$$

Where, $W_{can,max}$ is the canopy water holding capacity; p_{intc} is the coefficient of $W_{can,max}$; LAI is the projected leaf area index; W_{can} is canopy water content (mm water). Same parameters are used for rain and snow interception.

Canopy sublimation, evaporation and transpiration: Vegetation canopy surface is divided into wet and dry fractions according to Dickinson et al. (1993).

$$f_{can,wet} = \left[\frac{W_{can,rain}}{W_{can,\max}} \right]^{2/3} \leq 1$$

Where, $f_{can,wet}$ is the wet fraction of the canopy. $W_{can,rain}$ and $W_{can,max}$ are total rainfall intercepted by the canopy and the maximum canopy water holding capacity, respectively. The intercepted water by the canopy is estimated with the same methods described by Bonan (1996).

The model calculates evaporation and transpiration independently for wet and dry surfaces. For dry canopy surface, canopy evaporation is assumed to be 0. Plant transpiration for dry surface is thus calculated using a Penman-Monteith approach (Wigmosta et al. 1994):

$$E_t = \frac{\Delta \times R_{sw,abs} + \rho \times c_p \times (e_s - e) / r_a}{\lambda \times [\Delta + \gamma \times (1 + r_c / r_a)}$$

Where, E_t is transpiration rate (mm·m^{-2}·s^{-1}), Δ is the slope of saturated vapor pressure-temperature curve, $R_{sw,abs}$ is net short-wave radiation density, ρ is the density of moist air, c_p is the specific heat of air at constant pressure, e_s is the saturation vapor pressure, e is the vapor pressure, r_a is the aerodynamic resistance to vapor transport, λ is the latent heat of vaporization of water, γ is the Psychrometric constant, and r_c is the canopy resistance to vapor transport.

The potential canopy evaporation from wet surfaces can be estimated by setting r_c equal to zero (Wigmosta et al., 1994):

$$E_p = \frac{\Delta \times R_{sw,abs} + \rho \times c_p \times (e_s - e) / r_a}{\lambda \times [\Delta + \gamma]}$$

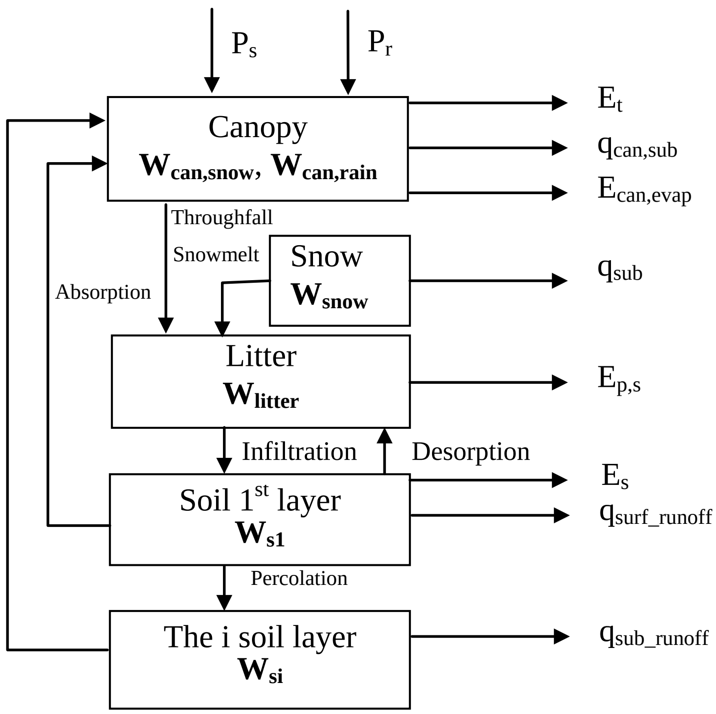

Figure 1. Structure of the hydrological module in DLEM. Note: The soil is represented by i layers: one litter layer (or above-ground water table) with varied depth, and other i -1 mineral soil layers with different depths. Precipitation is separated as snow (P_s) and rain (P_r). Canopy intercepts some of P_s and P_r into canopy snow storage ($W_{can,snow}$) and rain storage ($W_{can,rain}$), respectively. The intercepted water eventually evaporates ($E_{can,evap}$) or sublimates ($q_{can,sub}$) to the air. The remaining P_s and P_r enter into the ground snowpack (W_{snow}) and litter layer (W_{litter}), respectively as throughfall. When soil moisture in the first mineral layer (W_{s1}) exceeds the saturated soil water content, the excess water forms surface runoff ($q_{surf,runoff}$). The percolation from top soil layers to lower soil layers forms sub-surface runoff ($q_{sub,runoff}$).

Canopy ET includes three components: snow sublimation, wet surface evaporation, and transpiration. Sublimation is assumed to have the highest priority, followed by evaporation and transpiration. This means that the total daytime length ($t_{daytime}$, second) available for ET is first allocated for snow sublimation, then for wet canopy evaporation, and finally for transpiration.

Intercepted snow on canopy surfaces will be sublimed to the atmosphere (Coughlan and Running 1997). The left canopy snow that could not be sublimed within a day will enter into the soil surface snow pools.

$$q_{can,sub} = \min(W_{can,snow}, \frac{R_{sw}}{\lambda + k})$$

$$t_{sub} = \left(1 - \frac{q_{can,sub} \times (\lambda + k)}{R_{sw}}\right) \times t_{daytime} \quad for\ W_{can,snow} < \frac{R_{sw}}{(\lambda + k)}$$

$$t_{daytime} = 0.0 \quad for\ W_{can,snow} \geq \frac{R_{sw}}{(\lambda + k)}$$

Where, R_{sw} is the total daily incident short wave radiation (KJ m^{-2} day^{-1}); λ is latent heat of vaporization of water (KJ mm^{-1} m^{-2}); κ is latent heat fusion (KJ mm^{-1} m^{-2}); t_{sub} is the left available time after snow sublimation, which is equal to the daytime length minus the time used for sublimation.

$$E_{can,evap} = \min(W_{can,rain}, f_{wet} \times E_p \times t_{sub})$$

$$if\ E_{can,evap} < f_{wet} \times E_p \times t_{sub} \quad t_{evap} = \left(1 - \frac{E_{can,evap}}{f_{wet} \times E_p \times t_{sub}}\right) \times t_{sub}$$

$$if\ E_{can,evap} \geq f_{wet} \times E_p \times t_{sub} \quad t_{evap} = 0.0$$

$$ET_{can} = E_{can,evap} + \min(AWC, (1 - f_{wet}) \times E_t \times t_{evap} + f_{wet} \times E_t \times t_{evap})$$

Where, t_{evap} is the left daytime length after canopy evaporation; t_{can} is the left daytime length that is available for canopy transpiration; $q_{can,sub}$ is the daily snow sublimation; $E_{can,evap}$ is the daily evaporation from wet canopy surfaces; ET_{can} is the total canopy evapotranspiration in a day (kg m^{-2}); $W_{can,rain}$ is the intercepted rain storage at canopy surface (mm); and *AWC* (mm) is the total available water from all soil layers except the litter layer.

Soil surface evaporation: Soil surface evaporation is influenced both by energy, atmospheric drivers and a maximum infiltration rate as a function of soil properties at a given soil moisture (Philip, 1957; Wigmosta et al., 1994). When soil is wet, it may be able to provide water to the surface at a rate equal to or greater than the potential evaporation demand. This condition is termed climate-controlled (Eagleson, 1978). As soil moisture is depleted, the rate of delivery falls below the potential evaporation rate and this condition is termed soil-controlled (Wigmosta et al., 1994). Using this approach, Wigmosta et al. (1994) and Entekhabi and Eagleson (1989) revised the soil water evaporation (E_s) equation as follows:

$$E_s = (1 - f_{snow}) \times \min(E_{p,s}, F_e, W_{litter} + W_{avevap})$$
$$F_e = S_e \Delta t^{1/2}$$
$$S_e = \left[\frac{8\phi K(\theta_s)\psi_b}{3(1+3m)(1+4m)}\right]^{1/2}\left[\frac{\theta}{\phi}\right]^{(1/2m)+2}$$

Where, $E_{p,s}$ is the simulated soil potential evaporation; F_e is the desorption volume; S_e is the soil sorptivity; and W_{avevap} is the available soil water that can be evaporated. We assumed that surface evaporation can only affect the top soil layer to a depth of 50 cm, and that minimum water content after evaporation approaches at wilting point (i.e., soil metric potential is -1500 kpa) in the 5-50 cm soil layer and zero in the top 5 cm soil (Agam et al., 2004; Wythers et al., 1999); Φ is soil porosity; $K(\theta_s)$ is saturated hydraulic conductivity (mm/hour); Ψ_b is air entry (bubbling) pressure (mm water); *m* is the pore size index; θ is the relative soil moisture; if the water table is higher than the soil surface (e.g., wetland and paddy land), E_s is equal to $E_{p,s}$; and Δt is the time step of evaporation (hour).

Simulation experiments and methods

To address the effects of climate change, land use change, and land management practices, we designed six simulation experiments (Table 1): 1) climate change only (CLM) where only climate data change over time and other environmental factors keep constant during the study period. The land use, atmospheric CO_2 concentration and nitrogen deposition data in 1895 were used as input data and do not change over time; 2) land use change and land management (LUC_man) where land use types and land management (irrigation and fertilization) will change over time and other environmental factors keep constant during the study period; 3) land use change with N fertilization to cropland (LUC_fer) where land use and N fertilizer amounts change over time; 4) land use change with irrigation practice (LUC_irr) where land use and irrigation change over time; 5) land use change without land management (LUC_noman) where only land use changes over time; and 6) Combined land use, climate and land management change (COMB) where land use types, climate and land management change over time. In these experiments, the effect of climate change (CLM) on ET and water yield is represented by the difference between COMB and LUC_man (COMB – LUC_man); land use change and associated land management effect (LUC_man) is represented by COMB – CLM; irrigation effect (IRR) is represented by LUC_irr – LUC_noman; fertilization effect (FER) is represented by LUC_fer – LUC_noman; land management effect (MAN) is represented by LUC_man – LUC_noman; and combined effect (COMB) is represented by adding all the individual impacts (CLM + LUC_noman + FER + IRR).

Table 1 The designed model simulation scenarios

Scenario	Climate	Land use	N fertilization	Irrigation
CLM	1895-2007*	1895	None	None
LUC_man	1895	1895-2007	1944-2007	2000®
LUC_fer	1895	1895-2007	1944-2007	None
LUC_irr	1895	1895-2007	None	2000
LUC_noman	1895	1895-2007	None	None
COMB	1895-2007	1895-2007	1944-2007	2000

*Indicates model input data over this period are used. "None" indicates that no N fertilizer or irrigation is used in this simulation scenario. ® Irrigation is considered but has no changes over time (i.e., only one time-period irrigation map in 2000 is used).

DLEM was first run to an equilibrium state using the mean climate data (averaged over 1895 - 1924) to develop the simulation baseline for C, N, and water pools. Then a spin-up simulation (90 years) using detrended climate data (i.e., climate data is randomly selected from 1895-1924) was conducted to eliminate system fluctuations caused by shift of simulation modes (i.e., from the equilibrium mode to the transient mode).

Model parameterization

The DLEM has been parameterized and applied in several regional studies both in China and the United States using various field observational data for all plant functional types, and then validated with independent field observational data, inventory data and regional estimates from other models and remote sensing tools (Chen et al., 2006b; Ren et al., 2007a, b, 2010; Zhang et al., 2008; Tian et al., 2008, 2010a). In the SUS, the model is specifically calibrated for all plant functional types (including temperate deciduous broadleaf forest, temperate evergreen needleleaf forest, mixed needleleaf and broadleaf forest, deciduous shrubland, C_3 grassland, C_4 grassland, grass wetland, forest wetland, and cropland) to simulate C and water cycles (Tian et al. 2010a).

Model evaluation and performance

DLEM-simulated monthly and annual ET and water yield were validated against field observation data from multiple observation sites and watersheds. The simulated and observed annual total streamflow at watershed 18 in the Coweeta Basin of the western North Carolina, USA (35.05^o N, 82.42^o W) for the period 1937 to 2007 did not differ significantly in any specific year (Figure 2A). Very good agreement was found in annual changes in streamflow in both wet and dry years. Time series data for monthly streamflow also displayed generally good agreement in both the timing and magnitude of seasonal changes (Figure 2B and C), with the biggest difference occurring during spring. We performed a t-test and results indicated that both annual and monthly patterns of observed data had no significant difference ($P > 0.4$), which implied that DLEM could capture the inter-annual and monthly patterns of runoff under climate change. The regression line between predicted and observed water yield for all months (Figure 2C) fell very close to the 1:1 line and had a R^2 of 0.53.

The DLEM-simulated ET was also compared with the observed ET in the AmeriFlux sites (Figure 3). Daily ET simulated with DLEM fit very well with observations. The fitted slopes fell into the range of 0.83-1.14 and had a high correlation coefficient except for the Duke Forest loblolly pine.

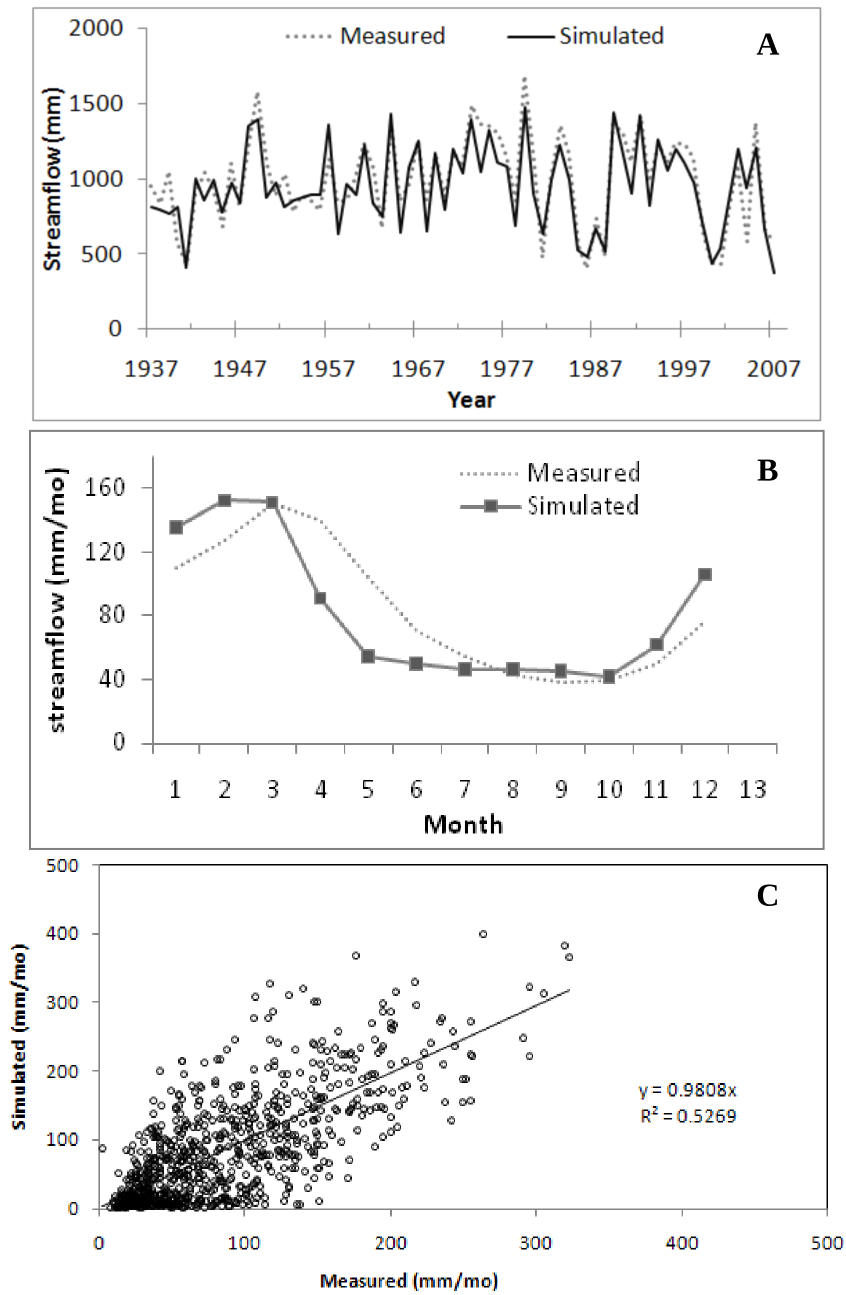

Figure 2. Comparison of simulated annual (A) and monthly (B, C) streamflow (mm mon^{-1}) against measured streamflow for watershed 18 at Coweeta Basin, North Carolina, USA (35.05° N, 82.42° W). Note: B is the mean monthly streamflow for all the observation years during 1937-2007, while C is the regression line between simulated and measured monthly streamflow for all the months.

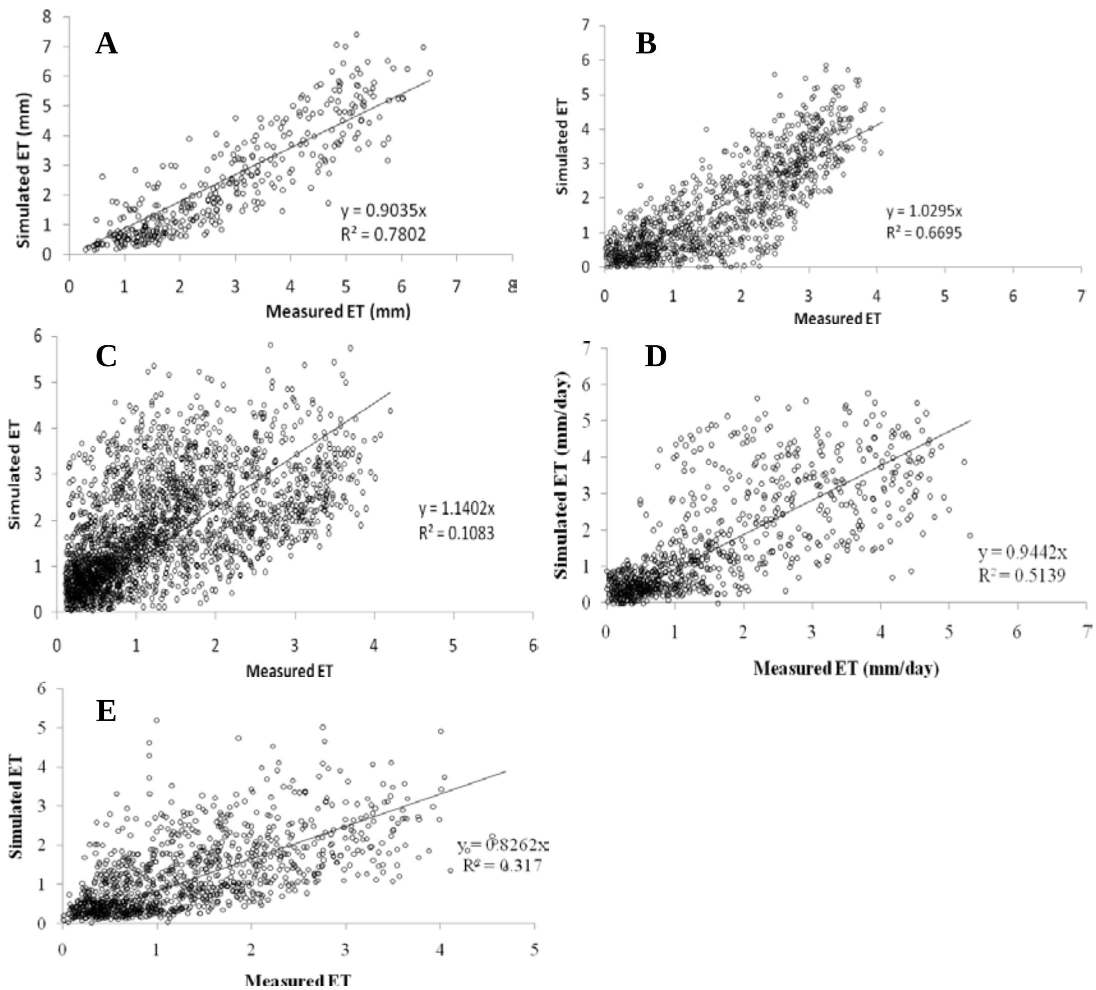

Figure 3 Evaluation of model-simulated against field-measured daily ET (mm day^{-1}) at sites with different land use types and climate. A: North Carolina Loblolly Pine (US-NC2, evergreen needleleaf forest) in 2005; B: Duke Forest Hardwoods (US-DK2, NC, USA, deciduous broadleaf forest) from 2003 to 2005; C: Duke Forest Loblolly Pine (US-DK3, evergreen needleleaf forest) from 2003 to 2005; D: Shidler Tallgrass Prairie (US-shd, OK, USA, C4 grassland) from 1998 to 1999; E: ARM SGP Main (US-arm, OK, USA, cropland) from 2003 to 2006.

RESULTS AND ANALYSIS

Changes in climate, land use, and land management

The mean annual air temperature in the SUS during 1895-2007 was 16.97 ^{o}C, ranging from 16 to 18 ^{o}C (Figure 4A). Although the global average air temperature increased during the study period, the mean annual air temperature in the entire SUS has not significantly changed. Air temperature displayed a slight increase during the first half century, decreased in the middle of the century, and then slightly increased during recent decades. The highest air temperatures were noted in the early 1950s. Precipitation showed an increase of 10 mm decade^{-1}, with a huge inter-annual variation ranging from 800 mm in 1954 to 1,316 mm in 1973 (Figure 4B). Precipitation in the SUS was generally increased except for some areas in the east (Figure 5). The largest increase in precipitation occurred in the central SUS with a rate of 40 mm decade^{-1}. Although the increase rate in precipitation in the arid and semi-arid regions was relatively small, it might have a significant impact on the regional water and C cycles. Urban and build-up land area has continuously increased from 1895 to 2007. Cropland area increased before 1940, then decreased until 1964 and remained relatively constant thereafter (Figure 4C). From 1895 to 2007, total forest land area decreased ~1.5%, cropland area increased ~3.3%, and wetland decreased slightly. Nitrogen fertilizer amount increased rapidly from 1945 to 1970s and then displayed a slight increase after 1975 (Figure 4D). About 2.5 tons per year of N were applied to the cropland during most recent 30 years.

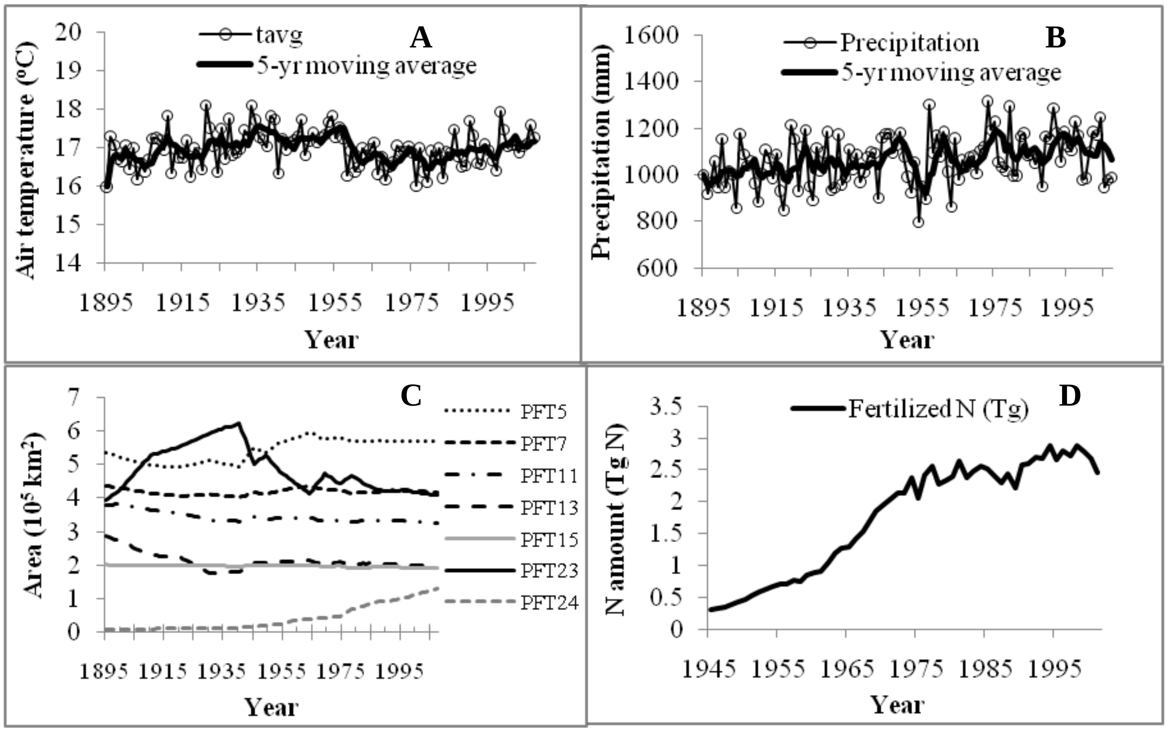

Figure 4 Inter-annual variations in climate, land use, and land management in the SUS. A) mean annual temperature (^{0}C) from 1895 to 2007; B) annual precipitation (mm); C) area of different biomes from 1895 to 2007 (PFT5: Temperate deciduous broadleaf forest; PFT7: Temperate evergreen needleleaf forest (including mixed forest); PFT11: deciduous shrubland; PFT13: C_3 and C_4 grassland; PFT15: grass and forest wetland; PFT23: cropland; PFT24: build-up land); and D) N fertilizer use (Tg N) for cropland from 1945 to 2001.

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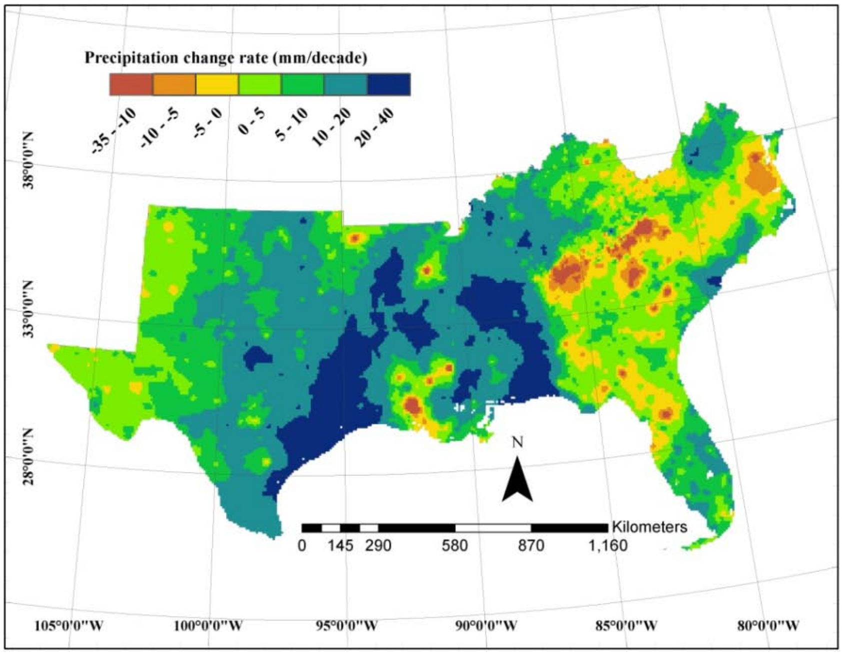

Figure 5 Precipitation change rate per decade (mm decade^{-1}) from 1895 to 2007 in the SUS.

