**Effects of Ozone Pollution and Climate
Variability/Change on Spatial and Temporal Patterns
of Terrestrial Primary Productivity and Carbon
Storage in China**

By

Wei Ren

A dissertation submitted to the Graduate Faculty of
Auburn University
in partial fulfillment of the
requirements for the Degree of
Doctor of Philosophy

Auburn, Alabama
December 18, 2009

Keywords: Carbon storage, Climate, Net Primary
Productivity, Ozone Pollution, Terrestrial ecosystem

Approved by

Hanqin Tian, Chair, Alumni Professor of Ecology
Art Chappelka, Professor of Forest Biology
Luke Marzen, Associate Professor of Geography
Ge Sun, Research Hydrologist

Abstract

Over the past several decades, China's terrestrial ecosystems have experienced severe air pollution and other environmental changes. Quantifying how these environmental changes have affected carbon (C) fluxes and storage in China's terrestrial ecosystems is crucial to understanding the global C cycle as well as China's sustainability. Using an improved Dynamic Land Ecosystem Model (DLEM), I assessed the spatial and temporal patterns of net primary productivity (NPP) and C storage in China's terrestrial ecosystems in response to tropospheric ozone (O_3) pollution and historical climate variability/ change in the context of multi-factor global change during 1961-2005. An overall evaluation has been implemented to investigate how elevated O_3 in combination with climate variability has affected the C cycle in terrestrial ecosystems of the nation. The modeled results showed that during 1961-2005, elevated O_3 resulted in a mean 4.5% loss in NPP and a 0.9% reduction in total C storage for China's terrestrial ecosystems as a whole, which has reduced the magnitude of terrestrial C sink during the same period. The reduction of C storage among different terrestrial ecosystems varied from 0.1 Tg C to 312 Tg C ($1\ T = 10^{12}$) with a decreasing rate ranging from 0.2% to 6.9%. For example, China's grassland ecosystems, distributed mainly in arid and semi-arid regions of North China, were the most vulnerable in response to climate variability/ change and elevated O_3. The effect of O_3 pollution on China's forest ecosystems could be accelerated by climate extreme events such as drought. For agricultural ecosystems, some rain-fed

cropland areas in arid and semi-arid regions of North China, which experienced high O_3 levels and frequent drought events, acted as a C source. However the effect of O_3 pollution and climate variability/change on C storage in cropland could be modified by land management (e.g. irrigation and fertilizer). Results from this study indicate that improved air quality could significantly increase productivity and C storage in China's terrestrial ecosystems and that optimized land management options could enhance the adaptation of terrestrial ecosystem to climate variability/change and air pollution.

Acknowledgments

My gratitude first goes to my advisor, Dr. Hanqin Tian, for his broad knowledge, great patience and constant support in my six and a half years' graduate study. His passion and insight on his research field have always inspired me, and will benefit me for my future academic career. His unique guidance to nurture a student's talent has encouraged me to dedicate myself to pursuing the beauty of science for my life. I also highly appreciate all the support and encouragement from my committee: Dr. Art Chappelka, Dr. Ge Sun and Dr. Luke Marzen. My committee's precious comments and suggestions are invaluable in conducting my PhD projects, and their instructions and advices have illuminated my academic path from a graduate student to an ecologist.

I also would like to thank Ms. Shufen Pan, who gave me a great deal of help on both my academic and non-academic life. I thank Dr. Mingliang Liu for his support in my researches. Furthermore, I thank Guangsheng Chen, Bo Tao, Xiaofeng Xu for their help and suggestion on Chapters 4, 5 and 7, respectively. Thanks also go to my colleagues, Chi Zhang and Chaoqun Lu, for their cooperation, and to Elizabeth Williams and Dafeng Hui for their help with the language. Furthermore, I want to express my heartily thanks to Ms. Staudenmaier Patti, Mrs. Marjorie Gentry, and Mrs. Pam Beasley, who offered me much help during my studies in School of Forestry & Wildlife Sciences.

Finally, and most importantly, I am deeply grateful to my parents and sister, who have been giving me unconditional love and constant support at all times. Without them, it would be impossible for me to be here and complete my Ph.D. study. I sincerely appreciate Bo, for his love, patience and support during the difficult yet exciting period of dissertation writing.

Thanks to all who love me and care about me, and I shall dedicate this dissertation to them.

Table of Contents

List of Tables

List of Illustrations

xiv

xvi

xviii

Chapter I

Introduction

Terrestrial ecosystems constitute a major player in the system of carbon- cycle-climate feedbacks by releasing or absorbing relevant greenhouse gases and controlling exchanges of energy, water and momentum between the atmosphere and the land surface (Heimann and Reichstein, 2008). Carbon in terrestrial ecosystems is gained through photosynthesis and is lost primarily as CO_2 through respiration in autotrophs (plants and photosynthetic bacteria) and heterotrophs (fungi, animals and some bacteria), and non-CO_2 losses such as methane or dissolved carbon. The net amount of carbon captured by plants through photosynthesis, defined as Net Primary Productivity (NPP), is of fundamental importance to human society, given that the largest portion of our food supply depends on productivity of plant life on land, as does our need from wood for construction and fuel. Carbon can be stored in trees for several hundreds of years and in soils for thousands of years (carbon storage, C storage), depending on the turnover time. The capacity of C storage in different carbon pools and its temporal and spatial patterns are mainly controlled by environmental change, including climate change, which is characterized as global warming induced by increasing atmospheric CO_2 concentration, and human activities, such as land use change and agricultural land management (Melillo et al., 1993, 1996; Vitousek et al., 1997; Tian et al., 1998, 1999, 2003; Cao and

1

Woodward, 1998; White et al., 1999; Falkowski et al., 2000; Chapin et al., 2002; Houghton, 2003, 2007; Grace, 2004; Field et al., 2007; Luo, 2007). Consequently, quantifying NPP and C storage (both interannual variations and spatial patterns) in terrestrial ecosystems is crucial for the assessment of ecosystem services and global carbon budget.

The carbon cycle in terrestrial ecosystems is influenced by a number of processes and environmental factors at different time-scales. During the last two centuries, atmospheric CO_2 concentrations have dramatically increased by 31 percent (Intergovernmental Panel on Climate Change, IPCC, 2007) due to the large amount of CO_2 released through human activities (e.g. fossil combustion, cement production, intensive farming and vast deforestation) (e.g. Ramankutty and Foley, 1999; Tilman et al., 2002). The global air temperature has increased at an unprecedented rate (Hansen et al., 2006); this is generally believed to be caused mainly by greenhouse gases (GHGs), with CO_2 as the major contributor. As the globe has continued to get warmer, precipitation trends have varied widely by regions and over time and the frequency and intensity of extreme climate events have increased (IPCC, 2007; Severinghaus et al., 2009). In addition to rapid changes in climate systems, atmospheric composition changes (e.g. nitrogen deposition, O_3, and aerosol) have received increased attention in the past decades (e.g. Akimoto, 2003; Galloway et al., 2003), and in recent years, elevated tropospheric O_3 has been generally recognized as one of the dominant pollutants at a global scale (Ashmore, 2005).

Evidence shows that the changing climate system and other environmental factors have significantly influenced the carbon cycle in terrestrial ecosystems; this includes positive effects, such as CO_2 fertilization stimulating plant growth and enhancing CO_2 accumulation (Free Air CO_2 Enrichment, FACE Projects), and negative effects such as O_3 pollution reducing carbon sinks (European OTC program, EOTC). However, environmental stresses usually do not operate independently, but often interact to produce combined impacts on ecosystem functioning (Schindler, 2001). For example, recent work indicates that the combined effects of O_3 pollution and its influence on the climate system could reduce the capacity of land-carbon sink (Sitch et al, 2007), which will eventually threaten environmental sustainability and economic development. Consequentially, it is necessary to conduct an integrated assessment on terrestrial ecosystem production and carbon sequestration capacity in response to both global climate change and environmental changes.

Due to the high spatial heterogeneity and the complex structures and feedbacks of terrestrial ecosystems in response to multiple environmental stresses, it is challenging to realistically quantify the magnitude, patterns, and variation of NPP and C storage in the context of global change. Understanding the underlying mechanisms at regional level through traditional controlled experiments, surveys, and empirical statistical models is especially difficult. An integrated process-based ecosystem model, which includes the biogeochemical and physiological responses of ecosystems to atmospheric and climate changes, has proved a powerful tool in such multiple stress studies, especially over large regions (Tian et al., 1998, 2003, 2008; Karnosky et al., 2005). An integrated

process-based ecosystem model is necessary to capture or characterize the response of carbon dynamics in terrestrial ecosystems to main environmental changes (such as elevated tropospheric O_3, climate variability/change, increasing atmospheric CO_2 concentration, nitrogen deposition, and land cover/use change).

China, home of more than 1.3 billion people and the 3rd largest country in the world, has seen great economic development and rapid industrialization over recent decades. In late 2006, China overtook the United States as the world's largest producer of CO_2, the chief greenhouse gas (Gregg et al., 2008). China's terrestrial ecosystems have experienced substantial changes in climate (Cao et al., 2003; Tian et al., 2003; Chen et al. 2006), atmospheric chemical compositions (e.g. O_3 pollution, nitrogen deposition) (Felzer et al. 2005, Lu and Tian 2007; Ren et al., 2007a, b), and land-use and land-cover (Liu et al., 2005a,b; Tian et al., 2003). Air pollution and its close correlation with economic development - primarily the burning of fossil fuels (coal, gasoline, etc.) has been one of the most pressing environmental concerns in China over the recent decades (Liu and Diamond, 2005). Site-level monitoring, remote sensing and models show that the concentrations of ground-level O_3 have significantly increased in past decades (Wang et al., 2007) and will continue to increase in the coming years, with a peak concentration 150 ppb level of O_3 in some locations (Elliot et al., 1997; van Aardenne et al., 1999). The effects of air pollution on ecosystems, including human health and natural ecosystems, are diverse and numerous. Studies have suggested that terrestrial ecosystems have distinct responses to elevated ozone and climate variability/change, because of their distinct structure and functioning characters within each functional type, also because of their

4

complex environmental and management conditions. The effects of O_3 on the terrestrial carbon budget are not well understood in China (Felzer et al., 2005; Ren et al., 2007a, b). Though several experiments investigating O_3 effects on croplands have been carried out in China for around 20 years (e.g. Wang et al., 1995, 2002; Guo et al., 2001; Bai et al., 2003), few have focused on the impacts of O_3 pollution on the carbon budget at national scale. Furthermore, China is dominated by a monsoon climate and climate variability/change has exerted significant impact on carbon dynamics in China's terrestrial ecosystems (Cao et al., 2003; Tian et al., 2003; Fang et al., 2007). To illustrate the O_3 effects on a continental scale, therefore, it is necessary to consider the interactive effects of O_3 and other environmental factors and the resulting impact on terrestrial ecosystem production and carbon storage.

In order to investigate the magnitude and variation of NPP and C storage in terrestrial ecosystems across China in responses to multiple environmental stresses during the period 1961-2005, this study applied an improved integrated process-based ecosystem model and refined gridded regional database. Emphasis is placed on examining the influences of O_3 pollution and climate variability/change, and the sensitivities of different functional types to these factors.

1. Objectives

The overall goal of this study is to examine the response of the terrestrial carbon cycle in China to tropospheric O_3 pollution and climate variability/change during the period 1961-2005. It is hypothesized that the carbon fluxes (i.e. NPP) and carbon pools (carbon storage in vegetation and soil pools) have been influenced by global change,

particularly the increasing tropospheric O_3 concentrations and climate variability/change; the responses in different terrestrial ecosystems (grass, forest and agriculture) have varied. The study is organized by several interlinked questions and hypotheses relating to O_3 pollution, climate change, and carbon cycle responses:

Question 1: What were the changes in tropospheric O_3 and climate in China over the past decades (1961-2005)?

Hypothesis: Tropospheric O_3 and climate variability/change show substantial spatial and temporal variations due to regional differences in economic development and environmental conditions.

Question 2: How has air pollution and climate variability/change affected the terrestrial carbon cycle in China over the past decades (1961-2005)?

Hypothesis: Direct and indirect effects of O_3 pollution and climate variability/change have resulted in NPP reduction and carbon loss from the terrestrial ecosystems since 1961.

Question 3: How have the carbon cycles of different functional types changed in response to O_3 pollution and climate variability/change?

Hypothesis: Natural (forest and grassland) and managed ecosystems (cropland) in the context of global environmental change have had different responses to O_3 pollution and climate variability/change.

Our strategy for answering these questions involves the following tasks:

i. Examine the spatial and temporal characteristics of tropospheric O_3 and climate variability/change;

ii. Assess the impacts of the tropospheric O_3, climate variability/change, combined with other changing environmental factors, on terrestrial carbon cycle in China;

iii. Further examine the impacts of changes in tropospheric O_3 and climate on different ecosystems including grassland, forest, and cropland.

2. Approaches

To address these objectives and answer the above questions, the study is conducted by applying process-based model simulation driven by refined gridded regional database. Two components are necessary: 1) the integrated ecosystem model, which can simulate the underlying mechanisms of the impacts of elevated O_3 concentrations, increasing temperature, and precipitation change on the carbon cycle; 2) the long term and regional database as inputs for the model, including historical information on O_3 pollution, climate variability/change and other environmental factors such as land cover/use change, land management (application of fertilizer and irrigation, harvest, rotation), and increasing CO_2. The study is carried on using the following steps and tasks:

- Model development
    1) Improve the process-based ecosystem model, Dynamic Land Ecosystem Model (DLEM), by developing a O_3 sub-model;

2) Develop an agricultural version of Dynamic Land Ecosystem Model (DLEM-Ag), by including agronomic practices that distinguish the natural and managed ecosystems;

3) Calibrate, parameterize and validate the model at the site and regional levels

- Data collection, development, application, and analysis

    1) Collect basic information and observed database at site level and regional level;

    2) Develop regional datasets such as irrigation, rotation maps and historical fertilizer applications;

    3) Apply and analyze the O_3 pollution, temperature and precipitation database

- Model application and regional assessment

    1) Assess NPP and C storage in responses to the combined effects of O_3 pollution, climate variability/change and other environmental factors for all the terrestrial ecosystems across China

    2) Explore the spatio-temporal variations of NPP and C storage in natural and managed ecosystems

    3) Examine the sensitivities of NPP and C storage, including their responses to the single effects of O_3 pollution and climate variability/change as well as combined effects through experimental simulations

- Model uncertainties analysis

    1) Examine the model responses to multiple environmental changes through conducting sensitivity experiments

    2) Validate the modeled results at site level

3) Compare the modeled results with historical survey records and contemporary observations from remote sensing at the regional level

4) Compare the modeling results from DLEM and TEM models

3. Dissertation Structure

Chapter 1 gives an introduction of the background, research questions and hypotheses, study objects and tasks, and approaches used in this study.

Chapter 2 presents a detailed literature review on multiple environmental factors and their impacts on the carbon cycle in terrestrial ecosystems. It emphasizes the significant impact of increasing O_3 concentrations on terrestrial ecosystem carbon sequestration in China over multiple decades, and points out the importance of assessing NPP and C storage and considering tropospheric O_3 pollution, climate variability/change and other environmental factors in the context of global change. It also points out that due to complex ecosystem feedbacks and the interactions among different environmental controls, large uncertainties exist in current assessments that use traditional methods. Finally, this chapter describes the advantages of process-based model simulation and its application in regional carbon cycle studies.

Chapter 3-6 includes four sections for the assessments of NPP and C storage in different ecosystems. It first makes the general assessment of NPP and C storage in the terrestrial ecosystems in China as a whole, considering tropospheric O_3 pollution in the context of global change. It further investigates the impact of tropospheric O_3 pollution and climate variability/change in combination with other environmental factors on natural

ecosystems (e.g. grasslands, forests) and managed ecosystem (croplands) across China. The general description of the process-based ecosystem model and all the input datasets are presented in Chapter 3, and the development of the model and refined databases are individually described in the following chapters. In Chapter 3, the DLEM model and the long-term high resolution gridded dataset are applied to assess the magnitude and variations of NPP and C storage across China's terrestrial ecosystems in responses to both O_3 effects and no O_3 effects, which helps to explore the different sensitivities of each ecosystem to O_3 pollution and promote further investigation of these three ecosystems. Chapter 4 and chapter 5 investigate the impacts of O_3 pollution combined with climate variability/change on natural ecosystems (grasslands and forests) in China. Without too much disturbance derived from human activities, it helped to examine the spatial-temporal variations of NPP and C storage in responses to a single O_3 influence and the combined effects of O_3 pollution and climate variability/change at regional level. Chapter 6 conducts a synthetic simulation study on the impact of multiple stresses including O_3 pollution, climate change/variability, land management and other environmental factors on croplands in China. This chapter also develops the agricultural version of the ecosystem model (DLEM-Ag) as well as the relevant input database (e.g. cropping system map, irrigation and fertilizer dataset) consistent with the complex characteristics of agricultural ecosystems. The effects of O_3 pollution and climate variability/change on crop NPP and soil carbon storage, and the interactive effects involved in land management were also investigated.

Chapter 7 validates the model results and analyzes the uncertainties in this study. This chapter presents the general responses of the model to environmental changes and particularly examined the sensitivity of our model to O_3 exposure through conducting sensitivity experiments. Potential problems contributing to uncertainties in the assessments in this study were also analyzed and quantified through comparing the simulated results with the measurements at site level, survey records and observations from remote sensing.

Chapter 8 summarizes the overall findings in the dissertation, and assesses uncertainties and proposes directions for future work.
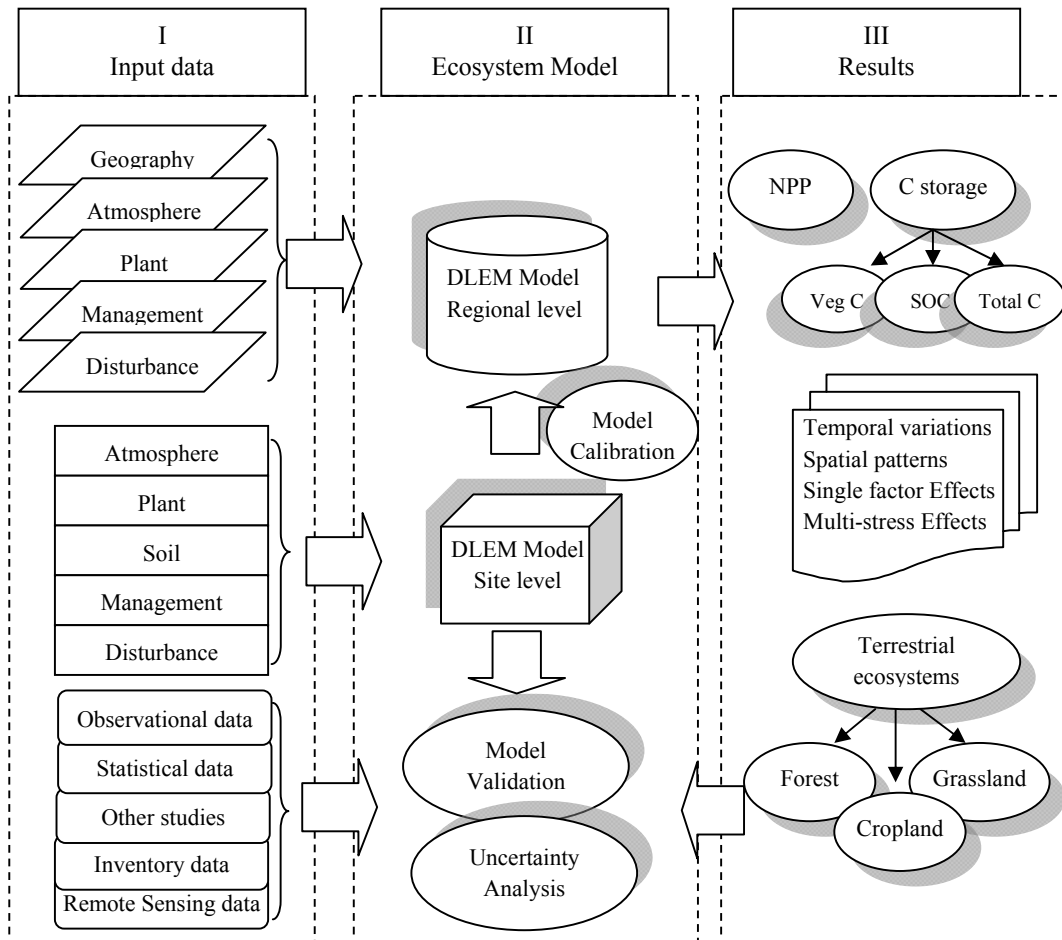
Figure 1-1 Framework of the research

Chapter 2

Literature Review

1. Carbon cycling in terrestrial ecosystems, climate change and environmental problems

1.1 Carbon cycling

Carbon is one of the most abundant elements in the universe and provides the structural basis for all known life forms. The carbon cycle in the terrestrial biosphere is a crucial biogeochemical cycle, providing basic food production, influencing climate regulation, and supporting other nutrient cycles, etc (Schimel et al., 2001). Carbon dioxide (CO_2), the major component of the carbon cycle, not only affects the rate of key ecological processes, such as plant photosynthesis and respiration, but as one of the most important green house gases (GHGs), also strongly affects the heat insulating capability of the atmosphere. The carbon cycle is changing rapidly as a result of human activities, substantially altering the Earth's climate (e.g. atmospheric temperature) and human living environments (atmosphere components). It has been reported that global warming and looming environmental problems (e.g. air pollution, water quality etc.) are among the major threats to human society, and both of these are closely related to the carbon cycle (Melillo et al., 1993, 1996; Tian et al., 1998, 2003; Cao and Woodward, 1998; White et al., 1999; Chapin et al., 2002; Field et al., 2007; Luo, 2007).

13

1.2 Climate change

Global warming has been attracting more and more attention since the 1980s. Average global temperatures increased by 0.74 ± 0.18 °C (1.33 ± 0.32 °F) during the 20th century, with the rate of 0.2 °C per decade since 1976 (Hansen et al. 2006). A further 1.1 to 6.4 °C (2.0 to 11.5 °F) increase in the global surface temperature during the twenty-first century was reported in the latest IPCC, based on climate models (IPCC, 2007). The IPCC (2007) report concludes that increasing GHGs concentration resulting from human activities such as fossil fuel burning and deforestation (Marland et al. 2002; Houghton and Hackler et al., 2003) caused a significant perturbation in the natural cycling of carbon between land, atmosphere and oceans, and is therefore responsible for most of the observed temperature increase since the middle of the 20th century. For example, every 1 ppm (10^{-6}) increase in atmospheric CO_2 concentration can increase global mean temperatures by about 0.01 °C (Prentice et al., 2001). By 2100, atmospheric CO_2 concentration will have risen from its current value of 380 ppm to between 500 and 1000 ppm; global mean temperatures would then rise by 1.5 °C to 5.8 °C due only to this change in CO_2.

1.3 Air pollution

Changes in atmospheric chemistry (e.g. increasing CO_2, N, and tropospheric O_3) have not only contributed to climate change but have resulted in increased air pollution, which is one of the most pressing environmental problems worldwide. Ozone (O_3) is the third most important greenhouse gas after CO_2 and CH_4 and is now also becoming a poisonous gas to human beings and plants as its concentrations in the troposphere

14

increase. It is reported that the tropospheric O_3 level has increased on multiple scales - local, national, continental, and even global (Akimoto, 2003). Streets and Waldhoff (2000) suggested that tropospheric O_3 levels might increase substantially in the future. Advection from the Asian continent increases pollutant levels over the Pacific Ocean (Jacob et al., 1999; Mauzerall et al., 2000), and will eventually influence North America and Europe by intercontinental transport (Jaffe et al., 2003; Wild and Akimoto, 2001). O_3 can influence both ecosystem structures and functions (e.g. Heagle, 1989, 1999; Ashmore, 2005; Muntifering et al., 2004). Over 90% of the damaging effects of air pollution on ecosystems on a regional basis may be the result of tropospheric O_3 (Adams et al. 1986), which can cause up to a 30% reduction in crop yield and forest production (Adams et al., 1989). Approximately 50% of global forests might be exposed to higher O_3 levels (>60 ppb, a value that has been employed in impact assessment research in USA) by 2100 (Percy et al. 2003). Consequently, the need to investigate the adverse effects of O_3 on terrestrial ecosystem production is urgent.

1.4 Environmental changes in China

China has experienced extensive environmental changes over the past four decades, including climate change, increasing O_3 pollution and land use change (Tian et al., 2003; Chen et al. 2006; Felzer et al. 2005; Liu et al. 2005; Liu et al., 2005a, b; Liu and Diamond 2005; Lu and Tian 2007; Ren et al., 2007a, b). Global warming and air pollution have become the most pressing environmental concerns in China in recent decades (Chen et al., 2005; Liu and Diamond, 2005). Chen et al., (2006) found that China experienced a large increase of $0.23^{o}C$ and 10.6 mm per decade for average air temperature and precipitation, respectively. But these variations in temperature and

15

precipitation had distinct regional differences (Sha et al., 2002; Zuo et al., 2003; Qian et al., 2009). The rate of change in air temperature in China was significantly higher than the corresponding global average during the same time period (about 0.1 $^{\circ}$C increase per decade from 1961to 2005, Brohan, et al. 2006; IPCC 2007). Documented evidence suggests that ground O_3 concentration increased significantly in three decades in China, based on site measurements, observations from remote sensing and simulated results (Feng et al., 2003; Wang et al., 2004; Wang et al., 2007); other studies predict that ground O_3 will continue to increase in the coming years with a peak concentration of 150 ppb in some locations (Elliot et al., 1997; von Aardenne et al., 1999). All the impacts of these environmental changes on China's terrestrial carbon cycle areas yet unknown (Fu et al., 1999; Cao et al., 2003; Tian et al., 2003; Felzer et al., 2005; Chen et al., 2006; Ren et al., 2007a, b).

2. Net primary productivity, carbon storage and carbon sink in terrestrial ecosystems

2.1 Caron cycling and terrestrial ecosystems

Documented evidence shows that the terrestrial ecosystems can respond rapidly to an altered carbon cycle. Land and oceans absorbed more than 50% of atmospheric CO_2 released from human activities and fossil fuel burning in the past two decades (IPCC 2007). The terrestrial biosphere interacts strongly with the climate, providing both positive and negative feedback. Carbon sink (carbon uptake and storage processes) can provide a negative feedback to the altered carbon cycle, maintaining the stability of the global ecosystem, atmosphere system and climate system (Pacala et al. 2007). Net primary productivity (NPP) is regarded as an important variable to measure energy input

to the biosphere and terrestrial CO_2 assimilation, and carbon storage in different pools (vegetation and soil) is generally used to measure the magnitude of carbon stored in different ecosystems. Both variables indicate the potency of the carbon sinks in terrestrial systems. The need to assess and understand the variability and controls of carbon sinks in responses to the combined effects of climate change induced by human disturbances and other environmental stresses, such as atmospheric components changes (e.g. increasing O_3 pollution), land-use change and land management, is becoming increasingly important in the 21st century (Denman et al., 2007).

2.2 Caron sinks in terrestrial ecosystems

Ciais et al (1995) suggests that there was a significant land sink in the Northern Hemisphere. A net sink of 1-2.5 Pg C/yr was estimated to be distributed evenly between North America and Eurasia (Prentice et al., 2001, Schimel et al. 2001, Gurney et al., 2002). However, the magnitude, geographical distribution, and causes of a northern, mid-latitude terrestrial carbon sink are uncertain (Houghton and Hackler, 2003). Agriculture and forestry are in a unique position due to their positive role in helping reduce the buildup of carbon dioxide in the atmosphere through sequestering or storing carbon in perennial vegetation (such as plant residues and wood) and by increasing soil carbon in the form of soil organic matter in croplands with conservation practices. For example, the introduction of conservation tillage in the USA is estimated to have increased soil organic matter (SOM) stocks by about 1.4 Pg C over the last 30 years. Schimel et al. (2000) suggested that U.S. agriculture could remain a modest sink for decades to come with the best management practices. Regional studies have confirmed that forest re-growth is one of the strongest mid-latitude sinks for C. Both data from the

17

eddy flux tower network (Curtis et al., 2002; Hollinger et al., 2004) and forest inventory

(Pacala et al., 2001; Woodbury et al., 2007) showed that forests on long-abandoned

former agricultural lands and in industrially managed forests take up significant amounts

of carbon every year.