Inter-annual variations in ET and water yield

ET has exhibited large inter-annual variations due to changes in climate, land use, and land management practices. From 1895 to 2007, combined changes (COMB) in climate, land use and land management have resulted in an increase of 35 mm decade^{-1} (0.35 mm per year) in ET, with a large inter-annual variation ranging from -84 mm in 1917 to 101 mm in 1949 (Figure 6A). ET increased before 1950s, followed by a slight decrease. This ET pattern was primarily determined by climate change, while land use change only altered the magnitude slightly. ET slightly increased during the study period due to the impacts of land use change and land management; this was attributed to the interactive effects of increased forest area, decreased cropland area after 1930s (Figure 4C), and enhanced crop growth due to N fertilizer use (Figure 4D).

Water yield has increased about 6.7 mm decade^{-1} during 1895-2007 under climate, land use and land management change (Figure 4B). Water yield showed an even larger variation than ET, ranging between -256 mm in 1954 and 227 mm in 1979. 1954 was characterized as one of the driest years in the history of SUS, which was associated with extremely high temperatures (Figure 6A, B), causing the lowest water yield and a serious drought in this year. Very low water yield was also found during drought periods of 1953-1956 and 2005-2007. Opposite to ET, water yield showed a decreasing pattern under the scenario of land use change and management.

Contributions of different factors to ET and water yield

To identify the impacts of individual factors and their contributions to ET and water yield, we conducted several simulations to isolate the impacts of each factor. The increased ET was primarily attributed to climate (67.1%), followed by irrigation (35.4%) and N fertilization (5.8%; Figure 7A). Land use change (land conversion and plant regrowth) negatively contributed to ET and resulted in a reduction of 8.4%. Overall water yield increased under climate, land use, and management changes (Figure 7B). Climate change and land conversion were positive contributors to water yield while irrigation and N fertilization were negative contributors. In DLEM, irrigated water to soil is assumed to not higher than the field capacity, so no runoff (or water yield) is produced during an irrigation event. In addition, DLEM assumes that irrigated water is from the local ecosystem. Irrigation enhanced ET through stimulating plant growth, so water yield (i.e., precipitation – ET) is decreased after irrigation. Increase in water yield was primarily attributed to climate change (59.8%), followed by land use change (LUC_noman, 6.8%). Decrease in water yield was attributed to irrigation (-28.7%) and fertilization (-4.7%).

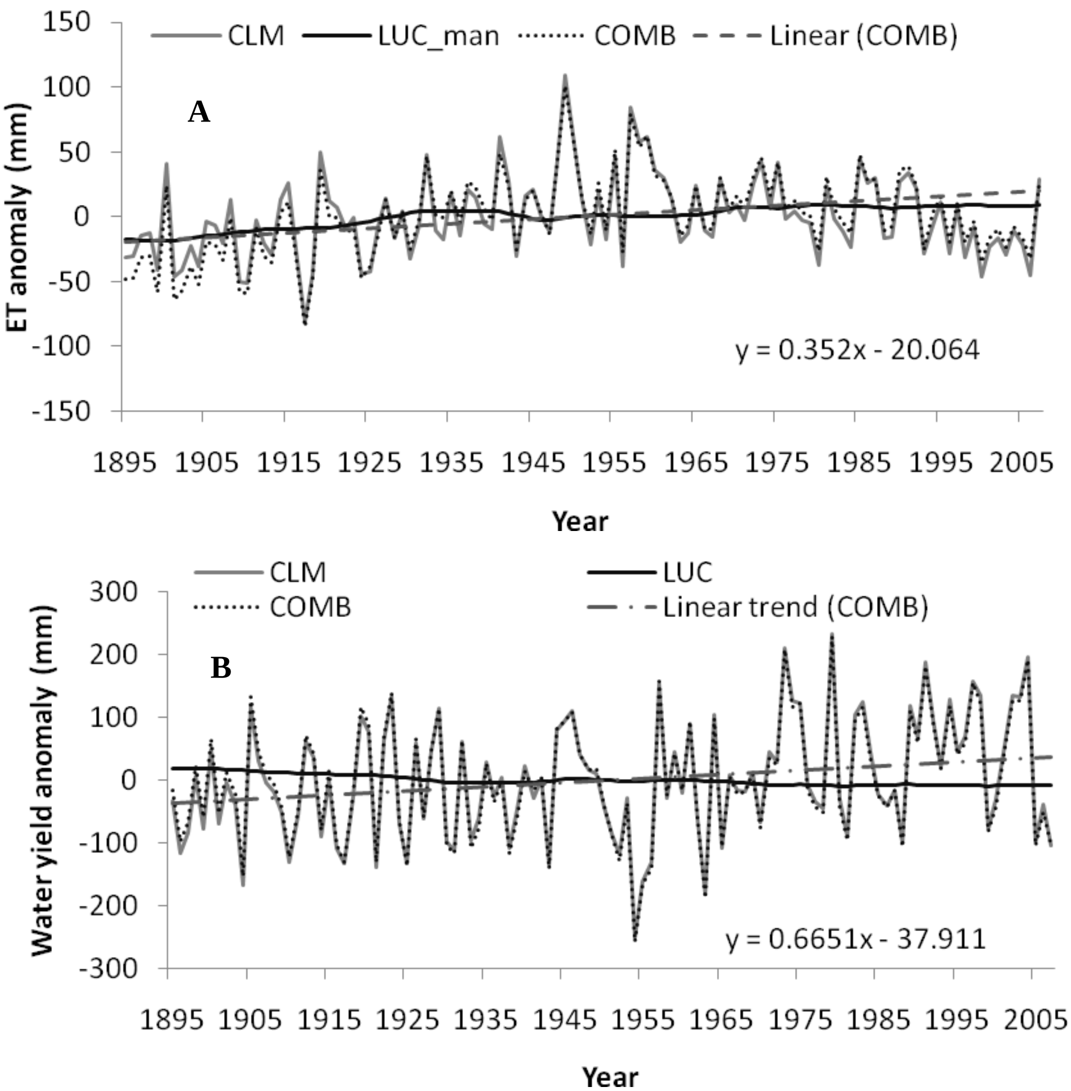

Figure 6 Evapotranspiration (A) and water yield anomalies (B) (relative to mean value during 1895-2007) under different scenarios (CLM, only climate change; LUC_man, land use change with N fertilization; COMB, combined climate and land use change with land management). Note: the dashed line is the linear trend of ET and water yield for the COMB scenario.

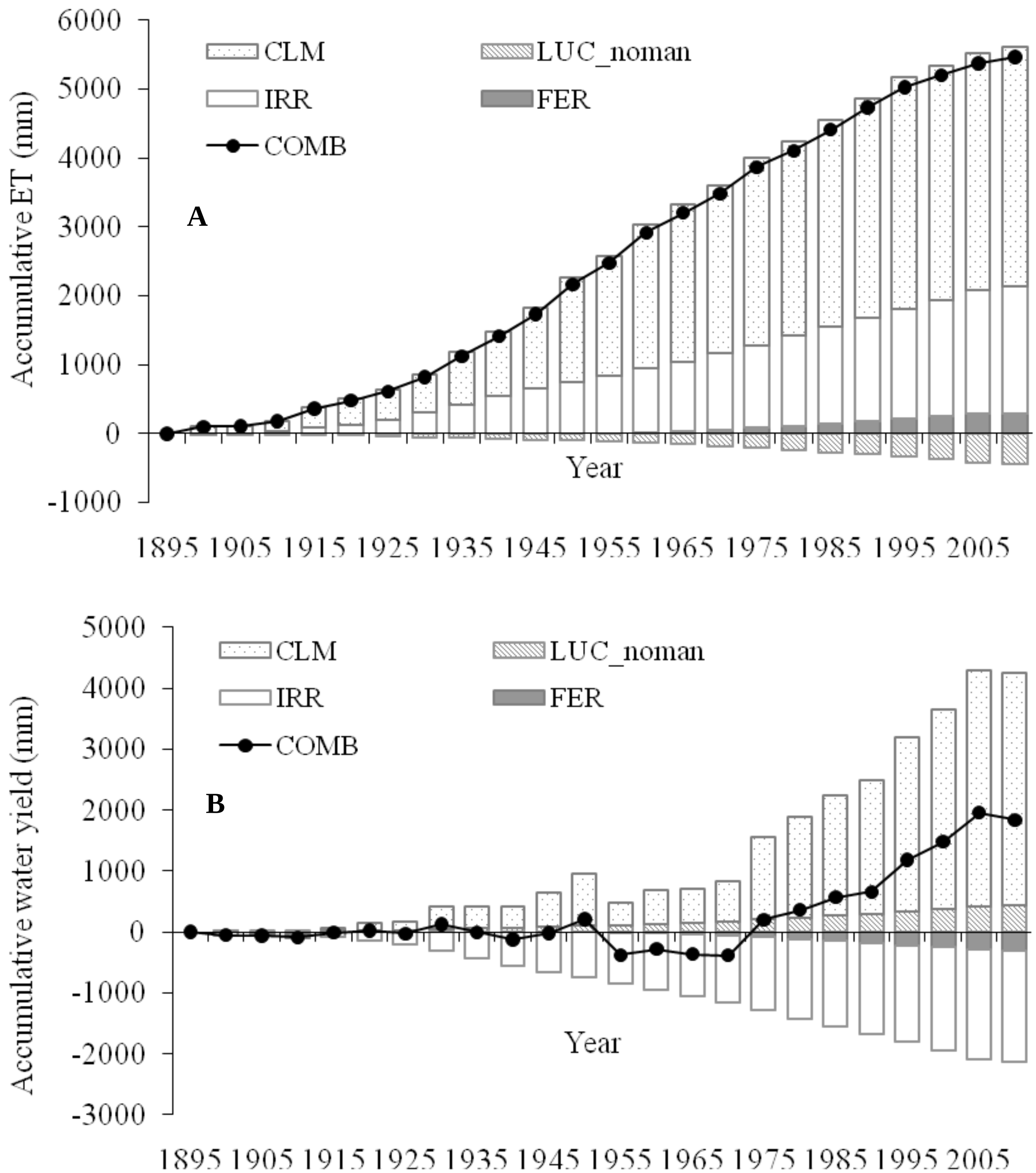

Figure 7 Contributions of different factors to accumulative changes in ET (A) and water yield (B) under different scenarios (FER, with N fertilization; IRR, with irrigation; CLM, climate change; LUC_noman, land use change without management; COMB, combined effects of all the factors).

Spatial variations in ET and water yield

The spatial variation in climate, land use, and land management practices have resulted in greatly varied spatial pattern of water yield and ET. Under climate change, water yield generally increased in the central SUS, where precipitation is at a medium level but increased the most compared to other regions (Figure 5). Water yield decreased in some parts of the eastern and western SUS (Figure 8A). Water yield could decrease as much as 32 mm decade^{-1} and increase as much as 40 mm decade^{-1}. Although precipitation in the western Texas has increased (Figure 5), water yield still decreased due to more ET in this region (Figure 8B). ET change rate displayed different spatial patterns from water yield due to climate change. Generally, ET increased in the western SUS and decreased in the south (Figure 8B). Under both land use and management changes, water yield displayed a complicated spatial variation (Figure 8C). Water yield decreased when land use type changed from natural vegetation to managed cropland and urban land and increased vice versa. This change rate in ET showed an opposite spatial distribution pattern compared to water yield (Figure 8D). Irrigation in cropland caused a vast increase in ET in cropland areas.

Although the impacts of climate change controlled the spatial pattern of water yield under the combined scenario, the impacts of land use change and management enhanced or counteracted the magnitude of change rate (Figure 8E). Overall water yield in the SUS increased only about 6.7 mm decade^{-1} from 1895 to 2007 (Figure 6B), which might not be a significant change for some areas having a high rainfall. However, huge spatial variability in water yield change rate was found. Water yield could increase as much as 50 mm decade^{-1} and decrease 40 mm decade^{-1} in the arid and semi-arid regions. Significant decreases in water yield could mean a serious water resource shortage in dry areas such as Texas and in some extreme dry years (e.g.

2002, 2007) for the entire SUS. Georgia declared a drought emergence in 2007 due to extreme drought and freshwater shortage despite this state is known as a high long-term mean annual precipitation during the historical period.

Land management could greatly decrease water yield in magnitude. Under only land use change, changes from non-cropland to cropland caused an increase in water yield especially in the western SUS (Figure 9B). However, water yield was greatly reduced after land management (Figure 9A). To increase crop productivity, irrigation and fertilizers were commonly applied in the SUS by most cropland managers. Although climate change has slightly mitigated the water shortage induced by land management, great decreases in water yield still occurred for some cropland areas especially in the western SUS (Figure 8E). From the perspective of regional water balance, cropland expansions in the western SUS could greatly reduce available water resources and result in more serious droughts.

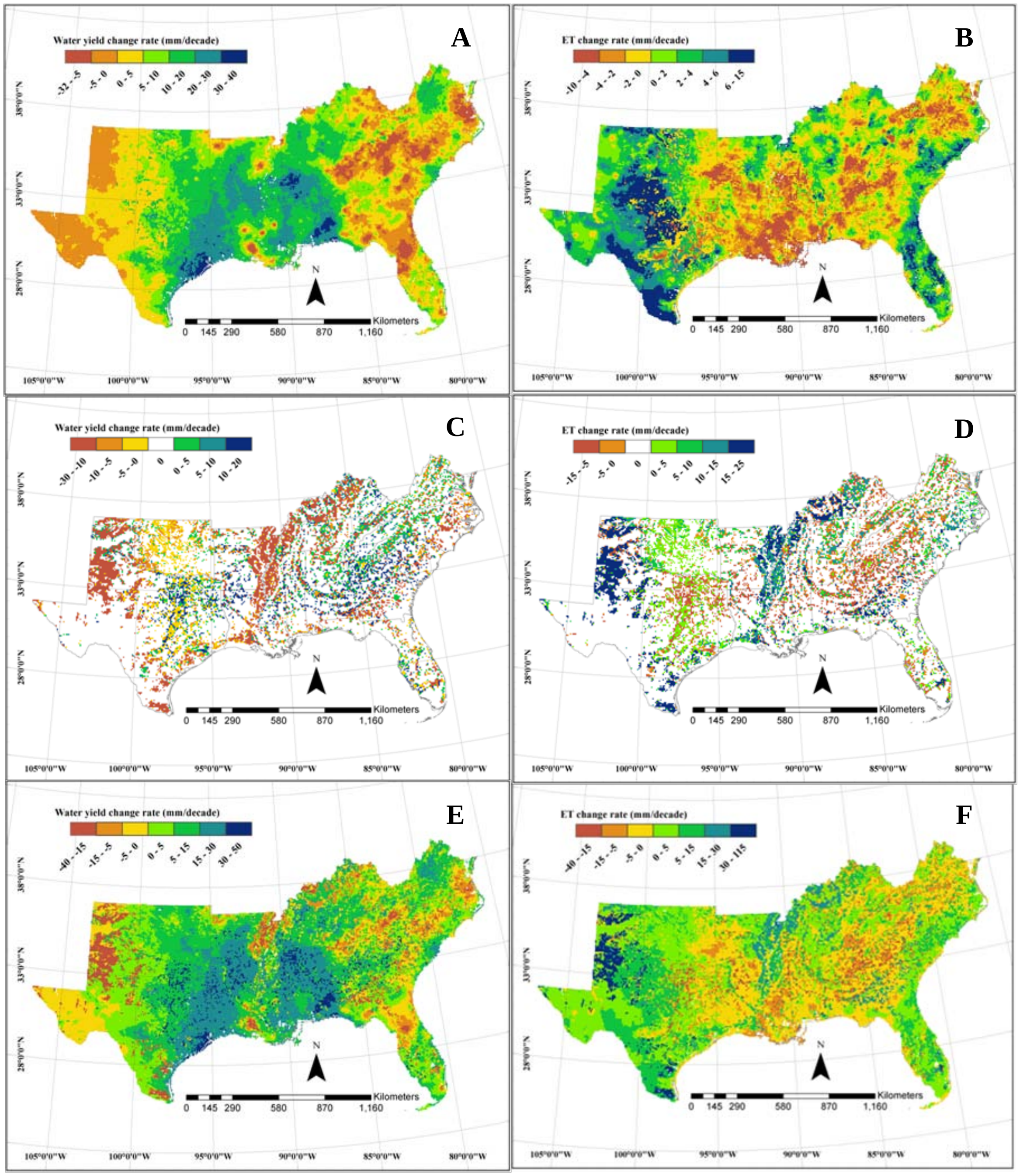

Figure 8 Decadal water yield (mm decade^{-1}) and ET (mm decade^{-1}) change rate during 1895-2007. A, C, and E: water yield change rate under climate change, land use change with land management, and combined scenarios, respectively. B, D, and F: ET change rate under climate change, land use change with land management, and combined scenarios, respectively.

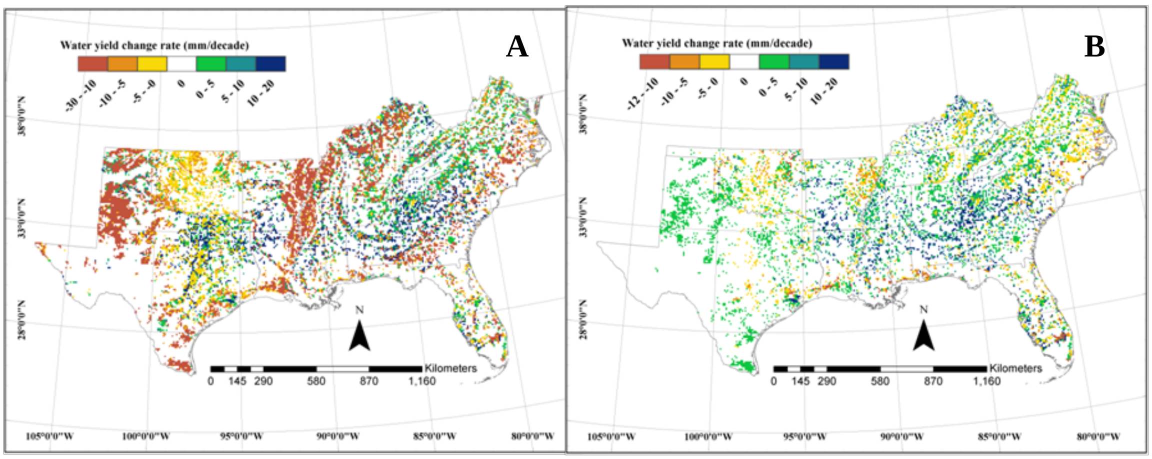

Figure 9 Water yield change rate under land use with management scenario (A) and land use change only scenario (B).

DISCUSSION

Comparisons with other studies

Several regional or watershed level studies have been done in the SUS in the past decade (e.g., Lu et al. 2003; McNulty et al. 2007; Sun et al. 2002, 2005, 2008). Most of these studies were based on observational or empirical (or statistical) modeling results and primarily focused on forests. Based on a large number of observations for water yield in 38 watersheds in the SUS, Sun et al. (2005) indicated that ratios of water yield to precipitation were located in a range of 18%-63% (~35% for mean values), and ET accounts for ~65% of the precipitation. This study found a mean water yield/precipitation ratio of 30% (a range of 5%-60%) for the entire SUS. The lowest water yield/precipitation ratio was located in the western SUS.

A processed based model Community Land Model has been used to estimate global ET with a spatial resolution by 1 degree (Rodell et al. 2004; Oleson et al. 2007). We selected the estimated ET for the SUS region. The mean ET (696 mm) estimated from DLEM for the 1979-

2007 period was close to the estimate of CLM (686 mm). The annual changing pattern of DLEM-modeled ET in the SUS was also similar with that estimated by CLM. Compared with CLM model, DLEM underestimated the extreme low ET in 1997. 1997 was found a very wet year (1229 mm precipitation), ET in this year may not lower than 500 as estimated by CLM. The coarse resolution of CLM (1 degree) might influence the estimated actual ET in the SUS, but could still accurately reflect the annual changing pattern.

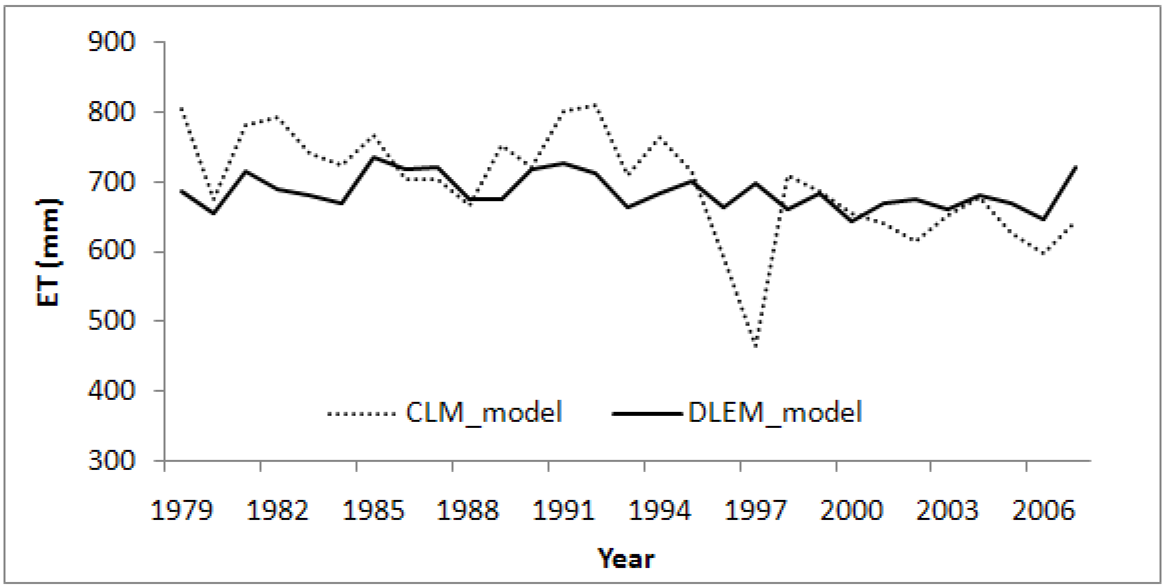

Figure 10 Comparison of DLEM estimated ET with CLM estimated ET for the period: 1979-2007 in the SUS.

Contribution of environmental factors to ET and water yield

Lu et al. (2003) developed a regression model that correlates watershed ET to watershed characteristics. Their study suggested that patterns of water loss from forests to the atmosphere were mostly controlled by air temperature and precipitation, and land cover playing a second role in the SUS. Based on future (2020) climate and land use data, Sun et al. (2008) also indicated the contribution of climate change was more important than land use change. In this study, we also found that the changing patterns in ET and water yield were primarily controlled by climate change. Aber et al. (1995) also noted that combination of increased temperature (+6^{o}C) and decreased precipitation (-15%) resulted in increases in NPP and decreases in water yield over the northeastern US. Increased precipitation during 1895-2007 has not only increased ET by elevating productivity and soil water availability, but also water yield because more water was not used by the ecosystems (Lu et al. 2003; Sun et al. 2005). Land use and management only slightly altered the ET and water yield magnitudes. We found that land use change from forest land to non-forested land, vegetated land to bare ground, and wetland to non-wetland could decrease ET and vice versa, which has also indicated by Lu et al. (2003), Sun et al. (2005) and Brown et al. (2005). Irrigation could not only increase ET through increasing soil water content and thus evaporation but also through promoting crop growth and thus canopy ET. However, unlike the function of increased precipitation which is from an outside source, water yield could be decreased with irrigation because irrigated water is drawn from the local ecosystem. In addition, DLEM assumes that soil water content will arrive at field capacity after irrigation, so no extra runoffs are yielded during irrigation events. Nitrogen fertilization indirectly increased ET by elevating crop productivity and leaf area index, and thus reduced water yield (Aber et al. 1995; Sun et al. 2005). Nitrogen fertilization in the SUS was found to increase NPP by 27%, which could result in a significant increase in ET.

Uncertainties and policy implications

Although the DLEM-simulated water yield and ET could be fitted well to field observational data, there are still many uncertainties. In this study, we did not consider the the underground water table and water movement among surrounding grid cells, although the new version of DLEM included those mechanisms and has the capability to make such simulations. Due to lack of historical irrigation data, the irrigated area distribution map in 2000 (Siebert et al. 2005) was used as the control of irrigated cropland distributions in the SUS. In addition, DLEM determines the irrigation time by soil water condition, i.e., the cropland will be irrigated when soil volumetric water content is lower than half of the field capacity and wilting point. Irrigation will stop when soil moisture is the same with field capacity. Thus, no runoff is produced during irrigation, and the irrigation effects on ET and water yield may not be fully reflected. Forest harvest and thinning were not considered in this study. Sun et al. (2005) and Brown et al. (2005) indicated that water yield decreases and ET increases with increasing forest age and coverage. Since most forests in the SUS are young-aged (Pan et al. 2010), the ignoring of forest harvest might overestimate the contemporary water yield. In addition, simplifications of some water cycle processes in the model could have failed to capture changing patterns of water yield and ET under some extreme conditions. Future improvements will be made based on the above-mentioned aspects.

As indicated by historical records (e.g., Seagers et al. 2009; Karl et al. 2009; Tian et al. 2010a) and projected by several Global Circulation Models (GCMs), the SUS is becoming wetter due to increased precipitation and water quality is degrading due to intensive agricultural practices, urban development, and industrial activities. Water resource managers demand practical tools to estimate water yield potentials from various ecosystem lands and evaluate how management, land use change, and climate change may affect water resources (Sun et al. 2005). The study results on the impacts of changes in climate, land use, and management on water yield could be a reference for the land managers and policy makers to better manage water resources in the Southern United States.

CONCLUSION

This study explored the impacts of climate, land use and land management changes on ET and water yield during 1895-2007. We found that both ET and water yield were increased during the study period. Compared to land use change and management, climate variability had the largest impact and contributed the most to the increase in both ET and water yield. Although the change in both ET (3.5 mm decade^{-1}) and water yield (6.7 mm decade^{-1}) were not significantly large, the much larger spatial and inter-annual variation could result in significant impacts on ecosystem structure and function, especially in the arid and semi-arid region. Climate change has a dominant impact on the changing rate and pattern of ET and water yield, while land management played the second role. Land use change resulted in a decrease in ET and increase in water yield. The modeled water yield and ET have been validated against field observation data, showing good agreement between simulated and measured results. The study results on the impacts of changes in climate, land use, and management on water yield could be a reference for the land managers and policy makers to better manage water resources in the SUS.