2.3 Caron sinks in China

China has the third largest land area in the world, most of it within northern

mid-latitudes. Studies suggest that China's terrestrial ecosystems have contributed to

global carbon sinks in the past two decades (1980-2000). A mean annual carbon

sequestration rate in forests was estimated to range from 57.7 g C/m^2 to 62.3 g C/m^2 with

a total of 0.068 to 0.075 Pg C for the period 1981-2000 based on forest inventory data

(Pan et al., 2004; Fang et al., 2007). Fang et al (2007) also estimated an annual carbon

sequestration rate in grassland ecosystems of 2.1 g C/ m^2/yr or a total of 0.007 Pg C for

the same period. Using field inventory data, Huang et al (2006) found 0.018-0.022 Pg

carbon per year has been sequestrated in the upper 20 centimeters of soil in China's

croplands since the 1980s. Annual crop NPP was estimated to continue to increase in the

second half of the 20th century, going from 146 Tg C/yr in the 1950s to 513 Tg C/yr in

the 1990s, which implied that the increasing crop residues could be attributed to soil

organic carbon (Huang et al., 2007). The estimation of annual forest NPP was 0.59 Pg per

year based on the inventory data (Fang et al., 1996), and modeled results showed

different estimations of annual NPP ranging from 0.76 Pg/yr to 3.01 Pg/yr (Xiao et al.,

1998; Feng et al., 2004; Tao et al., 2006). In addition, due to differences in research

methods (e.g. NDVI-biomass, forage yield, biomass and carbon density, inventory,

ecosystem models), the assessment of vegetation carbon storage (0.13-4.66 Pg C in

grasslands, and 4.38- 57.9 Pg C in forests) varied significantly among different studies (Fang et al., 1996, 2001; Ni, 2001, 2004; Piao et al., 2004, 2005; Peng and Apps, 1997; Liu, 2001; Wang et al. 2002). All the studies mentioned above suggest that China's terrestrial ecosystems have great potential for carbon sequestration, especially in agricultural ecosystem and forest ecosystems. The assessments of NPP and carbon storage in vegetation and soil pools in China's terrestrial ecosystems are helpful in recognizing the role of worldwide agriculture, forest and grassland in the global carbon budget. However, discrepancies in the results of various studies which used different methods indicate that further work need to be done to reduce the uncertainty of assessing NPP, carbon storage and carbon sink; these uncertainties are largely due to a lack of informative databases or a poor understanding of the mechanisms of carbon sink in the terrestrial ecosystems.

3. Factors influencing carbon sink in the terrestrial ecosystems

3.1 Main drivers and their effects on carbon sink in terrestrial ecosystems

Since carbon sink in terrestrial ecosystems has complex feedbacks to climate systems, it is critical that we understand the underlying mechanisms for carbon uptake and what is likely a trend in changing carbon uptake, which are of great importance for reasonable future prediction. Basically, the carbon cycle in terrestrial ecosystems is driven by the direct impacts of climate change (temperature, precipitation, and radiation regime etc.), atmospheric composition (e.g. increasing CO_2, nitrogen deposition, pollution damage), land use/cover change (deforestation, agricultural practices, and their legacies over time), and disturbances derived from nature and human activity (e.g. wild/

prescript fire, flooding, hurricane) (IPCC, 2007). Those drivers can directly, indirectly

and interactively affects the carbon cycle and could cause carbon release and carbon

uptake in the terrestrial ecosystems.

More than 50% of the terrestrial vegetation carbon is stored in forest ecosystems

(Dixon et al., 1994) and boreal forests soils account for about 26% of the total terrestrial

carbon stock. It has been suggested that forest re-growth is the major contribution to the

land carbon sink (e.g. Pacala et al., 2001; Schimel et al., 2001; Hurtt et al., 2002). The

20th century trend of increasing forest area at middle and high latitudes has led to carbon

sequestration by re-growing forests. Forest area in China accounts for approximately

18.2% of total national land area, which has continually increased during the 1990s, even

as total global forest area has decreased during the same period (FAO, 2005).

Consequently, land use change induced by human activity does directly influence the

carbon cycle by changing land cover area and land use cover category. In addition,

intensive land management, including fertilizer/irrigation, harvest, tillage, can also

enhance carbon accumulation in vegetation while stimulating green house gas emissions

from soil respiration (e.g. Cole et al., 1996; Smith et al., 1997). Climate change has both

positive and negative effects on the vegetation carbon balance. Warming trends could

lengthen growing seasons, thus increasing plant productivity (Nemani et al. 2003), and

accelerating soil decomposition resulting in carbon release instead of carbon

accumulation (Hobbie and Chapin 1998; Oechel et al., 2000; Rustad et al., 2001; Melillo

et al., 2002). Changes in atmospheric compositions, such as increasing CO_2 and nitrogen

deposition, were found to stimulate carbon assimilation and sequestration (e.g. Cramer et

al., 2001; Oren et al., 2001; Luo et al., 2004; DeLucia et al., 2005; Mellio and Gosz,1983; Schindler and Bayley,1993;Gifford etal.,1996; Holland *et al.*,1997; Neff *et al.*,2000; Matson *et al.*, 2002); however, their "fertilizer effects" may have been exaggerated by some models, as much smaller changes of NPP in response to increased CO_2 and nitrogen saturation and even NPP reductions were also observed (e.g. Nowak et al., 2004; Emmett et al.,1996; Magill et al.,2000; Guan et al., 2004).

3.2 Tropospheric O_3 and its effects on carbon sink in terrestrial ecosystems

Tropospheric O_3 is a major secondary air pollutant and a rapidly changing atmospheric component, whose level has been increasing across a range of scales - local, national, continental, and even global (e.g. Akimoto, 2003; Jacob et al., 1999; Mauzerall et al., 2000; Streets & Waldhoff, 2000; Jaffe et al., 2003). Documented evidence from numerous field experiments indicates that O_3 has significant adverse effects on plant growth, at the cell level, and on carbon sink, at the ecosystem level. O_3 pollution can directly and indirectly reduce the photosynthesis rate by injuring leaf structure and mesophyll tissue and influencing RubP and plant growth substances (Farage et al., 1991; McKee et al., 1995; Pell et al, 1997). O_3 pollution can also change stomatal conductance (either increasing or decreasing), which then alters stomatal response to irradiance, vapor pressure deficit (VPD) and internal carbon dioxide concentrations (Ci) (Tjoekler et al., 1995; Grulke et al., 2002). Increasing carbon allocation to leaves rather than roots may have been an attempt to maintain photosynthesis, but it also reduced water and nutrient uptake due to decreasing ground biomass (Woodbury et al., 1994; Bergman et al., 1995; grantz & Yang, 2000; Oksanen & Rousi, 2001; Karlssson et al., 2003b). In addition, O_3 can decrease soil decomposition by influencing enzyme and microbe activities and

changing the litter fall characteristics (Kim et al., 1998; Islam et al., 2000; Olszyk, 2001; Pregitzer & King, 2004). Therefore, increased tropospheric O_3 pollution can inhibit both plant productivity and soil respiration (Adams et al. 1986; Reich 1987; Chappelka and Samuelson 1998; Booker et al., 2009).

O_3 damage to crops and forests has been documented in North America and Europe in the past two decades (Heck, 1984, 1988; Heagle, 1989; Skelly et all, 1999; Fumigalli et al., 2001; Emberson et al., 2001; Orendovici et al., 2003; Chappelka et al., 2002, 2003; Vollenweider et al, 2003; Mills et al., 2007). In China, similar studies have been conducted to investigate crop response to O_3 pollution since the 1990s (Wang et al, 1995; Ji and Feng, 2001; Bai et al.,2002; Guo et al., 2003; Wang et al., 2006), though study of O_3 effects on forests and grasslands has been limited. Regional studies have indicated that the detrimental effects of tropospheric O_3 on plant growth may reduce carbon sequestration (Ollinger et al., 2002b; Felzer et al, 2004, 2005; Ren et al., 2007a, b). Felzer et al, (2004) estimated that CO_2 sequestration in the USA was reduced by 18 – 20 Tg C/yr possibly due to increasing ground O_3 concentrations since 1950. To reduce uncertainty, O_3 pollution should not be ignored when estimating the carbon budget between the atmosphere and the terrestrial ecosystems in climate system studies. However, the current generation of coupled carbon-climate models doesn't account for air pollution effects.

In China, few studies have been conducted to examine the impacts of O_3 on regional estimation of NPP, C storage and net carbon sinks; however, several field

experiments on O_3 effects in croplands have been carried out for about 20 years (Wang et al., 1995, 2002; Guo et al., 2001; Bai et al., 2003). Furthermore, China is characterized by a monsoon climate and the impacts of climate variability/change on carbon dynamics in terrestrial ecosystems are large (Tian et al., 2003; Cao et al., 2002; Fang et al., 2007). Therefore, to illustrate O_3 effects at a continental scale, it is necessary to consider the interactive effects of O_3, climate change and other environmental factors on terrestrial ecosystem production and carbon storage.

4. The ecosystem complexity and ecosystem model approach

Due to complex ecosystem responses to changing environmental factors (sunlight, temperature, soil moisture) (Clark, 2002; Ciais et al., 2005b; Dunn et al., 2007), large uncertainties still exist in assessing the terrestrial carbon budget, even though experiments and model simulations have been designed to study a wide variety of factors in the terrestrial cycles (Schlesinger 1997; Chapin III et al. 2002). There are several aspects of ecosystem complexity. Firstly, a single environmental factor can have positive, negative or both effects on ecosystems. For example, global warming can enhance carbon sequestration in mid-latitude temperate forests through increasing soil nitrogen mineralization rate (Melillo et al., 2002); however, it can also reduce the carbon accumulation in arid regions by reducing available water (Melillo et al. 1993). Secondly, environmental stresses usually interact to produce a combined impact on ecosystem functioning instead of operating independently (Schindler2001). For instance, the adverse effects of elevated O_3 pollution can be offset by increasing CO_2 through reduced stomatal conductance; O_3 uptake can also be accelerated by drought and fertilizer application (Ollinger et al., 2002; Felzer et al., 2005). Accordingly, ecosystem response to

multifactor environmental changes is the result of a complex combination of effects rather than simply the accumulation of the effects of a single factor (Norby and Luo 2004). In addition, inherent heterogeneity of the landscape at spatial scales ranging from microns to thousands of kilometers is a problem that can hardly be solved by field experiments. Accurate assessments and predictions of the carbon cycle in terrestrial ecosystems in responses to global changes, therefore, depends on the successful integration of a range of processes and time scales, using an ecosystem model, rather than traditional statistics and controlled experiments (Ollinger et al., 2002a; Hui and Luo 2004).

Quantitative assessments of NPP, carbon storage and carbon sinks in terrestrial ecosystems and their responses to increasing O_3 pollution in the context of global change are in need of O_3 concentration databases and process-based models that are able to address the mechanisms of effects within the ecosystems. Among the various indices used to determine O_3 effects (e.g. AOT40, SUM00, SUM60, W126, details in Mauzerall and Wang, 2001), AOT40 is widely used to define the critical level of O_3 exposure within ecosystems in terms of the hourly accumulated exposure over a threshold of 40 ppb. In Europe, however, there has been intensive debate about replacing the AOT40 index with modeled cumulative flux or uptake for regional risk assessment (Fuhrer et al., 1997). However, AOT40 is used since they can provide broad-scale assessments of the impact of O_3, and associated databases have been developed in past decades at multiple scales from local, regional to a global level (Felzer et al., 2004, 2005). For example, in China, some sites have included O_3 concentration observations and related field experiments have

24

been conducted since the 1990s (e.g. Wang et al., 2007). Therefore, the simulated AOT40 database derived from chemical-transport models is practical for regional assessment (Ren et al., 2007a).

Ecosystem-level models are another necessary tool for quantitative assessments on the carbon cycle in terrestrial ecosystems in response to O_3 pollution. An ecosystem model allows researchers to conceptualize and measure a complex system and to predict the consequences of an action that would be expensive, difficult, or destructive to an ecosystem (Haefner, 2005). Ecosystem modeling aims to pursue the integration of natural processes rather than isolate a particular component as traditional field control experiments tend to do. Also, ecosystem modeling is able to study complex systems involving many nonlinear interactions among multiple subsystems over long period of time. Both traditional and ecosystem modeling are necessary because ecosystem modeling relies on traditional field work that can provide both basic phenomenon observation and quantitative mechanisms. For example, the quantitative relationship of O_3 effects on photosynthesis was first incorporated into the ecosystem model based on sustainable field studies (e.g. Reich, 1987). The final modeled outputs of NPP, carbon storage and other variables are controlled by the type of ecosystem due to the diverse sensitivity of different functional types to O_3 effects, and environmental conditions such as light, temperature, water and nutrient supply from plant and soil. Therefore, the ecosystem model is a powerful tool to investigate the interactions among ecosystem components.

Since the 1980s, O_3 studies have been conducted using empirical or process-based dynamic simulations (Ren and Tian, 2007). Among well-documented empirical models, Weibull function, based on exposure indices and corresponding exposure-response relationships, has been used to assess crop and forest production loss, as well as economic losses (e.g. Heck et al., 1984; Heggestad et al.,1990; Chameides et al., 1994; Aunan et al., 2000; Kuik et al., 2000; Wang and Mauzerall, 2004). Several process-based models have attempted to study the effects of O_3 on vegetation and have begun regional assessments of carbon storage (eg. Reich, 1987; Ollinger et al., 1997, 2002a; Martin, 2001; Felzer et al., 2004, 2005). Reich (1987) modeled an empirical linear model that describes the response of crops and trees to O_3 and argued that crops were more sensitive to O_3 than other functional types. Ollinger (1997) used O_3-response relationships with the PnET-II model to simulate tree growth and ecosystem functions and addressed the combined effects of CO_2, O_3, and N deposition along in the context of historical land-use changes, but only focused on hardwoods in northeastern U.S. Martin et al. (2001) incorporated O_3 effects on photosynthesis and stomatal conductance into the functional-structural tree growth model ECOPHYS by using O_3 flux data; however, it is difficult to scale this up to an ecosystem study. Felzer et al. (2004, 2005) first incorporated algorithms from Reich et al. (1987) and Ollinger et al. (1997) for hardwoods, conifers, and crops into a biogeochemical model TEM (Terrestrial Ecosystem Model); while this model is lacking mechanisms of O_3 response to the links between photosynthesis and stomatal conductance, and the agriculture module is too simple to address the responses of crop types. In addition, it is important to conduct synthetic studies, assessing the dynamic responses of the carbon in terrestrial ecosystems to O_3

26

pollution combined with other environmental stresses (e.g. changing CO_2, climate) in the context of global changes. The model-based analyses of Ollinger et al. (2002a) and Hanson et al. (2005) indicate that the complex interactions between O_3 and other environmental factors could lead to great uncertainty for multivariate predictions of ecosystem response.

Chapter 3

Terrestrial Ecosystems, Ozone Pollution –

Effects of Tropospheric Ozone Pollution on Net Primary Productivity and

Carbon Storage in Terrestrial Ecosystems of China

Abstract

We investigated the potential effects of elevated ozone (O_3) along with climate variability, increasing CO_2, and land-use change on net primary productivity (NPP) and carbon storage in China's terrestrial ecosystems for the period 1961-2000 with a process-based Dynamic Land Ecosystem Model (DLEM) forced by the gridded data of historical tropospheric O_3 and other environmental factors. The simulated results showed that elevated O_3 could result in a mean 4.5% reduction in NPP and 0.9% reduction in total carbon storage nationwide from 1961 to 2000. The reduction of carbon storage varied from 0.1 Tg C to 312 Tg C (a decreased rate ranging from 0.2% to 6.9%) among plant functional types. The effects of tropospheric O_3 on NPP were strongest in east-central China. Significant reductions in NPP occurred in northeastern and central China where a large proportion of cropland distributed. The O_3 effects on carbon fluxes and storage are dependent upon other environmental factors. Therefore, direct and indirect effects of O_3, as well as interactive effects with other environmental factors, should be taken into account in order to accurately assess the regional carbon budget in

China. The results showed that the adverse influences of increasing O_3 concentration across China on NPP could be an important disturbance factor on carbon storage in the near future, and the improvement of air quality in China could enhance the capability of China's terrestrial ecosystems to sequester more atmospheric CO_2. Our estimation of O_3 impacts on NPP and carbon storage in China, however, must be used with caution because of the limitation of historical tropospheric O_3 data and other uncertainties associated with model parameters and field experiments.

Keywords: air pollution; carbon storage; China; climate change; net primary productivity; tropospheric ozone.

1. Introduction

The tropospheric ozone (O_3) level has been increasing across a range of scales - local, national, continental, and even global (e.g. Akimoto, 2003). Tropospheric O_3 levels might increase substantially in the future (Streets and Waldhoff, 2000). Advection from the Asian continent increases pollutant levels over the Pacific Ocean (Jacob et al., 1999; Mauzerall et al., 2000), and eventually influences North America and Europe by intercontinental transport (Jaffe et al., 2003; Wild and Akimoto, 2001). O_3 can influence both ecosystem structure and functions (e.g. Heagle, 1989, 1999; Ashmore, 2005; Muntifering et al., 2006). Over 90% of vegetation damage may be the result of tropospheric O_3 alone (Adams et al. 1986),   and it could cause reductions in crop yield and forest production ranging from 0% - 30% (Adams et al., 1989). Approximately 50% of forests might be exposed to higher O_3 level (>60 ppb) by 2100. Therefore, there is an urgent need to investigate the adverse effects of O_3 on terrestrial ecosystem production.

Air pollution is one of the most pressing environmental concerns in China (Liu and Diamond, 2005). The rapid urbanization and industrialization, and intensive agricultural management in the past decades, are closely related to increasing fossil fuel combustion and fertilizer application. Between 1980 and 1995, fertilizer use in China was 36% higher than the average in developed countries (where fertilizer use has been decreasing), and 65% higher than the average in developing countries (Aunan et al., 2000). Both fossil fuel consumption and N-fertilizer application will highly contribute to total emissions of NO_x, a main O_3 precursor, and consequently result in increased atmospheric O_3 concentration. It was estimated that China's emissions of NO_x might increase by a factor of four towards the year 2020, compared to the emissions in 1990 under a non-control scenario (von Aardenne et al., 1999), which would lead to a much larger increase of surface O_3 with 150 ppb level of O_3 in some locations (Elliot et al., 1997). Consequently, it is important to study the impacts of O_3 on terrestrial ecosystems in China. Although studies on O_3 have been carried out in China for about 20 years, observations of O_3 concentrations are still limited, and the records of most sites are discontinuous (eg. Chameides et al., 1999; Liu et al., 2004; Wang et al., 2007). Several experiments demonstrated the interaction of O_3 and CO_2 on locally grown species and cultivars in China (e.g. Wang et al., 1995, 2002; Guo et al., 2001; Bai et al., 2003). However, these studies rarely involved other plant functional types (PFTs), such as forests and grassland. An assessment of O_3 effects on different PFTs at regional level over a long-time period has not been done yet. To illustrate the O_3 effects at a continental

30

scale, it is necessary to consider interactive effects of O_3 with other environmental factors on terrestrial ecosystem production and carbon storage.

Quantitative assessment of O_3 effects on terrestrial ecosystem production has been conducted since the 1980s based on empirical or process-based dynamic simulations (Ren and Tian, 2007). Well-documented empirical models, such as the Weibull function, are based on exposure indices and corresponding exposure-response relationships, and have been used to assess crop and forest production loss, as well as economic losses (e.g. Heck et al., 1984; Heggestad et al.,1990; Chameides et al., 1994; Aunan et al., 2000; Kuik et al., 2000; Mauzerall and Wang, 2004). Process-based models allow plant growth responses to vary with dynamic environments, such as high O_3 concentration, elevated CO_2 concentration, and climate change (Tian et al. 1998a). Several process-based models have attempted to study the effects of O_3 on vegetation (eg. Reich, 1987; Ollinger et al., 1997, 2002; Martin, 2001; Felzer et al. 2004, 2005).   Reich's (1987) model is not actually a process-based model, but he generalized a linear model to describe the response of crops and trees to O_3 and argued that crops were more sensitive to O_3. Ollinger (1997) used O_3-response relationships with the PnET-II model to simulate tree growth and ecosystem functions. These models can apply the dynamic O_3 damage mechanisms in seedling and mature trees from leaf level to canopy level. Ollinger and his colleagues (2002) applied the model to study the effect of O_3 on NPP for specific sites within the northeastern U.S. (a reduction in NPP of between 3 and 16%) and the combined effects of CO_2, O_3, and N deposition along with the context of historical land-use changes for hardwoods in the northeastern U.S. with a new version of PnET

(PnET-CN). Felzer et al. (2004, 2005) incorporated the algorithms from Reich et al. (1987) and Ollinger et al. (1997) for hardwoods, conifers, and crops into a biogeochemical model (i.e., TEM). Their study across the conterminous U.S. indicated a 2.6-6.8% mean reduction in annual NPP in the US during the late 1980s and early 1990s. Unlike Ollinger's and Felzer's work, in which the effects of O_3 on stomatal conductance were not considered, Martin et al. (2001) incorporated O_3 effects on photosynthesis and stomatal conductance into the functional-structural tree growth model ECOPHYS (http://www.nrri.umn.edu/ecohpys) by using O_3 flux data. Not only did they combine the well-accepted equations from mechanistic biochemical models for photosynthesis (e.g., equations from Farquhar et al., 1980; Caemmerer and Farquar, 1981) and the equations from phenological models for stomatal conductance (Ball et al., 1987, adapted by Harley et al., 1992), but also explored the underlying mechanisms of O_3-inhibited photosynthesis models.   They found that O_3 damage could reduce both protective scavenging detoxification system (Vc max) and light-saturated rate of electron transport (J max) by the accumulated amounts of O_3 above the threshold of damage entering the inner leaves. Considering the advantages and disadvantages of different models in simulating O_3 effects, a coupled mechanistic model that fully couples energy, carbon, nitrogen, and water, as well as vegetation dynamics is needed in the near future (Tian et al. 1998a).

In this research, we used a highly integrated process-based model called Dynamic Land Ecosystem Model (DLEM) (detail description of this model can be found in Tian et al. 2005). The dynamic O_3 damage mechanisms were extrapolated from a small spatial scale (leaf level) and a short-term scale into the corresponding long-term mechanism at

the ecosystem scale. The O_3 module was primarily based on the work of Ollinger et al. (1997). The equations from Farquhar (1980) and Ball and Berry (1987) were used to simulate photosynthesis and stomatal conductance, similar to Martin et al. (2001). This module simulated O_3 damage on plant photosynthesis and NPP. We also developed the spatial datasets including historical climate, soil information, and land use change across China over a long period. The O_3 sensitivities for different PFTs including crops, coniferous trees, hardwoods, and other vegetation types, were based on the Reich's compilation of OTC experiments in the U.S., which we assume to be applicable to China as well.

More O_3 pollution in China is closely related to domestic food security and the global environment in the future (eg., Chameides et al., 1999; Akimoto, 2003). Unlike other studies in China (Aunan et al., 2000; Wang and Mauzerall, 2004; Felzer et al., 2005), we try to illustrate the effects of tropospheric O_3 pollution on terrestrial ecosystem productivity throughout the country between 1961 and 2000. We focus on the analysis of O_3 effects on NPP and carbon storage in the context of multiple environmental stresses including increasing O_3, changing climate, elevated CO_2, and land-use changes (including nitrogen fertilization and irrigation on croplands) across China. In this paper, we first briefly describe our model development, data preparation, and the experimental design, and then examine the relative effects of O_3 and other environmental factors on the spatiotemporal changes of total carbon sequestration across the country. The sensitivity of different PFTs to O_3 pollution is also examined. Finally, we discuss and analyze the simulation results and their uncertainty.
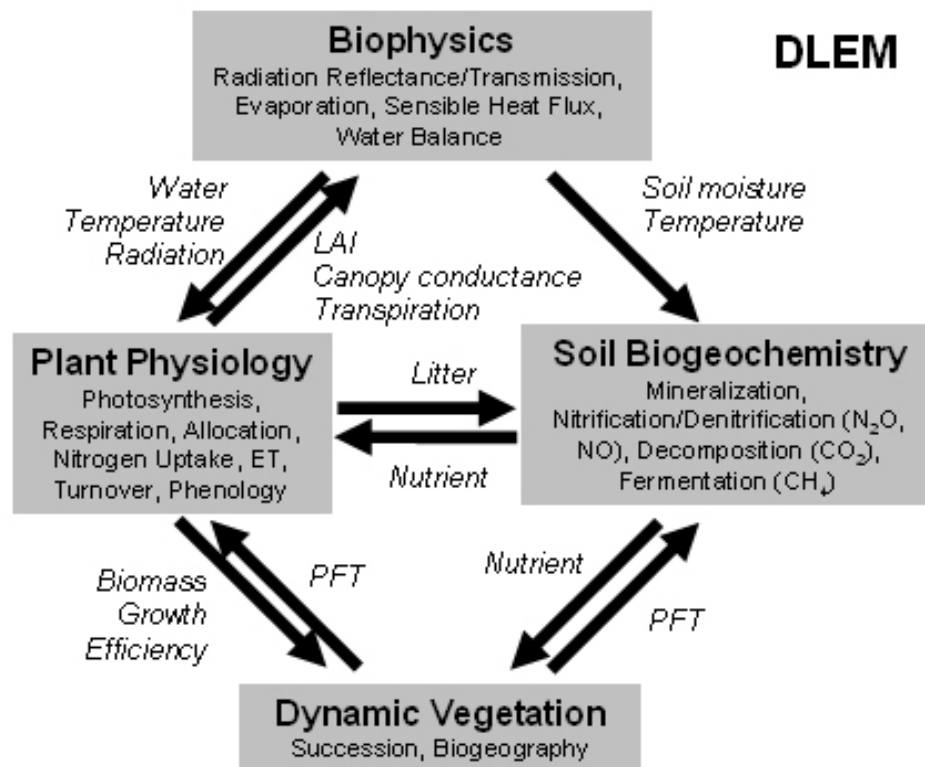
Figure 3-1 Framework of the Dynamic Land Ecosystem Model (DLEM). The DLEM model includes five core components: 1) biophysics, 2) plant physiology, 3) soil biogeochemistry, 4) dynamic vegetation and 5) land use and management (Tian et al., 2005). The DLEM is a process-based model which couples biophysical processes (energy balance), biogeochemical processes (water cycles, carbon cycles, nitrogen cycles, and trace gases (NO_x, CH_4)-related processes), community dynamics (plant distribution and succession), and disturbances (land conversion, agriculture management, forest management, and other disturbances such as fire, pest etc.) into one integral system. DLEM can simulate the complex interactions of multiple stresses such as climate change, elevated CO_2, tropospheric O_3, N deposition, human disturbance, and natural disturbances.
(See detail structure of DLEM in appendix II, Table 1)

2. Methods

2.1. The Dynamic Land Ecosystem Model (DLEM)

The DLEM couples major biogeochemical cycles, hydrological cycle, and vegetation dynamics to generate daily, spatially-explicit estimates of water, carbon (CO_2, CH_4), and nitrogen fluxes (N_2O) and pool sizes (C and N) in terrestrial ecosystems (See Figure 3-1). DLEM includes five core components: 1) biophysics, 2) plant physiology, 3) soil biogeochemistry, 4) dynamic vegetation, and 5) land use and management. The biophysical component includes the instantaneous exchanges of energy, water, and momentum with the atmosphere. It includes aspects of micrometeorology, canopy physiology, soil physics, radiative transfer, hydrology, surface fluxes of energy, moisture, and momentum influences on simulated surface climate. The component of plant physiology in DLEM simulates major physiological processes, such as photosynthesis, autotrophic respiration, carbon allocation among various parts (root, stem, and leaf), turnover of living biomass, nitrogen uptake and fixation, transpiration, phenology, etc. The component of soil biogeochemistry simulates N mineralization, nitrification/ denitrification (Li et al., 2000), NH_3 volatilization, leaching of soil mineral N, decomposition and fermentation (Huang et al. 1998). Thus, DLEM is able to simultaneously estimate emissions of multiple trace gases (CO_2, CH_4 and N_2O) from soils. The dynamic vegetation component in DLEM simulates two kinds of processes: the biogeographical redistribution when climate changes, and the plant competition and succession during vegetation recovery after disturbances. Like most DGVMs (Dynamic Global Vegetation Models), DLEM builds on the concept of PFT (Plant Functional Type) to describe vegetation distributions (figure 3-4). The DLEM has also emphasized the

simulation of managed ecosystems, including agricultural ecosystems, plantation forests, and pastures. The DLEM 1.0 version has been used to simulate the effects of climate variability and change, atmospheric CO_2, tropospheric O_3, land-use change, nitrogen deposition, and disturbances (e.g., fire, harvest, hurricanes) on terrestrial carbon storage and fluxes in China (Tian et al. 2005). This model has been calibrated against field data from various ecosystems including forests, grassland, and croplands. The simulated results with DLEM have also been evaluated against independent field data (Tian et al. 2005).

In DLEM, the carbon balance of vegetation is determined by the photosynthesis, autotrophic respiration, litterfall (related to tissue turnover rate and leaf phenology), and plant mortality rate. Plants assimilate carbon by photosynthesis, and use this carbon to compensate for the carbon loss through maintenance respiration, tissue turnover, and reproduction. The photosynthesis module of DLEM estimates the net C assimilation rate, leaf daytime maintenance respiration rate, and gross primary productivity (GPP, unit: g $C/m^2/day$). The photosynthesis rate is first calculated on the leaf level. The results are then multiplied by leaf area index to scale up to canopy level (Tian et al. 2005; Chen et al., 2006; Ren et al. 2007a, b; Zhang et al. 2007). Photosynthesis is the first process by which most carbon and chemical energy enter ecosystems so it has critical impacts on ecosystem production. The GPP calculation can be expressed as:

$$GPP_i = (A_i + Rd_i) \times LAI_i \times dayl \qquad (1)$$

$$Ai = f(PPFD_{leaf}i, g_i, leafN_i, T_{day}, Ca, dayl) \qquad (2)$$

36

where GPP (g C/m^2/day) is the gross primary productivity of ecosystems for leaf type i ; i is leaf type (sunlit leaf or shaded leaf); A (g/s/m^2 leaf) and Rd (g/s/m^2 leaf) are daytime photosynthesis rate and leaf respiration rate respectively; LAI is leaf area index; $dayl$ (s) is the length of daytime; $PPFD$ (μmol/m^2/s) is the photosynthetic photon flux density; g (m/s) is the stomatal conductance of leaf to CO_2 flux; T_{day} (ºC) is daytime temperature; Ca (ppmv) is the atmospheric CO_2 concentration; $leafN$ (g N/m^2 leaf) is the leaf N content.

Based on the "strong optimality" hypothesis (Dewar, 1996), DLEM allocates the leaf N to sunlit fraction and shaded fraction each day according to the relative PPFD absorbed by each fraction, to maximize the photosynthesis rate. In this study, NPP in an ecosystem and annual net carbon exchange (NCE) of the terrestrial ecosystem with the atmosphere were computed with following equations:

$$NPP = GPP - R_d \qquad (3)$$

$$NCE = NPP - R_H - E_{NAD} - E_{AD} - E_P \qquad (4)$$

Where NPP is the net primary productivity, Rd is the plant respiration, R_H is soil respiration, E_{NAD} is the magnitude of the carbon loss from a natural disturbance and is assigned as 0 here due to the difficulty of being simulated at present conditions, E_{AD} is carbon loss during the conversion of natural ecosystems to agricultural land, and E_P is the sum of carbon emission from the decomposition of products (McGuire et al., 2001;

Tian et al. 2003). For natural ecosystems, E_P and E_{AD} are equal to 0, and so NCE is equal to net ecosystem production (NEP). Unlike the other models which estimate the cropland C cycle based on the simulation of potential vegetation type replacing the agricultural grids (McGuire et al., 2001), the agricultural ecosystems in DLEM are not based on natural vegetation, but parameterized against several intensively studied agricultural sites in China (http://www.cerndata.ac.cn/).

To simulate the detrimental effect of air pollution on ecosystem productivity, an O_3 module was developed based on previous work (Ollinger et al., 1997; Felzer et al., 2004, 2005), in which the direct effect of O_3 on photosynthesis and indirect effect on stomatal conductance by changing intercellular CO_2 concentration were simulated. Here the ratio of O_3 damage to photosynthesis is defined as O_{3eff}, similar to Ollinger et al. (1997), and the sensitivity coefficient a for each different plant functional type is based on the work of Felzer et al. (2004). The range of α is $2.6 \times 10^{-6} \pm 2.8 \times 10^{-7}$ for hardwoods (based on the value used by Ollinger et al., 1997), $0.8 \times 10^{-6} \pm 3.6 \times 10^{-7}$ for conifers (based on pines), and $4.9 \times 10^{-6} \pm 1.6 \times 10^{-7}$ for crops which was calculated from the empirical model of Reich (1987). The errors are based on the standard deviation of the slope from the dose response curves and the standard error of the mean stomata conductance.

$$GPP_{O_3} = GPP \times O_{3eff} \qquad (5)$$

$$O_{3eff} = 1 - (\alpha \times g_s \times AOT_{40}) \qquad (6)$$

$$g_s = f(GPP_{O_3}) \qquad (7)$$

Here, GPP_{O_3} is limited GPP due to O₃ effect; g_s is the stomatal conductance

(mms⁻¹); AOT40 is a cumulative O₃ index (the accumulated hourly O₃ dose over a

threshold of 40 ppb in ppb-hr), and in this study we use a monthly accumulative index as

in Felzer et al., (2004).   The AOT40 index has often been used to represent vegetation

damage due to O₃ (Fuhrer et al., 1997). Because of limited O₃ data throughout China, we

use the model-developed AOT40 values from Felzer et al. (2005).

Our photosynthesis module, based on Farquahar model (Farquahar 1980), has the

potential ability to use O₃ concentration as input, similar to Martin et al. (2001), if the O₃

flux data are available in the future. In DLEM, the leaf C: N ratio is also affected by O₃.

We do not use this mechanism in the current study due to the ambiguous role of O₃ on

plant C:N ratio (Lindroth et al., 2001).

2.2. Input data

Input datasets include: 1) elevation, slope, and aspect maps which are derived

from 1 km   resolution digital elevation dataset of China (http://www.wdc.cn/wdcdrre); 2)

soil datasets (pH, bulk density, depth to bedrock, soil texture represented as the

percentage content of loam, sand and silt) which are derived from the 1:1 million soil

map based on the second national soil survey of China (Wang et al. 2003; Shi et al., 2004;

Zhang et al. 2005; Tian et al. 2006); 3) vegetation map (or land cover map) from the 2000

land use map of China (LUCC_2000) which was developed from Landsat Enhanced

Thematic Mapper (ETM) imagery (Liu et al., 2005a); 4) potential vegetation map, which

is constructed by replacing the croplands of LUCC 2000 with potential vegetation in

39

global potential vegetation maps developed by Ramankutty and Foley (1998); 5) standard IPCC (Intergovernmental Panel on Climate Change) historical CO_2 concentration dataset (Enting et al. 1994); 6) AOT40 dataset (see below for detail information) (Figure 3-4); 7) long-term land-use history (cropland and urban distribution of China from 1661-2000) which is developed based on three recent (1990, 1995 and 2000) land-cover maps (Liu et al., 2003, 2005a, 2005b) and historical census datasets of China (Ge et al., 2003; Xu, 1983); and 8) daily climate data (maximum, minimum, and average temperature, precipitation, and relative humidity). Seven hundred and forty six climate stations in China and 29 stations from surrounding countries were used to produce daily climate data for the time period from 1961 to 2000, using an interpolation method similar to that used by Thornton et al. (1997). To account for cropland management, we also used data from the National Bureau of Statistics of China, which recorded annual irrigation areas and fertilizer amounts in each province from 1978 to 2000 (Figure 3-6b). We did not construct an irrigation dataset due to lack of data. We simulated the effects of irrigation by refilling the soil water pool to field capacity whenever cropland soil reached wilt point. All datasets have a spatial resolution of 0.5°×0.5°, and Climate and AOT40 datasets have been developed on daily time step while CO_2 and land-use datasets on yearly time step.

2.2.1 Description of Ozone Data

The methods used for monitoring O_3 vary among the limited ground O_3 monitoring sites in China (Chameides et al., 1999; Chen et al., 1998), Therefore, it is difficult to spatially develop a historical AOT40 dataset based on the interpolation of site-level data like Felzer et al. (2004) for the U.S. In this study, the AOT40 dataset was derived from the global historical AOT40 datasets constructed by Felzer et al (2005).

This AOT40 index is calculated from combining geographic data from the MATCH model (Multiscale Atmospheric Transport and Chemistry) (Lawrence et al., 1999; Rasch et al., 1997; and von Kuhlmann et al., 2003) with hourly zonal O_3 from the MIT IGSM (Integrated Global Systems Model). The average monthly boundary layer MATCH O_3 values for 1998 are scaled by the ratio of the zonal average O_3 from the IGSM (Integrated Global Systems Model), which are 3-hourly values that have been linearly interpolated to hourly values, to the zonal O_3 from the monthly MATCH to maintain the zonal O_3 values from the IGSM (Wang et al., 1998; Mayer et al., 2000a). This procedure was done for the period 1977-2000. From 1860-1976, the zonal O_3 values were assumed to increase by 1.6% per year based on Marenco et al. (1994).

The AOT40 (Figure 3-2) shows significant increase of O_3 pollution in the past 40 years, and the trend accelerated rapidly since the early 1990s, possibly due to the rapid urbanization during that period in China (Liu et al., 2005b). The dataset shows seasonal variation of AOT40, with the first peak of O_3 concentration occurring in early summer and the second in September. Both peaks appear approximately at the critical time (the growth and harvest seasons) for crops in China. Thus, O_3 pollution may have significant impacts on crop production in China.

Although the AOT40 generally increased throughout the nation, the severity of O_3 pollution varied from region-to-region and from season-to-season (Figure 3-3). The central-eastern section of north China experienced severe O_3 pollution, especially in spring and summer. The greatest increase of AOT40 appeared in winter of north-west

China, probably due to the rapid industrialization and the transport of air pollution from Europe (Akimoto, 2003). In contrast, the change of AOT40 in south China is relatively low despite the large urban population and rapid industrial development in this region.
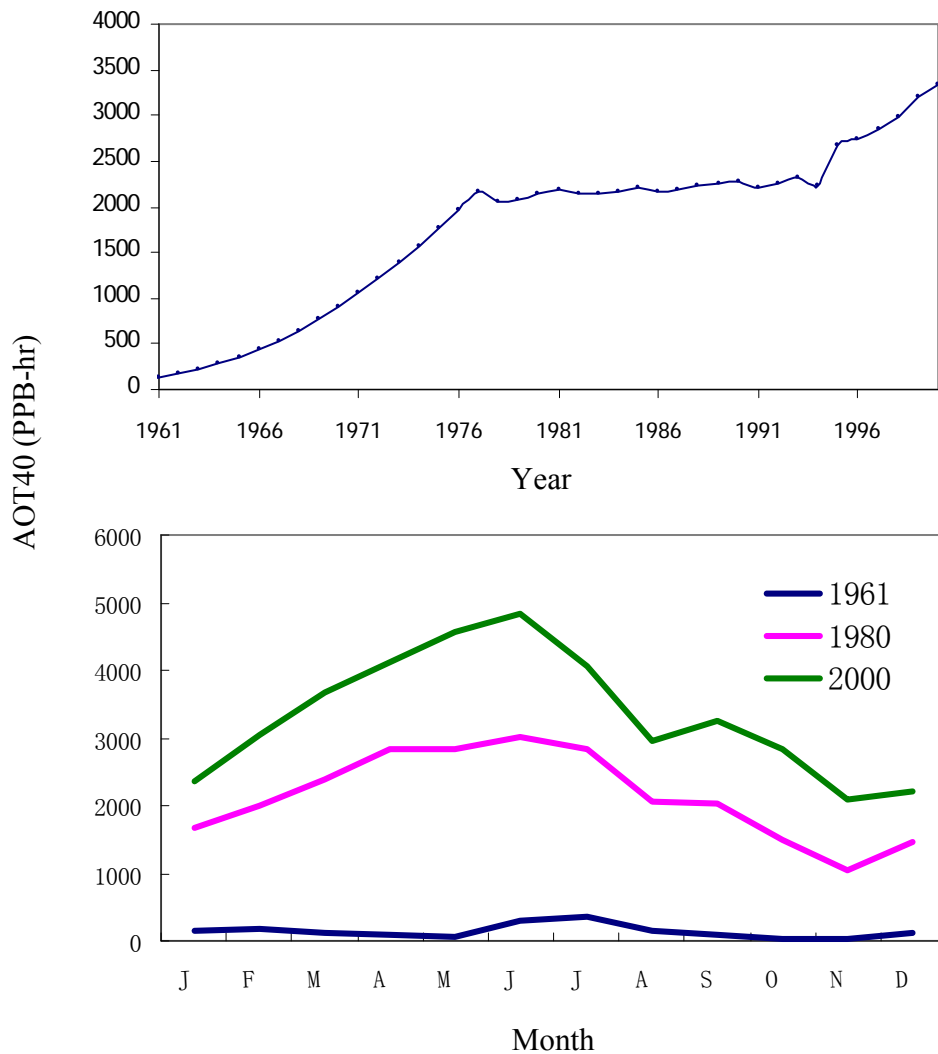
Figure 3-2 Annual monthly AOT40 (ppb-hr) mean from 1961 to 2000 (a) and Monthly AOT40 (ppb-hr) in 1961, 1980 and 2000 (b)

Note: From atmospheric chemistry model, MATCH (Multiscale Atmospheric Transport and Chemistry) (Lawrence et al., 1999; Mahowald et al., 1997; Rasch et al., 1997; and von Kuhlmann et al., 2003) and IGSM (Integrated Global Systems Model)(Wang et al., 1998; Wang and Prinn, 1999; Mayer et al., 2000a).
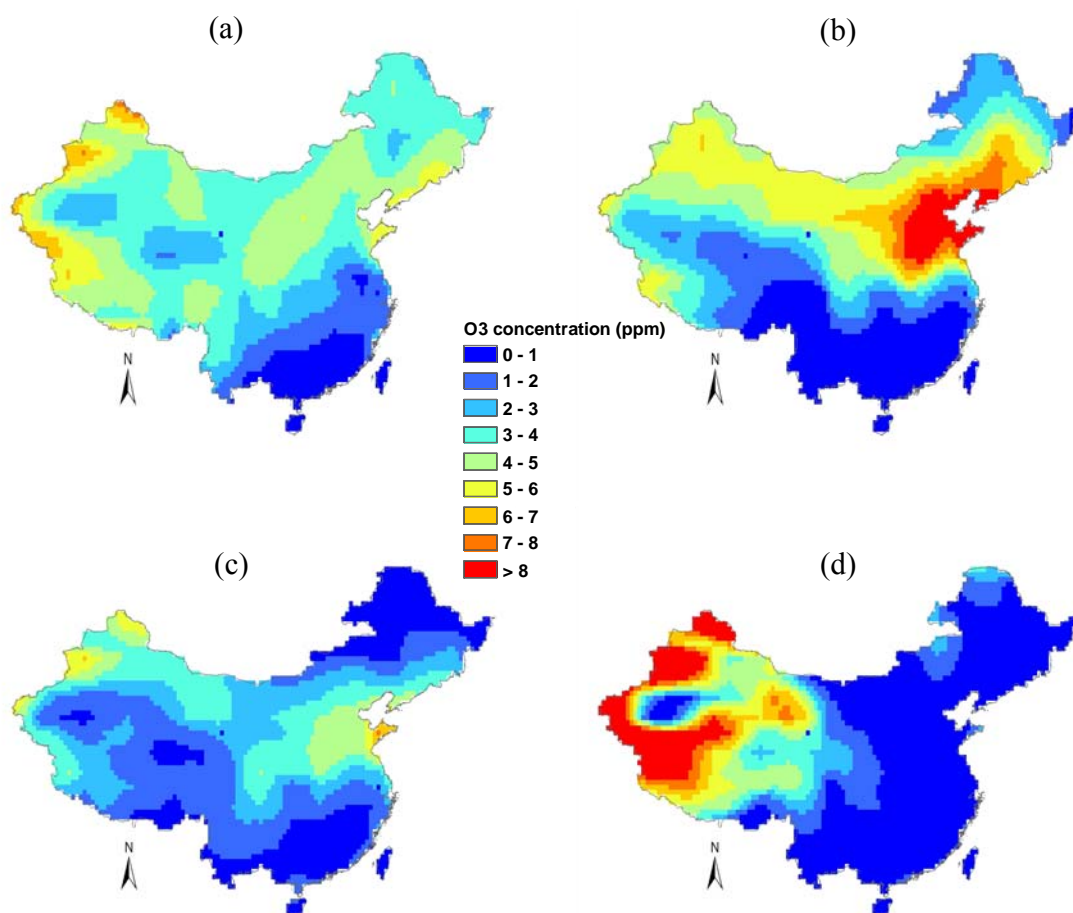
Figure 3-3 Average monthly AOT40 in spring(a) summer (b) autumn(c) and winter (d) from 1990 to 2000 in China (unit:1000 ppb-hr or ppm-hr)

Note: From atmospheric chemistry model, MATCH (Multiscale Atmospheric Transport and Chemistry) (Lawrence et al., 1999; Mahowald et al., 1997; Rasch et al., 1997; and von Kuhlmann et al., 2003) and IGSM (Integrated Global Systems Model)(Wang et al., 1998; Wang & Prinn, 1999; Mayer et al.,2000a).

Vegetation
- 1: Tundra
- 2: Boreal broadleaf deciduous forest
- 4: Boreal needleleaf deciduous forest
- 5: Temperate broadleaf deciduous forest
- 6: Temperate broadleaf evergreen forest
- 7: Temperate needleleaf evergreen forest
- 8: Temperate needleleaf deciduous forest
- 9: Tropical broadleaf deciduous forest
- 10: Tropical broadleaf evergreen forest
- 11: Deciduous shrub
- 12: Evergreen shrub
- 13: C3 grass
- 14: C4 grass
- 15: Dry farmland
- 16: Paddy land
- 17: Wetland
- 18: Bare land/Water
- 19: Built-up area
- No Data

500   0   500   1000   1500   2000   2500   3000   3500   4000 Kilometers

Figure 3-4 Contemporary plant functional types classified based on potential vegetation map and land use types in 2000 in China
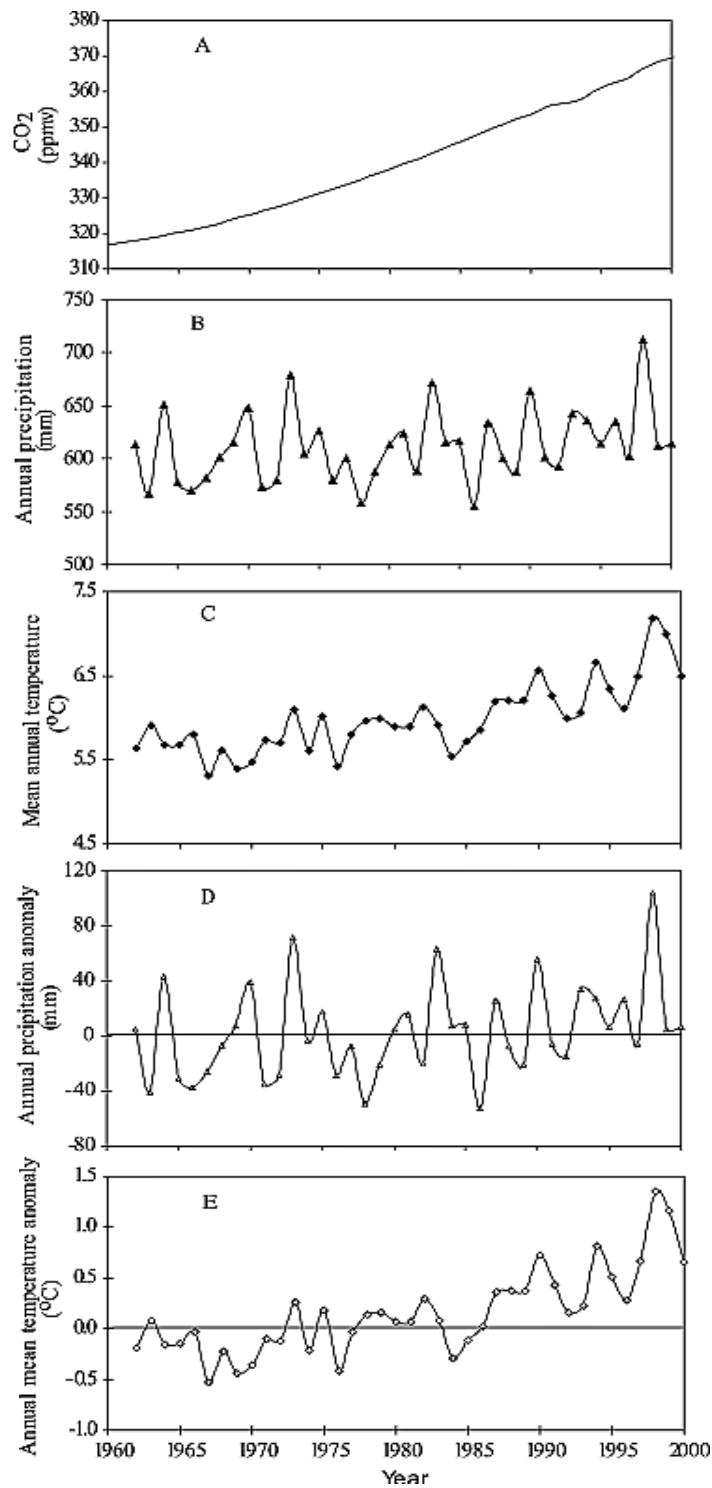
Figure 3-5 Variations in mean annual atmospheric CO_2 concentration (a); mean annual temperature (b); annual precipitation (c); annual precipitation anomalies (d) and annual temperature anomalies (e) (relative to 1961-1990 normal period) from 1961 to 2000
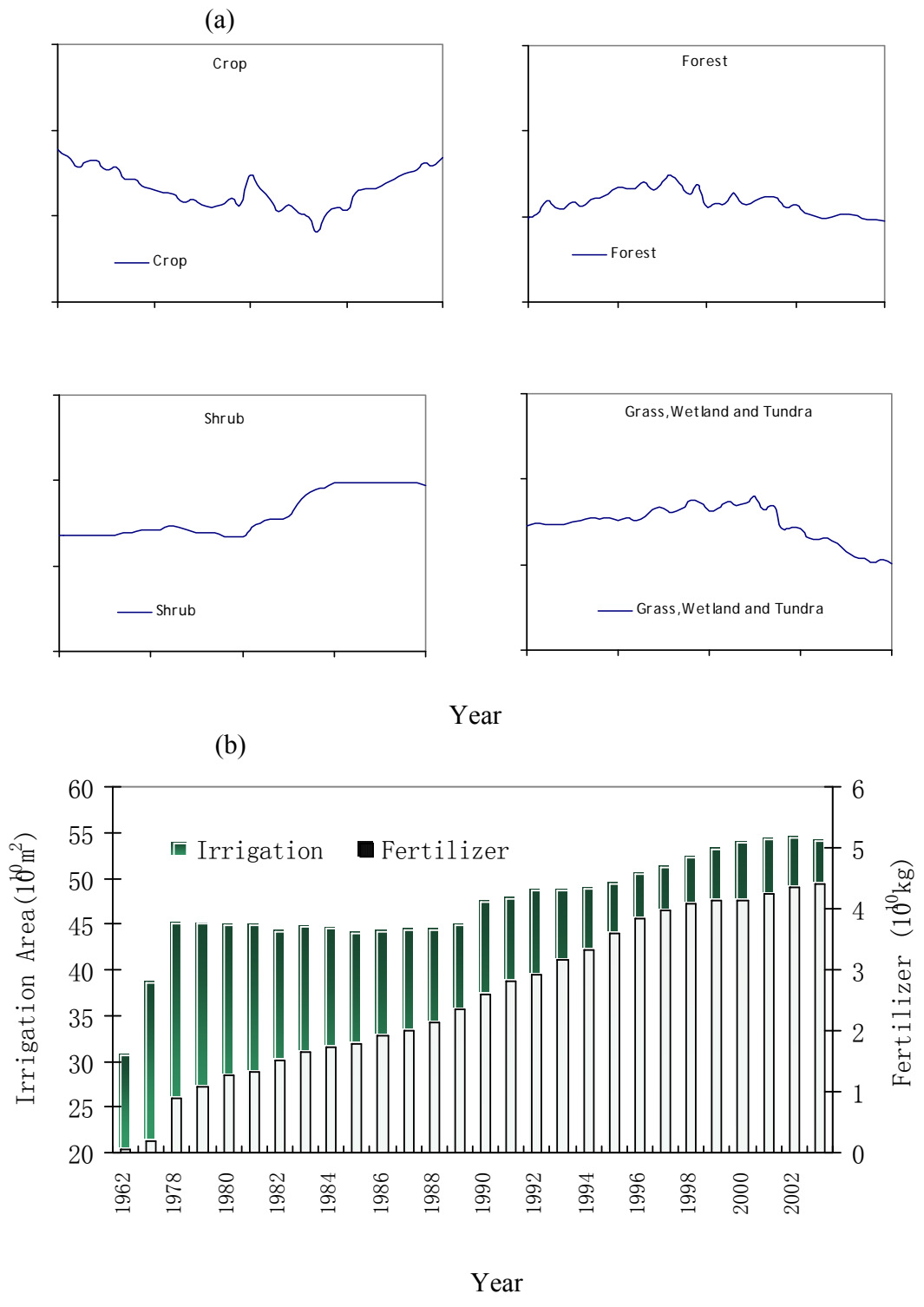
Figure 3-6 Variations of land use (a) and irrigation area ($10^{10}m^2$) and fertilizing amount (10^{10}kg) (b)

2.2.2. Description of Other input data

From 1961 to 2000, the CO_2 concentration steadily increased from 312 ppmv to 372 ppmv (Figure 3-5a), while temperature and precipitation fluctuated substantially (Figure 3-5b, c). Since the mid-1980s, China experienced an observable climate warming. The annual precipitation in the 1990s was higher than that in the 1980s. There was a relatively long dry period between 1965 and 1982, except for high annual precipitation in 1970 and 1974 (Figure 3-5c, e). Figure 3-6a shows that since the late 1980s, cropland expanded, while forestry and other land areas gradually decreased.

2.3. Simulation Design

In this study, six experiments were designed to analyze the effects of O_3 on NPP, NCE, and carbon storage in terrestrial ecosystems of China (table 3-1). Experiment I was used to examine the impact of transient O_3 on terrestrial ecosystem productivity while holding other environmental factors constant. Experiments II and III were used to analyze the combined effects of O_3 and CO_2 fertilization and of O_3 and climate change. Both experiments can help better determine the relative impacts of O_3, CO_2 and climate on the ecosystem. Experiments IV simulated the overall effect of climate change, atmospheric change, and land-use change. Experiment V was set up to study the overall combined effect without irrigation. The final experiment VI without O_3 effects is used for comparison against the other experiments.

The model simulation began with an equilibration run to develop the baseline C, N, and water pools for each grid. A spin-up of about 100 years was then applied if the

climate change was included in the simulation scenario. Finally, the model ran in transient mode driven by the daily or/and annual input data.

3. Results and analyses

3.1. Overall Change in Net Primary Productivity and Carbon Storage

In the simulation experiments, there were negative effects of O_3 on total average NPP and carbon storage during the study period (1961-2000). Average annual NPP and total C storage from the 1960s to 1990s in China increased by 0.66% and 0.06%, respectively, under the full factorial (climate, land use, CO_2, and O_3 were changed, hereafter referred to as OCLC), while they increased by 7.77% and 1.63%, respectively, under the scenario without O_3 (hereafter CLC) (Table 3-2). This difference indicates that under the full factorial, O_3 decreased NPP (about 1.64% in 1960s and 8.11% in 1990s) and total C storage (about 0.06% in 1960s and 1.61% in 1990s) in China's terrestrial ecosystems. Although NPP and total C storage in both scenarios increased over time, the soil and litter C storage decreased (-0.18% and -0.67%, respectively) under the full factorial, while they increased by 0.30% and 1.63%, respectively, under the scenario without O_3. Therefore O_3 reduced soil and liter C storage by about 0.03% and 0.16% in the 1960s and 0.52% and 1.84% in the 1990s, respectively, in China.

The model results show that NPP and carbon storage, including vegetation carbon, soil carbon, and litter carbon, decreased with O_3 exposure, and the reduced NPP was more than the decrease in carbon storage. The changing rates in the 1960s and 1990s indicate that increasing O_3 concentrations could result in less NPP and carbon storage,

Table 3-1 Experimental arrangement including CO_2, climate, land use and land management (fertilizer and irrigation)

	Scenarios	O_3	Climate	CO_2	Land use	Fertilizer	Irrigation
	Balance	0	Constant	Constant	Constant	0	0
I	O_3 only	Histroical	Constant	Constant	Constant	0	0
II	$O_3\_CO_2$	Histroical	Constant	Histroical	Constant	0	0
III	$O_3\_Climate$	Histroical	Histroical	Constant	Constant	0	0
IV	$O_3\_Climate\_Lucc\_CO_2$	Histroical	Histroical	Histroical	Histroical	Histroical	Histroical
V	$O_3\_Climate\_Lucc\_CO_2\_N$	Histroical	Histroical	Histroical	Histroical	Histroical	0
VI	$Climate\_Lucc\_CO_2$	0	Histroical	Histroical	Histroical	Histroical	Histroical

Note: Climate_lucc_CO_2: CLC O_3_climate_lucc_CO_2: OCLC

Table 3-2 Overall changes in carbon fluxes and pools during 1961-2000

Scenarios	$O_3\_Climate\_Lucc\_CO_2$			$Climate\_Lucc\_CO_2$			Difference with O_3 and without O_3	
	1960s (Pg)	1990s (Pg)	Net change (%)	1960s (Pg)	1990s (Pg)	Net change (%)	1960s (%)	1990s (%)
NPP	3.04	3.06	0.66	3.09	3.33	7.77	-1.64	-8.11
Veg C	26.02	26.32	1.15	26.03	27.38	5.19	-0.04	-3.87
Soil C	59.79	59.68	-0.18	59.81	59.99	0.30	-0.03	-0.52
Litter C	19.32	19.19	-0.67	19.35	19.55	1.03	-0.16	-1.84
Total C	105.14	105.2	0.06	105.2	106.92	1.63	-0.06	-1.61

which further implies that under the influence of O_3 alone, China's soil ecosystem would be a net C source, while without O_3, it would have been a net C sink (Table 3-2).