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Chapter 5

Increased Global Warming Potential by Nitrogen Fertilization in the Cropland of the Southern United States

ABSTRACT

Although increases in atmospheric CO_2 is t is a major contributor to global warming, more emphasis has been placed on the other trace greenhouse gases (i.e., CH_4 and N_2O), which have lower concentrations in the atmosphere, but have higher global warming potential (GWP). Recent progress in field observations, experiments, and modeling algorithms has greatly advanced our understanding to CO_2 and N_2O emissions. In this study, the effects of nitrogen (N) fertilization on CO_2, and N_2O fluxes from croplands of the Southern United States (SUS) were estimated using a process-based global biogeochemical model (Dynamic Land Ecosystem Model, DLEM). GWP was used to represent the overall greenhouse effect of these two gases as impacted from N fertilization. Results indicated that C storage has been greatly increased by ~296 Tg (4.70 Tg yr^{-1}) during 1945-2007, while N_2O emission was also significantly enhanced by 2.97 Tg N (0.047 Tg N yr^{-1}). The ratio of N_2O-N emission rate to fertilized N amounts was 2.5% ± 0.2%, indicating that ~2.5% of fertilized N was emitted as N_2O-N. Combining the GWP of these two gases, N fertilization was a net source that could enhance the GWP by 304.6 Tg CO_2 equivalents during this period. The GWP induced by N fertilization increased after mid 1970s and N fertilization exhibited a saturation effect for increasing C storage, suggesting that further increases in N fertilizer use would not significantly stimulate C sequestration. To

decrease GWP and maintain high crop productivity in the future, crop N use efficiency of added N needs to be increased rather than increasing N fertilizer amounts. Further research is needed to consider the impacts of other management practices in addition to N fertilization on GWP in the SUS.

Key words: Global warming potential (GWP); nitrous oxide (N_2O); nitrogen fertilization; Southern United States

INTRODUCTION

Carbon dioxide (CO_2) is the major contributor (36-72%) to global warming, while CH_4 and N_2O contribute 18% and 9%, respectively to the global warming in 2000 (Forster et al. 2007; IPCC 2007). Although atmospheric concentrations of N_2O in the atmosphere are relatively low, its warming potential is significantly higher than CO_2 (Denman et al. 2007). Similar to increases in atmospheric CO_2 concentration (increased by 13% since 1998, IPCC 2007), the concentrations of CH_4 and N_2O have increased at a rate of 1% and 0.25% per year, respectively, with significant inter-annual fluctuations (Forster et al. 2007; IPCC 2007; Rigby et al. 2008). Globally, agricultural byproducts and land use change contributed about 9%, and 88% to the increase in atmospheric CO_2 and N_2O, respectively (IPCC 2007). The United States Environmental Protection Agency (USEPA) greenhouse gas inventory report noted that agricultural soil and manure management is the most important source of N_2O emission (USEPA 2010), suggesting that cropland management could be very important for potentially reducing greenhouse gas emissions.

Managing cropland to optimize soil C storage and minimize N_2O could have a significant impact on the future atmospheric radiative forcing resulting from increased CO_2 and N_2O in the atmosphere and sustainable crop production (Mosier et al. 1998). Nitrous oxide can be produced naturally by soil microbial processes (i.e., nitrification and denitrification, Mosier et al. 1998) and its high variability is generally caused by the changes in environmental factors such as soil moisture, soil temperature, and nutrient availability (Li et al. 2000, 2005; Werner et al. 2006; Xu et al. 2010). Nitrification and denitrification require mineral and nitrate N as substrates (Mosier et al. 1998). The key process affecting nitrous oxide emission is denitrification, which is carried out by the population of denitrifying bacteria and is influenced by the level of oxygen present in soil, availability of nitrate as an electron acceptor, and availability of water-soluble organic C as an energy source (Franzlubbers 2005). Therefore, addition of N fertilizers to soil enhances N_2O emission (Clayton et al. 1997; Hall and Matson 2003; Karen and Smith 2003).

Uptake of atmospheric CO_2 by cropland soil can sequester CO_2, while normal crop production practices such as N fertilization, tillage and manure application generates N_2O and decreases the soil sink for atmospheric CH_4 (Moiser et al. 1998). The overall balance among these three gases forms the net global warming potential of a cropland. N fertilizer amounts applied in the U.S. cropland has greatly increased since the mid-1940s (Alexander and Smith 1990; Ruddy et al. 2006). Increases in N fertilizer use has significantly enhanced crop production and C sequestration in U.S. soils (Robertson et al. 2000; EIA 2001; USEPA 2010; Morgan et al. 2010); however, large amounts of N_2O have been reported to be released during this period. Based on statistical data, EIA (2001) found that N_2O amounts for 5% of the U.S. GWP-weighted greenhouse gas emission during 1990s and agricultural source accounts for 70% of the N_2O emission in the U.S. and among which, N fertilization associated N_2O emission accounts for 73% of agricultural emissions. Many previous large-scale estimates of N_2O emission induced by N fertilization were based on empirical up-scaling of limited databases (e.g., Mosier et al. 1998;

Sperow et al. 2001; Franzlubbers 2005; Christopher and Lal 2007; Xu et al. 2008; Gacengo 2008) which were unable to account for the spatial and temporal variability of soil-atmosphere exchanges (Kiese et al. 2005; Werner et al. 2006). In addition, the estimates of N_2O emissions are more uncertain than those either CO_2 or CH_4, since N_2O is not systematically measured and there are many sources of N_2O emissions (EIA 2001).

N fertilizer amounts in the SUS had also increased rapidly and remained a slow increasing rate in recent decades. Although there were many available data from field experiments and observations (e.g., Sainju et al. 2002; Franzlubbers 2005; Gacengo 2008; Smith et al. 2010) in the United States and SUS, little is known about how N fertilization in the SUS has affected both CO_2 and N_2O fluxes and what was the combined global warming potential (GWP) induced by long-term and high-dose N fertilization from a regional perspective (Franzlubbers 2005). Based on GIS-based, county-level cropland N fertilization data, and a process-based global biogeochemical model (Dynamic Land Ecosystem Model, DLEM), this study was to estimate changes in CO_2 and N_2O fluxes and to assess the overall GWP in response to N fertilization in SUS croplands. DLEM has fully coupled N, water and C cycles and has been widely evaluated for estimating N_2O and CO_2 fluxes in responses to multiple environmental changes including climate, land use, ozone, N deposition and land management in China, Monsoon Asia, the United States, and North America (Chen et al. 2006a, b; Ren et al. 2007; Liu et al. 2008; Zhang et al. 2008; Tian et al. 2008, 2010a, b, c, d; Xu et al. 2010). This study strives to greatly improve GWP estimates for croplands of the SUS and North America.

METHODS

Global warming potential calculation

The GWP of a greenhouse gas is defined as the ratio of the time-integrated radiative forcing from the instantaneous release of 1 kg of a trace substance relative to that of 1 kg of a reference gas (IPCC 2001). According to USEPA (2010), the GWP (100-yr time horizon) of N_2O is set as 298 as referenced to 1 GWP of CO_2. For example, GWP of 1 Tg C (1 Tg = 10^{12} g) emission = 1 / 12 × 44 Tg CO_2 equivalent; 1 Tg N emission = 1 / 28 × 44 × 298 Tg CO_2 equivalent for N_2O.

Model description

The Dynamic Land Ecosystem Model (DLEM) fully couples C, N and water cycles and its performance has been evaluated in different regions including China, Monsoon Asia, conterminous U.S., and North America (Zhang et al. 2008; Ren et al. 2007, 2010a, b; Tian et al. 2008, 2010a, b, c, d; Xu et al. 2010). DLEM has been fully described in several papers (e.g. Tian et al. 2010a, b, d; Xu et al. 2010). DLEM has a land use and land management module which assesses impacts of the land management (e.g., cropland and forest fertilization, irrigation, forest harvest, afforestation, etc.) on C storage and fluxes (Liu et al. 2008; Tian et al. 2010a, d; Xu et al. 2010; Ren 2010a, b). In addition, the full N cycling processes simulated by DLEM (Figure 1) has been described in detail in Lu et al. (2010) and soil N transformation processes have also been described in Xu et al. (2010).

Model parameterization, evaluation and simulation experiments

Mode parameterization for simulating CO_2 and N_2O fluxes in the SUS has been described by Tian et al. (2010a, b). The same parameter values as described in these two papers were used in this work. Evaluations of DLEM performance in simulating GPP, C fluxes, and N_2O fluxes in response to multiple environmental changes have been reported previously (Tian et al. 2010a, b, c, d; Xu et al. 2010; Lu et al. 2010).

To achieve study objectives, two model simulation experiments were designed: 1) LUC: considers land use conversion only, with no fertilizer use; 2) LUC_man: combines both land use conversion and changes in N fertilizer amounts. The nitrogen fertilization effects are thus estimated as the difference between LUC and LUC_man scenarios: LUC_man – LUC.

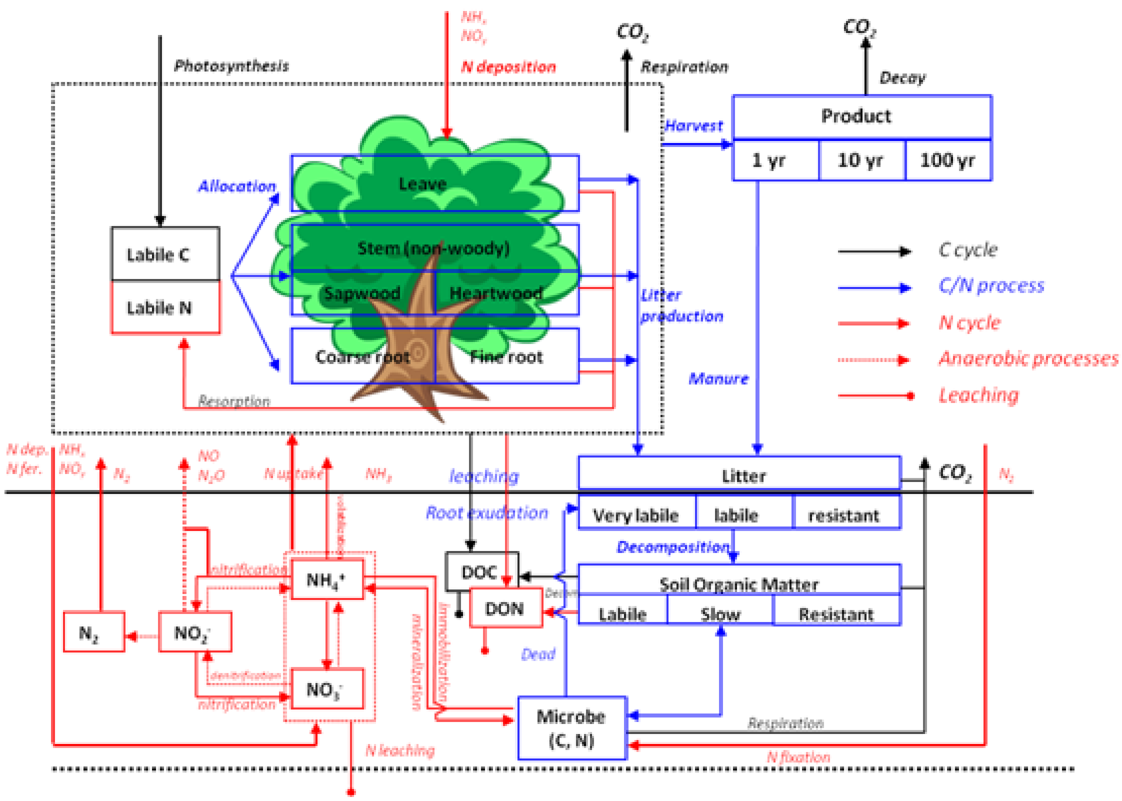

Figure 1 Schematic diagram of nitrogen cycling processes in the DLEM model (Lu et al. 2010).

Data description

Cropland data sets: Approaches similar to Chen et al. (2006b) and Zhang et al. (2008) were used to combine contemporary land-use maps derived from the USGS National Land Cover Datasets (http://edc.usgs.gov/products/landcover.html) with the historical census datasets of cropland, urban area, and population for reconstructing spatial maps (8 km × 8 km resolution) for cropland and urban/built-up areas from 1940 to 2005.

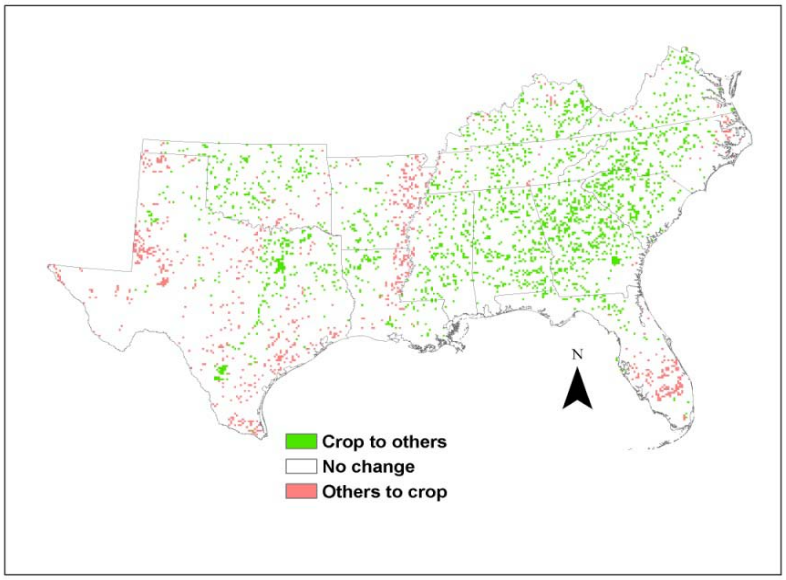

Figure 2 Spatial change pattern of cropland area during 1945-2007.

We first aggregated the 30 m resolution National Land Cover Map from USGS to 8 km resolution and recorded the fraction of cropland and urban/built-up areas in each grid cell. For historical cropland distribution data, we conducted temporal interpolation by calculating the cropland percentage for each grid cell in each year by using the cropland census data (Waisanen and Bliss 2002) as the control for changing trends. We used county-level relative change of cropland from the Census of Agriculture as controls to identify change rate of cropland so that the total area could match the county-level statistical data. The data from Census of Agriculture (Waisanen and Bliss 2002) covered the period from 1790 to 2002 and about a 5-year interval between two censuses after 1940. Thus, this cropland interpolation data relatively accurate reflected the cropland distribution and its change over time for each grid cell. Due to the lack of a recent land-use change dataset, we assumed no land use change took place in the study region from 2003 to 2007. Based on the generated data, we found that cropland area decreased ~18.6% (from 5.0×10^5 km^2 in 1945 to 4.1×10^5 km^2 in 2000s) during 1945-2007 in the SUS (Figure 3). This decline was attributed to the fast increase in urban/built-up areas and a slight increase in forest area during this period (Tian et al. 2010a). The spatial pattern of cropland change from 1945 to 2007 can be seen in Figure 2. The majority of this land use change type was characterized by cropland abandonment to other land cover types.

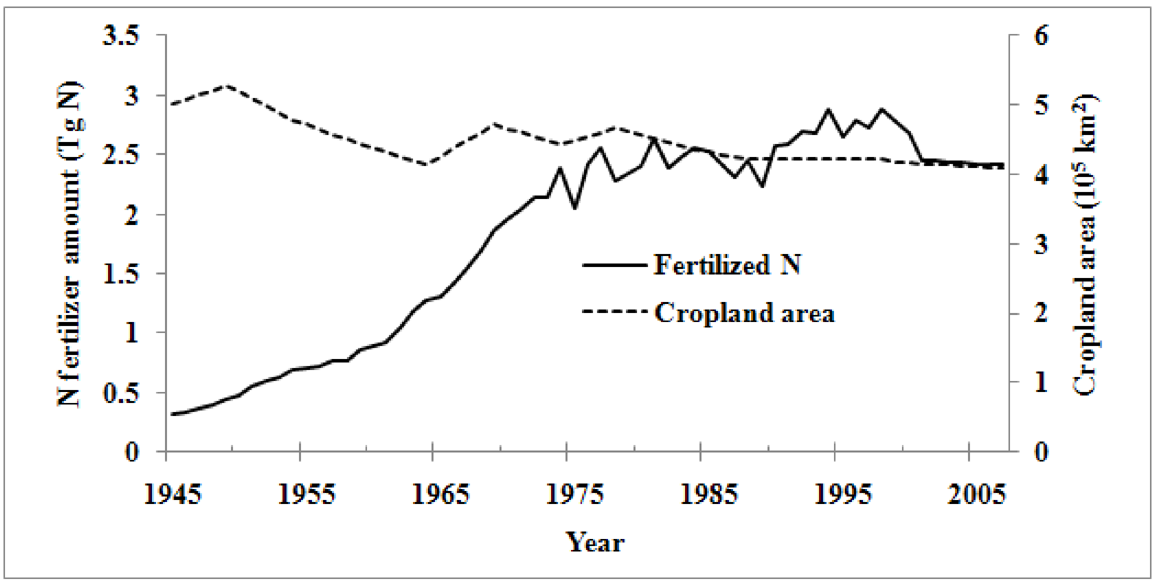

Figure 3 Cropland area (10^5 km^2) and fertilized nitrogen amounts (Tg N) in the SUS during 1945-2007

Major SUS crop types: Based on the global distribution maps of major crops at a spatial resolution of 5 arc minutes (~10 km) (Leff et al. 2004), transformed the projection information helped generate the distribution maps of major crops for the SUS at a spatial resolution of 8 km (Figure 4). Each major crop type has a distribution percentage in each grid cell. We compared the percentage of all the major crop types in a grid cell and then assign this grid as the crop type with the highest percentage, in other words, the crop type with the highest percentage in a grid cell will be assigned. In some areas of the SUS, double cropping systems (i.e. two crop types were continuously planted within a year) were used. However, in this study, only one of these (i.e., the crop type with the highest percentage) as the crop type in those areas since most of the SUS cropland areas only have one crop type within a year. In addition, due to lack of the historical crop type data, we assumed that crop types did not change with time. Only 7 major crop types in this study are selected for the SUS, which include soybean, winter wheat, corn, rice, cotton, peanut, and sugarcane (Figure 4).

Leaf phenology of major crop types: We used prescribed monthly (mid-month) leaf phenology to control crop growth life cycles. The MODIS 8-day LAI products (MOD15) during 2000-2008 (http://modis.gsfc.nasa.gov/data/dataprod) were collected and aggregated to monthly LAI (at the middle of each month). The 9-year LAI data were averaged and we selected the grid cells in which at least 80% were occupied by each major crop type. The selected grid cells for each crop type were then averaged to get the mean LAI for each month for a given crop type in the SUS. The month with the highest LAI was assigned a leaf phenology value of 1 and phenology in other months was calculated by the ratio of current month LAI to the highest LAI. Since phenology for a specific crop type does not change much during a short-term history and climate in the SUS is relatively uniform, the average phenology for all the grid cells for a given crop type could yield a good fit for actual crop phenology.

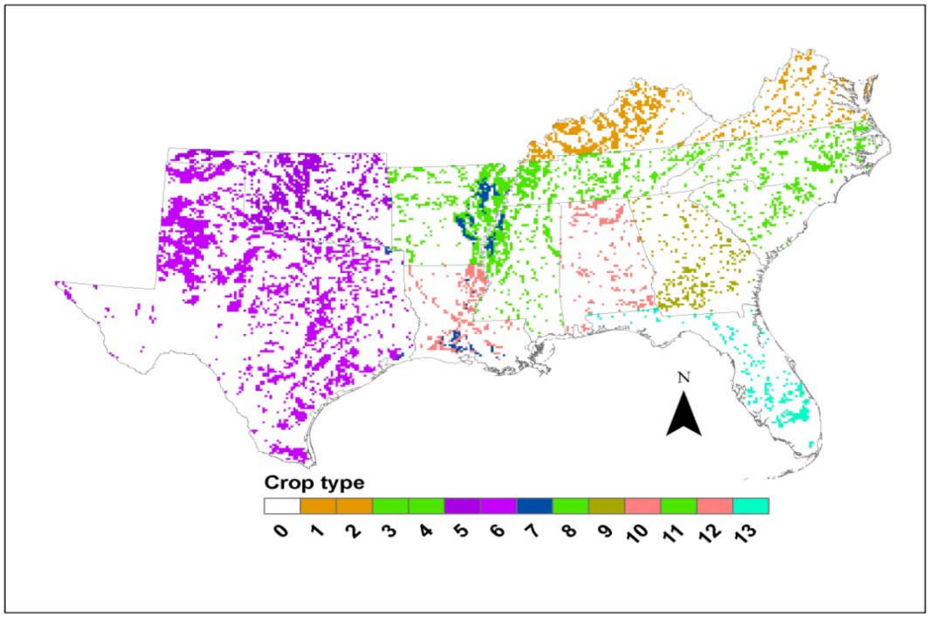

Figure 4 Major crop types in the SUS (Corn: 1, 2; Soybean: 3, 4, 8, 11; Winter wheat: 5, 6; Rice: 7; Peanut: 9; Cotton: 10, 12; Sugarcane: 13)

Nitrogen fertilization dataset: Alexander and Smith (1990) and Ruddy et al. (2006) developed county-level N fertilization tabular datasets for the contiguous U.S. from 1945 to 1985 and from 1987 to 2001. By assuming the N fertilization rate for 1986 was the average of 1985 and 1987 rates, the two datasets were combined to derive a county level N fertilizer tabular dataset from 1945 to 2001. Based on county-level cropland area census data (Waisanen and Bliss 2002), we then derived spatial maps of N fertilizer amounts (g N per m^2 cropland) for the SUS from 1945 to 2001. We assumed that there was no N fertilizer applied in the cropland before 1945. We further assumed that N fertilization application remained unchanged from 2001 to 2007 since no N fertilizer data were available after 2001. Based on generated data, we found that N fertilizer amounts increased rapidly from 1945 to the end of 1970s, and maintained a slow increase with fluctuations during the 1980s to 2000s (Figure 2). By assuming N fertilization time occurs on the first day of the growing season and fertilized N could take effects within the whole growing season, the annual amount of fertilized N is allocated for each day during the growing season (i.e., daily fertilizer amount = annual amount / growing season length). N fertilizer amounts increased from 0.31 Tg N yr^{-1} (0.32 g N m^{-2} yr^{-1}) in 1945 to 2.40 Tg N yr^{-1} (5.9 g N m^{-2} yr^{-1}) in 2001. The total fertilizer amounts during 1945-2007 were 115.5 Tg N. The spatial distribution of N fertilizer amounts for 1945 and 2001 can be seen in Figure 5. N fertilizer amounts have significantly increased for the SUS croplands. The highest increase rate was found in the Mississippi River Valley, where N fertilizer amounts were over 10 g N m^{-2} in 2001. The substantial increase in N fertilizer amounts could greatly change the C and N cycles in cropland ecosystems.

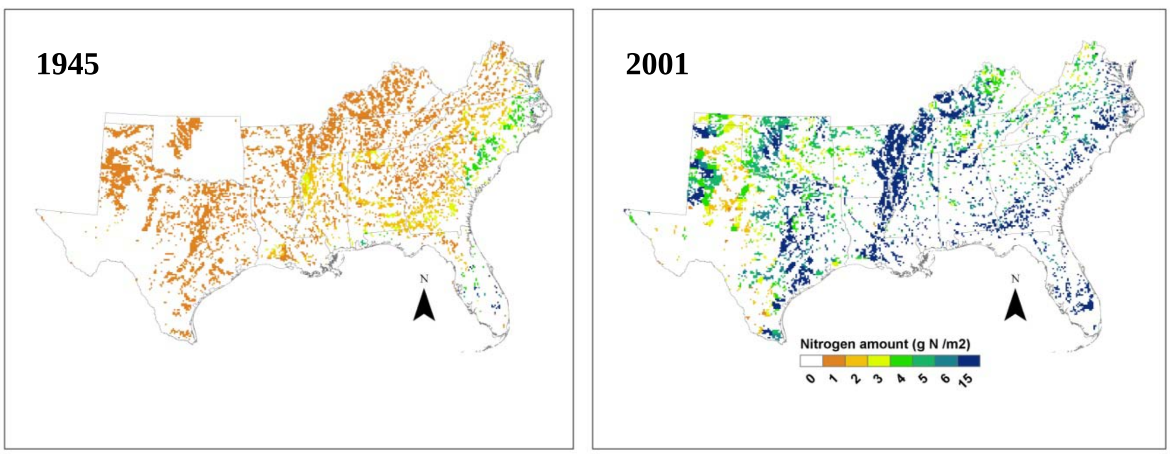

Figure 5 Spatial distribution of N fertilizer amounts (g N m^{-2}) in SUS croplands for 1945 and 2001.

RESULTS AND ANALYSES

Changes in CO_2 and N_2O fluxes, and global warming potential

Although N fertilization amount was low (about 0.3 Tg N, Figure 2) during the first several years of application, C storage in the SUS cropland soils were rapidly enhanced (Figure

6). With a continuous increase in N fertilizer, C storage also increased. However, C storage no longer increased and even decreased after N fertilizer amounts arrived at a certain level, indicating a sign of N saturation for C storage in the SUS during 1980-2007. During 1945-2007, N fertilization has accumulatively increased carbon storage by 296 Tg C (4.70 ± 1.51 Tg C yr^{-1}) in cropland soil.

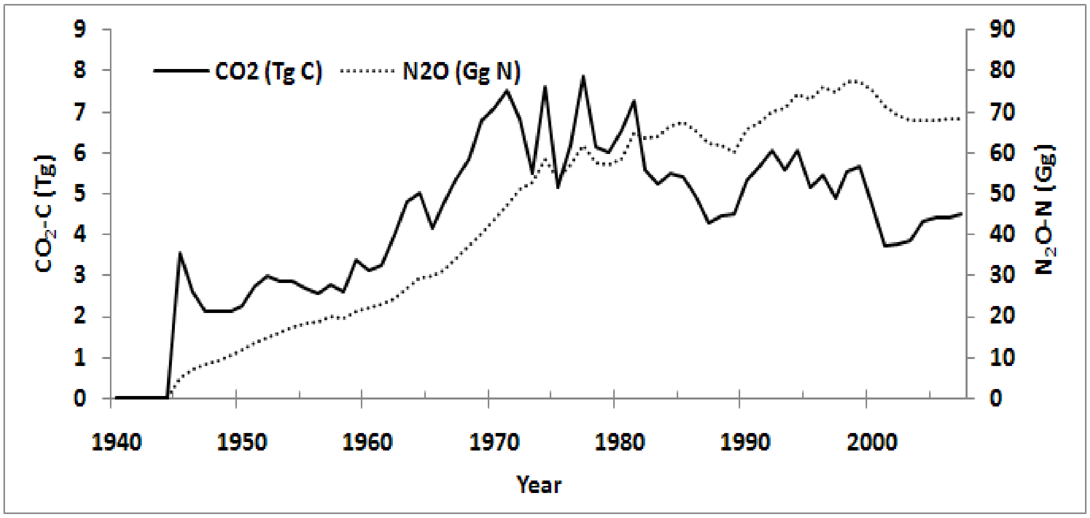

Figure 6 Changes in CO_2 and N_2O fluxes due to the impacts of N fertilization in SUS croplands. Note: Flux change is the difference between LUC_man scenario and LUC only scenario: LUC_man - LUC. Units for CO_2: Tg C, and N_2O: Gg N. 1 Tg = 10^{12} g; 1 Gg = 1000 tons = 10^9 g. Positive CO_2 values indicate a C sink, while positive N_2O values indicate a N source.