The results of the accumulated NCE across China under three simulation experiments, including O_3-only, combined effects without O_3 (CLC) and with O_3 (OCLC) from 1961-2000, indicate that O_3 effects could cause carbon release from the terrestrial ecosystem to the atmosphere (Table 3-3).   The accumulated NCE was -919.1 Tg C under the influence of O_3 only, 1,177.4 Tg C under the combined influence of changed climate, CO_2 and land use (CLC), and 677.3 Tg C under the full factorial (OCLC). These values imply that China was a CO_2 source when influenced only by O_3, but it was a sink under the influence of both CLC and OCLC scenarios. The accumulated NCE influenced by OCLC was 620.4 Tg C less (OCLC-CLC) than that influenced by CLC, implying that the interactions between O_3 and other factors (CO_2, climate, or land use change) were very strong.   These interactions decreased the emissions of CO_2 from terrestrial ecosystems. For the 1990s, a period with rapid atmospheric O_3 change, our simulated results show that the cumulative NCE under the full factorial (OCLC) throughout China decreased, compared to the results without O_3 influence (CLC) (Table 3-2 & Figure 3-7). In the central-eastern China and northeastern China, some places even released 150 g /m^2 more C into the atmosphere under O_3 influences during the 1990s.

(a)

(b)
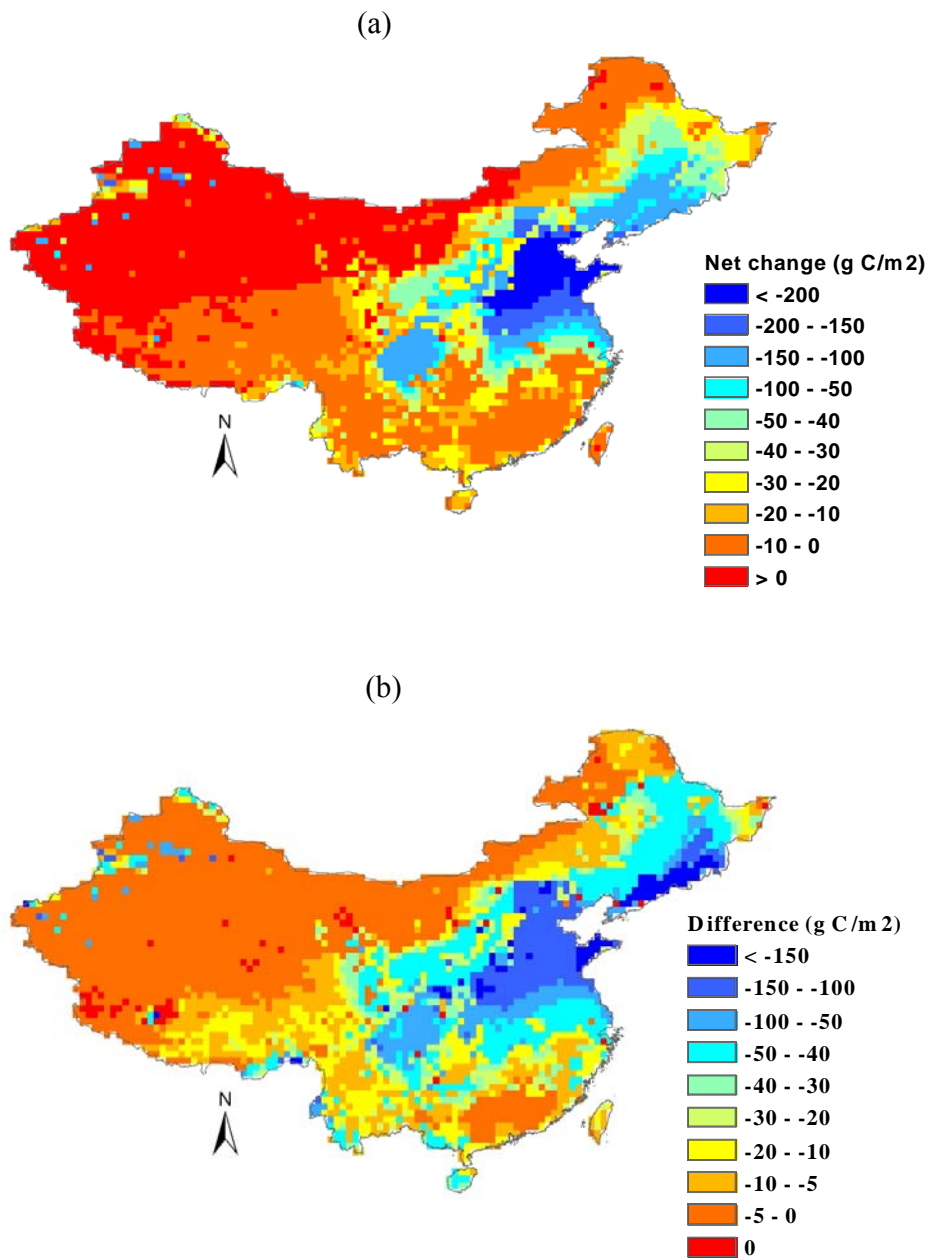
Figure 3-7 Difference in (a) cumulative net carbon exchange (NCE) and (b) net primary productivity (NPP) in 1990s between CLC with O_3 and without O_3 (g m^{-2})

Table 3-3 The accumulated NCE of different plant functional types under three scenarios including O$_3$ only, CLC, OCLC and the difference between CLC and OCLC from 1961 to 2000

Plant functional types	Scenarios			
	O$_3$ only	CLC	OCLC	OCLC-CLC
		Tg C(10^{12} C)		
1: Tundra	-30.6	-67.7	-130.2	-62.5
2: Boreal broadleaf deciduous forest	-1.2	-17	-18.3	-1.3
4: Boreal needleleaf deciduous forest	-9.9	94.9	85.1	-9.7
5: Temperate broadleaf deciduous forest	-615.9	-225.8	-547.0	-321.3
6: Temperate broadleaf evergreen forest	-26	231.8	206.5	-25.3
7: Temperate needleleaf evergreen forest	-72.1	720.4	720.4	43.2
8: Temperate needleleaf deciduous forest	-5.7	14.9	14.9	-4.0
10: Tropical broadleaf evergreen forest	-19.7	-48.1	-48.0	-19.8
11: Deciduous shrub	-4.9	17.9	17.9	-5.8
12: Evergreen shrub	-11.5	37.1	37.1	13.6
13: C$_3$ grass	-120.6	301.6	301.6	-136.9
14: C$_4$ grass	-0.9	24.4	3.8	-4.0
15: Wetland	-0.1	96.7	93.7	-30.1
16:Dry farmland		-3.6	-50.1	-46.5
17:Paddy farmland		-0.1	-10.1	-10.0
Total accumulated NCE since 1961	-919.1	1177.4	677.3	-620.4

Note: There is no dry farmland and paddy farmland under the scenario of O$_3$ only
Climate_lucc_CO$_2$: CLC; O$_3$_climate_lucc_CO$_2$: OCLC

Accumulated carbon storage of different PFTs under the three simulation experiments indicate that PFTs respond very differently to increasing O_3 concentration and its interaction with other environmental factors (Table 3-3). From 1961 to 2000, the accumulated NCE for different PFTs with O_3 exposure decreased by only 0.1 Tg C in wetlands up to 615.9 Tg C in temperate broadleaf deciduous forests.   Accumulated NCE under influences of CLC for different PFTs ranged from a decrease of 67.7 Tg C (Tundra) to an increase of 720.4 Tg C (temperate needleleaf evergreen forest), while accumulated NCE under influences of OCLC ranged from a decrease of 547.0 Tg C (temperate broadleaf deciduous forest) to an increase of 720.4 Tg C (temperate needleleaf evergreen forest). This range implies that temperate broadleaf deciduous forest was the biggest C source under the full factorial and that temperate needleleaf evergreen forest was the biggest C sink from 1961 to 2000. Compared with the combined effect with O_3 (OCLC) and without O_3 (CLC), the O_3-only scenario releases more C for all different PFTs. Compared with the combined effect without O_3 (CLC), the combined effect with O_3 (OCLC) resulted in less accumulated NCE for temperate broadleaf deciduous forest (321.2 Tg C) and dry farmland crops (46.5 Tg C) than other PFTs, which means that temperate broadleaf deciduous forest and dry farmland were more sensitive to O_3 than other PFTs. In general, we found that C_3 grass was more sensitive than C_4 grass to O_3, dry farmland was more sensitive than paddy farmland, and deciduous forest was more sensitive than needleleaf forest.

Overall results indicate that O_3 has negative impacts on terrestrial ecosystem production (Figure 3-2 and Table 3-2), and the negative effects become severe because

the O_3 concentration increased across China in the past decades, especially after the

1990s (e.g. Aunan et al., 2000). Biomes had complicated responses of carbon storage to

O_3 because of the different sensitivities of each PFT as well as different environmental

conditions. For example, some arid sites exhibit small O_3 effect on photosynthesis

because low stomatal conductance in arid plants leads to relatively low O_3 uptake. Some

other studies showed that O_3-induced reductions in photosynthesis were accompanied by

decreased water use efficiency (WUE), however, resulting in even larger reductions in

productivity, particularly at arid sites (Ollinger et al., 1997).   This fact may be the

reason that wetlands show relatively low reduction in carbon storage (Table 3-3), and dry

farmland crops are more sensitive to O_3 than paddy farmland crops. This might be also

because parameters of crops in the Reich model (Reich, 1987) are more sensitive to O_3

than those in deciduous and coniferous forests. In addition, in response to elevated O_3

concentration, plants allocate more carbon to leaves and stems than roots due to increased

defense mechanisms (e.g., Younglove et al., 1994; Piikki et al., 2004). This effect may

result in higher carbon storage loss in broadleaf deciduous forests than in evergreen

forests. It is clearly needed to address variations in biome-level responses to O_3 pollution.

3.2. Spatiotemporal Variations of C Flux and C Storage

Mean annual NPP changes from the 1960s to the 1990s under the O_3-only

scenario showed a significant spatial pattern (Figure 3-9). The mean annual NPP

decreased the most in the eastern China partially because the eastern China has

experienced faster development in urbanization, industrialization, and agricultural

intensification in the past several decades, than remote areas in the western China (Aunan,

2000; Wang et al., 2004; Liu et al., 2005). This development is closely related to an

increased use of fossil fuels and fertilizer. The imbalance of regional O_3 concentrations could also result in fluctuations of annual NPP.
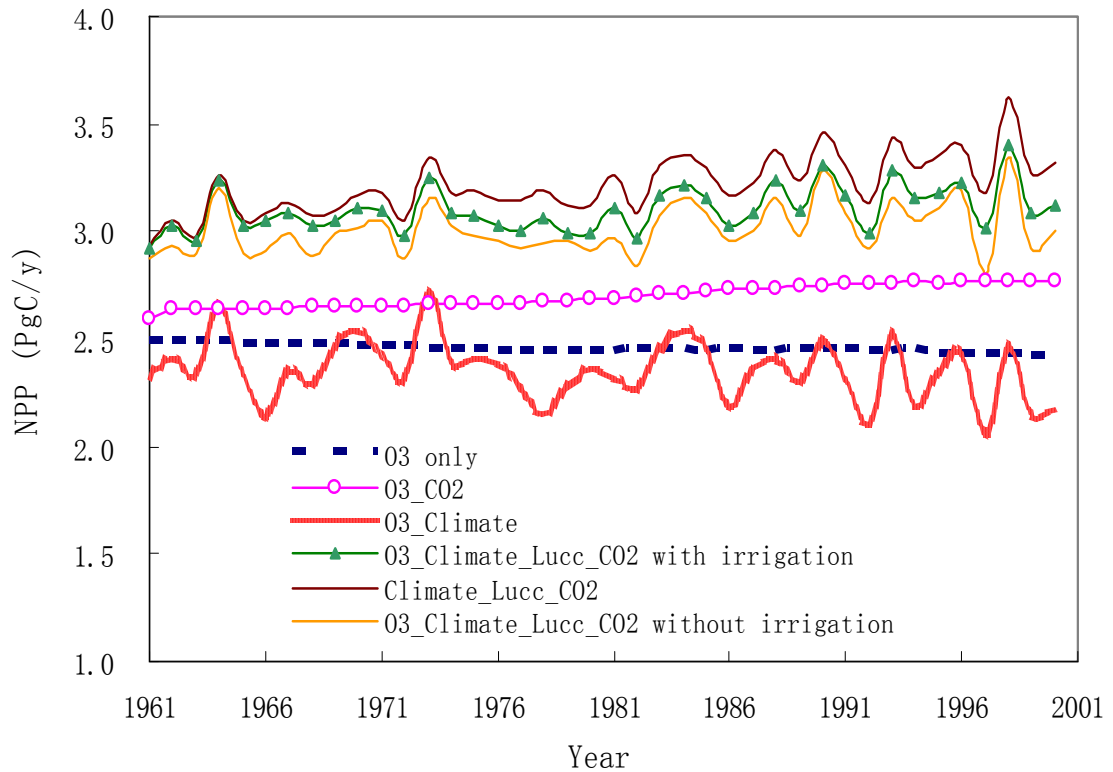
Figure 3-8 Annual net primary productivity (NPP) under 6 influencing factors from 1961 to 2000

Under the influence of O_3 and CO_2, the simulation results illustrate that mean annual NPP in the 1990s increased by 140.6 Tg C compared to the 1960s (Figure 3-8 and Figure 3-9); the total carbon storage increased by 46.3Tg over the past 40 years because of the accumulative increase in vegetation and soil C storage (Figure 3-10). The increased C storage may be attributed to the direct effects of increasing atmospheric CO_2 (Melillo et al., 1996; Tian et al., 1999, 2000), however, O_3 can partially compensate for the positive effects of CO_2 fertilization.

DLEM estimates that the total carbon storage for potential vegetation under O_3 and climate influences decreased by 15.9 Tg C. This decrease could mainly be attributed to the large decrease in soil C storage while vegetation C decreased relatively little from 1961 to 2000 (Figure 10). The interannual variation of NPP had a similar trend with the historical annual precipitation from 1961 to 2000 (Figure 3-8 and figure 3-5c).   For example, annual NPP decreased as less precipitation occurred in the late 1960s and the 1970s (Figure 3-8). From 1961 to 2000, due to the influence of the monsoon climate, precipitation and temperature in China exhibited large interannual variability during the study period. The results indicate that NPP was more sensitive to changes in precipitation than in temperature, while soil carbon storage was more closely linked to temperature through decomposition responses. In addition, the changing pattern of mean annual NPP from the 1960s to 1990s indicates that NPP in the areas of the eastern and northern China decreased more than other areas over the same time period (Figure 3-9b). Those variations might be related to the magnitude and spatial distribution of rainfall from seasonal to decadal (Fu and Wen, 1999; Tian et al., 2003). Therefore, the combined effects of changes in air temperature and precipitation with increasing O_3 concentration are complex, and the NPP loss might result from the balance among O_3, CO_2 and water uptake through changing stomatal conductance due to the combined effects of O_3, temperature and CO_2.
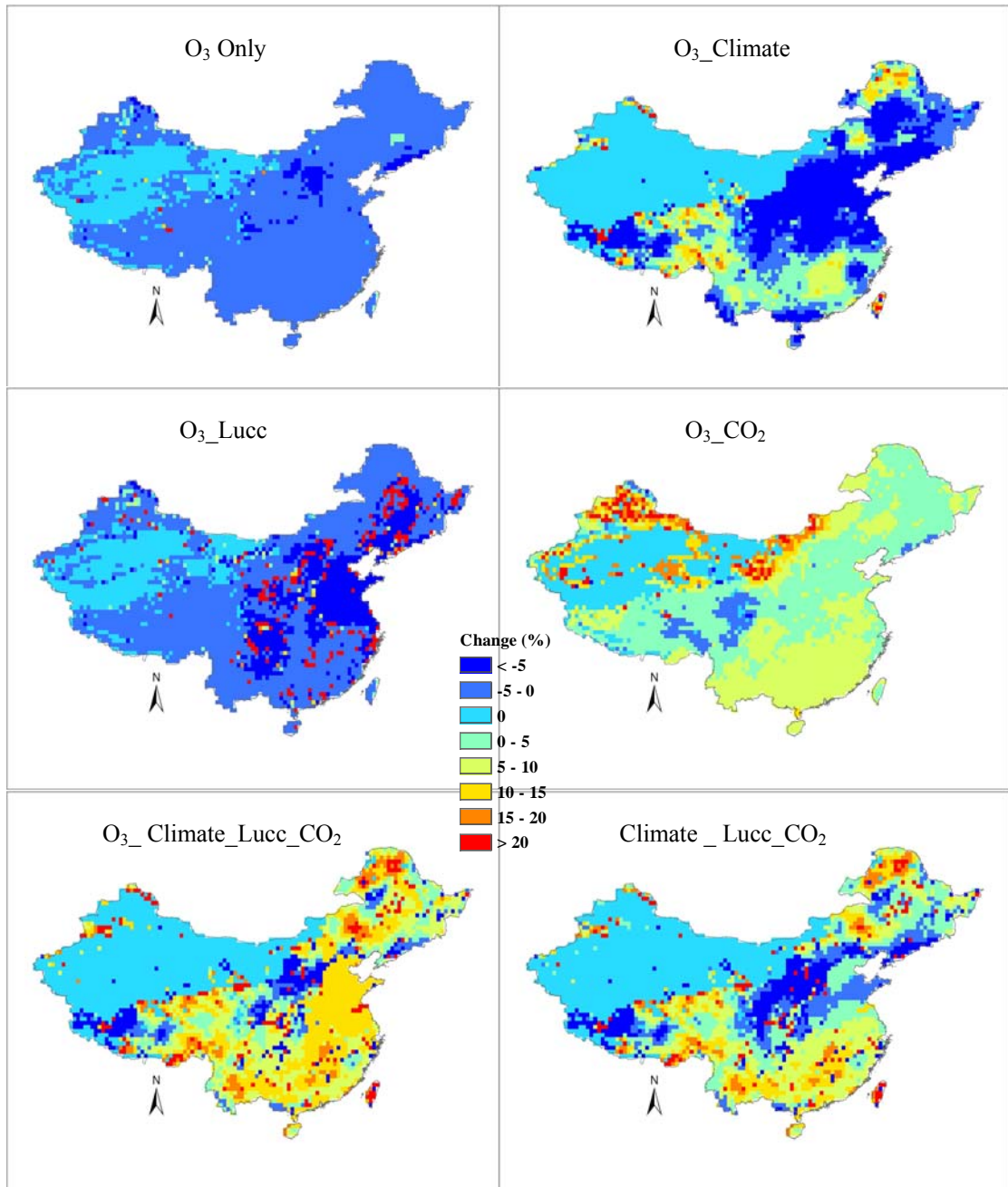
Figure 3-9 Change rate of average annual net primary productivity (NPP) from 1960s to 1990s under different influencing factors related to O$_3$ (%)

The above analyses address the response of potential terrestrial ecosystems to historical O_3 concentration, changing climate, and atmospheric CO_2 concentration. Land-use change as well as management, however, has substantially modified land ecosystems across China in the past 40 years (e.g. Liu et al., 2005). Based on the DLEM simulation, the total terrestrial carbon storage in China increased by 16.8 Tg C during 1960-2000 (Figure 3-10). Annual NPP over time in the OCLC (simulation experiment IV) increased slightly (mean 0.1%), and the variation of interannual NPP is similar to the result from climate change (Figure 3-8). The distribution of the annual NPP difference between the 1960s and the 1990s in OCLC scenario indicates that the NPP changes were smaller than in the O_3-only scenario, which could be caused by the modification of the interactive effects of changing climate, increasing CO_2, and land-use change. Compared with the effects of OCLC with irrigation and without irrigation (Figure 3-8), there was a mean 3% reduced rate of annual NPP over the time period from 1961 to 2000. The result implies that water conditions could alter the O_3-induced damage.

4. Discussion

4.1 Comparison with Estimates of Ozone Damage from Other Research in China

Research of O_3 effects on crop growth and yield loss in China has been conducted since 1990 by using field experiments and model simulations. Field studies have shown that O_3 exposure could result in crop yield reductions of 0%-86% under different experimental treatments with varying O_3 concentrations and duration (e.g. Wang and Guo, 1990; Huang et al.,2004). Estimates of crop yield loss on the regional scale, such as the Yangtze River Delta and national level, were established (Feng et al.,2003; Wang and

Mauzerall, 2004; Aunan et al., 2000) , and results indicate crop loss of 0-23% from historical O_3 in China at with potential future increases ranging from 2.3%-64% (Wang et al.,2007). Felzer et al. (2005) found that crops were more sensitive to O_3 damage at low O_3 levels both in China and Europe than in the U.S. All the above crop studies and assessments of yield loss were based on empirical exposure-response relationships for crop yield and O_3. However, there are few studies based on process-based ecosystem models to estimate the effects of O_3 damage on diverse PFTs on a national level. In our study, the DLEM model was used to address the influence of O_3 concentration on NPP and carbon storage across China during the past forty years from 1960-2000, and we estimated the influence of historical O_3 on different PFTs. Similar to previous studies, there was a reduction of NPP when considering O_3 effects. Also the O_3 effects on terrestrial ecosystem across China was consistent with the study of Felzer et al. (2005), that spatial variations were largely due to varied O_3 concentration, climate change and other stress factors. The large damage peaked in the eastern China with the greatest reduction in NPP over 70% for some places. Furthermore, our work shows different sensitivities for different PFTs, in part because we incorporated the Reich et al. (1987) dose-response functions into DLEM. Deciduous forests and dry farmland crops are relatively sensitive to O_3 than other PFTs. Dry farmland showed more reduction in yield than paddy farmland. It indicates an important need to further study variations among PFTs and the underlying mechanisms.

4.2 Ozone and its interactive effects on net primary productivity and carbon storage

In our study, average annual NPP decreased 0.01PgC/yr and accumulated NCE was -0.92PgC (Figure 3-8 and Table 3-3) from 1961 to 2000 with the effect of O_3 only.

Similar to most field experiments in US, Europe and China and other model results (e.g. Heagle, 1989 and references therein), our results show that O_3 has negative effects on terrestrial ecosystem production due to direct O_3-induced reductions in photosynthesis .
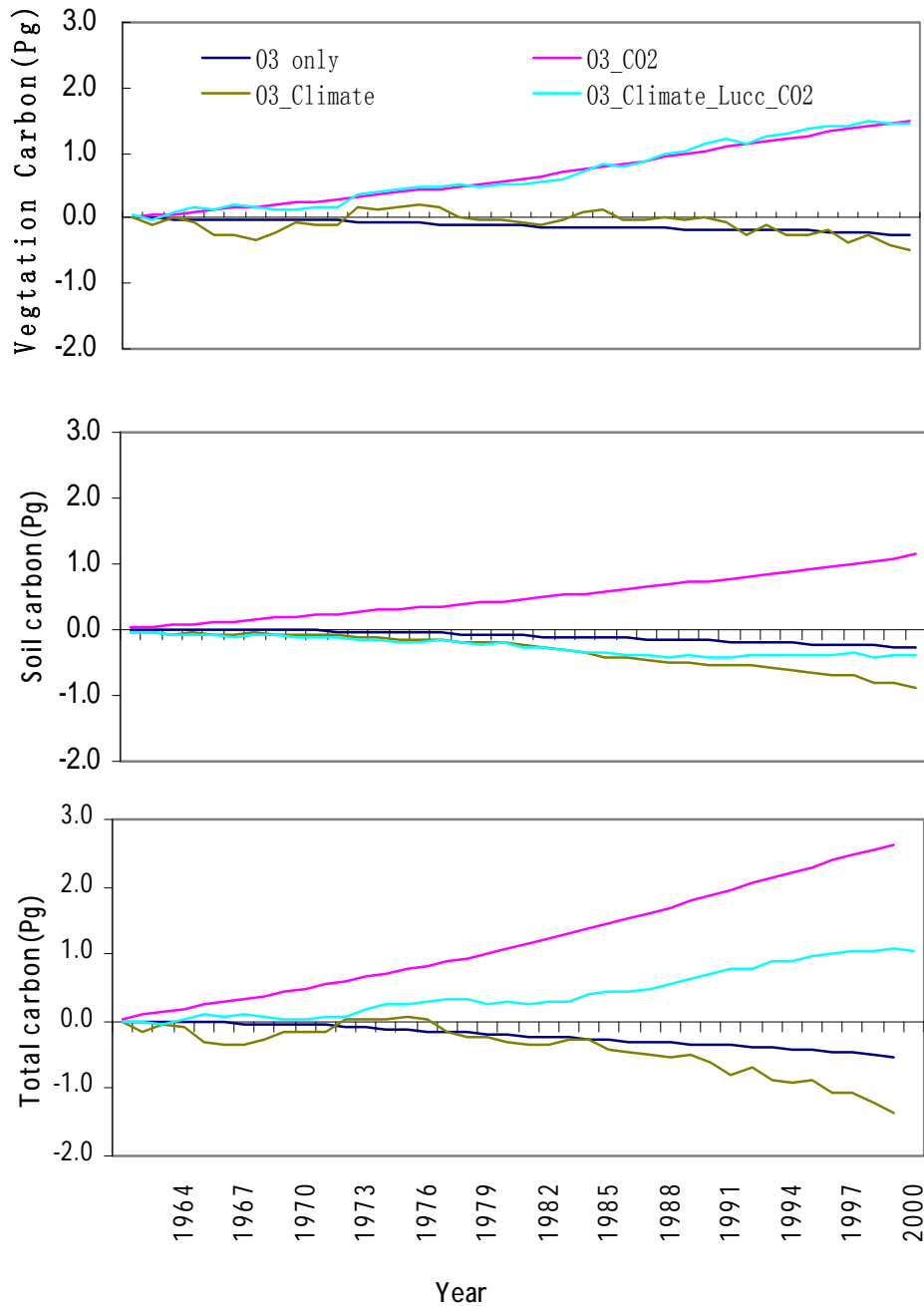
Figure 3-10. Annual changes in terrestrial carbon storage in China from 1961 to 2000 under four influencing factors

The combined effects of O_3, CO_2, climate, and land use show very different

results. O_3 may compensate for CO_2 fertilization and result in NPP losses in different

plant types over time across China (Figure 3-8, 3-9 and 3-10). Climate variability

increased the O_3-induced reduction of carbon storage and led to substantial year-to-year

variations in carbon fluxes (NPP and NCE). These results are consistent with many

previous studies (e.g. Cao and Woodward, 1998; McGuire et al., 2001; Tian et al., 1999;

2003). There is a direct positive effect of elevated CO_2 on photosynthesis and biomass

production (e.g. Agrawal and Deepak, 2003), as well as reduced stomatal conductance.

Climate warming increased decomposition, resulting in continuous loss of soil and total

carbon storage (Figure 3-10b, c) since 1990, although annual precipitation was

substantial during this period (Figure 3-5c,d). China is influenced by a monsoon climate

so that summer monsoons bring most of the annual precipitation (Fu and Wen, 1999).

The combined effects of changes in O_3 concentration and climate warming in arid areas

may result in larger variability in productivity.