Nitrous oxide emission increased rapidly from 1945 to the mid-1970s and then exhibited a slow increase with large inter-annual fluctuations. During 1945-2007, N fertilization has accumulatively stimulated N_2O emission by 2.97 Tg N (0.047 Tg N yr^{-1}), which accounts for about 2.5% ± 0.2% of the total fertilized N (115.5 Tg N) in the SUS. The changes of N_2O emission rate generally increased with N fertilizer amounts. However, the ratio of increased annual N_2O emission to annual fertilized N was not constant (Figure 7). We found for the first 5 years following N fertilization, N_2O/fertilized N ratio increased very fast, implying that N_2O emission could be greatly stimulated following the first several years of fertilization even although N fertilizer amounts were relatively low during this time. Nitrous oxide/fertilized N ratio then slightly decreased with increased N fertilization after the fast growing period and remained relatively constant thereafter.

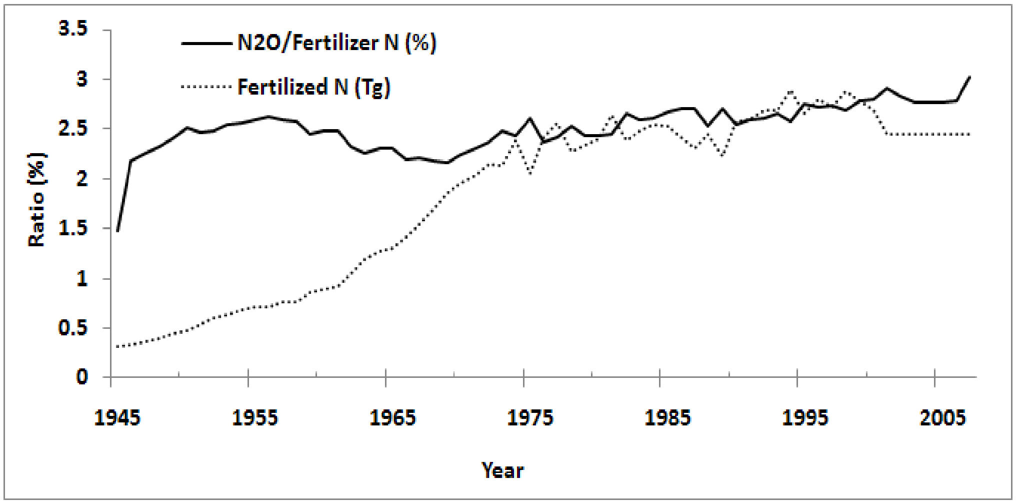

Figure 7 The ratio of elevated N_2O emission to fertilized N amounts during 1945-2007.

Combined GWP for CO_2 and N_2O fluxes

Combining the GWP for CO_2 and N_2O, the SUS cropland soil was a C source of 304.6 Tg CO_2 eq. as influenced by N fertilization during 1945-2007 (Figure 8). Although cropland soil C storage has increased by 296 Tg C (-1086 Tg CO_2 eq.), the higher GWP of N_2O (1390.6 Tg CO_2 eq.) made the SUS a C source. During 1945-1974, the annual GWP was generally a net carbon sink (-2.98 Tg CO_2 eq. yr^{-1}). However, the annual GWP was a net C source (11.94 Tg CO_2 eq. yr^{-1}) after 1974. The GWP generally increased with elevated N fertilizer amounts after 1974. This implied that lower N fertilizer amounts could decrease GWP, while higher N fertilizer amounts could increase GWP in the SUS. Carbon sinks induced by N fertilization might be saturated due to long-term N fertilizer applications.

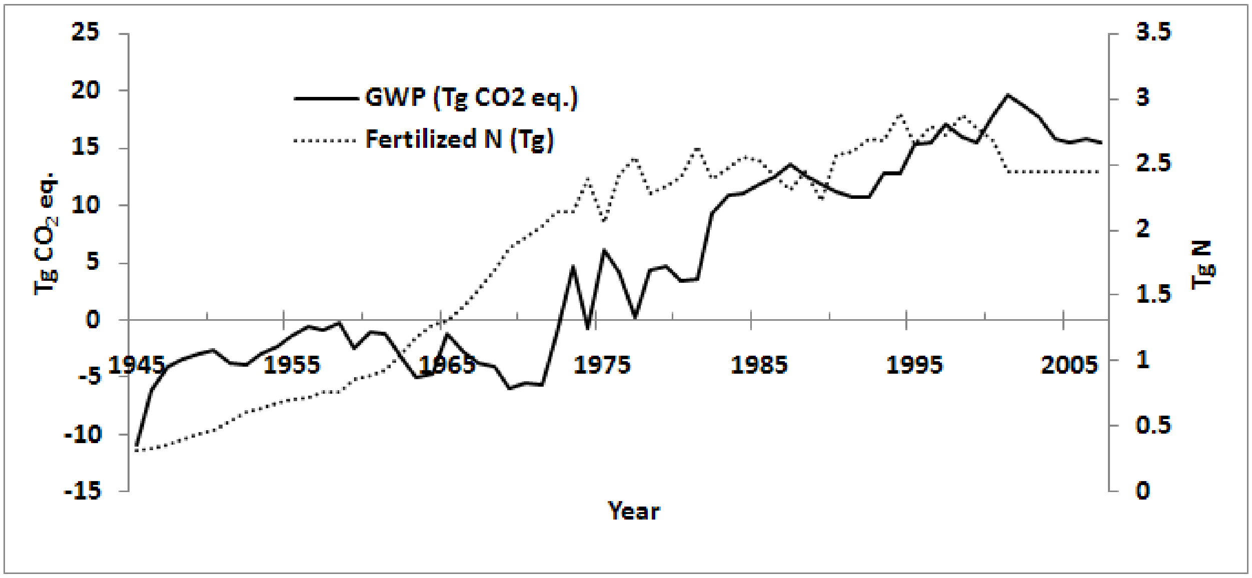

Figure 8 Combined global warming potential (GWP) of CO_2 and N_2O in the SUS and N fertilization rate (g N m^{-2} yr^{-1}) during 1945-2007 (negative GWP indicates a decrease in global warming potential).

Spatial variations in CO_2, and N_2O fluxes, and GWP

Carbon dioxide and N_2O fluxes over the SUS showed significant spatial variability (Figure 9). The spatial variation in CO_2 flux was primarily determined by N fertilizer amounts and the environmental conditions such as climate, soil texture, and topography. Cropland soil C storage increased for all fertilized areas. The greatest increase in C storage could be over 1100 g C m^{-2} during 1945-2007 (Figure 9A), which might due to continuous high N fertilizer amounts in these areas (Figure 5).

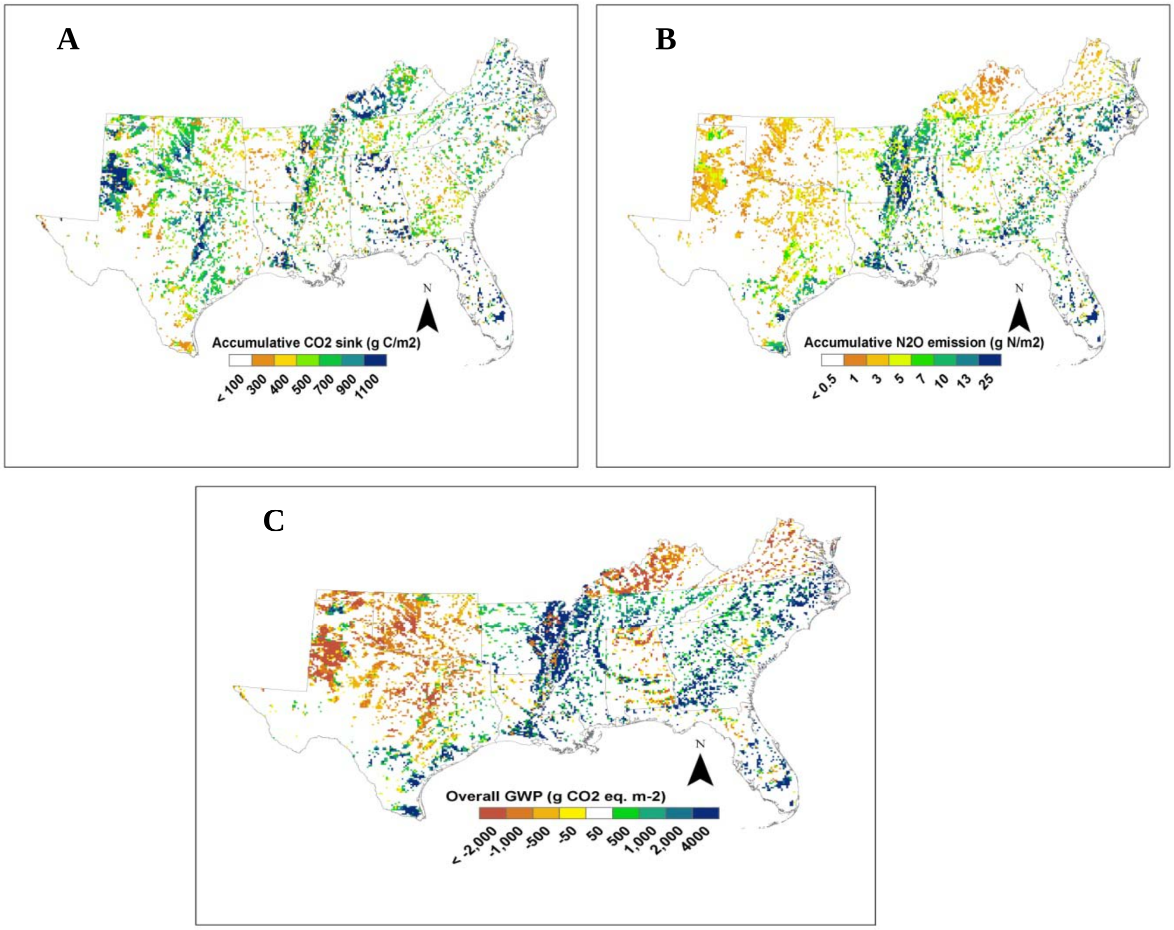

Figure 9 Spatial variations in cumulative CO_2 (g C m^{-2}) and N_2O (g N m^{-2}) fluxes, and overall GWP for CO_2 and N_2O (g CO_2 eq. m^{-2}) during 1945-2007.

All cropland area in the SUS acted as sources for atmospheric N_2O (Figure 9B). The highest N_2O emission (up to 25 g N m^{-2} during the study period) occurred in the Mississippi River Valley, where cropland had the highest N fertilizer amounts (Figure 5) and higher soil moisture due to the impacts of Mississippi River. The weak N_2O sources were observed in the western SUS. Although both N fertilizer amounts (Figure 5) and fertilizer-stimulated carbon storage (Figure 9A) were relatively high in west, accumulative N_2O emission was very low. This may be due to the region being very dry and having a soil texture with high sand fraction. In addition, lower soil water moisture may have greatly limited microbial N transformation activities.

The combined GWP for N_2O and CO_2 fluxes also varied across space (Figure 9C). Nitrogen fertilization could either increase or decrease GWP. Global warming potential increased the most in the central and southeastern SUS; these areas had the highest N fertilizer use (Figure 5), while GWP decreased in the west and northeast; these areas had the lowest N fertilizer use. This also suggested that lower N fertilizer amounts could decrease GWP, while higher N fertilizer amount could increase GWP.

DISCUSSION AND UNCERTAINTY

Carbon sequestration in cropland induced by N fertilization

We found that N fertilization resulted in a mean C sequestration rate of 10.63 ± 3.52 g m^{-2} yr^{-1} or 4.70 ± 1.51 Tg C yr^{-1} for the SUS cropland during 1945-2007. This estimate was comparable to the regional estimates (5-15 g C m^{-2} yr^{-1}) of Christopher and Lal (2007) for the entire United States. Based on a few sites with N fertilizer use in the SUS and IPCC N_2O emission factor (1.25%), Franzluebbers et al. (2002, 2005) indicated that optimized C sequestration rate could be 2.5 kg C kg^{-1} fertilizer-N applied. With this sequestration rate, the C sequestration rate in the SUS should be ~9.73 g C m^{-2} since average 3.89 g N m^{-2} was applied in SUS croplands during 1945-2007. This estimate is within our estimate range. Although most studies found that N fertilizer application could increase cropland C storage, few studies have also indicated no change or even a decreased pattern (e.g., Bremer et al. 1994; Potter et al. 1998; Halvorson et al. 1999, 2002; Sainju et al. 2002). The slight decrease or no change in C storage after N fertilization may be due to N effects on C storage being saturated, which has been indicated by this study. The effect of N fertilizer on C sequestration depends on the cropping system and tillage practice (Christopher and Lal 2007; Mosier et al. 1998). Our study indicated that soil C under those crop types with higher N fixation rate (e.g., soybean and peanut) increased less, while under those crop types with higher productivity (e.g., corn and sugar cane), soil C increased much more. The increased C storage in cropland soil was primarily attributed to increases in crop NPP due to N fertilization, which resulted in higher organic C inputs to cropland soil (Franzluebbers 2005). We found that annual crop NPP increased ~27.5% from 1945 to 2007 due to N fertilization.

Nitrogen fertilization rate, N_2O emission and N use efficiency

There are a few regional studies in the United States that have conducted the impacts of N fertilization on N_2O emission for the past decades (e.g., Kicke et al. 2004; Franzlubbers 2005; Crhistopher and Lal 2007; Tian et al. 2010b). However, no studies have specifically focused on the entire SUS. Based on data collection and statistical scaling-up, IPCC (1997) estimated that ~1.25% of the N input into agricultural system is directly emitted annually as N_2O. Cole et al. (1996) and Mosier et al. (1998) suggested including an additional factor of 0.75% to the IPCC emission factor to account for the indirect N_2O emission, implying a total 2.0% of fertilized N could be emitted as N_2O. In this study, we found that about 2.5% ± 0.1% of fertilized N was emitted as N_2O, indicating an inefficient use of fertilized N. Follett et al. (2005) suggested that 2% N emission implied that ~0.17 g C equivalent m^{-2} yr^{-1} is needed to balance the direct GWP and ~0.27 g C equivalent m^{-2} yr^{-1} for indirect GWP. In addition to emission as N_2O, the inefficiency of N fertilization is also reflected in a large amount of N leaching to ground or surface waters (Mosier et al. 1998; Halvorson et al. 2005; Christopher and Lal 2007). About 30 to 60% of N is reported to be leached in most crop production systems when excessive fertilizer is used (Kitchen and Goulding 2001). These leached N will pollute water bodies and partly release to atmosphere as N gases (Mosier et al. 1998). Therefore, it is necessary to improve the N use efficiency rather than N fertilizer amounts to increase C sequestration in cropland.

Combined GWP due to N fertilization in the SUS

The benefit of increasing C sequestration in cropland soil from N fertilization may be negated by CO_2 release during N fertilizer production and increased N_2O emission during nitrification and denitrification of applied N (Lal etal. 1998; Christopher and Lal 2007). Based on hundreds of field observational data, Liu and Greaver (2009) suggested that CO_2 reduction caused by N enrichment was offset by 53-76% due to increased N_2O and CH_4 emission from multiple ecosystems. In this study, we found that CO_2 reductions induced by N fertilization have already been negated by increased N_2O emission in croplands of the SUS. We found that lower N fertilizer amounts could decrease GWP and higher N fertilizer amounts could increase GWP. This implies that current N fertilizer uses in the SUS croplands are a probable source of GWP. Therefore, under current situation, N fertilization in SUS croplands could act as a better practice to increase crop yields or produce more biofuel energy rather than reducing GWP.

Uncertainties and policy implications

Although crop residue return to the cropland soil has been considered in this study, the return rate did not change over time due to lack of data. Crop residue return may be greatly decreased during past decades due to biofuel production (Lal 2005), resulting in lower soil nutrients and thus reducing N fertilization effects on N_2O emission and increasing the effects on

C storage. In addition, we assumed that fertilized N would be released over the whole growing season, which may underestimate the abrupt N_2O emissions right following N fertilization.

As reported in this study that higher N fertilizer amounts could increase GWP in the cropland ecosystems. To decrease GWP and maintain high crop productivity in the future, the best way is to increase plant N use efficiency to utilize the added N rather than increasing N fertilizer amounts. In addition, other management practices such as residue return (or organic amendments), conservation tillage, water management, and rotations of multiple cropping systems could effectively decrease GWP in the cropland (Lal et al. 1998, 1999; Franzlubbers 2005; Christopher and Lal 2007). Lal et al. (1998, 1999) indicated that conservation tillage, irrigation management, water table management, and elimination of fallow period may increase C storage by 20-40, 5-15, 5-15, and 20-60 g C m^{-2} yr^{-1}, respectively and these effects could maintain for 20-50 years. Currently, the challenge is to determine how much N fertilizer is required to maintain high crop productivity while minimizing GWP. Further research is needed to address the impacts of all major management practices in addition to N fertilization on GWP in the SUS.

CONCLUSIONS

Based on a global biogeochemical model and county-level N fertilizer data for the SUS, this study estimated the changes in CO_2 and N_2O fluxes as influenced by N fertilization. Results indicated that both soil C storage and N_2O emission were increased during 1945-2007. By combining the GWP of these two gases, we found that N fertilization was a C source and GWP has been greatly increased during the study period due to increased N_2O emissions. Global warming potential decreased in the western SUS and increased the most in the central SUS. This spatial pattern was primarily determined by the distribution of N fertilizer uses. We found that higher N fertilizer amounts could increase GWP while lower amounts could decrease GWP. To decrease GWP and maintain high crop productivity in the future, plant N use efficiency of added N needs to be increased rather than increasing N fertilizer uses. Further research is needed to consider the impacts of other management practices except for N fertilization on GWP in the SUS.

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Chapter 6

Effects of Continuous Forest disturbance on Carbon Storage and Fluxes

ABSTRACT

Forest ecosystems are dramatically altered by two major anthropogenic disturbances: timber harvesting and permanent conversion due to land-use change. As forest harvest is a common activity across the SUS, understanding in the impacts of harvest on carbon, water and nitrogen cycles is critical for sustainable managing forest ecosystems in this region. To achieve a full understanding, it is necessary to quantify the spatial forest change information and clarify the processes governing the impacts. Based on the forest disturbance data classified from Landsat TM/ETM+ images and a process-based global biogeochemical model (Dynamic Land Ecosystem Model, DLEM), this study estimated the impacts of continuous forest disturbance on carbon storage and fluxes during 1985-2007 in Mississippi and Alabama, USA. Although mean annual forest mortality rate for the two states was only 1.3%, the accumulative mortality rate could be 100% in some areas. This continuous but small forest disturbance has resulted in a large carbon source (0.20 Pg C, 8.3 Tg C yr^{-1}; 1 Pg = 10^{15} g, 1 Tg = 10^{12} g) in Alabama and Mississippi during 1985-2007 and the disturbance legacy effects could be larger than this number. Large decreases and slow recovery of forest biomass were the main cause of this C source. Although small disturbance events may not significantly change forest structure, the legacy effects on C storage could last over 100 years. To improve estimation accuracy of US C budget, impacts from small but continuous disturbance events should be taken into account.

Key words: Southern United States; forest disturbance; Landsat imagery; net carbon exchange; Dynamic Land Ecosystem Model (DLEM)

INTRODUCTION

The United States consumes 33% of the global wood production and produces only 25% of the world's roundwood supply (Salwasser et al. 1993). Along with the growing demand for wood products, there is a growing interest in sustainable forest resource use and the maintenance of long-term forest productivity. Forest disturbance (including harvest) have been reported to change forest structure, productivity, C sequestration, hydrological cycle and soil biogeochemistry (Houghton and Hackler 2000; Birdsey and Lewis 2003; Woodbury et al. 2007; Hansen et al. 2010; Pan et al. 2010), while the impact magnitudes are determined by the disturbance rate and frequency. Since forest biomass increases with stand age, delay harvesting to the age of biological maturity may result in the formation of a larger C sink (Alexandrov and Yamagata 2002; DOE 2007). In addition to the impacts on forest biomass and productivity, forest harvest could greatly influence organic C storage in the forest soils and wood products. Forest harvest for agriculture, forest management, and use of wood were reported to have a significant effect on terrestrial C stocks in the United States (Birdsey and Heath 1995; Houghton and Hackler 2000; Birdsey and Lewis 2003; Birdsey et al. 2006). Soil N is generally increasing in the undisturbed forest soil due to microbial or fungus fixation and atmospheric N deposition. After forest disturbance, the relative closed N cycle could be greatly disrupted. Soil N could be increased within a short-term (such as 1-5 years) since more litter may be added into the soil; however, the accumulated soil N in the soil could be soon lost through NO_3^--N leaching, biomass removal or denitrification (emitted as N_2O, NO and N_2 gases). This loss of N may cause future nutrient deficiency and thus productivity. The forest ecosystem C and N dynamic after disturbance generally follows a paradigm as described by Covington (1981), Allen et al. (1990), and Heath et al. (2003). Forest area in the Southern United States (SUS) has been greatly reduced from about 66% in 1630 to 40% at present (Smith et al. 2010). Forest area and structure changes subjected to disturbances such as land use change, hurricane, storm, wildfire, insects and diseases, and forest harvest as to meet the demand for wood products have greatly disturbed the forest ecosystems in the SUS.

With its advantages of high spatial resolution, short-time gapping, and long-term observations, Landsat TM/ETM+ images were used widely to address forest disturbance and harvest area at landscape scales and even expanded to a large scale such as a nation and a continent because of the recent technology advance in computation and modeling algorithms (Huang et al. 2009a, b; Goward et al. 2008; Masek et al. 2008). Based on the vegetation change tracker model, Huang et al. (2009a, b; 2010) and Li et al. (2009a, b) have generated the large scale forest disturbance maps based on Landsat TM/ETM+ images for most of the states in Northeast and Southeast U.S. In this study, we combine biogeochemical modeling, remote sensing, forest inventory, and eddy flux data to study forest net ecosystem productivity. Based on forest disturbance data classified from Landsat TM/ETM+ images (Huang et al. 2009a, b; Li et al. 2009a, b) and a process-based global biogeochemical cycle model: Dynamic Land Ecosystem Model (DLEM), our objectives in this study are to estimate changes in carbon storage and fluxes after forest disturbance in Mississippi and Alabama.

METHODS

Study region

In this study, we selected two states: Mississippi and Alabama in the United States as an example to apply and evaluate the DLEM model. These two states located at the central of the Southern United States (Figure 1). These two states have high forest coverage and a large area of forested wetland. Any disturbances to the wetland could significantly change the carbon and nitrogen cycles in this region. Most forests in the two states are young forest (i.e., less than 60 years old) since forests are often cut to obtain wood products. In addition, natural disturbances such as hurricanes occurred frequently, which could result in a large mortality.

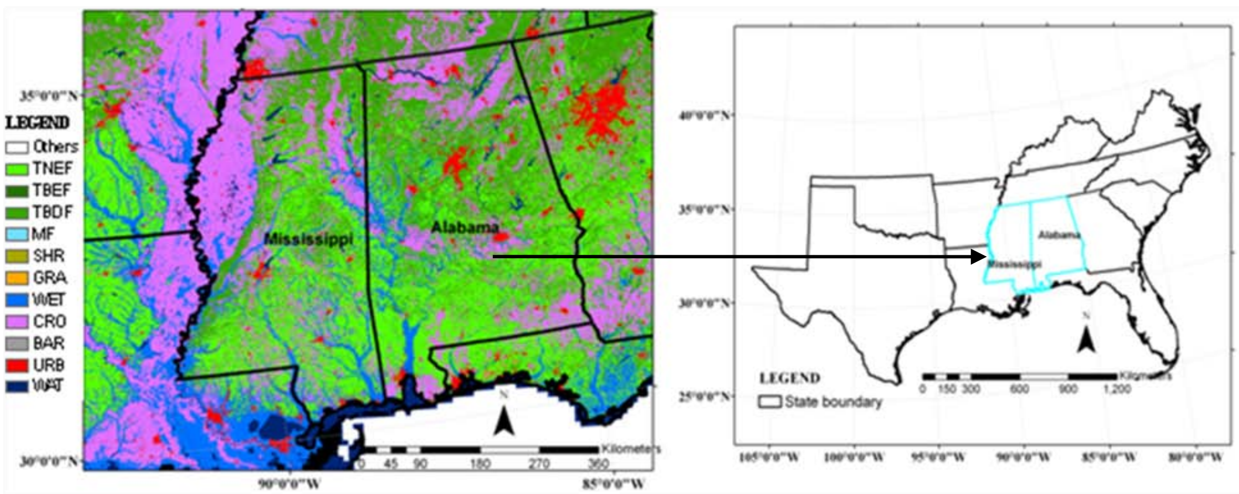

Figure 1 Boundary and land cover and land use map for the study region. TNF: temperate needleleaf forest; TBD: temperate broadleaf deciduous forest; MF: Mixed forest; SHR: temperate deciduous shrubland; GRA: grassland; WET: wetland; CRO: cropland; BAR: barren land; URB: urban and built-up land; WAT: water (Source: MODIS 2005 land cover map, http://www.cec.org).

Model description

The Dynamic Land Ecosystem Model (DLEM) is a highly-integrated process-based terrestrial ecosystem model that simulates daily carbon, water and nitrogen cycles driven by the changes in atmospheric chemistry including ozone and nitrogen deposition, climate, CO_2 concentration, land-use and land-cover types and disturbances (i.e., fire, hurricane, and harvest) (Figure 2). The DLEM has been extensively used in studying the terrestrial carbon, water and nitrogen cycles over Monsoon Asia, the continental U.S., North and South America (e.g., Tian et al., 2005, 2008, 2010a, b, c, d; Xu et al. 2010; Chen et al., 2006b; Ren et al., 2007a, b, 2010; Liu et al., 2007, 2008; Zhang et al., 2008; Schwalm et al. 2010).

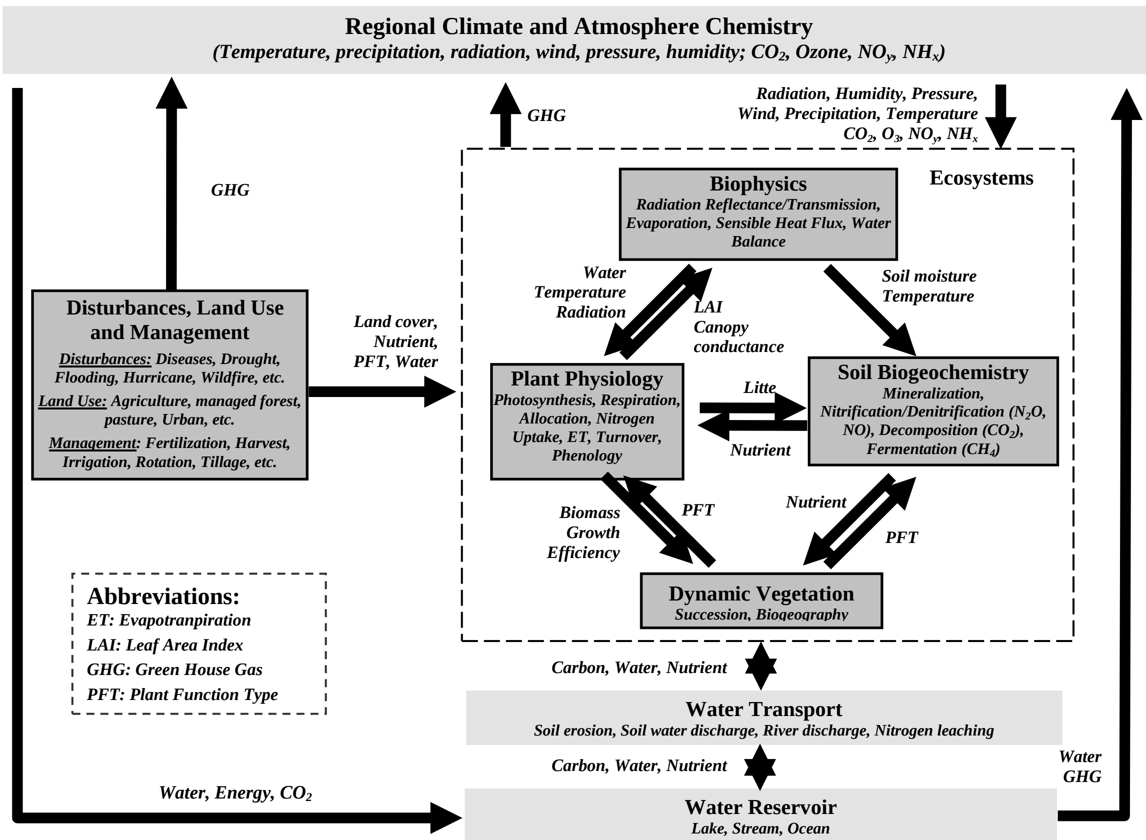

Figure 2 Conceptual model of the Dynamic Land Ecosystem Model (DLEM). Note: ET: Evapotranspiration; LAI: Leaf Area Index; PFT: Plant Functional Type (Tian et al. 2010a, b, c).