Similarly, the contribution of land-use change to the terrestrial carbon budget

varied over time and in different ecosystem types (e.g. Houghton and Hackler, 2003). In

our study, we took into account the combined effects of land-use change with historical

O_3 concentration, atmospheric CO_2 concentration, and climate variability. When

considering the effects of land use with O_3, three aspects need to be addressed. First,

transformations of different land use types, such as the conversion from forest to crop and

regrowth of natural vegetation after cropland abandonment, could result in carbon loss or

carbon uptake (e.g. Tian et al., 2003). The other two are the sensitivities of different

biomes to O_3 exposure and agriculture management. The former results in different carbon loss rates; however, the latter's effects combined with O_3 pollution on carbon storage are related to changing soil environment such as water and nitrogen conditions. In addition, dry farmland and C_3 grass are more sensitive than paddy farmland and C_4 plant types, which could better explain the field study results of the relationship between O_3 effects and photosynthesis and stomatal conductance. Reich's (1987) study indicated that a secondary response to O_3 is possibly a reduction in stomatal conductance, as the stomata close in response to increased internal CO_2. Tjoekler et al., (1995) found a decoupling of photosynthesis from stomatal conductance as a result of long-term exposure to O_3. Such a decoupling implies that O_3-induced reductions in photosynthesis would also be accompanied by decreased water use efficiency (WUE), resulting in even larger reductions in productivity, particularly at arid sites, although many studies indicate that drought-induced stress could reduce O_3 stress (Smith et al, 2003). Unlike C_4 photosynthesis, which adds a set of carbon-fixation reactions that enable some plants to increase photosynthetic water use efficiency in dry environments, C_3 grass and dry farmland in China are always water limited and are more sensitive to O_3 exposure. In addition, the modeling studies of Felzer et al. (2004) indicated that O_3 pollution can reduce more NPP with fertilizer application. In contrast to land-use change in the eastern U.S. (Felzer et al., 2004), it is necessary to consider how to manage irrigation in arid areas because there are a lot of moisture-limited regions that require irrigation in China. Reasonable irrigation management can both enhance the water use efficiency and modify the O_3 damage.

Besides the environmental factors discussed above, recent reviews of the global carbon budget also indicate that terrestrial ecosystem productivity could be affected by other changes in atmospheric chemistry, such as nitrogen deposition and aerosols (e.g. Pitcairn et al., 1998; Birgin et al., 2001; Lu and Tian, 2007). These changes can directly change carbon storage. For example, aerosols and regional haze may also reduce ecosystem productivity by decreasing solar radiation and changing climate conditions (e.g. Huang et al., 2006). Nitrogen deposition could also affect terrestrial carbon storage in a complex way (Aber et al., 1993). For example, many terrestrial ecosystems in middle and high latitudes are nitrogen limited (e.g. Melillo et al, 1995), and the increasing effect of elevated CO_2 on photosynthesis could be decoupled by limited nitrogen concentration. However, increasing nitrogen deposition could reduce total plant phosphorus uptake (e.g. Cleland, 2005) and nitrogen deposition can bias the estimates of carbon flux and carbon storage either high or low depending on the nature of the interannual climate variations (Tian et al., 1999). In this study, we ran the model with a closed nitrogen cycle because a database containing a time series of nitrogen deposition was not available for our transient analyses. To completely understand the effects of air pollution conditions on terrestrial ecosystem productivity, future work should take these atmospheric chemistry factors into account.

4.3 Uncertainty and future work

An integrated assessment of O_3 impacts on terrestrial ecosystem production at regional level basically requires the following types of information: (1) O_3 dataset that reflects the air quality in the study area and other environmental data such as climate (temperature, light and precipitation), plant (types, distribution and parameters) and soil

(texture, moisture, etc.) information; (2) mechanisms of O_3 impacts on ecosystem processes, which describe relationships between air-pollutant-dose and eco-physiological processes such as photosynthesis, respiration, allocation, toleration, and competition; and (3) an integrated process-based model, which is able to quantify the damage of O_3 on ecosystem processes. To reduce uncertainty in our current work, future work needs to address the following: First, we used one set of O_3 data from a combination of global atmospheric chemistry models, which have not been validated well against field observations because of limited field data. We found that the seasonal pattern of our simulated O_3 data (AOT40) was the same as the limited observation data sets (Wang et al., 2007), with the highest O_3 concentrations in summer. However, we still need observed AOT40 values to calibrate and validate our O_3 data set in the future. Second, the O_3 module in our DLEM focuses on the direct effects on photosynthesis and indirect effects on other processes, such as stomatal conductance, carbon allocation, and plant growth. The quantitative relationship between O_3 and these processes remains untested by field studies. Third, in our simulations, we simulate land-use change accompanying optimum fertilization and irrigation management. It is hard to separate the contributions of land-use change from their combination with fertilizer and irrigation. Especially, according to the studies of Felzer et al. (2004, 2005), the O_3 effect with fertilizer management could increase the damage to ecosystem production. However, irrigation management in dry lands could reduce the negative effect of O_3 (Ollinger et al., 1997). In our model, crops were classified as dry farmland and paddy farmland which improved the crops' simulation compared to previous process-based models, but different crop types, such as spring wheat, winter wheat, corn, rice, and soybean, have large differences in

sensitivity to O_3 and could result in different ecosystem production changes due to the effects of O_3 (eg. Heck, 1988; Wang et al., 2007). It is needed to improve the agricultural ecosystem module by including different crop types and varied managements (fertilizer, irrigation, tillage, and so on) to study the O_3 effect.

To accurately assess impacts of O_3 and other pollutants on NPP and carbon storage on a regional scale, it is needed to improve both observation data in China and ecosystem model. So the future research may focus on the following: (1) validation of simulation results with site data; (2) refinement of present O_3 data with field observations and comparison of present O_3 data with other simulated O_3 data; (3) development of the O_3 module by coupling the effects of O_3 on LAI and stomatal conductance; (4) improvement of the process-based agricultural ecosystem model to study the effects of air pollutants on different crop types; and (5) inclusion of other air pollutants (e.g., aerosols and nitrogen deposition) in the model.

5. Conclusions

This work investigated tropospheric O_3 pollution in China and its influence on NPP and carbon storage across China from 1961 to 2000 by using the Dynamic Land Ecosystem Model (DLEM). Our simulated results show that elevated tropospheric O_3 concentration has led to a mean 4.5% reduction in NPP nationwide during the period of 1961 –2000. Our simulations suggest that the interactions of O_3 with increasing CO_2, climate change, land use and management are significant, and that the interaction of O_3 with the climate and land use change can cause terrestrial ecosystems to release carbon to the atmosphere. O_3 effects on NPP varied among plant functional types, ranging from

0.2% to 6.9% during 1961 to 2000. Dry farmland, C_3 grasses, and deciduous forests are more sensitive to O_3 exposure than paddy farmland, C_4 grasses, and evergreen forests. In addition, following O_3 exposure experiments, we allowed crops to be more sensitive than deciduous trees and deciduous trees to be more sensitive than coniferous trees, and therefore had the highest mean reduction (over 14.5%) since the late 1980s. Spatial variations in O_3 pollution and O_3 effects on NPP and carbon storage indicate that eastern-central China is the most sensitive area. Significant reduction in NPP occurred in north-east, central, and south-east China where most crops are planted. Direct and indirect effects of air pollutants on ecosystem production, especially on agriculture ecosystem carbon cycling, should be considered in the future work in China. Lack of O_3 observation data sets and sensitivity experiments of different PFT_S could result in uncertainties in this study. To accurately assess the impact of elevated O_3 level on NPP and carbon storage, it is necessary to develop an observation network across China to measure tropospheric O_3 concentration and its effects on ecosystem processes, and then to enhance the capacity of process-based ecosystem models by rigorous field data-model comparison.

Chapter 4

Grassland Ecosystem, Ozone Pollution, Climate Variability/Change – Influence of Ozone Pollution and Climate Variability on Grassland Ecosystem Productivity across China

Abstract

Our simulations with the Dynamic Land Ecosystem Model (DLEM) indicate that the combined effect of ozone (O_3), climate, carbon dioxide and land use have caused China's grasslands to act as a weak carbon sink during 1961-2000. This combined effect on national grassland net primary productivity (NPP) and carbon storage was small, but changes in annual NPP and total carbon storage across China's grasslands show substantial spatial variation, with the maximum total carbon uptake reduction of more than 400 g/m^2 in some places of northeastern China. The grasslands in the central northeastern China were more sensitive and vulnerable to elevated O_3 pollution than other regions. The combined effect excluding O_3 could potentially lead to an increase of 14 Tg C in annual NPP and 0.11Pg C in total carbon storage for the same time period. This implies that improvement in air quality could significantly increase productivity and carbon storage in China's grassland ecosystems.

1. Introduction

Increasing air pollution by tropospheric O_3 is occurring globally. It has been documented by many researchers that elevated O_3 can reduce vegetation productivity (Heagle, 1989; Mauzerall and Wang, 2001; Ashmore, 2005). Our understanding of potential adverse effects of O_3 on semi-natural vegetation such as grasslands is relatively limited compared with many studies on growth and yield of cropland and forested ecosystems (e.g. Ollinger et al., 1997; Barnes and Wellburn, 1998; Felzer et al., 2004). The few available studies regarding O_3 impacts on natural grasslands, however, have indicated that O_3 can induce visible injury and detrimental effects on growth, reproductive development, and competition among different grass species (Farage et al, 1991; Davison & Barnes, 1998; Fuhrer and Booker, 2003; Bassin et al., 2006). Grassland ecosystems may be more vulnerable than agricultural and forested ecosystems to O_3 due to their distribution in more extreme climate zones, and the absence of intensive human management which occurs in agriculture, and long-term adaptation to environmental stresses in forests (Fuhrer and Booker, 2003). Most grassland regions are noted by substantial climatic variability and high frequency of drought events. Therefore, from the perspective of environmental policy and management, it is imperative to explore how net primary production and carbon storage of grassland ecosystems have been influenced by elevated tropospheric O_3 concentrations, and its combined effects with other factors of climate change, such as temperature, increases in CO_2 concentrations and alterations in rainfall patterns.

China's grasslands account for about 40% of the total land area in the country. As one of the major terrestrial ecosystems, grasslands play an important role in the carbon cycle in China. Most of China's grassland ecosystems are distributed in the arid and semi-arid areas of North China (Yang et al., 2002; Liu et al. 2005; Jin et al., 2005) where O_3 pollution has been documented (Aunan et al., 2000; Akimoto, 2003; Wang and Mauzerall, 2004; Felzer et al., 2005). Thus, grassland ecosystems in China are experiencing multiple stresses, including O_3 pollution and drought. Although research on O_3 pollution effects (e.g. Aunan et al., 2000) and the carbon cycle in grassland ecosystems (Xiao et al., 1995) have been carried out by either field experiments or model simulations, few studies have been conducted to assess the combined effects of elevated O_3 and climate variability on grassland ecosystems at the regional level. To address the complexity of O_3 effects on grassland ecosystem productivity at the national scale, we need to use spatially-explicit process-based ecosystem models with an O_3 sub-model to analyze the history and forecast the future of grassland productivity.

Based on many field experiments and observations, several process-based models have been developed to study O_3 effects on vegetation productivity by extrapolating its effects on individual plants to a plant community, an ecosystem and even a region (eg. Reich, 1987; Ollinger et al., 1997, 2002; Martin et al., 2001; Felzer et al., 2004). O_3 can affect ecosystem productivity through influencing leaf photosynthesis, respiration, stomatal conductance, carbon allocation, litter decomposition, water cycling and community properties such as species diversity, functional types and dominant vegetation

types (Neufeld et al., 1992, 2006; Chappelka et al., 2002, 2003; Fuhrer and Booker, 2003; Matyssek and Sandermann, 2003; Ashmore, 2005).

To assess the effects of O_3 on vegetation productivity, many process-based models simplify the influence mechanisms and focus on the fact that elevated O_3 exposure reduces CO_2 assimilation by either direct or indirect effects on photosynthesis and stomatal conductance (Pell et al., 1997; Torsethaugen et al., 1999; Fiscus et al., 2005). We followed these ideas and integrated a sub-model of the O_3 effect into the DLEM model, a highly integrated process-based model (Tian et al., 2005).

Our objectives in this study are: 1) to illustrate the effects of tropospheric O_3 pollution in combination with climate variability on productivity of China's grassland ecosystems from 1961 - 2000; 2) to distinguish the contributions of the main driving environmental factors; 3) to examine the temporal-spatial patterns of carbon pools and fluxes in China's grassland ecosystems from 1961 to 2000; and finally 4) to identify the uncertainties of present simulations and point out future directions and improvements for simulating O_3 effects.

2. Materials and methods

2.1. The Dynamic Land Ecosystem Model (DLEM) and input data

The same method as the description in detail in chapter 3 was used in this study.

2.2. Experimental design

In our study, we designed five simulation experiments to analyze the effects of O_3 only or climate only, and the combined effects of O_3 and climate on NPP, NCE and carbon storage in the grassland ecosystems of China (Table 4-1). In experiment I, we tried to examine the sole extent of O_3 impacts while other environmental factors were constant. In experiments II and III, we analyzed the contribution of climate variability only and the combined effect with O_3, respectively. The other two simulation experiments IV and V were designed to simulate a relatively realistic scenario to explore the effects of O_3 on ecosystem production. Here the important environmental factors, including climate variability, increasing CO_2, and land use change were considered.

Table 4-1 Experimental arrangement including O_3, climate, CO_2 and land use

Scenarios		Environmental Factors		
		O_3	Climate	CO_2 & Land use
I	Only O_3 (O)	H	C	C
II	Only Climate (Clm)	0	H	C
III	O_3_Climate (OClm)	H	H	C
IV	Climate_Lucc_CO_2 (ClmLC)	0	H	H
V	O_3_Climate_Lucc_CO_2 OClmLC)	H	H	H

Note: H is historical data, 0 means no data and C is constant data. Here we use CO_2 concentration (296ppm) in 1900 and mean climate data sets in 30 years from 1960 to 1990 and potential vegetation map as constant value.

The model simulation began with an equilibrium run to develop the baseline C, N, and water pools for each grid. Then a spin up of about 100 years was applied if climate variability is included in the simulation scenario. Finally, the model ran in transient model driven by transient data of climate, O_3, CO_2, and land use.
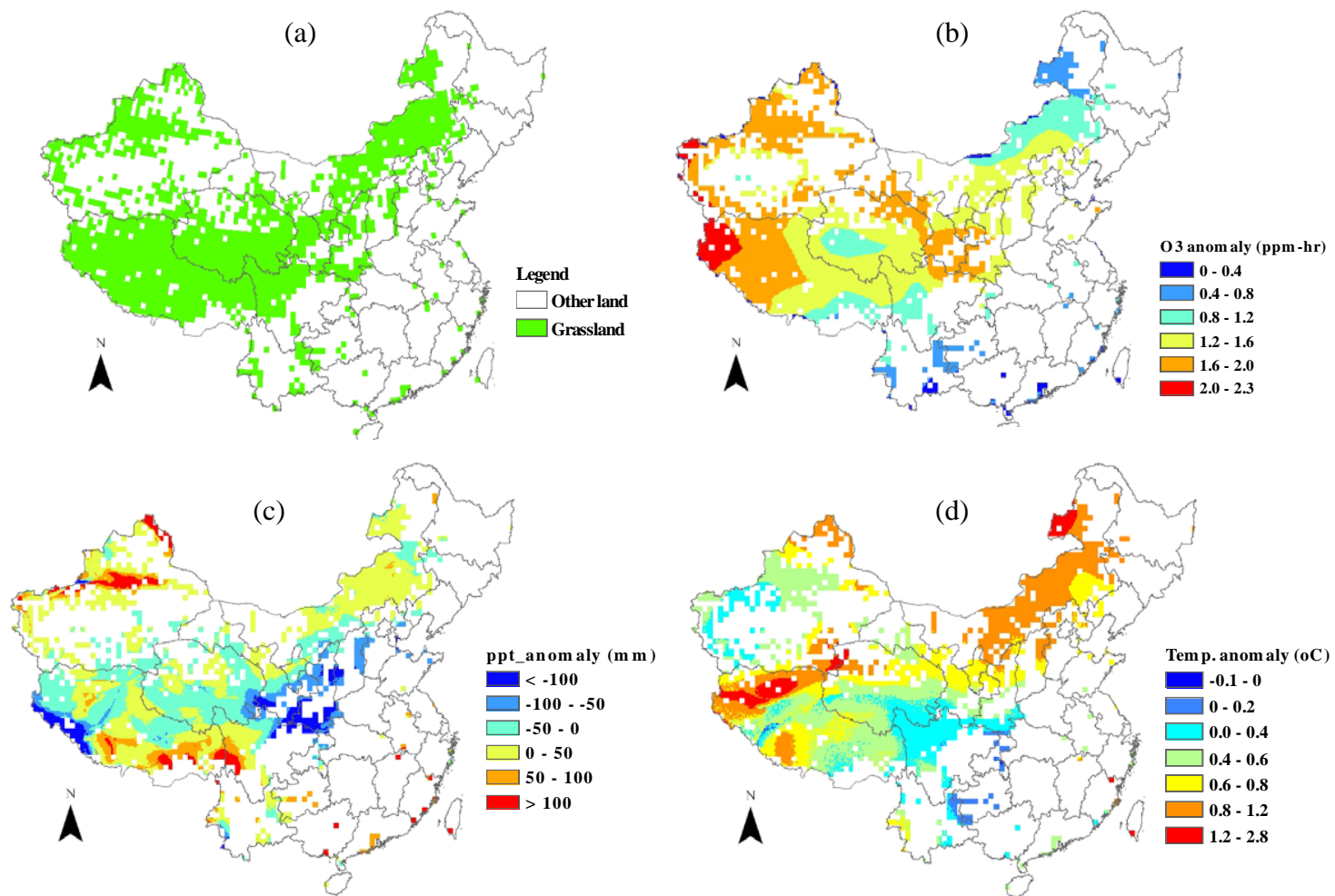
Figure 4-1 Map of grassland distribution in China (a) and maps of anomalies in the 1990s (relative to the average for 1961-1990) for (b) annual average AOT40 (ppm-hr), (c) precipitation (mm), and (d) temperature (ºC).
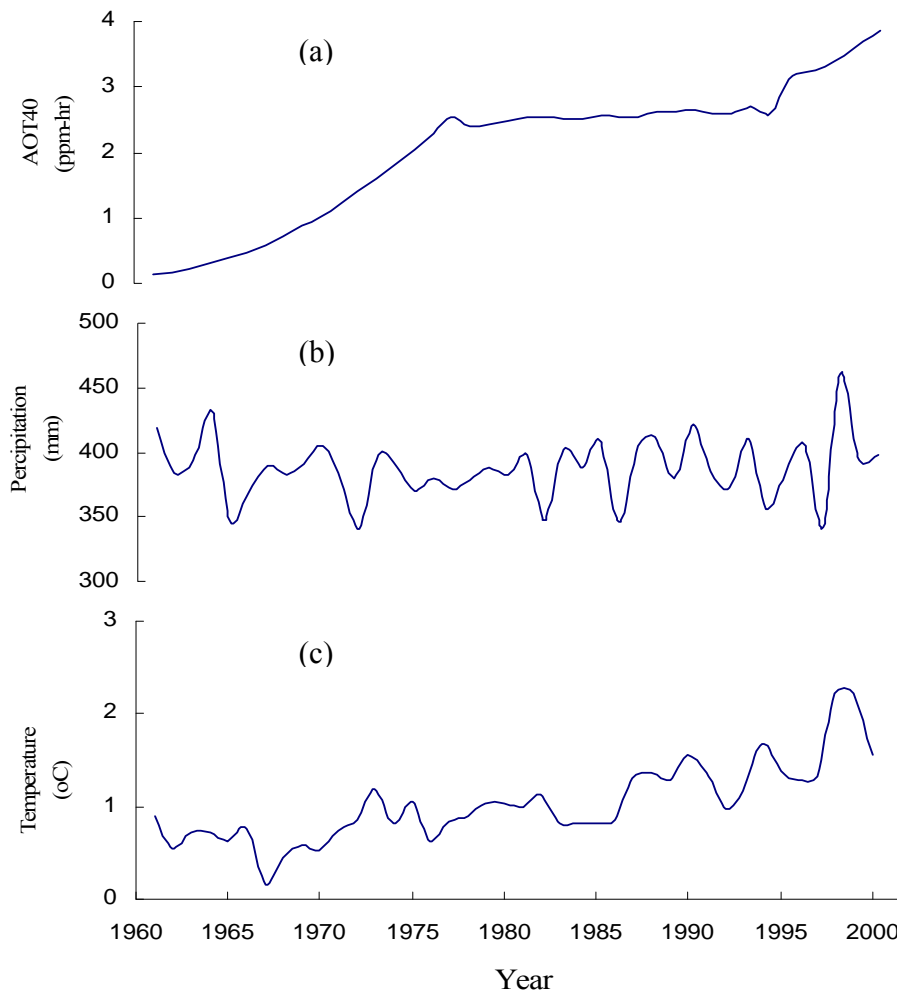
Figure 4-2 Changes in tropospheric ozone and climate across grassland areas of China: (a) annual average monthly AOT40 (ppm-hr/month), (b) precipitation (mm) and (c) mean annual temperature (ºC) for the grassland area in China.

3. Results and discussion

3.1 Tropospheric O_3 concentrations and climate variability in China during the past decades

The simulated AOT40 data set (Figure 4-2a) shows that from 1961 to 2000, O_3 concentrations have significantly increased across the entire grassland area of 327×10^6 ha as estimated in this study (Figure 4-3a). The most rapid increases occurred during

73

1970s and 1990s, which might be partly due to accelerated industrialization and urbanization in these two time periods (Liu et al., 2005b). From the map of spatially distributed annual average AOT40 (Figure 4-3b), we find an increasing trend of O_3 concentrations throughout the entire grassland area. The greatest rate of increase occurred in the Central North of China while the greatest increase of AOT40 values was in North-West China, probably due to the rapid industrialization in North China and the transport of pollutants from Europe (Akimoto, 2003). On the contrary, AOT40 was relatively low in the southeast of China, which might be that the area is small in the SE portion of China, and the way the data were derived by the model could result in lower than expected AOT40 values. So much monitoring data are needed in these sensitive areas in the near future.

Annual mean precipitation and temperature show substantial interannual and decadal variations (Figure 4-2b, c; Figure 4-3c, d). Since the late 1980s, the mean annual temperatures in the grassland ecosystems of North China have risen rapidly with the highest increasing rate of more than 2.8°C in some arid regions in western China and semi-arid regions in the west part of the northeastern China. Since 1990, annual precipitation has increased with a maximum of 461 mm in 1998 (Figure 4-2b). In addition, although annual average precipitation increased since 1990 in most northern and southern grassland areas, precipitation significantly decreased in some arid areas in the south of western China (Figure 3c) and in the semi-arid areas of central China, with a reduction of more than 100 mm/yr (the maximum decrease is 331 mm). These climate change patterns are consistent with some recent studies on climate variability in China

74

(Yang et al., 2002; Li et al., 2004). Over two-thirds of China's grasslands are located in temperate regions where the precipitation is extremely low, air temperature variation is greater and O_3 concentrations, except for Tibet are much higher compared to other grassland regions in China.

3.2 Spatiotemporal variations in carbon flux and storage as influenced by increasing O_3 pollution

Under the influences of the combined environmental factors of O_3, climate, land use and CO_2 (OClmLC) (Table 4-2), the total grassland productivity during the past 40 years changed only slightly, with an average increase of mean annual NPP from 400.5 Tg C/yr in the 1960s to 400.8 Tg C/yr in the 1990s. Ecosystem carbon pools including vegetation carbon (VC), soil carbon (SC) and total carbon (TC) increased 0.04Pg C (6.1%), 0.03Pg C (0.1%) and 0.07Pg C (0.3%), respectively from 1961 to 2000.

Through comparing the effects of OClmLC with ClmLC (Table 4-2, Figure 4-4 and Figure 4-5), we find that increasing O_3 concentrations could lead to a general decrease in NPP and total carbon storage, and these negative effects might continue to increase because of the rapid increase in O_3 concentrations since the 1990s (e.g. Elliot et al., 1997; Sims, 1999; Aunan et al., 2000). From the1960s to the 1990s, with O_3 included in the model (OClmLC senario), grassland NPP increased only 0.3Tg C, while without O_3 (under ClmLC scenario), NPP increased about 14.3Tg C, which indicates a net reduction of 14.0 Tg C in NPP induced by elevated O_3 (Table 4-2). The VC, SC and TC, accordingly, were about 0.03 Pg C, 0.08Pg C and 0.11Pg C lower, respectively from the 1960s to the 1990s.

Table 4-2 Overall changes in net primary productivity (NPP) between 1990s and 1960s and carbon pools including vegetation carbon (VC), soil carbon (SC) and total carbon (TC) between 2000 and 1961

	Scenarios								
	ClmLC				OClmLC				OClmLC-ClmLC
Carbon Flux (Tg C)	60s	90s	90s-60s	%	60s	90s	90s-60s	%	90s-60s
NPP	403.2	417.4	14.3	3.5	400.5	400.8	0.3	0.1	-2.2
Carbon Pool (Pg C)	1961	2000	2000-1961	%	1961	2000	2000-1961	%	2000-1961
VC	0.69	0.76	0.07	10.1	0.69	0.74	0.04	5.8	-0.02
SC	22.67	22.78	0.11	0.5	22.67	22.7	0.03	0.1	-0.34
TC	23.36	23.54	0.18	0.8	23.36	23.43	0.07	0.3	-0.36

Table 4-3 Overall changes in net primary productivity (NPP) between 1990s and 1960s and carbon pools including vegetation carbon (VC), soil carbon (SC) and total carbon (TC) between 2000 and 1961 under scenarios of O_3