DLEM includes four core components (Figure 2): 1) biophysics, 2) plant physiology, 3) soil biogeochemistry, 4) dynamic vegetation, and 5) disturbance, land use and management. The biophysical component includes the instantaneous exchanges of energy, water, and momentum with the atmosphere, which involves micrometeorology, canopy physiology, soil physics, radiative transfer, water and energy flow, and momentum movement. The plant physiology component in DLEM simulates major physiological processes such as photosynthesis, respiration, carbohydrate allocation among various organs (root, stem and leaf), nitrogen uptake, transpiration, phenology, etc. The component of soil biogeochemistry simulates mineralization, nitrification/denitrification, decomposition and fermentation so that DLEM is able to estimate simultaneous emission of multiple trace gases (CO_2, CH_4 and N_2O). The dynamic vegetation component simulates two kinds of processes: the biogeography redistribution of plant functional types under environmental changes, and plant competition and succession during vegetation recovery after disturbances. Like most Dynamic Global Vegetation Models (DGVMs), DLEM builds on the concept of plant functional types (PFT) to describe vegetation distributions. The disturbance, land use and management component in DLEM simulates the impacts of anthropogenic and natural disturbance, land use change, and land management on carbon, water and nitrogen cycles.

Forest harvest and disturbance module in DLEM

Harvest/thinning of forest ecosystems

Harvests/thinning are implemented by removing specific cohorts of specific species on sites selected for harvest. The framework of DLEM harvest/thinning module is shown in Figure 3. We separate the harvest/thinning regimes into 4 types. These harvest regimes vary in the number of entries required to complete a silvicultural treatment, and in whether they are applied to different harvest methods. In this version of model, only one entry is considered. So the harvest/thinning regimes include: 1) an entire stand harvest, 2) partial stand harvest (unit:

patches), 3) thinning (partial or whole-tree thinning), and 4) selective harvesting (unit: single lines/rows).

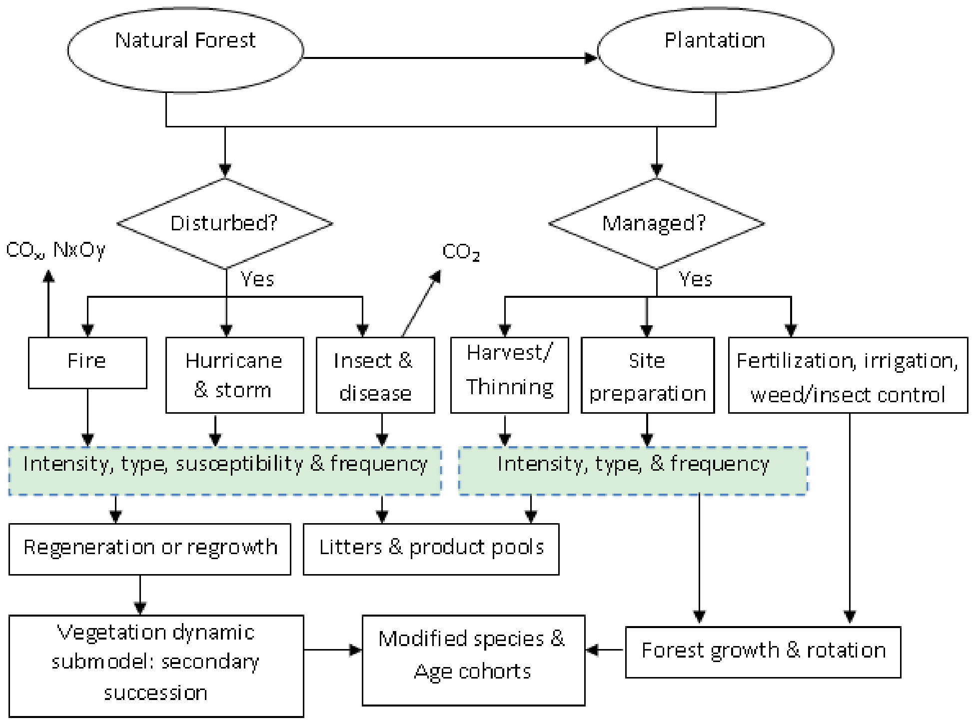

Figure 3 Structure of the forest management and disturbance module in DLEM

If a specific forest ecosystem under study is managed, part or all of the tree biomass will be removed through thinning, selective harvesting or clear-cutting. The harvested biomass is subtracted from the living biomass, and is allocated to different products pools and soil litter pools. Harvest in year *k* in age cohort *i* is defined as a fraction of the existing biomass in that age cohort (fHB_{ik}). The total cohort number is *N* ($i <= N$). If a forest stand is a pure forest land, *i* equal to 1. The harvested biomass fraction is divided into *j* components of foliage, stems, branches and roots. The harvested fraction (or intensity) of these *j* components could be the same (clear-cut and whole-tree selective harvesting/thinning) or different (partial thinning). Total harvested biomass (HB_{ik}) is thus calculated as:

$$HB_{ik} = \sum_{i=1}^{N}(B_{ijk} \times fHB_{ik})$$

Variation in the intensity of harvest activity is controlled by rules governing the removal of age cohorts of species found on the site being harvested. The rules for each harvest regime are specified by the user in the form of removal mask maps that for each species specify which age cohorts (if present on the site) are to be removed. The thinning methods, intensity and timing will be specified by the users and thinning only occurred in the managed forest plantations.

If the existing fraction of biomass after harvesting is less than 20% (default), the managed forest will be restart as forest age is set as 0 and forest type is the same with previous one.

Allocation of harvested/thinned biomass

The harvested biomass is first allocated to three pools: slashes, roundwood, and fuelwood (Figure 4). The roundwood will then be allocated different wood product pools, the slashes will be allocated different litter pools, and fuelwood will be burnt as energy and the C and N will be emitted immediately into atmosphere. The fuelwood may be used to replace fossil fuel as energy, partially reducing C emissions from fossil fuel. The fates of wood products are tracked in detail in the DLEM wood products module (Figure 4), while the slashes will be added into different existing litter pools that defined by DLEM (Figure 3):

$$When\ 1 \leq k \leq 3, LT_{ijk} += HB_{ik} \times fLT_{ijk}$$

$$When\ k = 4, CWD += HB_{ik} \times fLT_{ijk}$$

Where, *i* denotes the litter pool from 2 sources (1: aboveground litter, 2: belowground litter pool); *j* denotes the litter pools with different decomposition rate (1: labile, 2: middle, and 3: resistant litter pools); *k* denotes the allocation fraction of different components of removed tree biomass (1: leaf; 2: fine root; 3: coarse root; 4: stem) to litter pools with different decomposition rates. There are 3 types of litter pools as differentiated with decomposition rate: labile (decompose very fast), middle (decompose relatively fast) and resistant (decompose slowly) litters. Leaves and fine roots are allocated to these 3 types of litter pools according to a fixed proportion, coarse roots are only allocated to middle and resistant pools, and stems (including branches) that are not allocated to product pools will be allocated to coarse woody debris (CWD). In DLEM, two types of CWD pools are further differentiated according to their decomposition rate: fast decomposing CWD (including small size branches and barks) and slow decomposing CWD (including boles and large size branches).

The harvested roundwood will enter into different product pools:

$$PROD_{ij} = HB_{ij} \times (1 - fLT_i)$$

Where, $PROD_{ij}$ is the allocation of harvested forest biomass component *i* (leaf, stem, root) to product pool *j* (can be different end-product: sawnwood, paper, boards and firewood or different half-life product pools: long-term, medium-term and short-term); HB_{ij} is the allocation proportion of harvest biomass of component *i* to product pool j; fLT_i is the total allocation proportion of component *i* to litter pools.

A module for tracking the life cycle of wood products

The products module of DLEM tracks the C and N fates after the removed biomass is allocated into the product pools (Figure 4). In the same year as the harvest/thinning takes place, several intermediate processing and allocation steps are done, until the product resides in the end products, the millsite dump, or is released through energy. When the end products are discarded at the end of their lifespan, they might be either recycled, or deposited in a landfill, or be used for energy. C and N are released to the atmosphere through decomposition either at the millsite dump, or at the landfill, or via the energy. This module is developed based on CO2FIX Model which was developed by Schelhaas et al. (2004).

Stem and other parts of biomass are the inputs to the products module. The products module distinguishes three categories of end products: long term, medium term and short term products. Each of the commodities (sawn wood, boards & panels, and pulp & paper) is distributed over these end product categories. When end products are discarded, they can be recycled, be deposited in a landfill, or be used for energy. A product can only be recycled to the same life-span classes or lower classes. From the landfill and the mill site dump C and N are released directly to the atmosphere. The half-life span could be different when this product module is used in different regions. Due to the recycle of some end-use products, the product decay rate is not the same in each year. Generally, the simple exponential decay function is used to track the decay processes of different end products (including end products at landfill and mill site).

$$P_{i,t+1} = P_{i,t} \times (1 - \ln 2 \div HL_i)$$

Where, $P_{i,t}$ is the amount of C or N in product types of i at time t (years); $P_{i,t+1}$ is the amount of C or N in product types of i at time t+1. HL_i is the half life for product type i.

The parameter values of roundwood allocations to different intermediate wood products and end-use products are listed in Tables 1, 2 and 3.

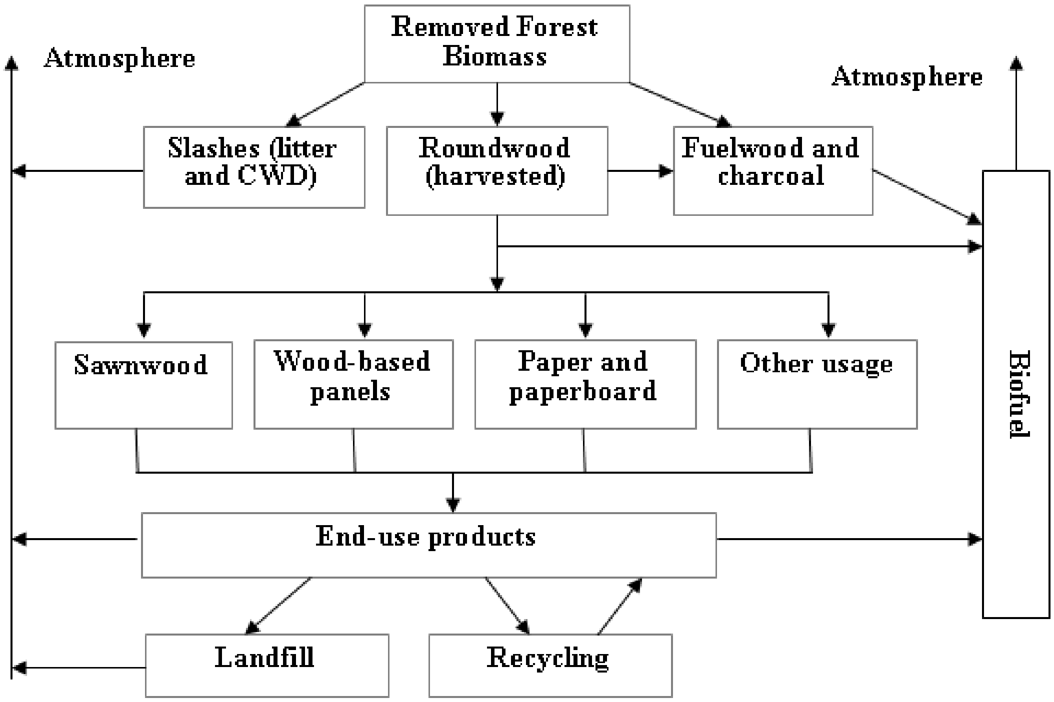

Figure 4 Flow diagram of DLEM forest product life cycle module as modified from Schelhaas et al. (2004) and Heath et al. (2003).

Site preparation

After harvest/thinning and before planting new seedlings, the harvested/thinned sites will be prepared to establish new plant species. Slash burning, fertilization, and slash fragmentation are often used before new vegetation establishment. The C and N in the aboveground litter, soil organic matter, and even the belowground litter will be released to the soil or to the atmosphere. Burnt C and N for different pools are calculated as:

$$BC_i = \sum_{j=1}^{4}(LT_{ij} \times BR_{ij})$$

$$BN_i = \sum_{j=1}^{4}(LT_{ij} \times BR_{ij})$$

$$BC = \sum_{i=1}^{3} BC_i$$

$$BN = \sum_{i=1}^{3} BN_i$$

Where, BC_i and BN_i are the burnt carbon and nitrogen from source i, respectively (3 sources: aboveground litter, belowground litter, and SOM). LT_{ij} is the litter biomass for j litter pool of decomposition rate (4 litter pools: CWD, labile litter, middle litter, and resistant litter pools). BR_{ij} is the burning intensity (percent of the burnt litter) for litter pool j. Litters include both previous litters on the forest floor or in the belowground soil, and the slashes left on the site after disturbance/harvest/thinning.

Some of the burnt C and N can be released as trace gases (e.g., CH_4, CO_2, CO, N_2O, and NO) or others return back to the soil C and N pools through ash deposition. The allocations of burnt carbon to different pools are calculated as (referenced from Andreae and Merlet 2001):

$$BC_m = BC \times E_m \times \frac{12}{W_m}$$

$$BN_n = BC \div 0.45 \times E_n$$

$$LEFTC = BC - \sum_{m=1}^{4} BC_m$$

$$LEFTN = BN - \sum_{n=1}^{4} BN_n$$

Where, m is the C-related gas type emitted from slash burning (Four types of C-related gases: 1, CO_2; 2, CH_4; 3, CO; and 4, NMHC); W_m is the molecular weight of C-related gas m; E_m is the C emission factor for species m (g kg^{-1} C); n is the N-related gas type from slash burning (Four types: 1, N_2O; 2, NO_y; 3, NH_3; and 4, N_2); E_n is the emission factor for N-related gas n. N-related gas emissions are related to the burnt dry matters (BC/0.45, DLEM assumes that C concentration in dry matters is 45%) as suggested by Andreae and Merlet (2001); BC_m and BN_n are released C and N for gas types *m* and *n*, respectively; *LEFTC* and *LEFTN* are C and N that have not emitted into atmosphere but return back to the soil or litter pools. *LEFTC* will enter in the resistant litter C pool, while *LEFTN* will enter soil available N pools.

The N that has not been emitted as N gases (*LEFTN*) returns to the soil as ashes with a format of NH_4-N. Therefore, the soil available N pools are changed to:

$$AV_{NH4} += LEFTN$$

Where, AV_{NH4} is the soil available NH_4-N. According to this, site preparation will cause a sudden increase in soil available N. However, due to lower vegetation coverage, higher leaching, and increased N immobilization due to increased litter inputs, soil available N will soon decrease too.

The major parameters and their values for the land management and disturbance module in DLEM are listed in Tables 1 to 5.

Table 1 Allocation of removed biomass to different middle product pools.

Raw materials	Production line			
	Sawnwood	Boards	Paper	Firewood
Logwood	0.8	0.15	0.05	0
Pulpwood	0	0.05	0.9	0
Slash	0	0	0	1

Table 2 Harvested wood allocations to different half-life span product pools.

Products	Long term (40 yrs)	Medium term (15 yrs)	Short term (1 yr)
Sawnwood	0.5	0.25	0.25
Boards wood	0.3	0.5	0.2
Paper & fire wood	0.01	0.1	0.89

Note: Landfill half-life span: 145 yrs

Table 3 Allocation for the end products.

Products	recycling	energy	Landfill
Long term	0.1	0.8	0.1
Medium term	0.1	0.8	0.1
Short term	0.2	0.75	0.05

Table 4 Proportion of biomass left on site after forest disturbance.

Harvest biomass	Leaves	Roots	Reproduction	Stem*
Temperate broadleaf forest	1.0	1.0	1.0	0.4
Temperate needleleaf forest	1.0	1.0	1.0	0.3

* Stem includes branches, boles and barks, most of large-size branches and all boles are removed.

Table 5 The burnt portion of slashes, and C and N gas emission factors after slash burning for forests (Parameter values are from Andreae and Merlet 2001).

Litters or residues	Slash burn proportion		Carbon gas emission factors after burning			
	Belowground litter	Aboveground litter	E_{CO2}	E_{CH4}	E_{CO}	E_{NMHC}
Temperate broadleaf forest	0.2	0.7	3484	10.7	236	12.7
Temperate needleleaf forest	0.2	0.8	3484	10.7	236	12.7

Table 5 Continued.

Litters or residues	N gas emission factors			
	E_{N2O}	E_{NOy}	E_{NH3}	E_{N2}
Temperate broadleaf forest	0.26	3.0	1.4	3.1
Temperate needleleaf forest	0.26	3.0	1.4	3.1

Model sensitivities to forest disturbances

To evaluate the model behaviors to forest disturbance, model sensitivity to key variables were tested. Two most important variables were selected: wood salvage rate after disturbance and forest disturbance rate.

Sensitivity to wood salvage rates

Wood salvage rates after forest disturbance could greatly influence forest floor litter C and soil C. Six levels of wood salvage rates (i.e., 0%, 20%, 40%, 60%, 80%, and 100%) were designed to evaluate the model behaviors. Under all salvage rates, soil C increased right after a disturbance event (50% mortality rate) and then soon decreased (Figure 5A). After a certain time span, soil organic C began to increase. This time span for soil C recovery was influenced by wood salvage rates. The less wood salvage, the faster of the soil C restored to pre-disturbance level. However, combining the total C storage in soil and wood products, higher wood salvage rate could maintain ecosystem C sink in a longer time period (Figure 5B).

Sensitivity to different disturbance rates

Five forest disturbance (mortality) rates were designed: 10%, 40%, 70%, 100% (only once), and continuous 2% mortality rates. Vegetation C gradually recovered to its pre-disturbance level, and recovery rate was faster for the smaller disturbance rates (Figure 6A); however, if continuous disturbance occurred (2% mortality), vegetation was not able to recover to its pre-disturbance level, implying continuous small disturbance events could result in more C losses in vegetation in the long run. Soil C storage decreased faster for higher disturbance rates and took a longer time for recovery (Figure 6B). One-time small disturbance events (e.g., 10%) did not significantly change total ecosystem C storage, while higher disturbance rates could significantly decrease total ecosystem C storage and even after as long as 100 years, the clear-cut forests could not recover to its pre-disturbance level. Continuous small disturbance events could reduce more ecosystem C storage than that of 100% one-time disturbance rate (Figure 6C), suggesting small continuous small disturbance rate could result in larger C emissions and small disturbance events could not be ignored in simulating ecosystem C storage.

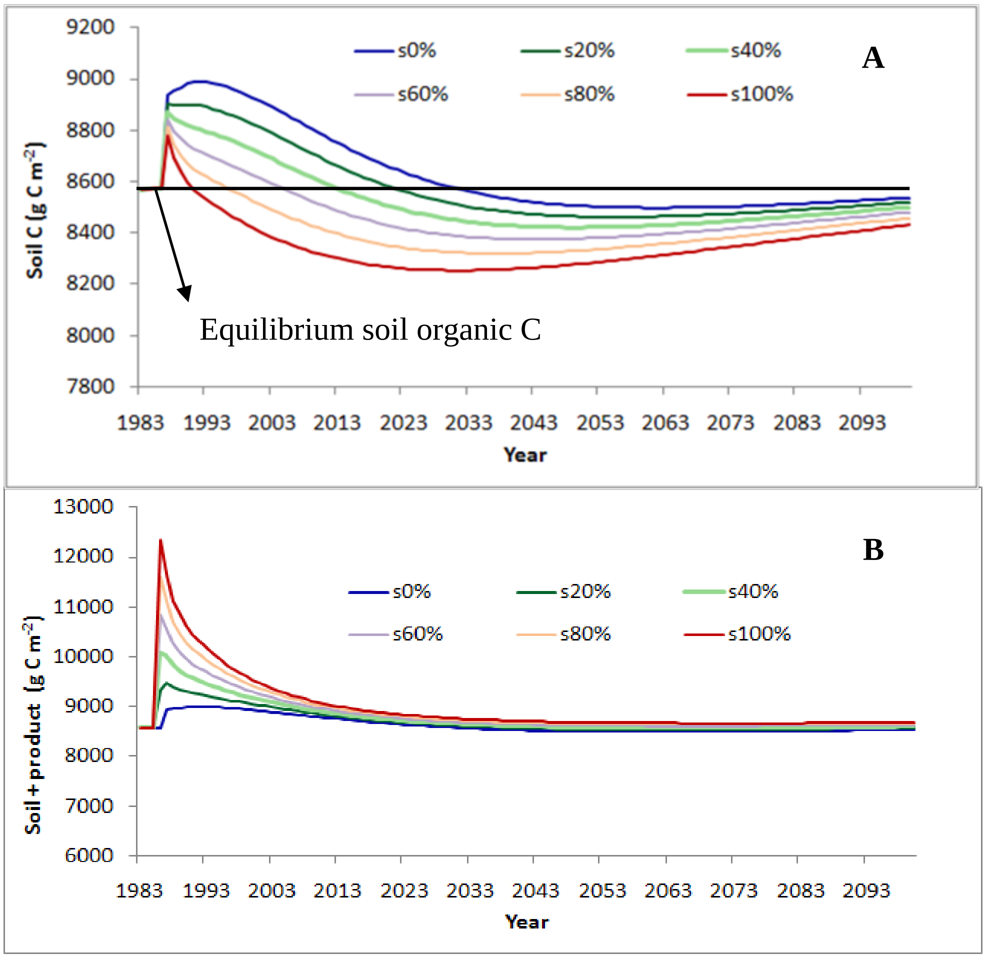

Figure 5 Sensitivities of soil organic C storage (A) and soil + in-use product C pool (B) to different wood salvage rates (i.e., salvage harvest after disturbances) after a disturbance event (50% mortality rate). Different wood salvage rates: s0%, s20%, s40%, s60%, s80%, and s100% represent that 0%, 20%, 40%, 60%, 80%, and 100% dead stems are removed as wood products after a disturbance event. The lower wood salvage rate, the faster is the soil organic C to recover to its pre-disturbance status.

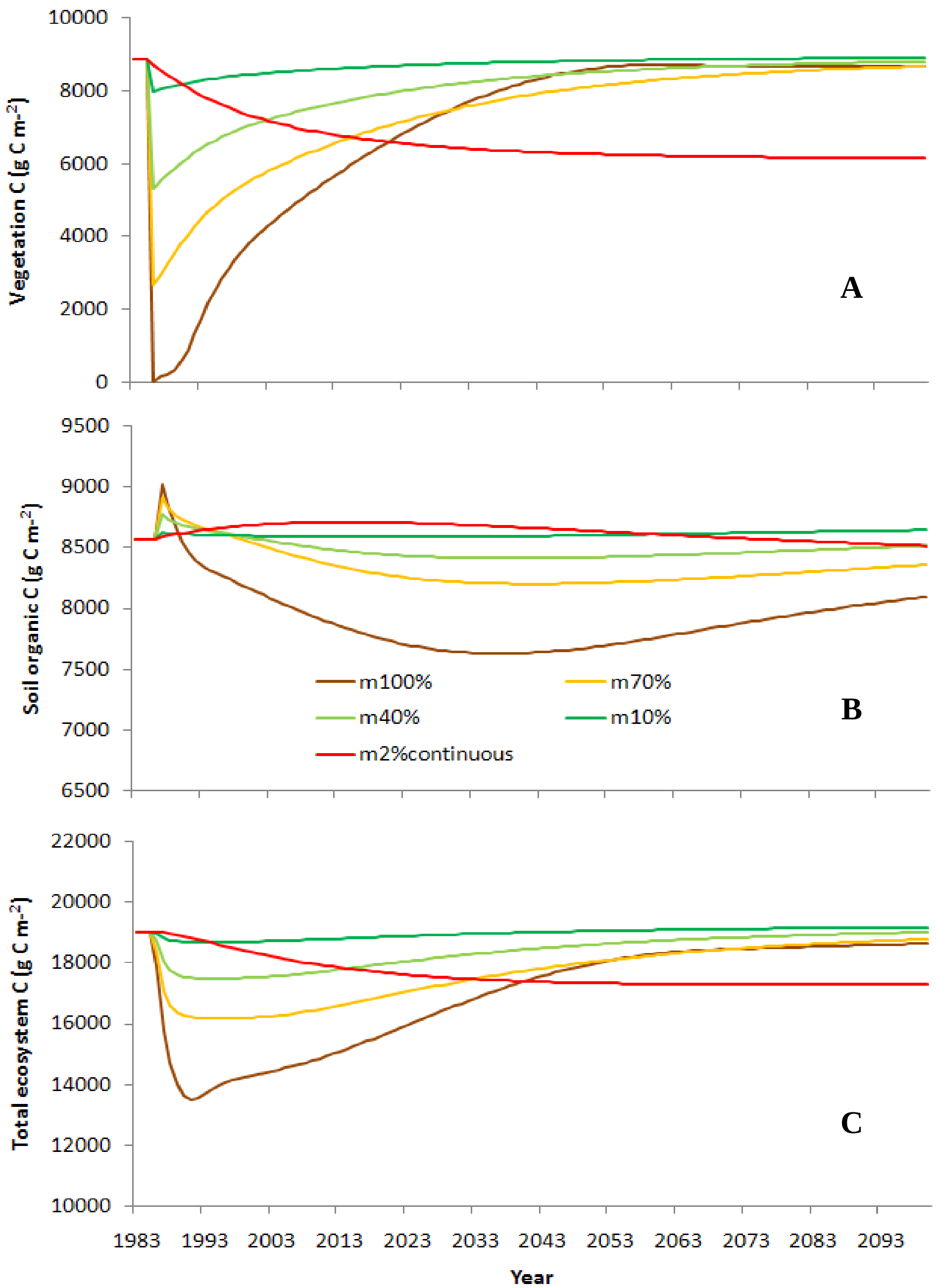

Figure 6 Sensitivities of ecosystem vegetation C (A), soil organic C (B), and total C (C; i.e., soil organic C + vegetation C + litter C + in-use product C) to different disturbance rates. Disturbance rates: m10%, m40%, m70%, m100%, and m2%continuous represent that 10%, 40%, 70%, 100% (only once in 1986), and 2% (every year since 1986) mortality rate in disturbance events.