	Scenarios								
	O				OClm				OClm-O
Carbon Flux(Tg C)	60s	90s	90s-60s	%	60s	90s	90s-60s	%	90s-60s
NPP	346.2	337.7	-8.5	-2.5	463.0	452.3	-10.7	-2.3	-2.2
Carbon Pool(Pg C)	1961	2000	2000-1961	%	1961	2000	2000-1961	%	2000-1961
VC	0.29	0.26	-0.03	-10.3	0.03	-0.02	-0.05	-166.7	-0.02
SC	25.58	25.54	-0.04	-0.2	23.05	22.67	-0.38	-1.6	-0.34
TC	25.88	25.80	-0.07	-0.3	23.07	22.64	-0.43	-1.9	-0.36
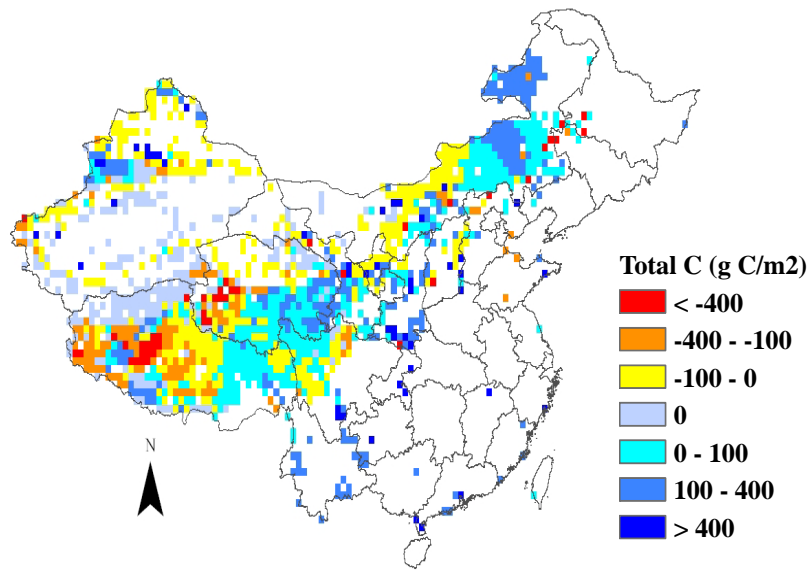
Figure 4-3 Carbon source and sink during 1961-2000 across China's grassland area as a simulation result of combined effect of climate, land use, O_3 and CO_2 by DLEM (g/m^2)
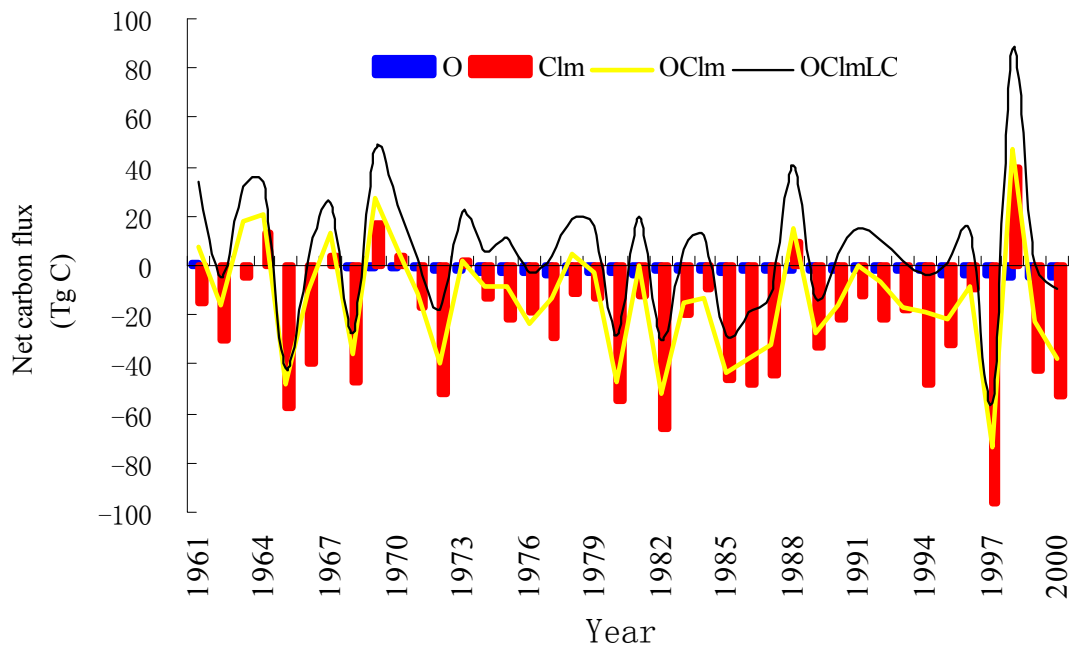
Figure 4-5 NCE responses from 1961-2000, showing the effect of ozone and climate disturbance (Pg C yr^{-1}), O (O_3), C (climate), OC (O_3_climate) and OClmLC (O_3_Climate_Lucc_O_3).
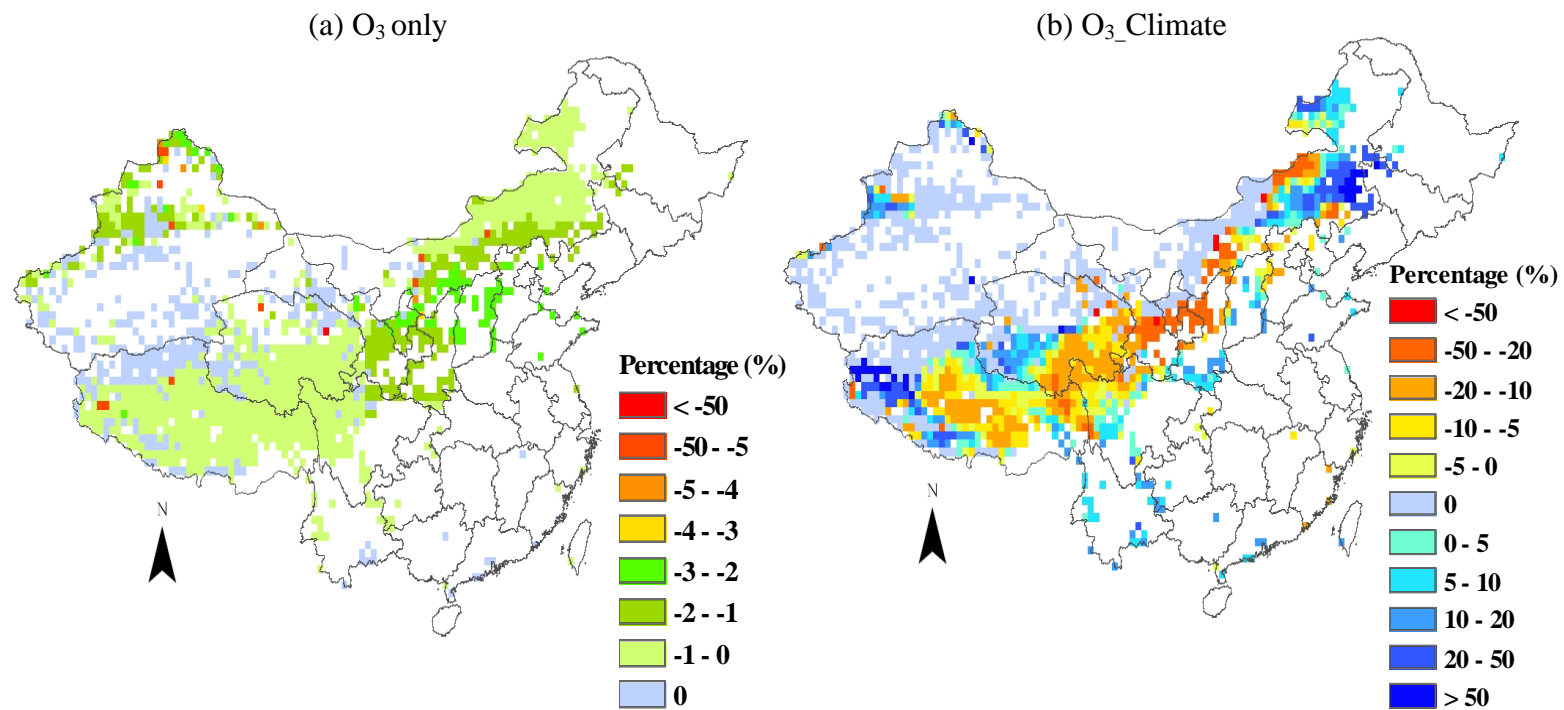
Figure 4-4 Net primary productivity (NPP) change rate (%) between 1990s and 1960s in (a) O$_3$ only and (b) O$_3$_Climate, showing the different effects of ozone alone and combination effects of ozone and climate change on NPP of China's grassland during the past four decades.

Although the temporal changes in TC were small at the national level (0.07Pg C increase in TC) from the 1960s to the 1990s under OClmLC (Table 4-2), significant spatial variability in TC is observed in some grassland areas (Figure 4-4). We found that total carbon storage decreased in most of the grassland areas in the southwestern and the northwestern portions of China, with a maximum carbon release of more than 400 g/m^2 in the places of southwestern China. Total carbon storage increased in most of the grassland areas in the west of northeastern China and the east of southwestern China. Some areas in these regions showed a maximum carbon sink of more than 400 g/m^2. By comparing the spatial pattern of O_3 (Figure 4-2b) and accumulated NCE (Figure 4-6), we found that the influence of O_3 was compounded by other environmental factors (e.g. climate, land use change and CO_2 change).

3.3 Combination effects of O_3 with changing climate on carbon flux and storage

Since grasslands are mostly located in arid and semi-arid regions, climate thus becomes a main limiting factor in controlling net primary production and carbon storage of grassland ecosystems. To study the effects of O_3 pollution under climate constraints on China's grassland ecosystem production, we conducted factorial analyses with three scenarios, including O_3 only (O), climate only (Clm), and the combination of O_3 and climate variability (OClm). In response to a combination of historical O_3 and climate variability including air temperature and precipitation (Table 4-3), the average annual NPP across China's grasslands from the 1960s to the 1990s decreased by 10.7 Tg C , The VC, SC and TC were reduced by 0.05 Pg C, 0.038 Pg C and 0.43 Pg C, respectively from 1961 to 2000. The OClm scenario resulted in 2.2 Tg C more NPP loss than that under the O scenario from the 1960s to the 1990s. The VC, SC and TC of the OClm scenario were

0.02 Pg C, 0.34Pg C and 0.36Pg C lower, respectively, than the results of the O scenario. We also found that spatial and temporal patterns of NPP under combined factors were consistent with those resulting from climate change alone, which indicates that most of the interannual variation in NPP was due to climate variability. We further found that spatial and temporal patterns of annual NPP were strongly controlled by the precipitation pattern (Figure 4-2b). Our results are consistent with other studies as reported by Tian et al. (1999; 2003). In addition, interannual variability in NPP and R_H led to substantial interannual variability in net carbon exchange between the atmosphere and grassland ecosystems (Figure 4-6). For all those three scenarios (Clm, OClm and OClmLC), there was a maximum carbon release in 1997 when precipitation was relatively low and a maximum carbon sink in 1998, the year having the highest precipitation (Figure 4-2a, b).

The effects of O_3 and climate on NPP varied from region to region under the O scenario (Figure 4-5a) and OClm scenario (Figure 4-5b) from the 1960s to the 1990s. We found a mean reduction of 2-3% in NPP in the entire grassland under the influence of O_3 alone (experiment I); however, when adding the climate change scenarios (experiment III), a mean reduction of 10-20% in NPP was found, with a maximum reduction rate of more than 50% in some places. Under both scenarios, the greatest reduction in NPP occurred mostly in central China, which possibly was due to the unique environment in central China where O_3 concentrations are higher (Figure 3b), denser grass coverage, higher productivity, and significantly changed climatic conditions than other grassland areas (Yang et al., 2002; Liu et al., 2005; Piao et al., 2004). The most rapid rate change

also occurrred in the east of southwestern China, which was induced primarily by climate change rather than O_3.

We found that vegetation carbon (-10.3%) is more sensitive than soil carbon (0.2%) in response to elevated O_3 (O and OClm, Table 4-3). This response may be due to the fact that O_3 and climate have greater effect on the living vegetation than the decomposition and accumulation of soil organic matter. In terms of previous studies, soil carbon storage is mainly controlled by the combined effects of O_3, soil moisture and nitrogen (Nussbaum et al., 2000). In other field observations it was reported that soil moisture has been decreasing in the past 100 years, despite increased precipitation in these areas (Li et al., 2004). Thus, soil carbon storage has decreased less than vegetation carbon due to the lower respiration rate caused by both drought stress and O_3 in the arid and semi-arid grassland areas. Although it was suggested that O_3 stress was less significant in arid environments due to decreasing stomatal conductance (Ollinger et al., 1997), it is well know that the ratio of shoot to root could increase and reduced root biomass can result in serious drought stress with the decline of water uptake (Cooley and Manning, 1987). In addition, recent reports (Grulke et al., 2007; McLaughlin et al., 2007a) suggest that O_3 may be causing stomatal sluggishness in certain species. This factor could have a profound influence on plant water relations and future modeling efforts (McLaughlin et al., 2007b). Further physiological studies regarding stomatal response to O_3 in grassland species is warranted.

Our results indicate that interannual variation in net carbon flux during 1961-2000 is primarily controlled by climate variability while O_3 leads to an increasing reduction of NCE (Figure 4-6). However, the combined effects of O_3 and climate cannot explain all of the variation in net carbon flux, which implies roles for CO_2, land use and grassland management.

3.4 Simulation results comparison

Similar to most field experiments (e.g. Heagle, 1989; Davison and Barnes, 1998; Fuhrer and Booker, 2003; Bobbink, 1998; Mclaughlin and Percy, 1999; Volk et al., 2006) and other regional model simulations (Ollinger et al., 1997; Felzer et al., 2004), our results show that O_3 has negative effects on semi-natural grassland ecosystem production. Due to differences in research methods (e.g. NDVI-biomass, forage yield, biomass and carbon density) as well as grassland area estimates (from 299 to 570 10^6 ha), the assessment of vegetation carbon storage (0.13-4.66 Pg C) and carbon density (0.06-1.15Pg C) vary significantly among different studies (Fang et al., 1996b; Ni, 2004; Piao et al., 2004) as shown in Table 4-4. Our estimates on vegetation carbon and carbon density are close to two recent analyses of Piao et al. (2004) and Ni (2004), but much lower than other previous studies (Fang et al., 1996b; Ni, 2001).

Uncertainties, including input data sets and model parameters, might result in imprecise estimation of the effects of O_3 and climate variability on grassland ecosystems. To better estimate regional carbon budgets and to better understand the underlying mechanisms, it is essential to examine how the structure and function of grassland ecosystems have changed as a result of multiple stresses, and interactions among those

stresses, including land-use change, climate variability, atmospheric composition (carbon dioxide and tropospheric O_3), precipitation chemistry (nitrogen composition), and fire frequency. Model estimates along with spatial and temporal patterns of carbon fluxes and storage need to be further evaluated through comparisons with the results of field studies, vegetation and soil inventories within China's grasslands.

Table 4-4 Estimates on Carbon storage in vegetation (VC) of grasslands in China

Source	Area (10^6 ha)	VC (Pg C)	Carbon density (g C/m^2)	Method
This study	327	0.69-0.74	0.21-0.23	Process-based modeling
Piao et al. (2004)	331	1.04	0.31	NDVI-biomass
Ni (2004)	299	0.13	0.06	Forage yield
Ni (2002)	299	3.06	1.15	Carbon density
Ni (2001)	406	4.66	1.15	Carbon density
Fang et al.(1996b)	570	1.23	0.22	Biomass

4. Conclusions

We studied the influences of elevated O_3 and climate variability on grassland ecosystem productivity across China from 1961 to 2000 by applying a process-based dynamic land ecosystem model. In this study, the analysis of temporal and spatial changes under different scenarios explains the contributions of O_3 and changing climate. Our results showed that with the combined effects of elevated O_3 concentrations and other environmental factors, including climate, land use and CO_2, the total grassland ecosystem productivity across China during the 1960s-1990s had a small increase (0.3Tg C) in annual NPP, and carbon storage in vegetation, soil and the total ecosystem from

1961 to 2000 increased 0.04 Pg C, 0.03 Pg C and 0.07 Pg C, respectively, indicating that China's grasslands acted as carbon sinks during the past 40 years . The results of simulation experiments indicate elevated O_3 leads to a general decrease in NPP and carbon storage, and these negative effects became more evident since the 1990s. Due to spatial heterogeneity in O_3 and climate distribution, simulated results show some places were a carbon source while other places were a carbon sink during the past four decades. Our simulation experiments indicate that China's grasslands could potentially increase annual NPP by 14 Tg C and total carbon storage by 0.11 Pg C.

Most of China's grasslands are located in arid, semi-arid areas and in the Tibetan plateau, where they have experienced significant environmental changes including elevated O_3 and substantial climate variation in the last 40 years of the 20th century. To accurately estimate response of carbon dynamics in these areas, future O_3 monitoring stations need to be established. Experiments on the effects of O_3 and climate variability on diverse grass species are required to be conducted. In addition, the model needs to be improved to simulate the mechanisms of ecosystem response to multiple stresses.

Chapter 5

Forest Ecosystems, Ozone pollution, Climate Variability/Change –
Influences of Ozone Pollution and Climate Variability on Forest
Ecosystems' Productivity and Carbon Storage in China

Abstract

We investigated the potential effects of elevated ozone (O_3) along with climate variability on net primary productivity (NPP) and carbon storage in China's forest ecosystems for the period 1961-2005, using the Dynamic Land Ecosystem Model (DLEM). Simulated results showed that elevated O_3 could result in a 0.2%-1.6% reduction in total NPP and a 3.5%-12.6% reduction in carbon storage, respectively. Interannual variations and spatial patters of annual NPP and carbon storage across China's forestlands were controlled by climate variability/change, and the combined effects of O_3 pollution with extreme dry conditions could lead to more carbon loss. The reduction rates of NPP and carbon storage ranging from 0.1% - 2.6% and from 0.4% - 43.1%, respectively, indicated varied sensitivity and vulnerability to elevated O_3 pollution among different forest types. Our preliminary results imply that improvement in air quality could significantly enhance the adaption and reduce the vulnerability of forests in China to climate variability/change.

1. Introduction

China has experienced one of the most rapid environmental changes in the past three decades and is likely to undergo further rapid development in the coming years. However, severe air pollution and frequent droughts have been the most serious environmental problems that have threatened the sustainability of China's ecosystems as well as its economy (China's National Climate Change Programme, 2007). Between 1980 and 1995, fertilizer use in China was 36% higher than the average in developed countries (where fertilizer use has been decreasing), and 65% higher than the average in developing countries (Aunan et al., 2000). Both fossil fuel consumption and N-fertilizer application will be major contributors to total emissions of NO_x, a main O_3 precursor, and consequently result in increased atmospheric O_3 concentration. It was estimated that China's emissions of NO_x might increase by a factor of four as the year 2020 approadches, compared to the emissions in 1990 under a noncontrol scenario (van Aardenne et al., 1999); this would lead to a much larger increase in surface O_3 with a 150 ppb level of O_3 in some locations (Elliott et al., 1997). In addition, the accelerated global warming has become a challenge faced by scientists, policy makers and public in every nation. China has experienced clear warming trends in the past two decades, and the 1990s was one of the warmest decades in the last 100 years with the large temporal and spatial variations of temperature and precipitation (Sha et al., 2002; Zuo et al., 2003; IPCC, 2007).

Forest ecosystems play dominant roles in the terrestrial carbon budget because they store a large amount of carbon in vegetation and soil, and its interactions with climatic change and atmospheric process (Goodale et al. 2002). Forest productivity, governed by both natural factors (e.g. climate, succession, and disturbance etc.) and human activities, is regarded as a key indicator of changes in forest ecosystem structure and function (Brown et al., 1999). Many studies have been conducted to estimate vegetation/soil carbon stock, biomass, NPP, and NEP, including inventory-based methods in China (e.g. Zhao et al., 2004; Feng et al., 1980; Li et al., 1981; Kang et al., 1996; Ma et al., 1996; Fang et al., 1996a, b; Luo et al., 1998;1999; Fang, 2000; Wang et al., 2001a; 2001b), process-based models (Xiao et al. 1998; Ni J, 2001; Pan et al., 2001; Li and Ji., 2001; Cao et al. 2003; Zhang et al., 2003; Feng, 2004; Tao, 2005; Huang et al. 2006), and remote sensing-based methods (e.g. Gong et al. 2002; Piao et al.,2005) at different spatial and temporal scales. Most studies indicated that NPP in China's forest ecosystems, including plantation and natural forests, experienced an continually increasing trend over past decades. Climate change could increase forest NPP (Tao, 2005) and would have the most obvious impact on the geographical distribution of NPP (Liu et al., 1998). Many studies showed that water regimes could be the key factor controlling forests NPP variations under the background of climate change (Zhou and Zhang, 1996; Liu et al. 1998; Gong et al. 2002; Cao et al. 2003). Gong et al. (2002) and Liu et al. (2000) estimated that China's forest ecosystems served as a carbon source of 0.03 Pg C from 1982 to 1998 and a carbon source of 0.06 Pg C in the period 1982-1988. The estimations of forest carbon sink in the 1990s varied from 0.039-0.068 Pg C based on model simulation (Tao, 2005) and inventory data (Pan et al., 2004).

Recent negotiations on the Kyoto Protocol to the UN Framework Convention on Climate change (UNFCC) have focused considerable attention on forests in the context of climate change (IISD, 2001). Forest both influence and are influenced by climate change, and play an important role in the global carbon cycle. More than 50% of terrestrial vegetation carbon is stored in forest ecosystems (Dixon et al., 1994) and boreal forests soils account for about 26% of total terrestrial carbon stock. Meanwhile, forest structure, function, and distribution could significantly affect the course of global warming in the 21st century. Recently more and more studies have focused on the impacts of potential climate change on forests ecosystem and feedback between climate and forests (Gate, 1990; Bonan et al., 1992; Mellilo et al., 1993; Smith et al., 1995; Joyce et al., 1995; Braswell et al., 1997; Shafer et al., 2001; Hansen et al., 2001; Logan et al., 2003; Hogg et al., 2005; Biosvenue and Running, 2006). In China, approximately 18.21% of the landbase is forested as reported in the 6th National Forest Resources Inventory, 1999-2003 (Xiao, 2005), which makes forest ecosystems prominent natural resources that contribute to biodiversity, absorbing air pollutants, and carbon sequestration. Most of the forested area**s** are distributed in the Monsoon climate zone with high O_3 pollution in some places. However, little is known about how elevated O_3 concentrations and drought stress derived from global warming have influenced China's forest ecosystems. Therefore, understanding the responses of China's forest ecosystems to climate change and air pollution in the context of global change is of great significance to regional sustainable development.

In this study, we investigated the potential effects of elevated O_3 along with climate variability on net primary productivity (NPP) and carbon storage in China's forested ecosystems for the period 1961-2000 by using a process-based Dynamic Land Ecosystem Model (DLEM). In addition to historical information concerning O_3 pollution and climate change, we considered the major environmental factors as model input, including atmosphere CO_2, nitrogen deposition, land use change and regrowth in the context of global change to reduce the uncertainties. Our objectives in this study are: 1) to illustrate the effects of tropospheric O_3 pollution in combination with climate variability on productivity and carbon storage of China's forest ecosystems from 1961 - 2000; 2) to look into the temporal-spatial patterns of carbon pools and fluxes in China's forest ecosystems from 1961 to 2000; 3) to examine the varied sensitivities of different forest types in response to O_3 pollution ; 4) and to investigate the effects of O_3 pollution combined with extreme climate conditions (wet and drought) on forest NPP and carbon storage.

2. Materials and methods

2.1. The Dynamic Land Ecosystem Model (DLEM) and input data

The same method and input data as the detailed in the description in chapter 3 were used in this study (Figure 5-1). The refined nitrogen deposition data has been applied to the study.
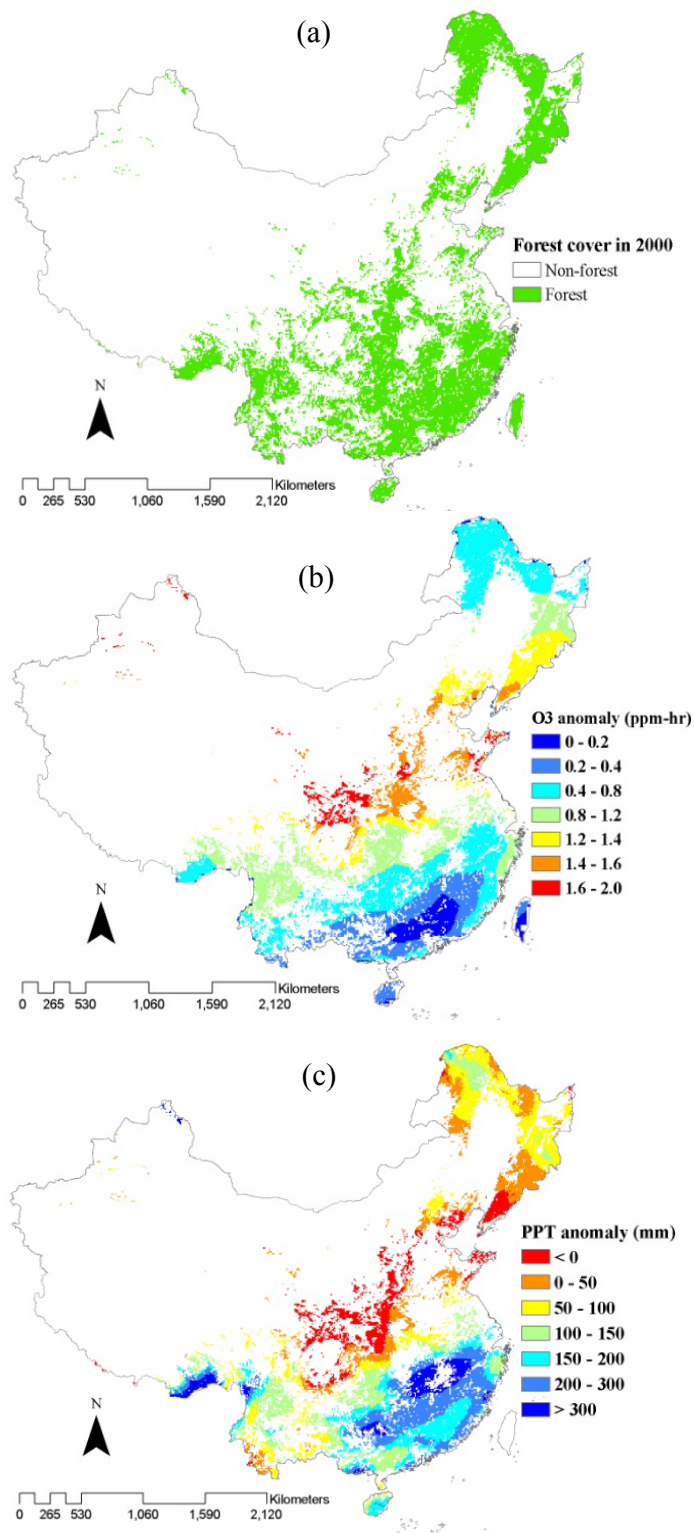
Figure 5-1 Map of forests distribution in China (a) and maps of anomalies in the 1990s (relative to the average for 1961-1990) for (b) annual average AOT40 (ppm-hr), (c) precipitation (mm).

2.2. Experimental design

Table 5-1 Experimental arrangement including O_3, climate, nitrogen deposition, CO_2 and land use

Scenarios	Environmental Factors		
	O_3	Climate	CO_2,Ndep & Land use
I Only O_3 (O_3)	H	C	C
II Only Climate (Climate)	0	H	C
III O_3_Climate (Clm_O_3)	H	H	C
IV Climate_Lucc_Ndep_CO_2(All_O_3)	0	H	H
V O_3_Climate_Lucc_Ndep_CO_2(All)	H	H	H

Note: H is historical data, 0 means no data and C is constant data. Here we use CO_2 concentration (296ppm) in 1900 and mean climate data sets in 30 years from 1960 to 1990 and potential vegetation map as constant value.

In our study, we designed five simulation experiments to analyze the effects of O_3 only and climate only, and the combined effects of O_3 and climate on NPP and carbon storage in the forest ecosystems of China (Table 5-1). In experiment I, we tried to examine the sole extent of O_3 impacts while other environmental factors were constant. In experiments II and III, we analyzed the contribution of climate variability only and the combined effect of climate variability and O_3, respectively. The other two simulation experiments IV and V were designed to simulate a relatively realistic scenario to explore the effects of O_3 on ecosystem production. In these simulations, the important environmental factors, including climate variability, increasing CO_2 and nitrogen deposition, and land use change were considered.

The model simulation began with an equilibrium run to develop the baseline C, N, and water pools for each grid. Then a spin up of about 100 years was applied if climate

variability was included in the simulation scenario. Finally, the model ran in transient

model driven by transient data of climate, O_3, CO_2, nitrogen deposition and land use.

3. Results

3.1 Temporal variations of annual NPP and carbon storage

Table 5-2 Decadal mean and change rates of average annual NPP and NCE in the 1960s, 1990s, and recent five years (2000-2005) under the combined effects with and without ozone pollution.