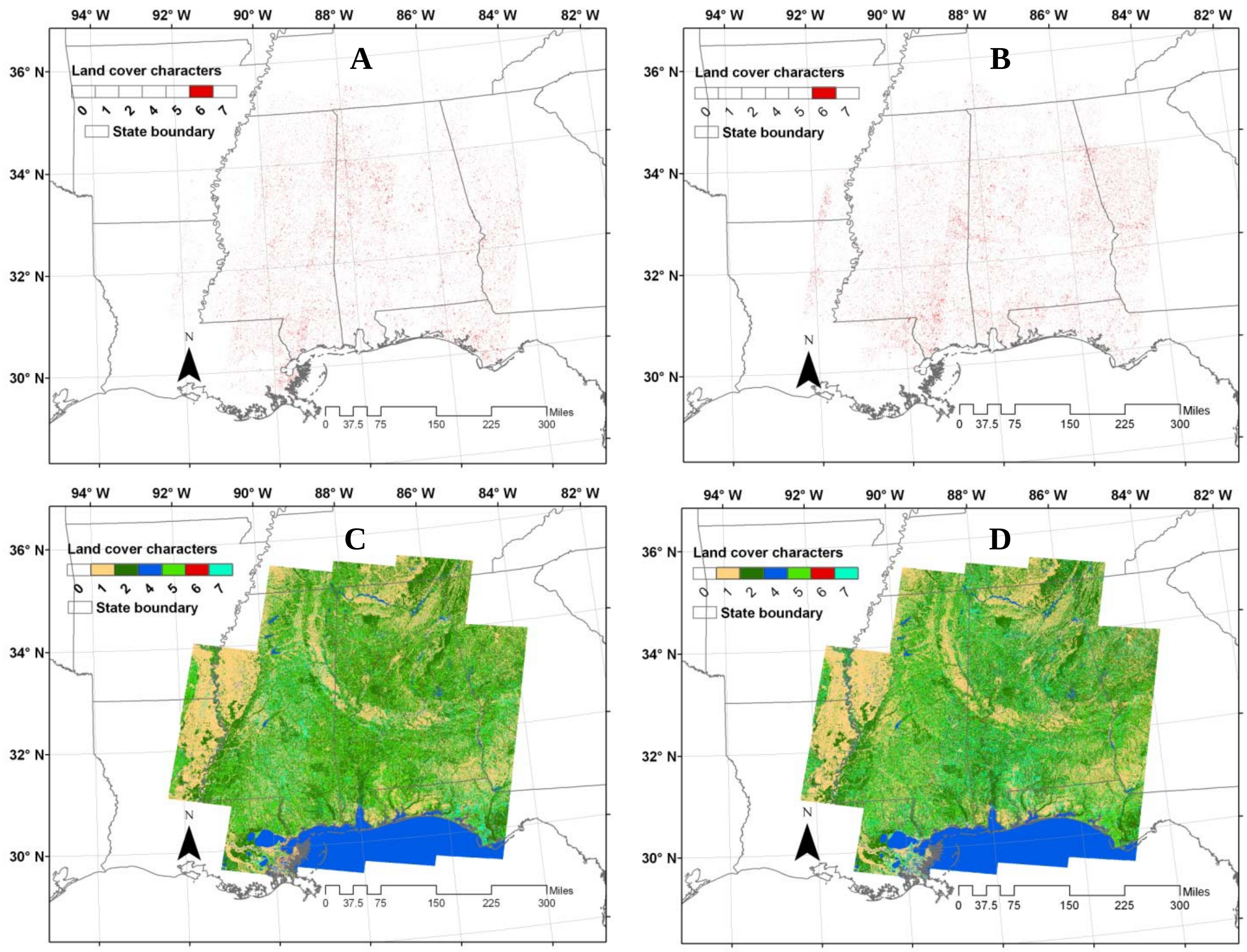

Figure 7 Forest disturbed areas (read colored areas) in 1986 (A) and 2007 (B, include 5 scenes of images from 2005 and 2006) and other land cover characters in the two years (C: 1986; D: 2007) in AL and MS based on classification of Landsat TM images (Li et al. 2009a,b; Huang et al. 2009 a, b; 2010). Note: 17 scenes of Landsat TM/ETM+ images were used to cover the study region and the spatial resolution is 30 m. Major categories of land cover characters include: 0 – background area; 1 – persisting nonforest; 2 – persisting forest; 4 – persisting water; 5 – previously disturbed but looked like forest by this year; 6 – disturbed in this year; 7 – post-disturbance nonforest.

Data description

The data collection and generation for forest harvest and disturbance intensity have been described in Huang et al. (2009a, b) and Li et al. (2009a, b). Based on a large collection of many scenes of Landsat TM images and a vegetation change tracker (VCT) model, which is an automated forest change mapping algorithm designed for analyzing dense time series stacks of Landsat images, Li et al. (2009a, b) and Huang et al. (2009a, b) derived the forest change information for Mississippi and Alabama. Through the developed forest change information, Huang et al. (2009a, b) identified the forest disturbance area in each investigation year. 7 categories were classified (Figure 7).

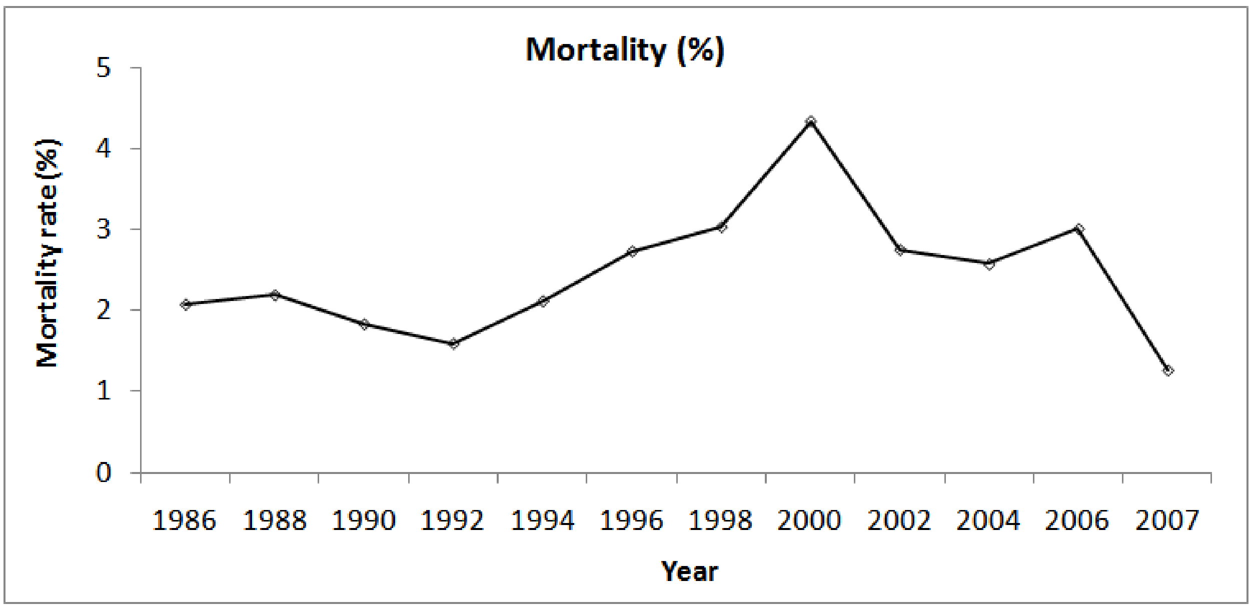

Figure 8 Mean forest disturbance rate (%, per two years) for Alabama and Mississippi, USA during 1985-2007.

RESULTS AND ANALYSIS

Forest mortality rate during 1985-2007

Forest mortality rate induced by harvest or disturbance increased during 1985-2000 and then decreased for the entire study region (Figure 8). During the study period, annual forest mortality rate is about 1.3% and could be accumulatively about 100% for some areas since there were multiple disturbance events occurring during the study period. Although the annual mean mortality is relatively low, the accumulative disturbance events could greatly disturb the forest ecosystems and result in significant changes in C, N and water cycles. At the spatial scale, maximum forest mortality rate increased over time during 4 periods (1985-1989, 1990-1994, 1995-1999, and 2000-2007. Figure 9). The maximum mortality rate could be up to 50% during 2000-2007.

Changes in carbon storage after disturbance

Although mean annual forest mortality (1.3%) is relatively low, this small but continuous forest disturbance induced a large C emission from the forest ecosystems in Mississippi and Alabama. From 1984 to 2007, in-use wood product and soil organic C continuously increased, vegetation C continuously decreased, while litter C (Aboveground + belowground litter) first increased and then decreased (Figure 10). During 1984-2007, forest disturbance has totally resulted in C emission of 199.75 Tg (8.32 Tg C yr^{-1}, Table 6). Among four C pools, product and soil carbon pools were C sink and increased by about 135.86 and 8.58 Tg C, respectively. Litter and vegetation carbon pools were carbon source and decreased by 21.44 and 322.74 Tg C, respectively. Wood product was the biggest C sink induced by forest disturbance, implying a significant role of wood salvage in preserving ecosystem C storage after forest disturbance events. The results also suggested that the recovery of vegetation C pool after disturbance was slower than that of soil organic C pool, which resulted in a continuous decrease in forest biomass during 1984-2007.

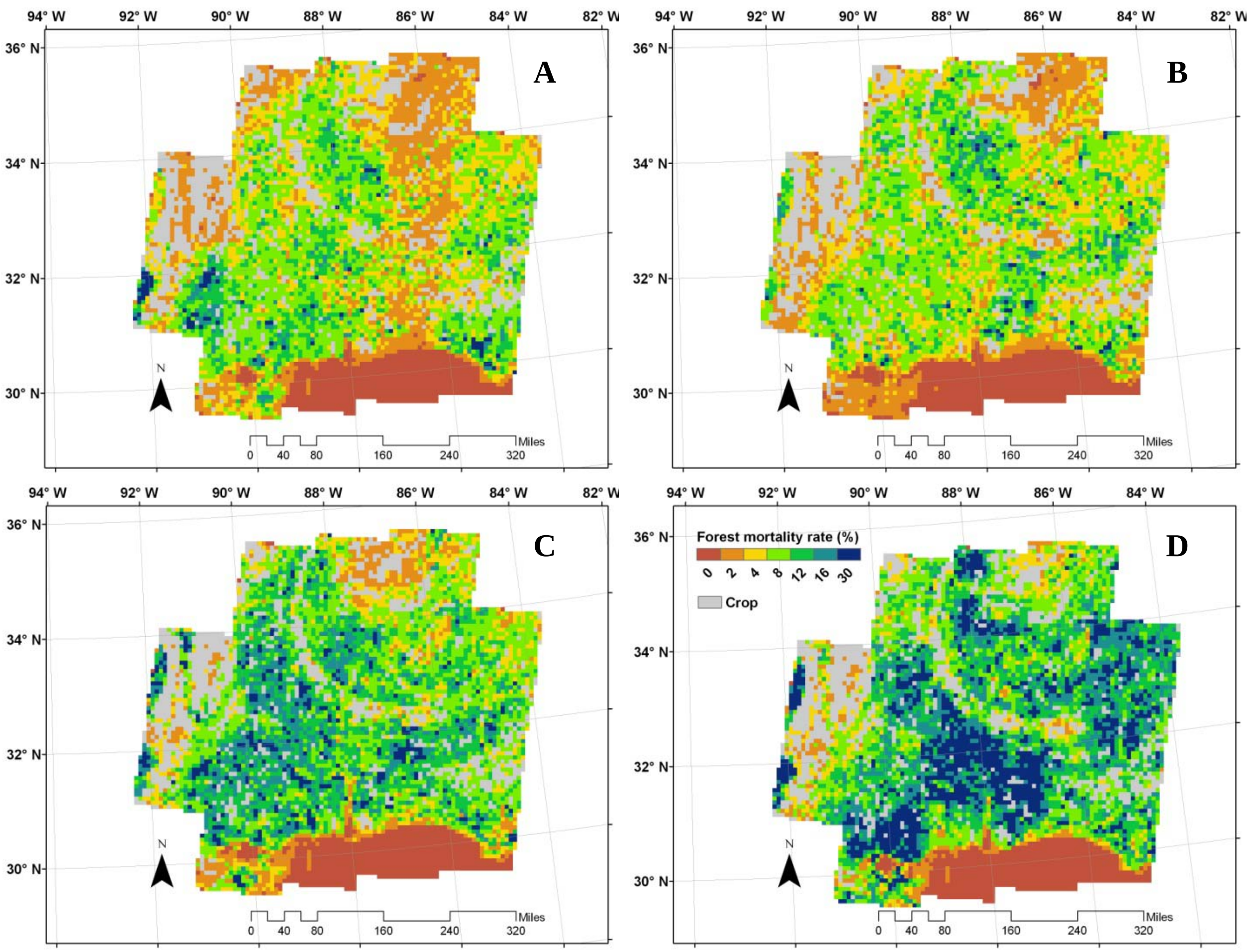

Figure 9 Spatial distribution of forest disturbance rate (%) for different time periods during 1985-2007 (A: 1985-1989; B: 1990-1994; C: 1995-1999; D: 2000-2007). Note: the data spatial resolution is 8 km which is aggregated from the 30 m classified Landsat TM/ETM+ images.

Table 6 Changes in different C pools caused by forest disturbance in Alabama and Mississippi from 1984 to 2007 (Tg C).

Carbon pools	Litter C	SOC	Vegetation C	Product C	Total
1984	572.76	4463.16	3140.25	0.05	8176.22
2007	551.31	4471.73	2817.51	135.91	7976.47
Difference	-21.44	8.58	-322.74	135.86	-199.75
Change rate (%)	-3.74	0.19	-10.28		-2.44

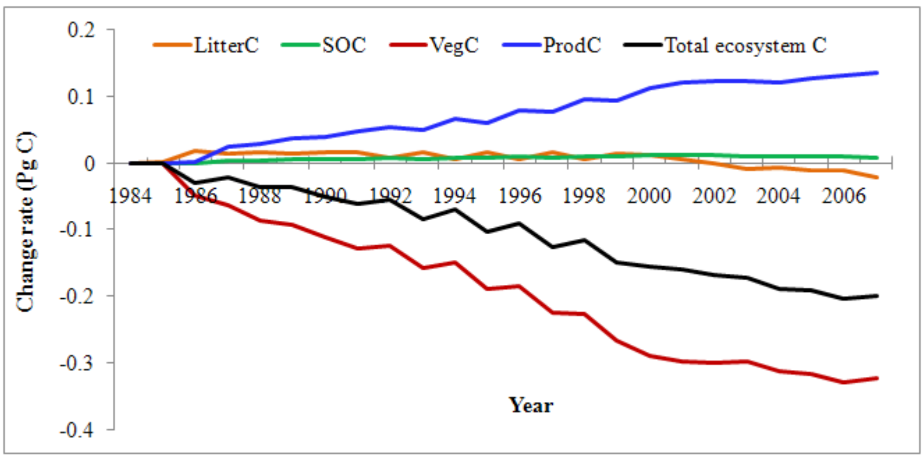

Figure 10 Inter-annual variations in different C storage pools (Pg C, 1 Pg = 10^{15} g) caused by forest disturbance in AL and MS from 1984 to 2007. Positive values indicated a C sink whereas a C source.

Spatial variations in carbon fluxes caused by forest disturbance

The mean annual mortality rate induced by forest disturbance varied spatially and temporally (Figures 9 and 11B), resulting in a large spatial variation in total net C exchange during 1984-2007. Although tree mortality rate is relatively small in some areas, the accumulated impacts are very large (Figure 11A). The accumulated carbon emission could be up to 1000 g m^{-2}, which means about 10% of the forest C storage in aboveground pool (aboveground litter and biomass) could be emitted during the 23 years. Compared Figure 11A with Figure 11B, we found that the higher mortality rate could induce more C emission, which further implied that the forest ecosystem C could not be recovered in a short term especially after continuous forest disturbance events.

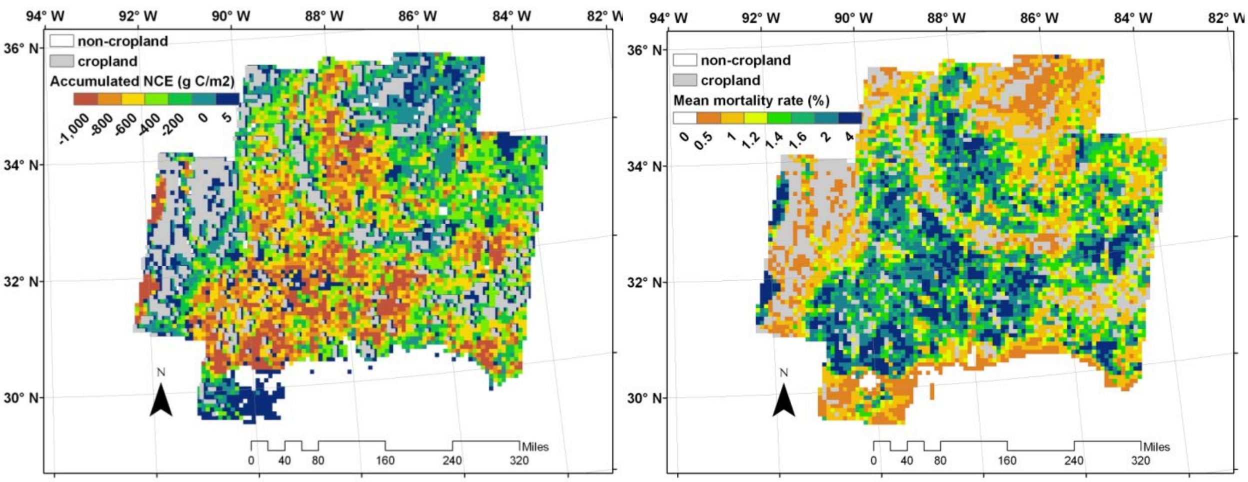

Figure 11 Accumulated net carbon exchange (NCE, positive indicate carbon sink, g C m^{-2}) and mean annual forest mortality rate (%) during 1984-2007 in Mississippi and Alabama.

DISCUSSION

Small continuous disturbance and forest C sink in the United States

Due to lack of high resolution data, the impacts of forest disturbance on C fluxes at a relatively large scale are very difficult to study. However, many researchers have recognized the important role of small-scale disturbance in C fluxes (e.g., Woodbury et al. 2007; Goward et al. 2008; Pan et al. 2010; Birdsey et al. 2006). On a large scale such as national or continental, forest disturbance is generally assumed to occur at a large extent and has a long returning interval (such as 20 or 30 years); but under most situations, forest disturbance may frequently occur on a small scale, and only a small part of the forest ecosystem is disturbed. So generally, model simulations with aggregated large-scale forest disturbance data could either underestimate or overestimate C fluxes. Our estimation found that continuous small-scale disturbance in Mississippi and Alabama could result in a C source of 0.20 Pg (1 Pg = 10^{15} g) or 0.01 Pg yr^{-1}. The forest ecosystems in the United States were reported as a C sink ranging from 0.11-0.15 Pg yr^{-1} (Pacala et al. 2001). If take into account small-scale forest disturbance such as those in Alabama and Mississippi, this C sink could be greatly reduced. FAO report (2005) estimated that 104 million ha per year of the world's forests, or 3% of the total area, were disturbed each year by fire, pests, and weather, though this was a significant underestimate of the disturbance rate because of incomplete reporting by countries. This suggested that small-scale (high spatial resolution) disturbance data could be of importance for accurately estimating C storage changes following forest disturbance in the United States and the North America (Pan et al. 2010; Huang et al. 2009a, b).

Change patterns of different carbon pools and legacy effects following forest disturbance

Forest disturbance (including harvest) and subsequent recovery could have a strong legacy effect on the forest ecosystem C, water, and N dynamics. Since biomass increases with stand age, delay harvesting disturbance to the age of biological maturity may result in the formation of a larger C sink (Alexandrov and Yamagata 2002; DOE 2007). We found that forest disturbance had significantly decreased forest biomass in Mississippi and Alabama (Figure 10). Disturbance had a long-term legacy effect on forest biomass accumulation. If no disturbance events occurred after 2007, forest biomass could not be fully recovered even after about 50 years (from 2007 to 2050, Figure 12).

The classic Covington’s curve described differences in organic matter storage in the forest that had been harvested at different time (Covington 1981). Through collection of a large number of field experiment data, Johnson et al. (1992, 2001) conducted a meta-data analysis to find the relationships between forest harvest and soil carbon storage. They concluded that soil C could decrease or increase after forest harvest and which depends on the harvest intensity and regimes. Yanai et al. (2003) revisited the Covington’s curve and the relationships found from Johnson et al. (2001) and concluded that forest harvest has a much smaller effect on forest floor and soil C pools than was predicted from early interpretations of Covington’s curve. In this study, we noted that forest soil C slightly increased by 8.58 Tg C due to continuous forest disturbance during 1985-2007. This is due to the continuous litter input into the soil. However, the increasing trend in soil organic C could be stopped due to decreases in litter C input in a long-term period (Figure 12). This implied that soil C could increase in a short term due to continuous and small disturbance rate but could decrease in a long-term period or after large disturbance events.

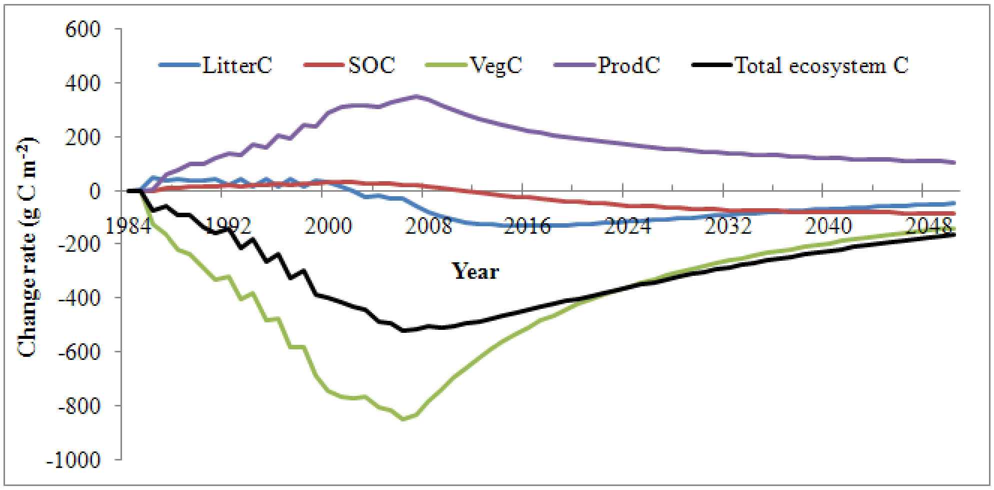

Figure 12 Changes of different C pools after continuous forest disturbance during 1985-2007 and disturbance legacy effects during 2008-2050 in AL and MS. Note: total ecosystem carbon includes vegetation, litter, soil organic, and in-use wood product C. We assume no forest disturbance occur after 2007.

Uncertainties

Currently, it is still a great challenge to assess disturbance impacts in estimates of forest carbon budgets (Pan et al. 2010). Two major uncertainties might exist: disturbance data accuracy and representative of modeling algorithms. In this study, the use of ~30 m high-resolution forest mortality data as classified from Landsat TM images could ensure a relatively high input data accuracy. In addition, the DLEM model used in this study has fully coupled ecosystem-level C, water and N cycles, and a full tracking module for carbon budget after forest disturbance. This could have improved the assessment accuracy for disturbance impacts. However, many uncertainties still exist. For example, DLEM assumes trees will regenerate right after a disturbance events and no competition among different biomes during forest succession. In addition, the allocation parameters were assigned based on average conditions in the Southern

United States for different wood product pools (i.e., 1-, 10-, and 100-yr product pools, and landfill), wood product salvage rate (i.e., the proportion of harvested biomass to slashes and wood products), and site preparation intensity (such as, slash burning portion and burning intensity). These uncertainties could alter the drawn conclusions in a certain degree.

CONCLUSIONS

Based on high resolution (30 m) forest mortality rate data and a process-based global biogeochemical model, this study estimated the changes of C storage in Mississippi and Alabama after continuous forest disturbance. Results indicated that mean annual forest mortality during 1985-2007 is about 1.3% with a large spatial variation. The accumulated forest mortality during 1985-2007 for some areas could be up to 100%. The continuous forest mortality has resulted in a large carbon source (0.20 Pg C or 8.3 Tg C yr^{-1}) in Alabama and Mississippi, which accounts for about 2.4% of the total carbon storage in this region. Forest ecosystems in the United States was reported as a large carbon sink during the 1980-present; however, if small-scale disturbance is considered in the estimation, this C sink could be greatly reduced. Although there are many uncertainties, a roughly estimation in this study to the changes of carbon budget could be of great implications to the related studies in the entire United States and North America.

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Chapter 7

Evaluating Potential Impacts of Increased Forest Plantation Area and Intensive Forest Management Practices on Carbon Storage and Nitrous Oxide Emissions during 1925-2040 in the Southern United States

ABSTRACT

The Southern United States (SUS) produces about 60% of the U.S. timber products and comprises one-half of the world's industrial plantations. Forest management practices are commonly used to improve productivity. The increases in forest plantation areas and more intensive management practices could have significant imprints on terrestrial ecosystem structure and function. It is of importance to identify and apply best management strategies to intensify C storage and reduce global warming potential in this region. In this study, based on generated regional level data sets and a global biogeochemical cycle model, we strived to quantify and evaluate the impacts of increased forest plantation areas and different intensive management practices on C storage and nitrous oxide (N_2O) emissions during 1925-2040 in the SUS. Results suggested that increased areas and improved genetics for the forest plantations could not ensure an increased C pool and reduced N_2O emissions if management practices were not appropriately employed. Longer forest rotation length could increase ecosystem C storage and reduce N_2O emission, while shorter rotation length could cause large amounts of CO_2 and N_2O emissions. An extension of 20 years above current forest rotation length could result in an additional C sequestration of 1200 g C m^{-2} during 1925-2040 in the SUS. Nitrogen fertilization resulted in high plantation productivity and ecosystem C storage; however, the concomitant increases in

N_2O partially offset its functions for global warming potential mitigation. Slash burning could slightly increase NPP and forest biomass in a short period; however, the decreases in litter and soil C make slash burning a C source. Although large uncertainties exist both in data sets and modeling mechanisms, this regional level study could help the local policy makers and land managers to understand the consequences of different management practices in a large-scale view and offer valuable guidance.
Key words: Forest plantation; forest management; carbon storage; nitrous oxide; global warming potential; Southern United States

INTRODUCTION

The Southern United States (SUS) produces about 60% of the U.S. timber products and more industrial timber than any other region of the world, where includes almost one-half of the world's industrial forest plantations (Prestemon and Abt 2002; Fox et al. 2007). In part, the role of the SUS as a woodbasket of the world is due to increasing productivity of pine plantations (Prestemon and Abt 2002; Samuelson et al. 2008), resulting from improved genetics and use of more intensive management practices, such as N fertilization, thinning, weed control, and herbicide use (Johnsen et al. 2001; Fox et al. 2004, 2006). Intensive forest management has been widely applied in the SUS, which accounts for ~33% of the country's fast-growing industrial wood plantations (Roise et al. 2000; Wear and Greis 2002). By 2007, there were ~18 million ha of pine plantations, accounting for ~20% of the forest land in the SUS (Smith et al. 2009). Loblolly pine (*Pinus taeda* L.) is the most commonly planted and intensively managed species in this region (Allen et al. 1990; Fox et al. 2007). Forest plantations generally have higher productivity than unmanaged forests. Excluding management practices, this is primarily due to better seedlings and genetic improvement (Buford and Burkhart 1987; McKeand et al. 2006). So even not managed after planting, forest plantation productivity should be higher than natural forests. Most of the pine plantations in the SUS were genetically improved types as indicated by McKeand et al. (2003).

For many forest landowners in the SUS, intensive managements to forest plantations have become routine practice (McKeand et al. 2006). Silvicultural activities such as site preparation, planting genetically improved seedlings, management of plant competition, and fertilization are routinely applied during the rotation (Allen et al. 2005; Jokela et al. 2004a).One of the largest gains in biomass production in southern pine plantations comes from fertilization (Stanturf et al. 2003; Leggett and Kelting 2003). In 1990, about 0.08 million ha of pine plantations were fertilized whereas over 0.49 million ha were fertilized in 2004 (Fox et al. 2007; Albaugh et al. 2007). From 1969 to 1991, fertilized forests increased from 0.01 to 0.08 million ha yr^{-1} and a total of ~6.47 million ha forested areas were fertilized during this period in the SUS. Nitrogen fertilization was reported to significantly increase net primary productivity (NPP), leaf area index (LAI), and biomass by many experimental studies in the SUS (e.g., Hynynen et al. 1998; Albaugh et al. 1998, 2004; Leggett and Kelting 2003; Samuelson et al. 2008). Some studies have found that LAI could be doubled with fertilization for loblolly pine plantations in the SUS (Albaugh et al. 1998, 2004; Lal et al. 2002). In addition, fertilization has been proven to enhance coarse root biomass production (Albaugh et al. 1998; Retzlaff et al. 2001). Soil C may also increase as a beneficial result of enhancing biomass production and returns of litter (Leggett and Kelting 2003; Samuelson et al. 2009). The increases in biomass production with fertilization transfer the increased inputs to the forest floor and soil. Through a 7-yr fertilization experiment, Samuelson et al. (2009) indicated that although soil CO_2 fluxes increased significantly after N fertilization, the much higher increases in NPP could still make the fertilized plantation be a net C sink. However, due to higher productivity and biomass, plantations were generally harvested at a younger age than natural forests. The choice of rotation length is considered to be an effective forest management activity for controlling the carbon stocks of forests (Liski et al. 2001; Harmon and Marks 2002; Kaipainen et al. 2004). It affects the carbon stocks of vegetation, soil, and wood products (Kaipainen et al. 2004). Carbon stock of trees increases with increasing rotation length but it is not necessary for soil and wood products (Liski et al. 2001). The decreases in the carbon stocks of soil and wood products would make the practice of increasing rotation length less efficient in sequestering carbon. The shorter forest rotation might greatly disrupt the normal ecosystem C and nutrient cycles (e.g., Lockaby et al. 1997; Johnson and Curtis 2001; Yanai et al. 2003). C storage in forest plantation has been reported to either increase or decrease, which may primarily depend on the rotation age, harvest regimes and the field environmental conditions (Johnson et al. 1992, 1995, 2001; Yanai et al. 2003). It is still unclear how the decreases in forest rotation age have influenced ecosystem C and N cycles in the SUS.

Although most experiments have proved that increased N fertilization could significantly increase C storage in the forest plantation ecosystems, the C losses during forest harvesting activities due to site preparation (e.g., slash burning and litter raking) and short rotation might offset most of this C sink. Slash burning, litter raking, and biomass removal in short-rotation forests could significantly reduce the C input to soil ecosystem, thus potentially reducing soil C (Klemmedson 1976; Covington and Sackett 1992; Johnson and Curtis 2001). Since soils serve as the largest terrestrial reservoir of C (Pennock and van Kessel 1997; Percival et al. 2000) and forest soils contain 50 to 63% of all soil C (Kimble et al. 2003), identifying and applying management strategies to intensify C storage in soil pool could have major impacts on future terrestrial C sequestration. In the other hand, forest managements such as slash burning, harvesting, and N fertilization may significantly increase N input to the soil in a short term but decrease soil N in a long term (Covington and Sackett 1992; Johnson and Curtis 2001; Carson et al. 2003), which may cause abnormal changes in soil N transformation and thus result in changes in nitrous oxide (N_2O) emission. Although many studies worldwide have reported the impacts of forest disturbance and harvest on N cycles (e.g., Vitousek and Matson 1985; Melillo et al. 1989; Bowden et al. 1990; Aber et al. 1998; Johnson and Curtis 2001; Wolf and Brumme 2003; Tate et al. 2006; Kellman and Kavanaugh 2008), less attention has been paid to the SUS, especially for regional-scale studies.

Given the important roles of forest plantations and management practices in the SUS, understanding the effects of increased forest plantation area and management practices on the C pools and N_2O emissions is very important. The objectives of this study were to 1) explore the effects of increased forest plantation area on C storage and N_2O emissions in the SUS; 2) determine the roles of N fertilization, site preparation, and plantation rotation lengths in terms of C storage and N_2O emissions; and 3) assess global warming potential (GWP) of increased forest plantation area and intensified management practices.

METHODS

Model description

A forest management and disturbance module was specially developed to incorporate into the Dynamic Land Ecosystem Model (DLEM), in order to address the effects of forest management practices (e.g., slash burning, prescribed fire, seedling planting, forest harvest, N fertilization, and irrigation) and disturbance (e.g., wildfire, hurricane, insect and disease) on forest ecosystem C, N, and water cycles. Some of the modeling processes for this module have been briefly described in Chapter 6. Here, we specifically described some modeling algorithms related to this study.

Forest fertilization

Fertilized N will directly enter into the soil available N pools (i.e., NO_3-N and NH_4-N). Then it could be taken up by plant roots or immobilized by microbes. The soil available N in managed ecosystems (i.e., pasture, forest plantation, and cropland) will be changed to:

$$AV_{NO3} += Nfer \times fratio$$

$$AV_{NH4} += Nfer \times (1 - fratio)$$

Where, *Nfer* is the daily N fertilizer amounts (g N m^{-2}); *fratio* is the fraction of NO_3-N in fertilized N. The default value for *fratio* in DLEM is 0.5. This value will be changed in terms of different fertilizer types and management types (e.g., prescribed fire, N fertilization). Due to lack of fertilization date data on the regional scale, DLEM assumes N fertilizer is applied with a constant slow release rate during the growing season. Thus, *Nfer* is calculated as the total annual N fertilizer amounts divided by total growing season length (days). The growing season length is calculated as the total days from sprouting to full leaf expansion (i.e., the time with highest leaf area index).

Nitrogen uptake

The applied N in the soil will be taken up by the plant roots. Currently, DLEM only considers plant uptake of soil inorganic N ions (NH_4^+ and NO_3^-) through root systems, but does not include uptake of nitrous gases (such as NH_3, NO, NO_2) by leaf stomata or bark. DLEM assumes that the N uptake (N_{uptake}) is influenced by plant N requirements ($N_{deficit}$), N availability (N_{av}), and plant potential uptake rate (N_{pup}):

$$\mathrm{N_{uptake}} = \min(\mathrm{N_{pup}}, \mathrm{N_{av}}, \mathrm{N_{deficit}})$$

N_{pup} is influenced by soil temperature (T_{soil}), soil moisture (*W*), and fine root biomass (*froot*):

$$N_{pup} = K_{max\,up} \times f(T_{soil}) \times f(W) \times f(froot)$$

Where, k_{maxup} is the maximum N uptake rate for each plant functional type. In DLEM, the valid temperature for N uptake ranges from -5 to 30°C, and plants will reach the maximum uptake at 15°C. Nitrogen uptake would not occur if daily evapotranspiration is equal to zero.

Nitrification

Nitrification, a process converting ammonium into nitrate, is simulated as a function of soil temperature, moisture, and soil NH_4^+ concentration (Lin et al. 2000).

$$N_{nit} = \min(N_{pot,nit}, N_{NH4})$$

$$N_{pot,nit} = V_{nit,max} \times \frac{N_{NH4}}{N_{NH4} + Km_{nit}} \times f_{nit}(T_{soil}) \times f_{nit}(vwc)$$

$$f_{nit}(T_{soil}) = Q_{10,nit}^{\left(\frac{T_{soil} - T_{opt,nit}}{10}\right)}$$

$$f_{nit}(vwc) = \begin{cases} 1.17 \times \frac{vwc}{vwc_{fc}} + 0.165 & vwc < vwc_{fc} \\ 1 - 0.1 \times \frac{vwc}{vwc_{fc}} & vwc \geq vwc_{fc} \end{cases}$$

Where, N_{nit} is the nitrification rate (g N m^{-3} d^{-1}); $N_{pot,\,nit}$ is the potential nitrification rate (g N m^{-3} d^{-1}); N_{NH4} is the concentration of NH_4^+ in the soil (g N m^{-3}); $V_{nit,\,max}$ is a parameter describing potential nitrification rate without limitation (g N m^{-3} d^{-1}); Km_{nit} is the half-saturation concentration of soil NH_4^+ for the maximum nitrification rate (g N m^{-3}); $f_{nit}(T_{soil})$ is a multiplier that describes the effect of soil temperature on nitrification; T_{soil} is the soil temperature (°C); $f_{nit}(vwc)$ is a multiplier that describes the effect of water content on nitrification (Lin et al. 2000;

Riedo et al. 1998); $Q_{10,nit}$ is the temperature sensitivity of nitrification, which is set as 2; $T_{opt,nit}$ is the optimum temperature for nitrification, which is set as 20°C following Rideo et al. (1998) and Lin et al. (2000); *vwc* is the volumetric water content; and vwc_{fc} is the soil field capacity.

Denitrification

Denitrification, through which the nitrate is converted into nitrogen gas, is simulated in the DLEM as a function of soil temperature, moisture, and soil NO_3^- concentration (Lin et al. 2000).

$$N_{denit} = \min(N_{pot,denit}, N_{NO3})$$

$$N_{pot,denit} = V_{denit,max} \times \frac{N_{NO3}}{N_{NO3} + Km_{denit}} \times f_{denit}(T_{soil}) \times f_{denit}(vwc)$$

$$f_{denit}(T_{soil}) = Q_{10,denit}^{\frac{T_{soil} - T_{opt,denit}}{10}}$$

$$f_{denit}(vwc) = \begin{cases} 0.0 & vwc < vwc_{fc} \\ \frac{vwc}{vwc_{fc}} & vwc \geq vwc_{fc} \end{cases}$$

Where, N_{denit} is the denitrification rate (g N m^{-2} d^{-1}); $N_{pot,\,denit}$ is the potential nitrification rate (g N m^{-2} d^{-1}); N_{NO3} is the concentration of NO_3^- in the soil (g N m^{-2}); $V_{denit,\,max}$ is a parameter describing potential denitrification rate without limitation (g N m^{-2} d^{-1}); Km_{denit} is the half-saturation concentration of soil NO_3^- for the maximum denitrification rate (g N m^{-2}); $f_{denit}(T_{soil})$ is a multiplier that describes the effect of soil temperature on denitrification; $f_{denit}(vwc)$ is a multiplier that describes the effect of water content on denitrification (Lin et al. 2000; Riedo et al. 1998); $Q_{10,denit}$ is the temperature sensitivity of denitrification, which is set as 3; and $T_{opt,denit}$ is the optimum temperature for denitrification, which is set as 25°C following Lin et al (Lin et al. 2000).

N_2O emission

All the products of nitrification and denitrification are nitrogen-containing gases. The empirical equation reported by Davidson et al (Davidson et al. 2000) is used to separate N_2O from other gases (mainly NO and N_2).

$$F_{N2O} = (0.001 * N_{nitrif} + N_{denitrif}) \times \frac{10^{vwc/\emptyset \times 0.026 - 1.66}}{(1+10^{vwc/\emptyset \times 0.026 - 1.66})}$$

where F_{N2O} is the fluxes of N_2O from soil to the atmosphere (g N m^{-2} d^{-1}), 0.001 is the proportion of nitrification product released as gaseous nitrogen (Lin et al., 2000), and it is converted to fluxes in the unit area (g N m^{-2} d^{-1}) by multiplying the depth of the first soil layer (0.5m); $\emptyset$ is the soil porosity.

Nitrogen effect multiplier for photosynthesis

DLEM estimates the C assimilation rate following the modified Farquhar Equation in which the leaf-level assimilation rate is limited by photosynthetic enzyme (RuBP), light and export or utilization of products (Farquhar et al. 1980; Collatz et al. 1991; Sellers et al. 1992). The actual maximum carboxylation rate (V_m) is related to leaf N content:

$$V_m = f(N)V_{max}$$

$$f(N) = min(foln + (1 - foln) \times \frac{N_{leaf} - N_{lfmin}}{N_{lfsat} - N_{lfmin}}, 1)$$

$$N_{lfsat} = N_{lfmin} + folnmx \times (N_{lfmax} - N_{lfmin})$$

Where, V_{max} is a function of maximum carboxylation rate at 25 °C (*Vmax25*, a constant for each PFT). *f(N)* indicates the magnitude of N control on photosynthesis ranging between 0-1. Given no limitations from other factors, photosynthesis rate will be the highest at the optimum leaf N concentration (N_{lfsat}), i.e., *f(N)* is equal to 1. If *f(N)* is higher than 1, carboxylation activities are saturated and not affected by leaf N content. N_{lfsat} is a value between the minimum and maximum leaf N content (N_{lfmin} and N_{lfmax}). Both *foln* and *folnmx* are PFT-specific parameters that will be calibrated.

Genetic improvement on plant photosynthesis

$$V_m = (1 + f(g))V_{max}$$

Where, *f(g)* is the impact factor from genetic improvement (> 0.0). Genetic improvement of forest tree species could greatly increase photosynthesis rate for forest plantations. In DLEM, the impact of genetic improvement on photosynthesis will be through the increasing in maximum carboxylation rate (*Vmax*).

Slash burning and resultant carbon and nitrogen dynamics

The modeling processes have been described in Chapter 6. The parameters for the equations were recalibrated in this study (See model parameterization).

Wood product module

The modeling processes have been described in Chapter 6.

Model parameterization

The DLEM has been parameterized and validated in the SUS (Tian et al. 2010a, b; Zhang et al. 2010; Xu et al. 2010). We specifically recalibrated the forest management and disturbance module for this study. Most parameters were obtained from literature reviews and few were kept default value given by the DLEM model due to lack of measurement data. The parameters and their values were shown in Tables 1, 2, 3, 4 and 5.

Table 1 Allocation of removed biomass to in-use product pools

Raw materials	Production line			
	Sawnwood	Boards	Paper	Firewood
Logwood	0.8	0.15	0.05	0
Pulpwood	0	0.05	0.9	0
Slash	0	0	0	1

Table 2 Parameters and their values for product allocation after harvest (Data source: Office of Policy and International Affairs 2007).

Plant functional type	Saw log				Pulpwood			
	In use	Landfill*	Energy	Emitted without energy	In use	Landfill	Energy	Emitted without energy
ENF	0.64	0.00	0.26	0.10	0.55	0.00	0.28	0.17
DBF	0.61	0.00	0.22	0.17	0.59	0.00	0.22	0.19

* No products are allocated to landfill at the first year. During decay of in-use products, 2.5% (saw log) and 2.3% (pulpwood) of in-use products will be allocated to landfill for hardwood; and 1.7% (saw log) and 2.4% (pulpwood) for softwood (OPIA 2007).

Table 3 Allocation for the in-use products to different half-life product pools.

Products	Long term (50 yrs)	Medium term (15 yrs)	Short term (1 yr)
Saw log	0.5	0.25	0.25
Boards wood	0.3	0.5	0.2
Paper & firewood	0.01	0.1	0.89

Note: The products in landfill have a half-life of 145 years.

Table 4 Forest plantation parameters and their values

Plant functional type	Rotation* age (pulpwood)	Rotation age (sawtimber)	fvcmax$^{\delta}$	Fertilization age$^{®}$	Pulpwood$^{\Phi}$	Saw log
ENF	30	35	0.5	Mid-rotation	0.5	0.5
DBF	40	50	0.2	Mid-rotation	0.3	0.7
MF	30	35	0.5	Mid-rotation	0.5	0.5

* Sources: Sedlo (1983), Winjum and Lewis (1993), and Foley (2009). $^{®}$ Sources: Fox et al. (2007), Albaugh et al. (2007), Jokela (2004b), and Carlson et al. (2008). $^{\Phi}$ Approximately estimated from the fraction of growing-stock volume (Foley 2009). $^{\delta}$ A multiplier for the productivity of genetically improved plantation (0.5 indicates that the plantation productivity is increased by 50% after improvement).

Table 5 Proportion of biomass left on site after forest harvest

Harvest biomass	Leaves	Roots	Reproduction	Stem*
Temperate broadleaf forest	1.0	1.0	1.0	0.3
Temperate needleleaf forest	1.0	1.0	1.0	0.3

* Stem includes branches, boles and barks, most of large-size branches and all boles are removed.

Data description

Forest plantation distribution and area

The plantation area and distribution data were obtained from reports of USDA Forest Service for forest resources in U.S. (Smith et al. 2000, 2004, 2009) and southern forest resource assessment (Wear and Greis 2002). The state level plantation area data were available for the period: 1952, 1962, 1970, 1982, 1989, 1999, 2007, and 2040. County-level forest plantation area data were obtained from the 2007 forest resources report (Smith et al. 2009). The Forest Inventory and Analysis (FIA) plot-level data for the period: 2000-2007 were collected to generate the spatial distribution pattern of forest plantation. Although FIA's plot data have been swapped for the geographic location, the data are accurate enough for this study since we have a spatial resolution of 8 km. According the plot distribution data, the plantation percentages in each grid cell were aggregated at 8 km spatial resolution. Since DLEM requires only one PFT for each grid cell, we need to generate 0, 1 forest plantation distribution data (0: non-plantation; 1: plantation). We assumed that the grid cells of higher percentage of plantation distribution will have higher priority to be assigned as forest plantation. Based on the county-level plantation area data in 2007, we first generated the plantation distribution data for 2007. The distribution area in 2007 was used as a boundary layer to control the assignments of plantation distribution grids in other years. After the generation for plantation distribution data for 1952, 1962, 1970, 1982, 1989, 1999, 2007, and 2040 was done, the plantation distribution data between these periods were linearly interpolated. We also collected the annual planted forest plantation data (for the entire SUS) from Wear and Greis (2002), in which annual planted forest area from 1925 to 2003 are available. Based on this data, we set forest plantation area in 1925 as 0.

Based on these collected data, we generated the historical forest plantation distribution data in the SUS (Figure 1; Figure 2). We found that forest plantation area increased the fastest during recent 20 years (1989-2007. Figure 1). Most forest plantation areas in this region were pine forests. Forest plantation area will continue increase till 2040. Forest plantations expanded gradually during 1925-2040 (Figure 2). Forest plantations were mostly distributed in the lower

SUS and had a spreading tendency toward the upper SUS. Although most of these plantation areas were converted from previously natural or semi-natural (regenerated) forests, intensive management practices to forest plantations could have greatly altered the C, water, and N cycles in the these ecosystems.

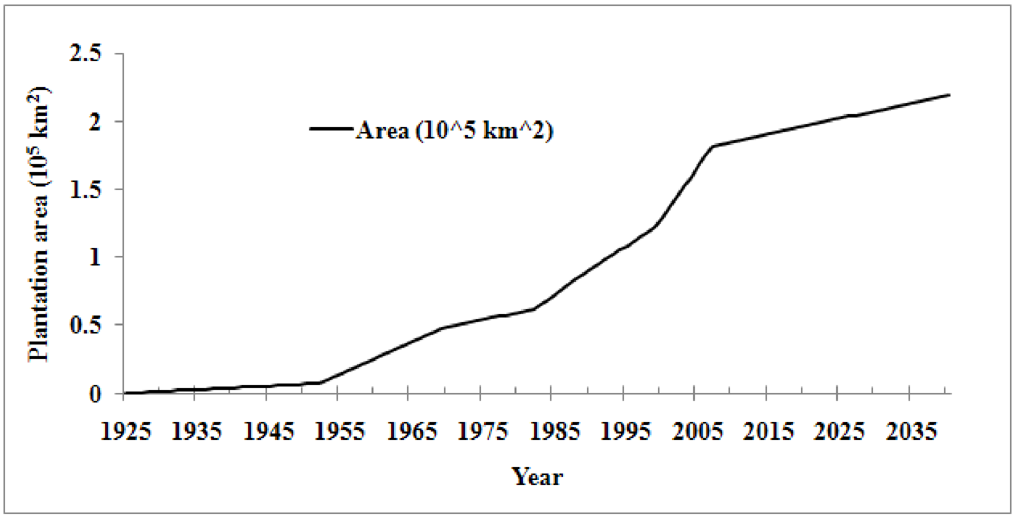

Figure 1 Forest plantation area during 1925-2007 (data source: Smith et al. 2000, 2004, 2009).

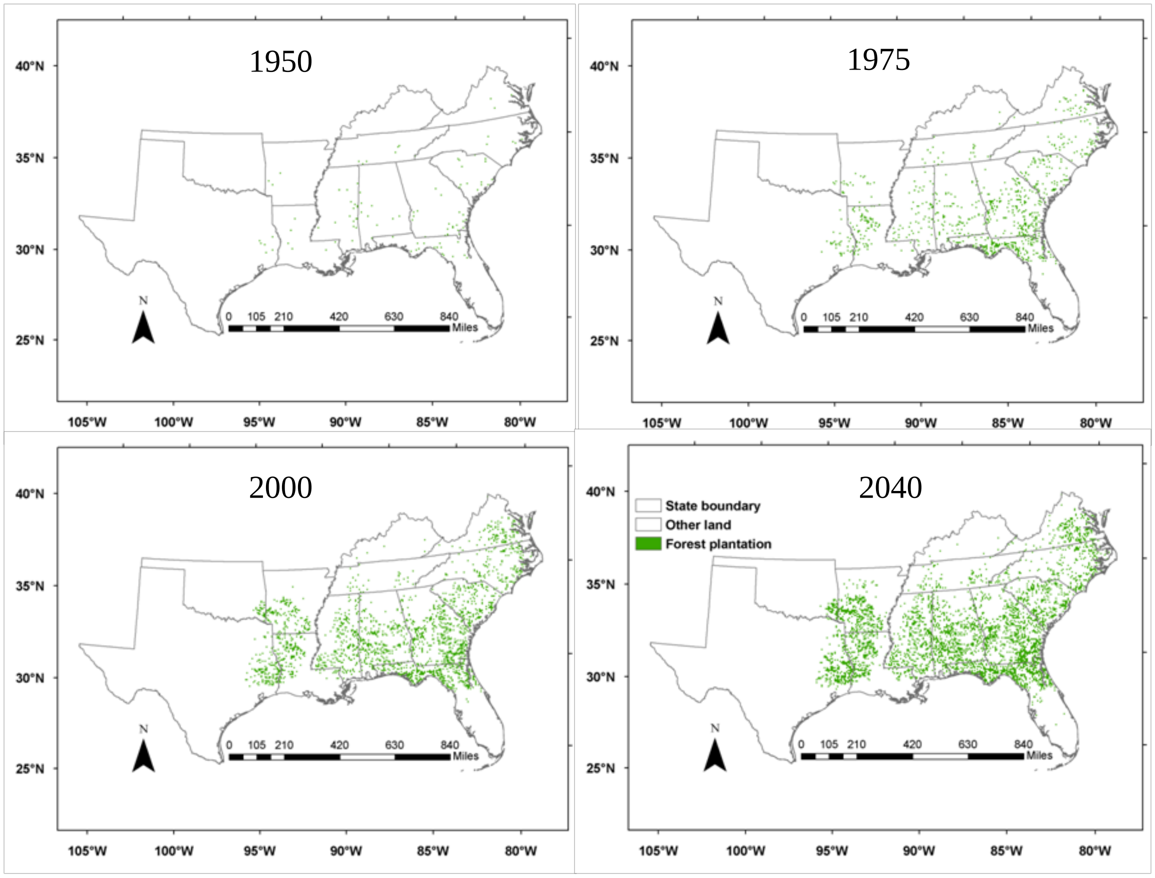

Figure 2 Spatial distribution of forest plantation in 1950, 1975, 2000, and 2040. Data sources: Wear and Greis (2002); Smith et al. (2000, 2004, 2009).

Nitrogen fertilizer amounts and fertilized plantation areas

Plantation areas receiving N fertilization during 1969-2004 have been reported by Fox et al. (2007) and Albaugh et al. (2007). From this data set (Figure 3), we found that the areas receiving N fertilization increased more rapidly during 1993-2004. The largest fertilized plantation area was 6.1×10^3 km^2 (1,510 thousand acres) in 2003, accounting for about 5.7% of the total forest plantation areas in this year. Fertilized areas were small before 1985. As indicated by Fox et al. (2007) and Carlson et al. (2008), most N fertilization occurred in the mid-rotation stage. Therefore, based on this data set, we random assigned N fertilization for grid cells at its mid-rotation age (default: 15. May change in terms of the rotation lengths for a specific PFT). We assumed there was no N fertilization before 1969 since no data were available. Due to lack of spatial distribution data for N fertilization, the N fertilization rates during 2000-2004 as indicated by Allen et al. (2005) and Fox et al. (2007) were used as reference to set up the N fertilization rates in this study.

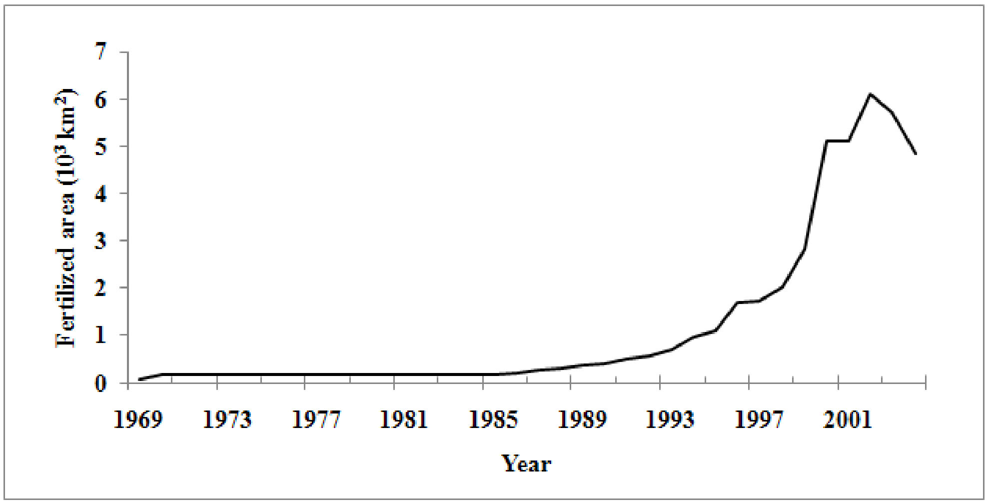

Figure 3 Forest plantation areas receiving N fertilizer during 1969-2004 (Source: Albaugh et al. 2007; Fox et al. 2007).