	NPP (Tg C/yr, 10^{12} g C/yr)				Carbon storage (Tg C/yr, 10^{12} g C/yr)			
	O_3	All	All_O_3	(All-All_O_3)%	O_3	All	All_O_3	(All-All_O_3)%
1960s	1296.2	1438.8	1442.0	-0.2	-3.9	81.0	83.9	-3.5
1970s	1288.5	1474.6	1484.2	-0.6	-10.2	90.1	98.4	-8.4
1980s	1291.3	1534.0	1544.1	-0.7	-6.3	108.7	116.2	-6.4
1990s	1287.4	1598.7	1613.9	-0.9	-8.4	112.2	122.7	-8.6
00-05	1280.4	1668.0	1694.8	-1.6	-18.4	120.9	138.2	-12.6
90s-60s%	-0.7	11.1	11.9		-116.3	38.6	46.3	
05-60s%	-1.2	15.9	17.5		-371.8	49.3	64.8	

In the simulation experiments, there were significant negative effects of O_3 on

total NPP and carbon storage during the study period from 1961-2005 (Table 5-2). With

the O_3 only effects, annual total NPP and carbon storage from the 1960s to the 1990s

decreased by 0.7% and 116.3%, respectively. When considering other environmental

factors of climate change, land cover and land use, nitrogen deposition and CO_2 together

with O_3 pollution (All) , the total forest NPP was estimated as 1438.8 Tg C/yr, 1474.6 Tg

C/yr, 1534.0 Tg C/yr, 1598.7 Tg C/yr, and 1668.0 Tg C/yr in the 1960s, 1970s, 1980s,

1990s and recent five years (2000-2005), respectively, with the rates increasing 11% in

the 1990s and 15% from 2000-2005 compared to the 1960s. The annual variations of

NPP under different scenarios indicate that O_3 pollution has uniform negative effects on forest production (Figure 5-2a); total NPP could be reduced 0.2% to 1.6% from the 1960s to five recent years. Climate is the dominant factor controlling the interannual changes in total NPP (Figure 5-2a).
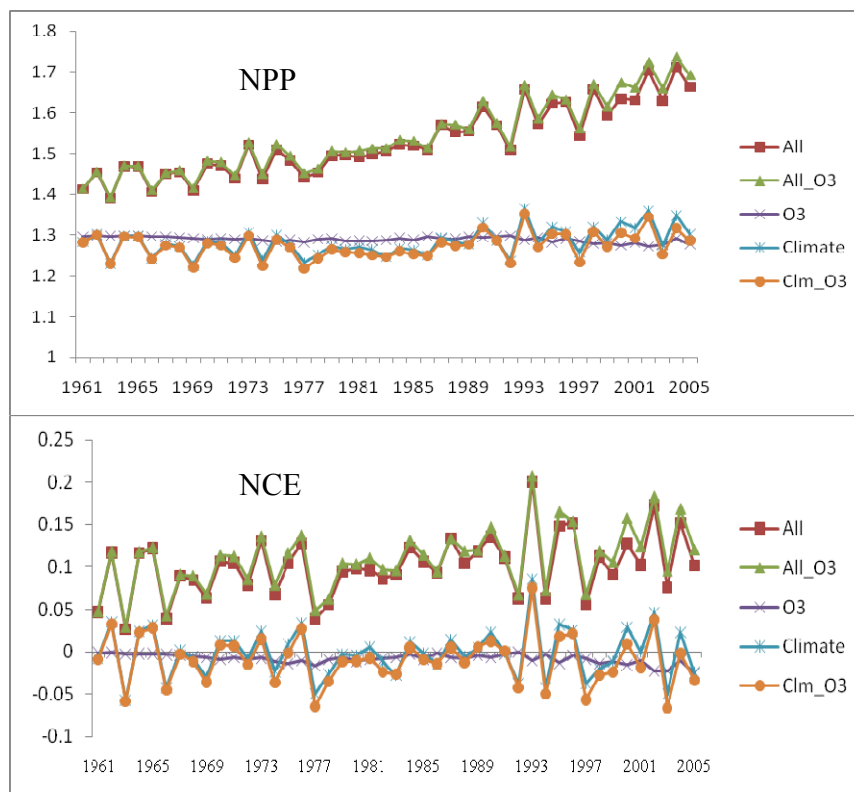
Figure 5-2 Changes of annual net primary productivity (NPP) (a) and net carbon exchange (NCE) (b) (Pg C/yr) from 1961to 2005 under different scenarios

With O_3 only effects, there was continuous carbon release from the 1960s to five recent years (2001-2005) in China's forest ecosystems ranging from 3.9 Tg C per year to 18.4 Tg C per year (Table 5-2). Under the combined influences (All), China's forest ecosystems were simulated as carbon sinks over the past forty five years and carbon
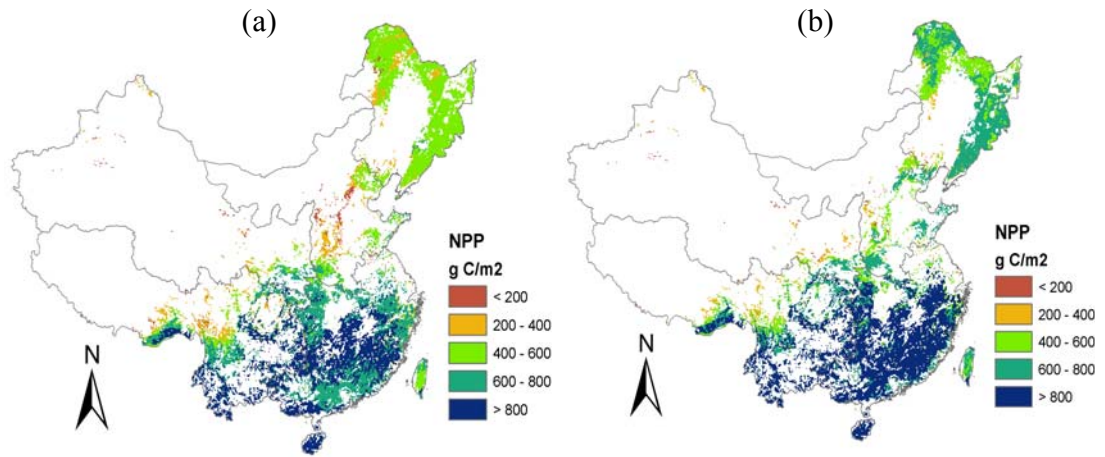
Figure 5-3 Decadal mean net primary productivity (NPP) (g C/m^2/yr) in the 1960s (a) and 1990s (b)
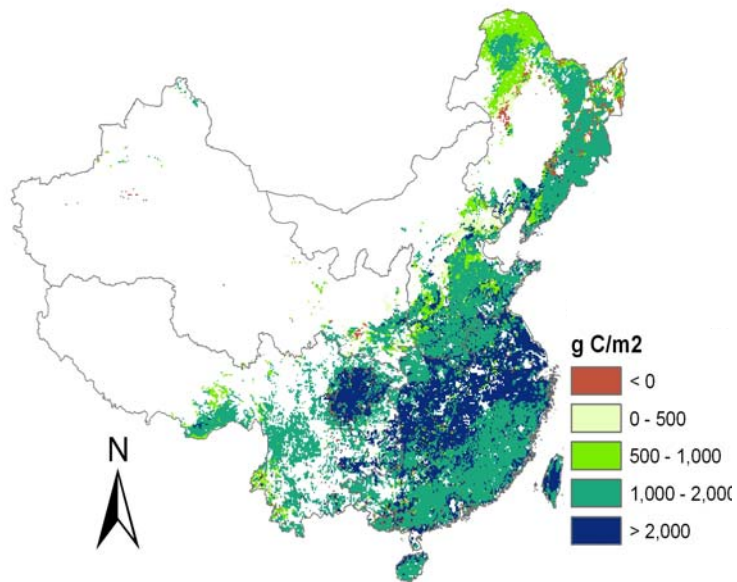
Figure 5-4 The total accumulated net carbon exchange (carbon storage) (g C/m^2) in forest ecosystems during the period 1961-2005.

uptake increased from 81.0 Tg C per year in the 1960s to 90.1, 108.7, 112.2, 120.9 Tg C per year in the 1970s, 1980s, 1990s and the most five recent years, respectively, with rates increasing 38.6% in the 1990s and 49.3% from 2000-2005 compared to the 1960s.Similarly, O_3 pollution had negative effects on carbon uptake and led to carbon storage reduction of about 3.5% in the 1960s to12.6% in five recent years, respectively. Climate could cause carbon uptake or release, and resulted in interannual changes of net carbon exchange (NCE) (Figure 5-2b).

3.2 Spatial variations of annual NPP and carbon storage

Since forestlands are mostly located in the SE, SW and NE regions across China, monsoon climate in most areas and severe air pollution in part of those regions resulted in large spatial variations of NPP and carbon storage of forest ecosystems (Figure 5-3 and Figure 5-4). The highest NPP occurred in Southeast (SE) China with the highest NPP of more than 1500 g C/m^2/yr occurring in some areas. Annual NPP increased across China's forest ecosystems in the 1990s compared to NPP level in the 1960s with the largest increase occurring in the SE forest area and then in the NE region (Figure 5-3a,b). However, the spatial distribution of NPP in the 1990s and 1960s showed that NPP was very low (less than 200 g C/m^2/yr) in some places in the Mid-north (MN) where these areas experienced frequent drought and high air pollution. Across China's forest ecosystems, the SE had the largest carbon storage with a carbon uptake of more than 2000 g C/m^2/yr between 1961 and 2005, followed by the NE. However, carbon release appeared in some places of the NE and SW regions (Figure 5-4).

95

3.3 Biome analysis of annual NPP and carbon storage

Table 5-3 Changes of average annual net primary productivity (NPP) and carbon storage in different forest types during the period 1961-2005 across China's forest ecosystems under the scenarios of the combined effects with (All) and without ozone pollution (All_O_3).

	NPP All-All_O_3 (%)	Carbon storage All-All_O_3 (%)
Boreal broadleaf deciduous forest	-1.1	-21.8
Boreal needleleaf deciduous forest	-0.3	-5.2
Temperate broadleaf deciduous forest	-2.3	-21.1
Temperate broadleaf evergreen forest	-0.6	-7.6
Temperate needleleaf evergreen forest	-0.1	-0.4
Temperate needleleaf deciduous forest	-0.4	-6.2
Tropical broadleaf deciduous forest	-2.6	-43.1
Tropical broadleaf evergreen forest	-0.6	-9.6

Annual mean NPP and carbon storage of different PFTs under the two simulation experiments indicated that forest types responded differently to an increasing O_3 concentration and its interaction with other environmental factors (Table 5-3). From 1961 to 2005, the reduction rates of annual mean NPP for different forest types with and without O_3 exposure ranged from 0.1% in temperate needleleaf evergreen forest to 2.6% in the boreal broadleaf deciduous forest, while the reduction rates of annual carbon sequestration rates ranged from 0.4% in temperate needleleaf evergreen forest to 43.1% in tropical broadleaf deciduous forest. This range implies that tropical broadleaf deciduous forest could be the largest carbon source under the full factorial effects, while the temperate needleleaf evergreen forest was less sensitive to O_3 pollution.

3.4 Relative contributions of O_3 pollution and climate change on NPP and carbon storage
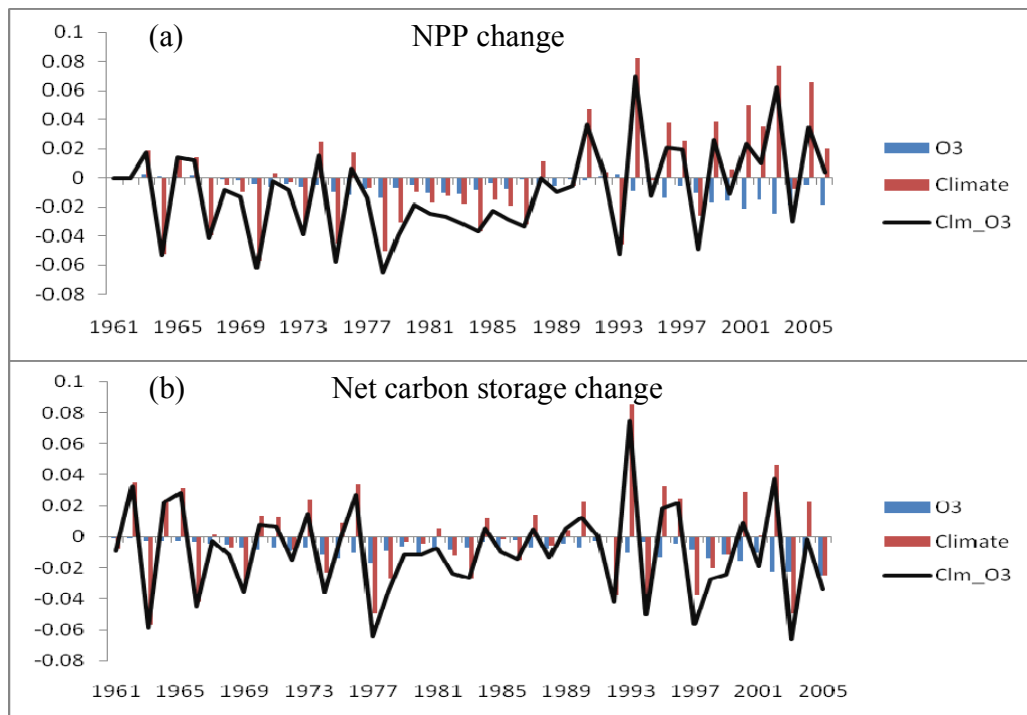
Figure 5-5 Annual changes in relative contributions of ozone pollution, climate change and interaction to (a) annual net primary productivity (NPP) and (b) net carbon storage (Tg C/yr)

Temporal variations of relative contributions (Figure 5-5) indicated that O_3 pollution had negative effects on both NPP and carbon storage during the period 1961-2005, with two periods (between the late 1970s and the early 1980s and after the 1990s) of high reduction in NPP and carbon storage. Climate change had both negative and positive effects on NPP and carbon storage, and was the major factor causing interannual variations of NPP and carbon storage. Between the late 1970s and the early 1980s, climate change had continuously reduced NPP and carbon storage, while after the 1990s it contributed much to NPP increase. The negative effects of O_3 pollution were either accelerated or offset when combined with climate change, therefore, the interactive

effects of O_3 pollution and climate change showed continuously aggregated effects on the reductions in NPP and carbon storage between the 1970s and 1990s.

Similarly, we found that spatial patterns of NPP change under only the O_3 pollution scenario between the 1990s and 1960s and varied from region to region (Figure 5-6a), with reduction rates ranging from less than -5% to 0%. Combined with climate effects, the rate of decrease was accelerated or offset, and ranged from < -10% to 10% (Figure 5-6b), which indicated that climate change was the dominant factor to control the spatial pattern. In further analysis we found that and the patterns were strongly consistent with the precipitation pattern in (Figure 5-1c), with the high NPP reduction rate of more than 15% in some place in the NE and MN regions which experienced low precipitation and high O_3 concentrations, and high NPP increase rates of more than 10% were found in some place in the SE region which experienced high precipitation and low O_3 concentrations. We examined the effects of O_3 pollution combined with climate change in extreme weather conditions including wet year and dry year, which were defined simply according to the annual total precipitation. The results indicated that O_3 pollution could result in NPP reduction in most forest areas even in extreme wet condition with high reduction rates in some parts of MN and NE regions; additionally, forest ecosystems with extreme dry conditions with a reduction rate of more than 40% (or 40 g C/m^2 reduction) in some places with extreme drought.
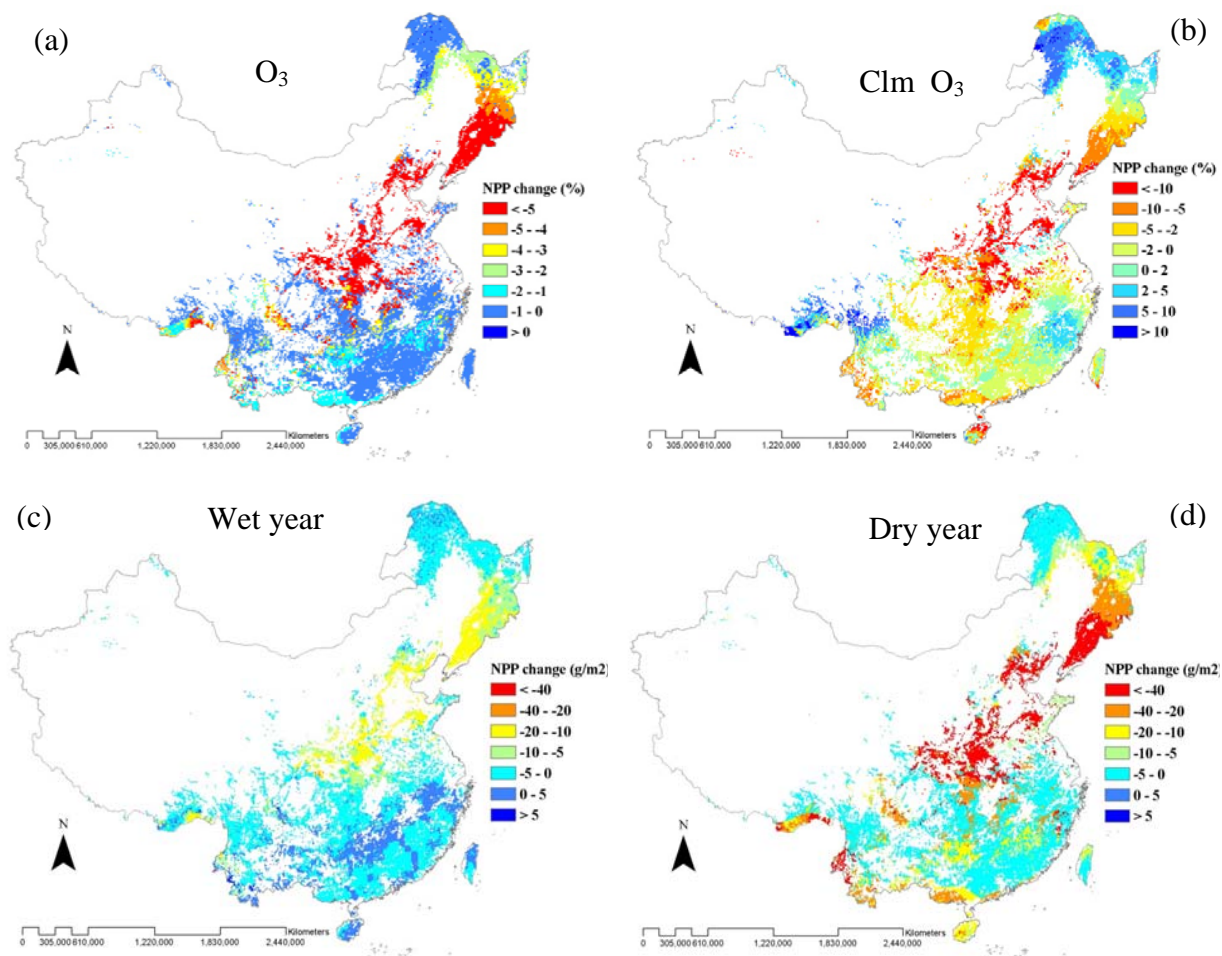
Figure 5-6 Net primary productivity (NPP) change rate (%) between 1990s and 1960s under scenarios of (a) O_3 only and (b) O_3 + Climate; difference of annual mean NPP between under the effects of O_3 pollution combined with climate change and the effect of climate only in wet year 2002 (c) and dry year 1986 (d).

4. Discussion

4.1 Comparisons of simulated net primary productivity (NPP) with other studies

Table 5-4 Estimations of mean NPP, total NPP, and carbon sequestration rate in different forest types and the whole forest ecosystems using models, inventory-based and RS methods

	Mean NPP ($Mg\ ha^{-1}\ year^{-1}$)			
Reference	This study		Ni et al. 2001	Jiang et al. 1999
Method	DLEM model		Biomass	RS estimation
	1989-1993	1994	1989-1993	1994
Boreal broadleaf deciduous forest	4.3	4.4	5.4-15.1	6.3
Boreal needle leaf deciduous forest	5.5	5.6	5.4-15.1	6.7
Temperate broadleaf deciduous forest	8.2	8.3	8.72-23.12	9.0-12.4
Temperate broadleaf evergreen forest	8.9	9.1	8.5-13.7	10.1-17.4

Our estimations of annual mean NPP in different forest types were close to other studies from inventory-based (Ni et al., 2001) and RS estimations (Jiang et al., 1999). For example, the simulated annual NPP of 5.5 $Mg\ ha^{-1}\ year^{-1}$Mg and 8.9 $Mg\ ha^{-1}\ year^{-1}$ in boreal needle leaf deciduous forest and temperate broadleaf evergreen forest, respectively, were compared to 5.4-15.1 $Mg\ ha^{-1}\ year^{-1}$ and 8.5-17.4 $Mg\ ha^{-1}\ year^{-1}$ from biomass estimations (Ni et al, 2001). However, estimation of total annual NPP varied significantly among different studies ranging from 0.59 Pg per year to 3.02 Pg per year during the period 1981-2000 (Fang et al., 1996; Xiao et al., 1998; Feng et al., 2004; Tao et al., 2007). The uncertainties possibly are due to differences in research methods and forest area. For example, methods based on forestry inventory or statistical/empirical models depend on plots numbers and whether their spatial distribution is homogeneous or not,

consequently, the different estimates derived from different inventory data, regression equations, and influencing indicators.

4.2 Comparison with estimates of ozone damage from other studies

While little work has been reported in China concerning the effects of O_3, our results like those studies on O_3 effects on forests for the short-term and long term based on field experiments (e.g. Alvarado et al., 1993; Andersen et al., 1997; Saleem 2001; Kim et al., 1998; Chappelka et al., 2002; Oksanen, 2003), and model simulations at the ecosystem scale and the tree physiological scale (e.g. Ollinger et al., 1997, 2002; Felzer et al., 2004; Hanson et al., 2005; Martin et al., 2001; Yun et al.,2001) in US and Europe, show that O_3 has negative effects on forest ecosystem production and carbon storage. Only considering the combined effects of O_3 and climate at the ecosystem scale, both Ollinger et al. (1997) and Felzer et al.(2004) found an annual NPP reduction ranging from 3%-16% in the NE USA for the period 1987-1992, using PnET-II model and TEM 4.3 model, respectively. In our research using DLEM model we found an annual NPP reduction of 2%-9% for the same time period, which agreed with the findings of Ollinger et al. (1997) and felzer et al.(2004) regarding adverse effects of O_3 on forests at the ecosystem scale but our results fell in the lower begin and end of their range, that is, the lower NPP reductions derived from O_3 in China than that in US in the same study period. The discrepancy between results is possibly caused by O_3 input data and the gaps among models, although the same O_3 sensitivity coefficients were used in all models.

4.3 Uncertainty analysis and future work

Our results indicate that major input data (e.g. O_3 and climate) and key parameters such as a sensitivity coefficient (see detailed model description in Chapter 3) can lead to great uncertainties in results. In order to conduct regional database evaluation using site observations, it is critical that the amount of monitoring stations be increased. The database development of land use and land cover including natural forests and managed forests is necessary to accurately estimate the regional carbon budget in forest ecosystems. Managed forests could enhance the ability for adaption to climate change and reduce the O_3 damage through land management by altering the interactions among carbon, water, and nutrient cycles (e.g. Ollinger et al., 2002; Hanson et al.,2005; Sun et al., 2008; Felzer et al., 2009). Further evaluation of model estimates along with spatial and temporal patterns of carbon fluxes and storage is greatly needed. More specifically, there is an urgent need to conduct short-term and long-term observations on air quality in forested areas, and also to conduct field experiments to investigate the effects of O_3, climate and other factors on forests in China.

5. Conclusions

We studied the influences of elevated O_3 and climate variability on forest ecosystem productivity and carbon storage across China from 1961 to 2005 by applying the DLEM model. Our results showed that both the total NPP and carbon storage in forest ecosystems could be reduced by elevated O_3 concentrations during the study period, and that these negative effects have become more evident since the 1990s. Climate variability/change was the dominant factor controlling interannual variations and spatial

patterns of total NPP and carbon storage, which in turn could reduce or increase NPP and carbon storage in different temporal-spatial scales; the positive effects on NPP have also become more evident since the 1990s. We found that the interactive effects of climate change and O_3 pollution could reduce carbon uptake or even increase more carbon release in extreme dry conditions in forest ecosystems. Though the combined effects of climate change, O_3 pollution and other environmental factors (nitrogen deposition, increasing CO_2 and land use change) led to carbon sink in China's forest ecosystems in the past forty five years, our simulation experiments showed that China's forestlands could sequestrate mean 8 Tg C per year more or increase the carbon uptake rate by 7.7% across forestlands without O_3 pollution effects. The study indicates that in the future O_3 pollution should be taken into account in order to reduce the uncertainty in assessing the ecosystems vulnerability to climate change in the context of rapid climate change and frequent extreme weather. Our preliminary results also imply that improvement in air quality could significantly increase productivity and the capacity of carbon sequestration in China's forest ecosystems.

Chapter 6

Agricultural Ecosystems, Ozone pollution, Climate Variability/Change,

Multiple Stresses –

Responses of Net Primary Productivity and Soil Organic Carbon Storage to

Multiple Environmental Changes in Agricultural Ecosystems of China

Abstract

Carbon sequestration in agricultural ecosystems plays a critical role in regulating global carbon cycling and feedback to global climate change. Thus, it is important to understand the contribution of the main environmental factors to carbon flux and storage in croplands, as well as the underlying mechanisms. In this study, we quantified the changes in net primary production (NPP) and soil carbon storage (SOC) in China's croplands in responses to multiple environmental factor changes from 1961 to 2005. A process-based ecosystem model DLEM-AG and the newly developed historical database of climate variability/change, ozone (O_3) pollution, land cover and land use change(LCLUC), fertilizer application and so on with 10 km resolution were used in model simulation. Results showed that both NPP and SOC in China's croplands increased from 1961 to 2005 with rates of 0.036 Pg C a^{-1} and 0.045 Pg C a^{-1}, respectively. These results were consistent with observation database and other studies. Between 1961 and 2005, the highest total annual NPP (on average 0.26 Pg C a^{-1}, 36.3% of the total NPP)

occurred in the Southeast (SE) region, while the largest annual soil carbon storage (mean 2.14 Pg Ca^{-1}, 35.2% of the total soil carbon storage) was found in the Northeast (NE) region with an increase rate of 0.023 Pg C a^{-1}. Land cover and land use change (LCLUC) increased both NPP and soil carbon storage, accounting for more than 80% of the total increase induced by multiple environmental changes. Elevated atmospheric CO_2 and nitrogen deposition contributed 16% and 14% of the total NPP increase, and 17% and 12% of the total soil carbon storage increase, respectively. Increasing O_3 pollution caused ~9% reduction in total NPP and ~2% soil carbon loss. Climate change resulting in NPP reduction and soil carbon loss was equivalent to ~9% and ~5%, respectively.

Keywords: agroecosystems, carbon storage, China; cropland, ecosystem model, multiple stresses, net primary productivity (NPP)

1. Introduction

An increasing concern on how agricultural ecosystems respond to multiple global changes has been raised, due to its importance in global food security and global carbon cycle (Paustian et al., 1997, 1998; Lal, 2004; Janzen et al., 1998; Bondeau et al., 2007; Schimel et al., 2000; Huang et al., 2009). Crop net primary production (NPP) and soil organic carbon storage (SOC) are two important variables, which are closely related to food production and C sequestration capacity in agricultural ecosystems. Therefore, in order to assess the role of agricultural ecosystems in global food production and the carbon cycle in response to multiple global environmental changes, it is critical to estimate the magnitude and variations of crop NPP and SOC.

Many environmental factors such as climate change, atmospheric CO_2 concentration, O_3, nitrogen deposition, land cover/use and land management can influence NPP and SOC in an intricate way, because environmental stresses usually interact with each other to produce combined impacts on ecosystem functioning rather than operate independently (Schindler, 2001). For example, global warming combined with adequate rainfall could lead to carbon accumulation in the vegetation C pool through enhanced photosynthesis (Cao and Woodward, 1998), or more carbon release from soil C pool through an elevated soil respiration rate (Melillo et al, 1993; Tian et al., 1998b; Granthe and Nalder, 2000; Vukicevic et al., 2001; Scheller and Mladenoff, 2005). Land cover /land use change (LCLUC) was recognized as the most important factor in enhancing or reducing the total crop NPP and SOC by expanding or decreasing the arable land area, and changing the physical and chemical soil characteristics, ecosystem structure and functioning, and microclimate (Houghton, 1995; Post and Kwon, 2000; Heal et al.,1997; Tate et al.,2000; Islam and Weil, 2000); this could result in high carbon sequestration or increased carbon release (Cole et al., 1995; Houghton, 2001; Li et al., 2002, 2005, 2006) when combined with varied agronomic practices. Nitrogen deposition/ fertilizer application could result in high NPP and C storage at the early stage when ecosystems are nitrogen limited (Melillo and Gosz,1983; Schindler and Bayley,1993;Gifford et al.,1996; Holland et al.,1997; Neff et al.,2000; Matson et al., 2002), but also a significant NPP reduction at the late stage when ecosystems are nitrogen saturated (Emmett et al.,1995; Magill et al.,2000). Similarly, elevated tropospheric O_3 concentrations could reduce crop yield and wood production through direct or indirect influences on photosynthesis and stomata conductance (e.g. Farage et al., 1991; Tjoekler

et al.,1995; Pell et al.,1997; Martin et al., 2000), or even reduce more NPP and C storage due to O_3 pollution in combination with extreme climate conditions such as drought or intensive management such as excessive fertilizer application (Ollinger et al., 2002; Felzer et al., 2005; Ren et al., 2007b). Therefore, it is important to conduct a comprehensive assessment of multiple environmental changes on NPP and C storage and explore the underlying mechanisms.

Although substantive work has been conducted to investigate the regional NPP and C storage and their responses to environmental changes (e.g. Melillo et al., 1993; Tian et al., 1998a, b, 2003; Houghton et al. 1999; Running et al., 2000, 2004; Cao et al., 2003; Fang et al., 2006; Felzer et al., 2004, 2005), most of those studies focused on natural ecosystems such as forest and grasslands but oversimplified or totally ignored agricultural ecosystems. Several recent studies have attempted to investigate the carbon dynamics in agricultural ecosystems at multiple regional scales using inventory, remote sensing and ecological models (e.g. Schimel et al., 2000; Li et al., 2003; Tao et al., 2004, 2008, 2009; Huang et al., 2006, 2007; Zhang et al., 2007; Wu et al., 2007; Bondeau et al., 2007). However, few studies have attributed the relative contributions of multiple environmental changes to NPP and SOC in agricultural ecosystems. Such work cannot be easily conducted easily using controlled experiments, survey, and statistical models due to the spatial heterogeneity, complex structure and feedbacks of terrestrial ecosystems to the multiple environmental stresses (Ollinger et al., 2002). Integrated process-based ecosystem models, which include the biogeochemical and physiological responses of ecosystems to atmospheric and climate changes, have been proved to be a powerful tool

in such multiple stress studies especially over large regions (Tian et al. 1998, 2003, 2008; Karnosky et al., 2005). Using ecosystem models to carry on the integrated study, plenty of time-series regional datasets that represent the historical information are needed and the mechanisms of important components and processes in response to environmental factors need to be coupled into ecosystem models. For example, Huang et al. (2007; 2009) and Bondeau et al. (2007)'s recent studies promoted the development of agroecosystem models; however, no work has been conducted on regional assessment of crop NPP and SOC in response to multiple stresses at the country or global level.