Experimental designs

To address the effects of increased forest plantation areas, forest harvest (rotation), slash burning, and N fertilization. We designed several simulation experiments.

1) S1, control: no forest plantation area; 2) S2, effects of increased plantation area: only change in plantation area. No fertilization, high-level slash burning rate (70%), and normal rotation age (30 years for pine; 40 years for hardwood) were applied; 3) S3, effects of rotation age: changed plantation area and four levels of rotation ages (20, 30, 40, and 60 for pine; 30, 40,

50, and 70 for hardwood); 4) S4, effects of slash burning: changed plantation area and three levels of slash burning rates (0%, 30%, and 70%); 5) S5, N fertilization effects: changed plantation area and four levels of N fertilizer uses (0, 2.8, 11, and 25 g N m^{-2}). The N fertilizer uses were referred to the values given in Fox et al. (2007).

RESULTS AND ANALYSES

Effects of increased plantation areas

Most of plantation tree species were cultured or genetically improved. We found that transforming natural forests to forest plantation significantly reduced soil and vegetation C since forest biomass is accumulating with increasing forest age and soil C could be emitted during harvest practices (Figure 4). Increases in forest plantation areas in the SUS could still decrease C storage in this region by 946 g C m^{-2} (0.2 Pg C, 1 Pg = 10^{15} g) compared to the natural forest ecosystems. Most of this reduction was caused by the decreases in vegetation C. Forest plantations are harvested in ~30 yrs for pine and ~40 yrs for hardwood in the SUS (Winjum and Lewis 1993; Foley 2009). Although plantation NPP was significantly higher than that of natural forests, vegetation C could not recover in such a short time. The increases in wood products could not offset the vegetation and soil C losses after harvesting.

Although ecosystem C storage decreased compared to natural forests, it increased significantly compared to regenerated forests (Regenerated forests were assumed to have the same rotation length with forest plantations) (Figure 4). This was primarily due to the higher NPP for forest plantations (Table 6). In the current SUS, most forests were regenerated forests with very young age structure due to industrial harvesting activities (Pan et al. 2010; Smith et al.

2009). Under this condition, the increases in forest plantation areas in the SUS could significantly increase C storage in this area. This C storage strength could be determined by the rotation length and management practices for the forest plantations.

Increases in forest plantation areas and harvest activities had resulted in a 10% increase in N_2O emission compared to natural forests (0.01 g N m^{-2} yr^{-1} or 0.26 Tg N. Table 6). N_2O in regenerated forests was significantly higher due to high litter biomass input in the soils following harvest. Since slash burning for forest plantations are generally used, a large amount of N in the litter biomass was emitted as N gases. Thus, high uptake rate for N due to increased NPP and less available N for forest plantations may cause lower emissions in N_2O than that of regenerated forests. Slash burning may cause N_2O emissions, but we did not account for it in this study. Increased forest plantation areas during 1925-2040 could significantly increase GWP by 0.77 Pg CO_2 equivalents in the SUS.

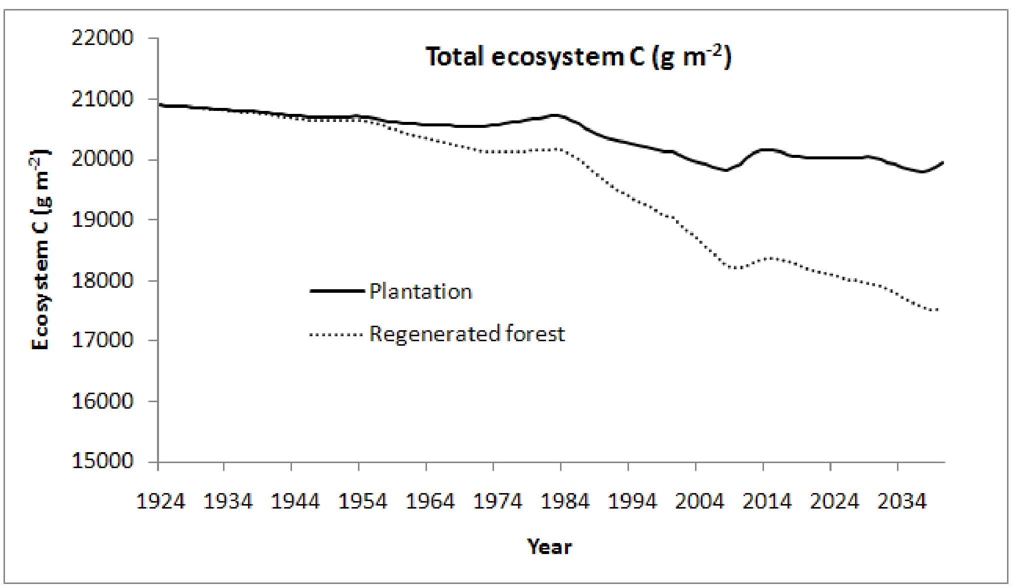

Figure 4 Total ecosystem C after natural forests are regenerated or transformed to forest plantations. Note: Compared to forest plantations, regenerated forests have the same harvesting cycles but no slash burning after harvest.

Table 6 Ecosystem C pools (g C m^{-2}), NPP (g C m^{-2} yr^{-1}), N_2O emissions (g N m^{-2} yr^{-1}), and GWP (Pg CO_2 equivalent) induced by increased forest plantation areas.

Variables	Natural forest	Regenerated forest	Forest plantation
NPP	683	661 (23)	732 (46)
Vegetation C	7611	3674	4911
Soil C	11670	10313	11194
Litter C	1456	762	706
Total C $^{\delta}$	20897	17535	19951
Change in total C	0	-3362	-946
N_2O *	0.11	0.17 (0.05)	0.12 (0.02)
GWP$^{©}$	0	2.72	0.77

$^{\delta}$ Total C includes vegetation, in-use product, soil, and litter C. *Only include N_2O emission from soils. $^{©}$ Combined GWP for both CO_2 and N_2O (relative to natural forests). N emissions from wood products were not counted. Values in the parenthesis are standard deviation. Natural forests were assumed no harvesting, while regenerated forests were assumed to have the same rotation age with forest plantation.

Effects of forest rotation lengths

Forest rotation was considered as an important management practice for C sequestration. We found that forest rotation lengths significantly changed the C storage in the SUS (Figure 5; Table 7). Total ecosystem C only decreased by 247 g C m^{-2} (or 0.054 Pg C) for 40-yr rotation length (for pine, hardwood rotation age is 10 years more than the corresponding pine plantations) plantations during 1925-2040, and forest plantations could be a C sink of 535 g C m^{-2} (or 0.12 Pg

C) for 60-yr rotation length. Although only 10 years difference, 20-yr rotation length could result in more significant decrease in C than that for 30-yr rotation length. The longer the rotation length, the higher is the forest biomass and soil C. Both vegetation and soil C could greatly increase if increasing 10 years rotation length. Vegetation C was more sensitive to increase in rotation length.

N_2O emission could increase by 0.02 g N m^{-2} yr^{-1} (or 0.26 Tg N during 1925-2040; Table 7) in short-rotation (20-yr) forest plantations, while it could be 0 in the 40-yr rotation plantations and even lower emission in the 60-yr rotation forests. This suggested that more frequent forest harvests could greatly increase N_2O emissions. Combining the GWP for both N_2O and CO_2, we found that GWP was enhanced for less than 40-yr rotation lengths, while reduced for 50-yr and 60-yr rotation forest plantations.

Table 7 Changes in C pools (g C m^{-2}), N_2O emissions (g N m^{-2} yr^{-1}) and overall GWP (Pg CO_2 equivalent) for different forest rotation ages compared to natural forest ecosystems.

Variables	20-yr *	30-yr	40-yr	50-yr	60-yr
Vegetation C	-3572$^{\delta}$	-2700	-2015	-1033	-507
Soil C	-636	-476	-300	-235	-197
Litter C	-770	-749	-498	-415	-358
Product C	3294	2941	2386	1881	1580
Total C	-1548	-946	-247	261	535
N_2O	0.02	0.01	0.00	-0.003	-0.006
GWP	1.25	0.76	0.20	-0.21	-0.43

* These rotation ages are for pine plantations; hardwood plantations are 10 years more than these; [δ] The values are calculated as the difference between forest plantations and natural forests in 2040.

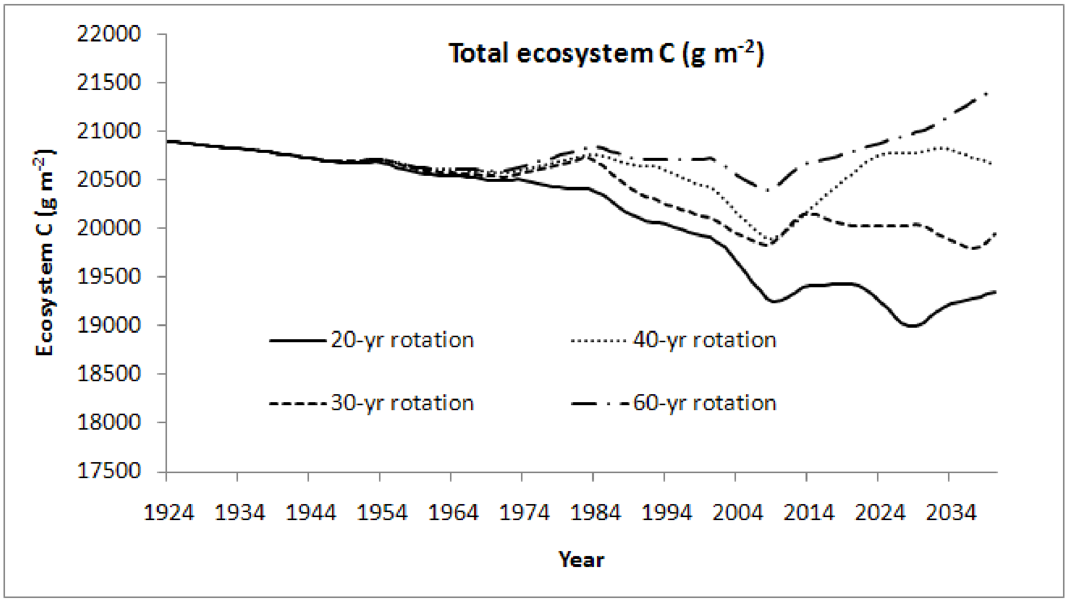

Figure 5 Changes in ecosystem C storage for forest plantations with different rotation lengths.

Effects of N fertilization

Due to N deficiency in the SUS forest soil, N fertilization could greatly increase forest plantation productivity and C storage (Table 8). NPP, vegetation, soil, and total C were increased with increased N fertilizer amounts. Due to N fertilizer only applied once at the mid-rotation age, the mean effects of N fertilization on NPP and C pools were diluted for the entire SUS. N fertilization reduced the GWP in the SUS. The highest N fertilization rate could reduce GWP by 639 Tg CO_2 equivalents.

Although most plantations in the SUS were proved to N deficiency, N fertilization could still induce large N_2O emissions. If the highest level N fertilization rate (25 g N m^{-2} yr^{-1} or 225 lb ac^{-1} yr^{-1}) was used for the mid-rotation forest plantations, 0.52 Tg N_2O-N (0.02 g N m^{-2} yr^{-1}) could be emitted compared to no fertilizer treatment (Figure 6). This is equivalent to the GWP of 9.6 g CO_2 m^{-2} yr^{-1}(245 Tg CO_2 equivalent), indicating that N_2O emission could offset a C sink of 9.6 g CO_2 m^{-2} yr^{-1}. About 0.8% of the fertilized N for the highest N fertilization level was released as N_2O. Combining both N_2O and CO_2 effects, N fertilization could reduce GWP by 55, 378 and 394 Tg CO2 equivalents, respectively.

Table 8 Carbon pools (g C m^{-2}), N_2O emissions (g N m^{-2} yr^{-1}) and GWP (Tg CO_2 equivalent) for different fertilization levels.

Variables	No fertilizer	Low fertilizer	Medium fertilizer	High fertilizer
NPP	732	746	761	775
Vegetation C	4911	5009	5107	5205
Soil C	11194	11317	11442	11505
Total C	19951	20050	20547	20845
N_2O	0.121	0.123	0.130	0.142
GWP for N_2O*	0	24	101	245
GWP for CO_2	0	-80	-479	-639
Overall GWP	0	-55	-378	-394

* The GWP is relative to the no fertilizer scenario. Negative GWP denotes reduced GWP.

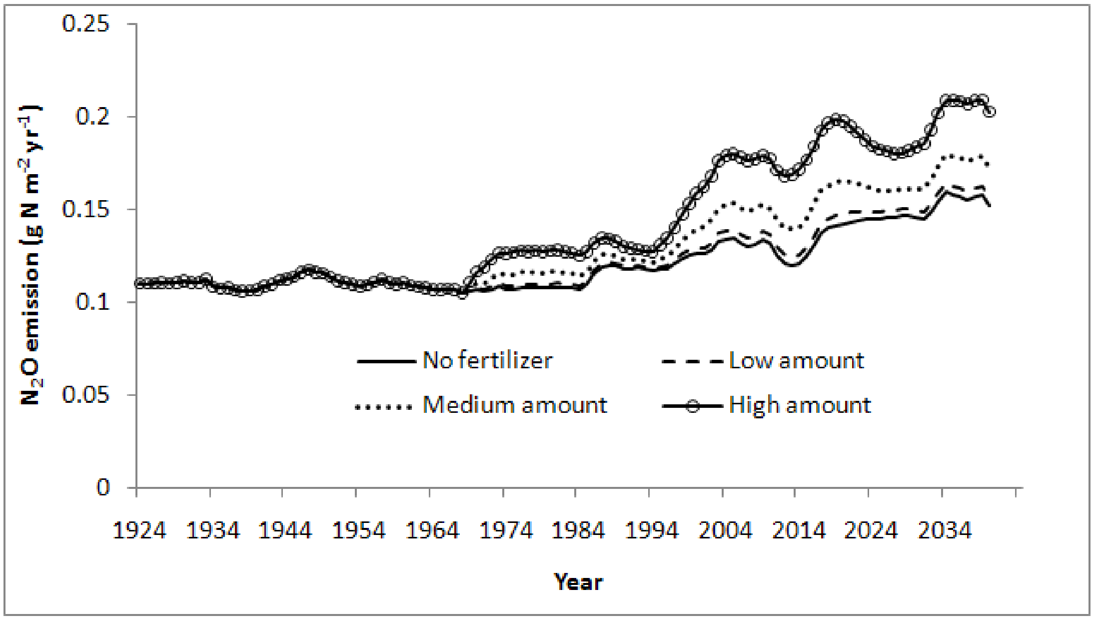

Figure 6 Annual N_2O emissions after different level N fertilization for the mid-rotation forest plantations during 1969-2040. Low amount: 2.8 g N m^{-2}; medium amount 11.0 g N m^{-2}; high: 25 g N m^{-2}. Fertilization levels were designed according to Albaugh et al. (2007) and Fox et al. (2007).

Effects of slash burning rates

Slash burnings are often used before establishing new plantation forests. Slash burning could not only remove competitions from other vegetation but also provide more available nutrient resources for new plantation forests in a short term. Although slash burning could increase NPP of the seedlings in a short term, we found that long-term NPP may not significantly increased (Table 9). During 1925-2040, average NPP for the 70% slash burning scenario only increased ~2 g C m^{-2} compared to the 0% slash burning scenario. This suggested that slash burning may not directly increase NPP, though plantation tree growth could be stimulated by removing competition plants. Vegetation C increased slightly with higher slash burning rate due to the slight increase in NPP. Slash burning significantly reduced total C storage. 70% slash burning rate could result in 518 g C m^{-2} (or 0.11 Pg C) emissions during 1925-2040 (Table 8). Most of these emissions were from the decreases in soil organic C (297 g C m^{-2} or 0.06 Pg C) and forest floor litter C (253 g C m^{-2} or 0.05 Pg C). In this study, we did not consider understory vegetation biomass, which could increase after slash burning and thus increasing ecosystem C storage.

Table 9 Effects of slash burning (site preparation) on C flux (g C $m^{-2}yr^{-1}$), storage (g C m^{-2}), N_2O emissions (g N m^{-2} yr^{-1}), and GWP (Pg CO_2 equivalent) for the forest plantations in the SUS.

Variables	0% slash burning	30% slash burning	70% slash burning
NPP	730	731	732
Vegetation C	4879	4899	4911
Soil C	11491	11368	11194
Litter C	960	852	707
Total C	20469	20259	19951
Change in total C*	0	-210	-518
N_2O	0.113	0.116	0.121
Total GWP*	0	0.20	0.49

* Changes relative to 0% slash burning.

Slash burning could cause organic N in the litters to release in the short time, thus higher soil available N. The higher soil available N could not only increase productivity but also change soil N_2O emissions. Nitrous oxide emission could be increased by 0.01 g N $m^{-2}yr^{-1}$ (or 0.21 Tg N; 1 Gg = 10^9 g). This included the indirect emissions from soil N transformation and direct emission from the burning spots. Combining the GWP of CO_2 and N_2O emissions, we found slash burning could increase GWP by 0.20 and 0.49 Pg CO_2 equivalents under 30% and 70% slash burning rate, respectively during 1925-2040.

DISCUSSION

Planted forests or natural generated forests?

If longer rotation lengths were used to harvest forests in the SUS, both natural and managed forests could be used to maintain ecosystem C storage. If shorter rotation lengths were applied, managed forests could be better to store more C in the SUS. In current SUS, except for pine plantation forests which account for ~20% of the total forest areas, other natural forests are also routinely harvested, which results in relatively evenly-distributed age groups less than 60 yr old for more than 80% of the forested areas in the SUS (Pan et al. 2010). This implied that most forests in the SUS were not primary forests. Under this situation, managed plantation forests with higher productivity could be better choice to distribute in this region from the perspective of increasing C storage. Intensive management practices have been proved to enhance C storage and thus regarded as potential solutions to mitigate climate change (Birdsey et al. 2006; Malmsheimer et al. 2008). However, intensive management practices could also induce other GHG emissions (such as CO, and N2O). In this study, we found that N fertilization in forest plantations could significantly enhance N_2O emissions under high-dose N fertilization rate, which resulted in enhanced GWP. This suggested that some management practices should be first evaluated for their consequences before applying them.

Forest rotation length and C sequestration

Currently, most forest plantations in the SUS have a rotation length ~30 yrs for pine and 40 yrs for hardwood (Sedlo 1983; Winjum and Lewis 1993; Foley 2009). With this rotation length, the increases in forest plantation areas in the SUS could release 0.95 Pg C during 1925-2040 (Table 6) compared to the natural forest ecosystems, but could be a C sink compared to regenerated forests. When the rotation length was increased, soil and vegetation C could significantly increase (Kaipainen et al. 2004). Increasing rotation length by 20 years (50-yr and 60-yr rotation for pine and hardwood, respectively) from currently applied had considerably different effects on C storage and N_2O emissions. The average C storage of forest plantations was estimated to increase ~1200 g C m^{-2} (Table 7) during 1925-2040 if increasing rotation length by 20 years. Kaipainen et al. (2004) also estimated that 20 years extension for rotation length could increase C stock of biomass in studied pine forests by 480-930 g C m^{-2} and in spruce forests by 950-2460 g C m^{-2} in Europe. IPCC (2000) estimated that carbon sinks resulting from the increased rotation length to be 2.2-3.6 g C m^{-2} yr^{-1} (or 257-421 g C m^{-2} during 1925-2040) in Canada, USA and The Netherlands. Our estimate is significantly higher than the IPCC estimates. One reason for our larger estimates is that we assumed a 20 years extension whereas IPCC assumed it for 80 years, indicating our estimate should be 4 times higher than IPCC estimate. This means our estimate is also within the range of IPCC estimates.

Although shorter rotation forests could produce more wood products (Table 7) which could be used as biofuel and GWP reduction, the larger decreases in vegetation C could offset this effect.

Fertilizer amount, productivity and GWP

Although annual fertilized plantation areas are only ~4% (the highest rate in 2003) of all the current forest plantations, it is expected to increase rapidly in the near future (Fox et al. 2007; Albaugh et al. 2007). One reason is the forest plantation areas still increase rapidly as predicted by Wear and Greis (2002). Another one is that soil in the SUS is deficient in N nutrients due to short harvesting cycle (short-rotation) upon request of the increasing demands of wood products. Experiments in the SUS have found that N fertilization could double LAI and productivity for some poor sites (Hynynen et al. 1998; Albaugh et al. 1998; Lal et al. 2002), which confirmed the N deficiency in most areas of the SUS. Our study implied that high-dose N fertilization could result in large amounts of N emissions, which partly offset the GWP reduction through increasing C storage. In the future, appropriate N fertilization practices including fertilization timing, site selections, and amounts should be taken into account.

CONCLUSION

Based on series of temporal and spatial data sets and much previous work, we evaluated the potential impacts increased forest plantation area and intensive management practices on NPP, C storage and N_2O emissions during 1925-2040 in the SUS, through using a global biogeochemical model which couples a forest management and disturbance module. Results suggested that increases in forest plantation areas in the SUS could decrease C storage if compared to the natural forest ecosystems. Most of this reduction was caused by the decreases in vegetation C. However, if compared to the regenerated forests, increases in forest plantations could increase C storage due to its higher productivity. Longer forest rotation length could store more C and reduce N_2O emission, while shorter rotation length could cause large amounts of CO_2 and N_2O emissions. Nitrogen fertilization resulted in high plantation productivity and ecosystem C storage; however, the concomitant increases in N_2O partially offset its functions for global warming potential mitigation. Slash burning could slightly increase NPP and forest biomass in a short period; however, the decreases in litter and soil C make slash burning a C source. In the future, increases in forest rotation length, forest plantation areas, and appropriate N fertilizer could continue increasing C storage and reduce N_2O emissions in the SUS.

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Chapter 8

General Conclusions and Future Directions

General conclusions

The overall objective in this research is to quantify the impacts of disturbance (e.g. drought, land use change, and forest mortality) and land management (e.g. N fertilization in cropland and forest, and forest harvest) on terrestrial C, water and N dynamics in the Southern United States. Based on this research, several general conclusions were drawn:

1) Using the standard precipitation index as a tool, we characterized the drought intensity and duration in the SUS during 1895-2007. No significant changes in drought intensity and duration were found for the study period. However, we found that the area of the SUS experiencing extreme high rainfall events appeared to be increasing, which might imply an increasing flooding frequency during the study period. NPP was greatly reduced during dry years and could induce up to 40% decrease in some areas. Although no significant trend in drought intensity and duration was found in the SUS, Climate change in the SUS has resulted in a net release of 0.33 Pg C (1 Pg = 10^{15} g) into the atmosphere, while changes in precipitation and temperature patterns induced C emissions of 0.035 and 0.14 Pg, respectively during 1895-2007. The interactions between precipitation and air temperature induced a C emission of ~0.15 Pg, suggesting that changes in air temperature could significantly enhance drought impacts in the SUS. In total, C emission from drought impacts induced both by precipitation and temperature could be ~0.19 Pg. The drought impacts on NPP and C storage could be enhanced by changes in other climatic

factors; therefore, the study for drought impacts should also consider the interactive effects among drought events and other climatic factors such as solar radiation and air temperature.

2) The impacts of climate, land use and land management changes on ET and water yield during 1895-2007 were modeled with a global biogeochemical model. We found that both ET and water yield were increased during the study period. Compared to land use and management change, climate variability had the largest impact and contributed the most to the increase in both ET and water yield. Although the change in both ET (3.5 mm decade^{-1}) and water yield (6.7 mm decade^{-1}) were not significantly high, the much larger spatial and inter-annual variation could result in significant impacts of this change on the ecosystems especially in the arid and semi-arid region. The modeled water yield and ET have been validated against field observation data, showing a good agreement between simulated and measured results. The study results for impacts of climate change, and land use and management change on water yield could be a reference for the land mangers and policy makers to better manage water resources in the SUS.

3) Based on a global biogeochemical model and county-level nitrogen fertilizer amount data for the SUS, this study estimated the changes of CO_2 and N_2O fluxes as influenced by nitrogen fertilization. Results indicated that both soil carbon storage and N_2O emission were increased during 1945-2007. By combining the GWP of these two gases, we found that N fertilization was a C source and GWP has been greatly increased during the study period due to rapidly increased N_2O emissions. GWP decreased in the western SUS and increased the most in the central SUS. This spatial pattern was primarily determined by the distribution of N fertilizer amounts. We found that higher N fertilizer amounts could

increase GWP while lower amounts could decrease GWP. To decrease GWP and maintain high crop productivity in the future, the best way is to increase plant N use efficiency to utilize the added N rather than increasing N fertilizer amounts.

4) Based on high resolution (30 m) forest mortality rate data and a process-based global biogeochemical model, this study estimated the changes in C storage in Mississippi and Alabama after continuous forest disturbance. Results indicated that mean annual forest mortality during 1985-2003 was ~1.3% with a large spatial variation. The accumulated forest mortality during 1985-2007 for some areas could up to 100%. The continuous forest mortality has resulted in a large C source (0.20 Pg C or 8.3 Tg C yr^{-1}) in Alabama and Mississippi. Forest ecosystems in the United States was reported as the largest C sink during the 1980-present; however, if small-scale disturbance is considered in the estimation, this C sink could be greatly reduced.

Uncertainties and future directions

Although the Dynamic Land Ecosystem Model (DLEM) has been applied and validated over different regions (e.g., China, North America, and Monsoon Asia), many uncertainties still exist for many aspects. The first uncertainty is from model input data. In this study, we generated or utilized a large number of large-scale and long-term data sets, such as climate, nitrogen fertilizer amount, and land use data during 1895-2007 for the entire SUS. The accuracy of these data could significantly influence the model results. In the future, with more available experimental, observational or regional inventory data, input data accuracy could be greatly improved. In addition, due to lack of enough observational and experimental data, some of the model parameters used to run the model might not represent the actual conditions in the SUS.

For example, the parameters for wood product allocation after harvest, nitrogen fixation, C allocation among different components, harvest crop residue return rate, and maximum stomata openness. Finally, some of important modeling algorithms were too simplified or even ignored in the model. These uncertainties limit the applications of model results within a certain confidence level. Despite of these, model results could still offer relatively accurate estimations to environmental changes and their impacts on ecosystem structure and function.

In the near future, following work needs to be done:

1. Improve the DLEM mechanisms and model input data to more accurately represent C, N, and water dynamics in the SUS;
2. Some analyses (deep analysis) will be expanded in the future (e.g., impacts of different forest management types and impacts on CH_4 emission);
3. Simulate the impacts of other major management practices (such as prescribed fires, forest thinning, and tillage) and disturbances (such as hurricanes, southern pine beetles, and wildfires) on C, N, and water dynamics in the SUS;
4. Connect model results and conclusions with policy making;
5. Extend the studies in the SUS to the entire US and North America.