Agriculture in China plays a key role in both food security and global carbon cycling. With 10% of the world's arable land, China has to support 20% of the world's total population (NBS, 2005). Over the past decades, China's terrestrial ecosystems have been affected by a complex set of changes in climate, the chemical compositions of the atmosphere (e.g. O_3 pollution, nitrogen deposition), and land-use and land-cover. As one of the most intensively studied regions in the world, therefore, the substantive previous studies benefit help provide a regional database and improve our understanding of the influence mechanisms of changing environmental factors (Tian et al., 2003; Chen et al. 2006; Liu et al. 2005 a, b; Yan et al., 2007; Lu and Tian, 2007; Felzer et al., 2005). The data from these previous studies can be used as model inputs, and for model validation.

In this study, a new agricultural version of Dynamic Land Ecosystem Model (DLEM-Ag) was applied to investigate the effects of multiple environmental factors on NPP and SOC in croplands in China. DLEM-Ag, coupling the processes of crop growth,

hydrological and biogeochemical cycles in agricultural ecosystems, is integrated into

terrestrial ecosystems through the process of land use change. Not only is DLEM-Ag

driven by natural environmental factors (climate change, atmospheric CO_2, O_3, nitrogen

deposition) similar to those affecting natural functional types (grasslands and forests), but

unlike the DLEM version, it has been improved to address the effects of many agronomic

practices compared to the DLEM version (Tian et al., 2005; Ren et al., 2007b).

The specific objectives were: 1) estimate the magnitude of crop NPP and SOC in

croplands of China; 2) analyze the spatial and temporal patterns of crop NPP and SOC in

China's agricultural ecosystems; 3) attribute the relative role of environmental factors to

crop NPP and SOC; and 4) indentify the major uncertainties associated with this study.

Through this study, we intend to provide an integrated assessment of how multiple

environmental factors have affected cropland NPP and soil carbon storage in China

during the past half century.

2. Materials and methods

The basic framework of Dynamic Land Ecosystem Model (DLEM) has been

described in detail in Chapter 3. For the study of agricultural ecosystems in response to

multiple environmental stresses, an agricultural model of DLEM was developed and the

newly database related agronomic practices were build up.

2.1. Introduction of the Dynamic Land Ecosystem Model and its agricultural module

The agricultural module of DLEM (DLEM-Ag) was built on the well-established

DLEM (Figure 6-1) (Tian et al., 2005; 2008). By tightly coupling crop growth with soil
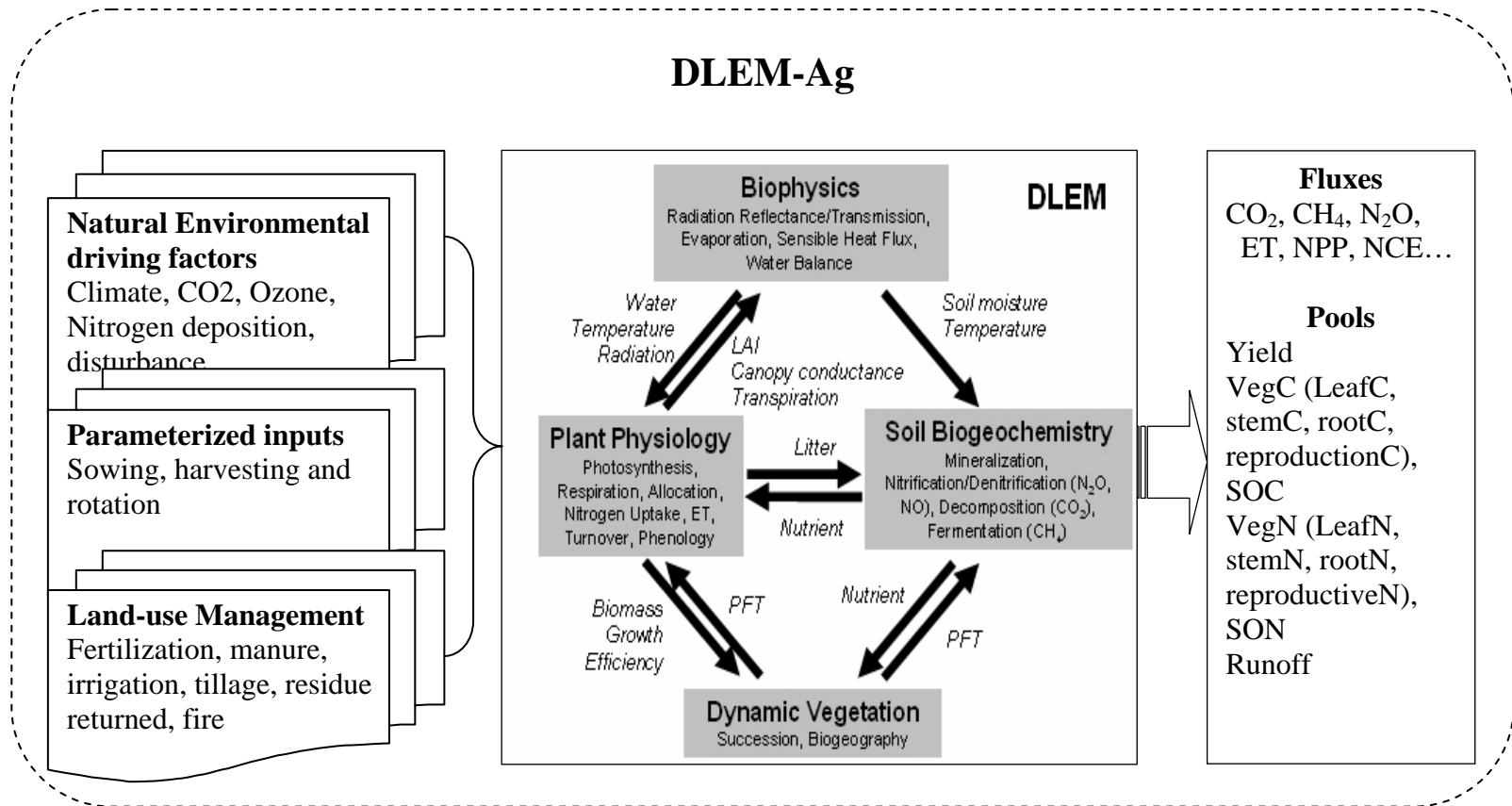
Figure 6-1 Framework of the Agricultural module of the Dynamic Land Ecosystem Model (DLEM-Ag) includes four parts: 1) specific parameterized inputs 2) process-based biophysics, plant physiology and soil biogeochemistry, 3) natural environmental driving and 4) human management

biogeochemical processes and considering the changing environmental factors (climate, air pollution, disturbance etc.) and land management, the model is able to simulate carbon, nitrogen, and water cycles in an agricultural ecosystem as a relatively independent complete system but in an integrative terrestrial ecosystem. Though the processes of growth (e.g. photosynthesis, respiration, allocation) and soil biogeochemistry (e.g. decomposition, nitrification, fermentation) are simulated similarly to the natural functional types in DLEM with daily time-steps, all crops in DLEM-Ag are parameterized specifically according to each crop type. In addition to natural environmental driving factors, differing from natural functional types, the impact of agronomic practice on crops growth and soil biogeochemical cycles in cropland are simulated as the main controllers in DLEM-Ag, including irrigation, fertilization application, tillage, genetic improvement, rotation and so on.

Crop parameters and crop systems. The model structure is suitable for all crops and cropping systems. In this study, we will focus on six major agronomic crops in China representing dry farmland and paddy land, C_3 plants and C_4 plants including irrigated and nonirrigated corn, wheat, barley, soybean and rice, with three major cropping systems including single cropping system, double cropping system (corn-wheat; rice-rice), and triple cropping system (rice-rice-rice). The main crop categories in each grid were identified according to the global crop geographic distribution map with a spatial resolution of 5 minutes based on the work of Leff et al. (2004), modified with regional agricultural census data that are available for every country in FAOSTAT and local agricultural census data such as the Chinese Academy of agricultural sciences

(http://www.caas.net.cn). The rotation type in each grid was developed by using phenology characteristics and census data at the state and national level (Figure 6-3).

Phenology. The phenology information obtained from MODIS LAI (with a spatial resolution of 1km) was used to help identify the rotation type and was also calibrated by using census data and site data before application. For our study at the site level, we simulate phenological development based on thermal time (Richie, 1991; Jones and Kiniry, 1986) like CERES: DR=Dtt/Pi, where the daily development rate (Ddt) is determined by the total thermal time needed (Pi) for completing a given stage i and actual daily thermal time is Dtt. Dtt is calculated based on canopy temperature and temperature parameters for each crop type. For our historical synthesis study at regional level, we simulated crop growth according to the phenological development information, which was developed based on substantive observation in hundreds of agriculture meteorological stations and observation by remote sensing (Yan et al., 2006). The role of remote sensing in phenological studies is increasingly regarded as the key to understanding seasonal phenomena over a large area. Phenologic metrics, including the start of season, end of season, duration of season, and seasonally integrated greenness, could be obtained from MODIS time series data and Advanced Very High resolution Radiometer (AVHRR), which is very useful for studying the historical patterns at a regional level (Yu et al., 2005; Reed, 2006). We have developed the gridded phenology database according to LAI information from MODIS, which was validated and modified by field observations in different study areas. This database has been incorporated into studies on the Southeast US, North American continent and Asian area (Tian et al., 2009

112

a,b). In this study, due to substantial observation data from China's agriculture meteorological stations, we aggregated the inventory data to develop the phenology for each cropping system.

Agronomic practices. In this study, the major farming practices were identified and developed according to the available data sets. Fertilizer application in China increased dramatically in the 1970s, which stimulated crop yield increase and changed the soil biogeochemical cycles. The historical fertilizer information was build up based on the survey database at the county and province level. A current irrigation map was developed, and in this study, it was assumed that the soil moisture would arrive at field capacity when irrigated and that irrigation would automatically occur when the soil moisture reaches the wilting point in the identified irrigated grids. (Detail in the description of input data section).The cropping systems in China are very important and complex that directly influence estimations of crop production; therefore in this study, the rotation map was developed to catch the current spatial pattern though it is unable to represent historical change. In addition, this study considered the residue and root conversion to SOC after harvesting.

2.2. Input data

The development of input data sets is of critical importance for regional assessment. At a minimum, five types of data sets are needed (Table 6-1 and Figure 6-2): 1) dynamic crop distribution map, 2) topography and soil properties (elevation, slope, and aspect; pH, bulk density, depth to bedrock, soil texture represented as the percentage content of loam, sand and silt), 3) climate and atmospheric chemistry (e.g. surface O_3,

atmospheric CO_2, and nitrogen deposition), 4) agronomic practices (fertilization, irrigation, harvest, rotation, residue treatment).

Table 6-1 Input data description including time step, unit and reference information

Input data	Time	Unit	Description
Climate data	Daily	T: °C/d; PPT: mm/d	1961-2005
LUCC	Annual	0/1 value	1961-2005
O_3	Daily	AOT40: ppb.h	1961-2005
Nitrogen deposition	Annual	Kg N/ha/yr	1961-2005
CO_2	Annual	ppm/yr	1961-2005
Fertilizer	Daily	gN/m^2	1961-2005
Irrigation	Daily	mm/m^2	1961-2005
Cropping map	Annual	1-13 value	
Other dataset	Soil map, geophysical database		

Note: T means temperature including maximum, minimum and average temperature; PPT is precipitation; other dataset include soil information, vegetation

Elevation, slope, and aspect maps are derived from 1 km resolution digital elevation datasets of China (http://www.wdc.cn/wdcdrre). Soil data was derived from the 1:1 million soil maps based on the second national soil survey of China (Wang et al. 2003; Shi et al., 2004; Zhang et al. 2005; Tian et al. 2006). Daily climate data (maximum, minimum, and average temperature, precipitation, and relative humidity) were developed by the Ecosystem Dynamics and Global Ecology (EDGE) Laboratory based on seven hundred and forty six climate stations in China and 29 stations from surrounding countries were used to produce daily climate data for the time period from 1961 to 2000, using an interpolation method similar to that used by Thornton et al. (1997). Historical
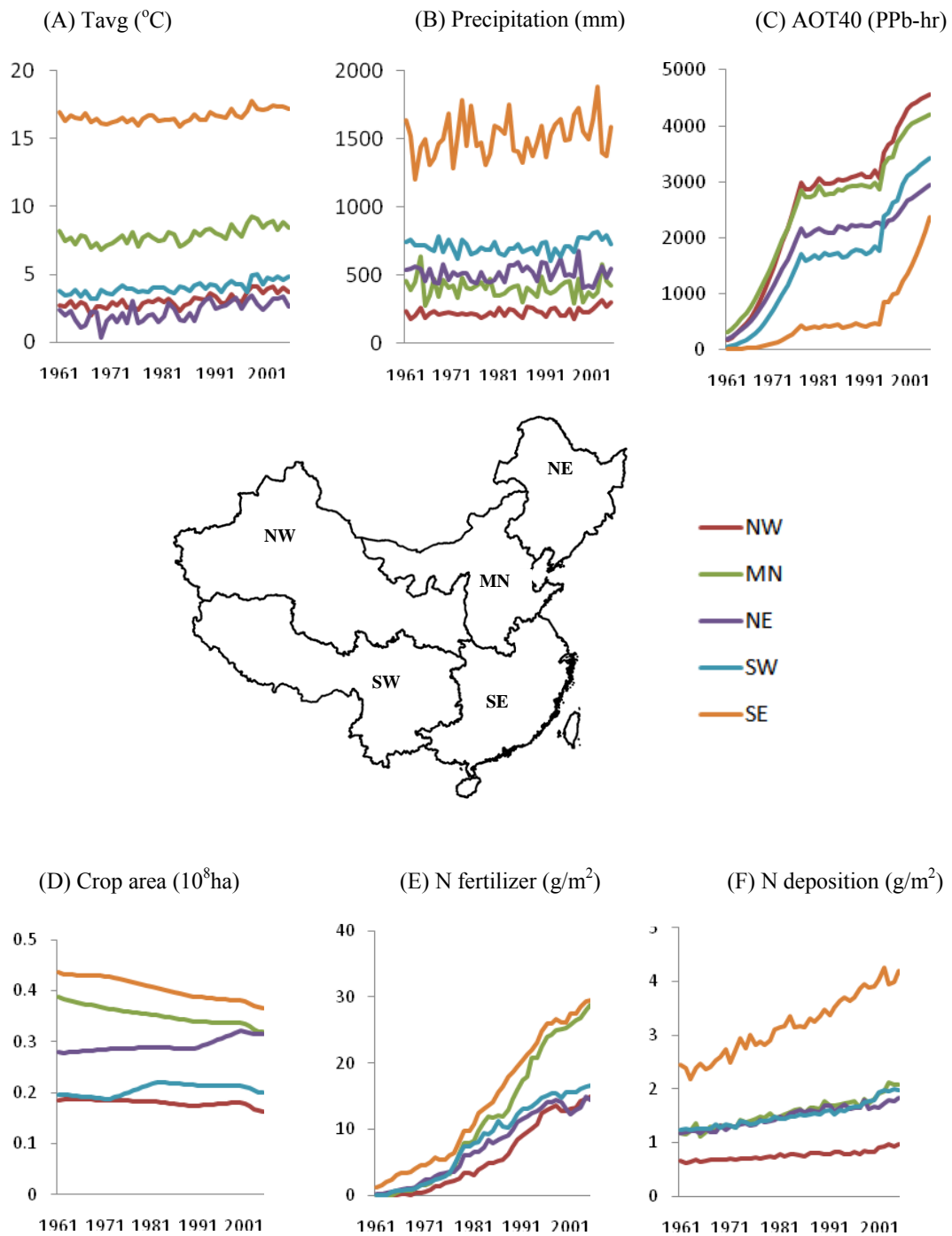
Figure 6-2 Changes in multiple environmental stresses including annual mean average temperature (A), precipitation (B), AOT40 (C), total cropland area (D), nitrogen fertilizer application (E), and nitrogen deposition (F) in five regions of China during the period 1961-2005.
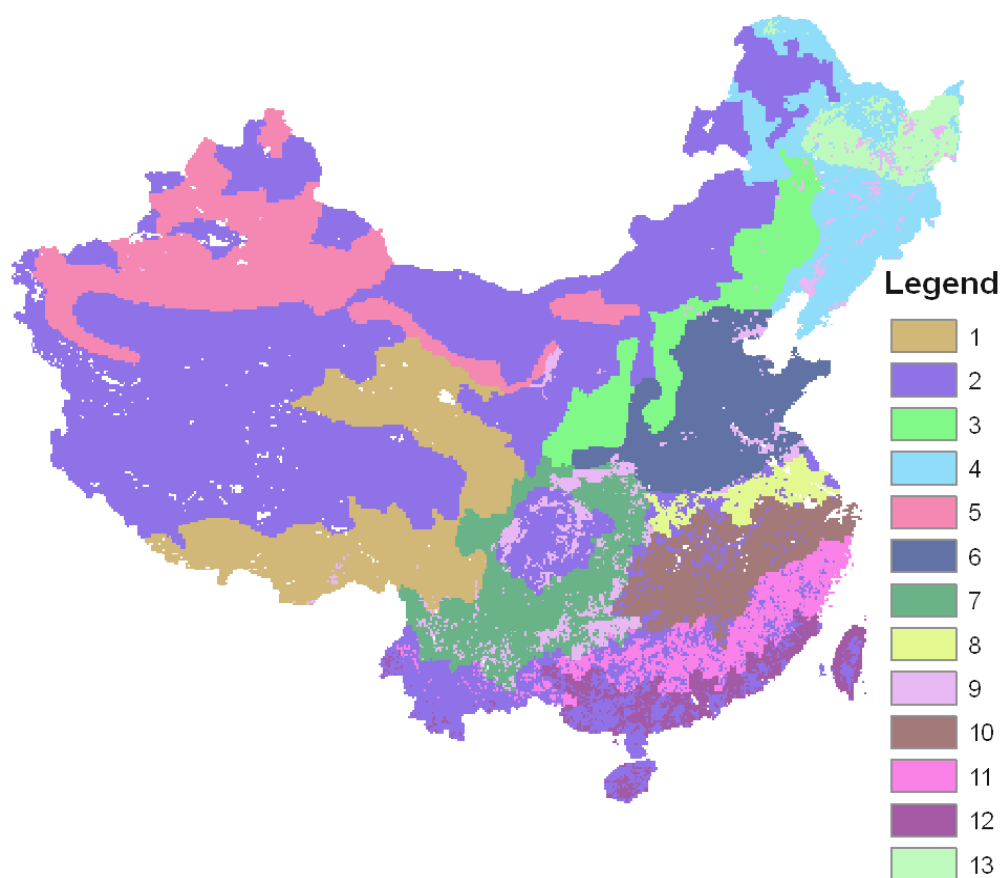
Number	Name of agricultural regions	Cropping System	Crop Types
1	Tibet Region	Single	Spring wheat
2	North High Plateaus	Single	Barley
3	North Low Plateaus	Single	Spring wheat
4	Northeast Region	Single	Corn
5	Northwest Region	Single	Spring wheat
6	Huang-huai-hai Region	Double	Corn/ winter wheat
7	Southwest Region	Double	Corn/ winter wheat
8	Jiang-huai Region	Double	Rice
9	Sichuan Basin Region	Double	Rice
10	Yangtze Lower-middle Region	Double	Rice
11	Southeast Region	Double	Rice
12	South Region	Triple	Rice
13	Northeast Region	Single	Soybean

Figure 6-3 Cropping systems in China.

CO_2 concentration dataset is from standard IPCC (Intergovernmental Panel on Climate Change) (Enting et al. 1994).The AOT40 dataset was derived from the global historical AOT40 datasets constructed by Felzer et al (2005) and was described in detail in the study by Ren et al. (2007a, b). Nitrogen deposition database was developed by Ecosystem Dynamic and Global Ecology (EDGE) Laboratory (Lu and Tian, 2007).

A crop distribution map was derived from the 2000 land use map of China (LCLUC_2000) and was developed using Landsat Enhanced Thematic Mapper (ETM) imagery (Liu et al., 2005a); our potential crop distribution map was constructed by replacing the croplands of LCLUC 2000 with potential vegetation as given in global potential vegetation maps developed by Ramankutty and Foley (1998); long-term land-use history (cropland and urban distribution in China from 1661-2000) was developed based on three recent (1990, 1995 and 2000) land-cover maps (Liu et al., 2003, 2005a, 2005b) and historical census datasets of China (Ge et al., 2003; Xu, 1983). Based on county-level census irrigation data (1990), nitrogen fertilization data from 1981 to 2005, and the provincial tabular data for 1950-2005 from the National Bureau of Statistics (NBS), we constructed a historical nitrogen dataset. For the irrigation strategy, because we lacked a solid database, we assumed that soil moisture arrives at field capacity when irrigated and irrigation automatically occurs when soil moisture reaches the wilting point in the identified irrigated grid. All datasets have a spatial resolution of 10km×10km, and climate and AOT40 datasets were developed on a daily time step and CO_2 and land-use datasets on a yearly time step.

117

2.3. Experimental design

We designed a series of eleven experiments to analyze the relative contribution of each environmental factor to the crop NPP and SOC in agricultural ecosystems in China (Table 6-2). In simulations one to six, we tried to capture both the direct effects of an environmental factor and its interactive effects with other environmental factors on NPP and SOC in croplands. We conducted five additional simulations, seven to eleven, to test the sensitivity of each factor, and allowed a particular environmental factor to change over time while we held the other environmental factors constant at the initial levels.

Table 6-2 Simulations including climate, carbon dioxide (CO_2), nitrogen deposition (NDEP), ozone (O_3), land-cover and land-use (LUCC)

	Scenarios	Climate	CO_2	NDEP	O_3	LCLUC
	Equilibrium	C	C	C	C	C
1	Combined	H	H	H	H	H
2	Com-Climate	C	H	H	H	H
3	Com-CO_2	H	C	H	H	H
4	Com-O_3	H	H	C	H	H
5	Com-Ndep	H	H	H	C	H
6	Com-LCLUC	H	H	H	H	C
7	Climate only	H	C	C	C	C
8	CO_2 only	C	H	C	C	C
9	N deposition only	C	C	H	C	C
10	Ozone only	C	C	C	H	C
11	LCLUC only	C	C	C	C	H

Note: H means historical data are used and C means the variable(s) is set as a constant.

The model was run using a daily time scale to simulate crop development and growth. For the model run at the regional level, the model simulation began with an equilibrium run to develop the baseline carbon, nitrogen, and water pools for each grid with a maximum tolerance of 20,000 years. A spin-up of about 100 years was applied if the climate change was included in the simulation scenario, and a spin-up of about 1,000 years was used if no irrigation was applied in the simulation scenario. Finally, for the above eleven experiments, the model was run in transient driven by the daily or/and annual input data.

3. Results

3.1. Spatial and temporal variations in NPP and carbon storage

Both NPP and carbon storage in vegetation and soil carbon pools increased from 1961 to 2000, and leveled off or even slightly decreased during the latest five years 2001-2005 in China's agricultural ecosystems (Table 6-3 and Figures 6-4 & 5). The decadal mean of annual NPP increased from 0.42 Pg C/yr in the 1960s to 0.98 Pg C/yr in the 1990s. Annual NPP showed a large increase rate between the 1960s and 1980s and the highest NPP was found in the 1990s (Table 6-3, Figure 6-4C).

Regional analysis showed large spatial and temporal variations in NPP. Annual NPP in five regions increased over a 45 year period of the simulation. The range varied from 0.06 Pg C/yr (8% of total NPP) in Northwest (NW) to 0.26 Pg C/yr (36.3%) in the Southeast (SE) China. The high NPP increase occurred in the SW followed by the MN

Table 6-3 Decadal mean carbon flux and carbon pools (Pg C/yr）

	NPP	Veg C	SOC	Total C
1960s	0.418	0.30	5.63	5.93
1970s	0.599	0.45	6.08	6.53
1980s	0.876	0.65	6.42	7.07
1990s	0.978	0.74	6.87	7.61
00-05	0.961	0.70	7.08	7.78
45y-average	0.745	0.57	5.96	6.53
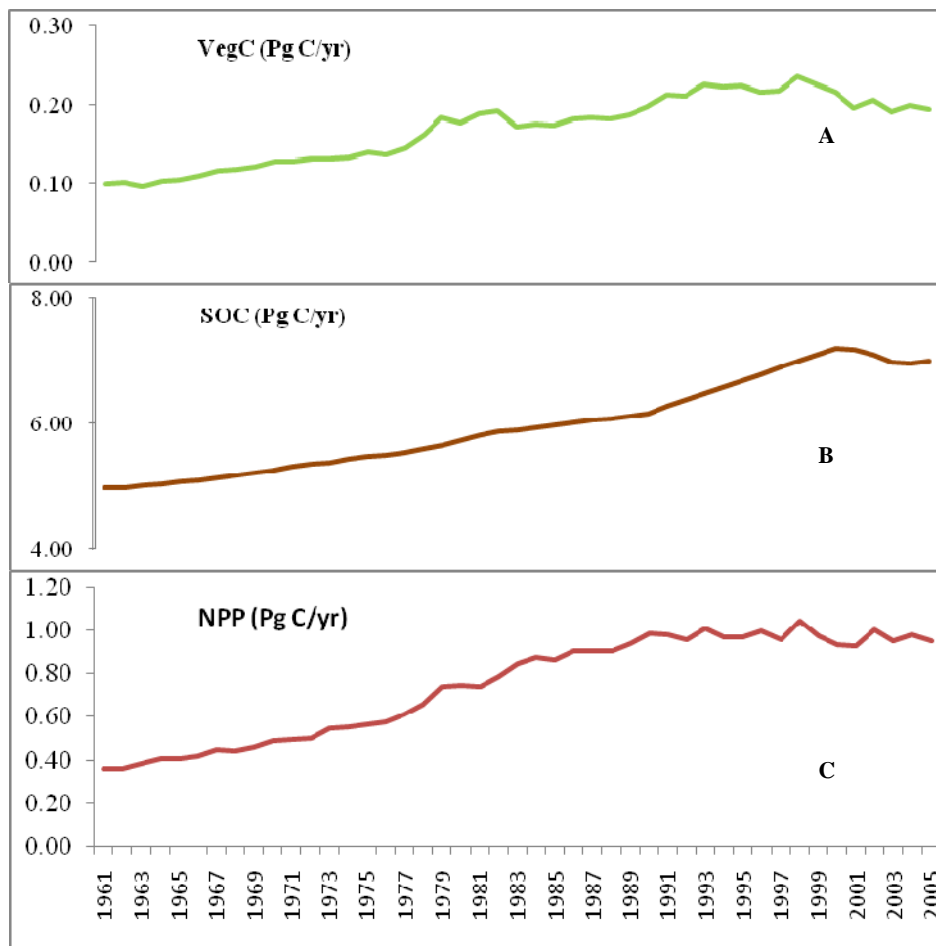
Figure 6-4 Annual changes of different carbon pools in China's agriculture
ecosystems during 1961-2005.

(Table 6-4). However, the spatial distribution of the NPP difference between the 1990s and 1960s showed that NPP decreased in some places in the MN that experienced frequent drought and high air pollution (Figure 6-5).

Carbon storage in the vegetation pool increased from 0.30 Pg C/yr in the 1960s to 0.74 Pg C/yr in the 1990s, with a slightly decrease during 2001-2005 (Table 6-2). Annual vegetation carbon increased with high increase rates in the late 1970s and the late 1990s (Figure 6-4A). Among the five regions in China, the SE had the largest vegetation carbon storage with an average of 0.2 Pg C·yr^{-1} (35%) between 1961 and 2005, followed by MN with 0.12 Pg C/yr (20.3%). Vegetation carbon reached the peak value in all regions in the 1990s. When compared to that in the 1960s, the total vegetation carbon in the 1990s increased by 1.27 times in the NW, 1.92 times in the MN, 1.18 times in the NE, 2.33 times in the SW, and 1.08 times in the SE. Most likely, the changes in the total vegetation carbon were primarily caused by the changes in crop land areas in 1990s (Table 6-3).

Soil carbon storage continued to increase during 1961-2005, ranging from 5.63 Pg C/yr to 7.08 Pg C/yr. Accordingly, the total carbon storage in agriculture ecosystems increased from 5.93 Pg C/yr in 1961 to 7.78 Pg C/yr in 2005 (Table 6-2). Unlike natural ecosystems, carbon sequestration in agriculture ecosystems mainly occurred in soil carbon pool due to annual vegetation harvest. Increased vegetation carbon could also provide more litter fall as input into the soil carbon pool, which could result finally in more soil carbon storage. Soil carbon storage in the NE was highest among the five regions, accounting for 35% of the total soil carbon storage in China's croplands.

Percentage of SOM Change between 1990s and1960s
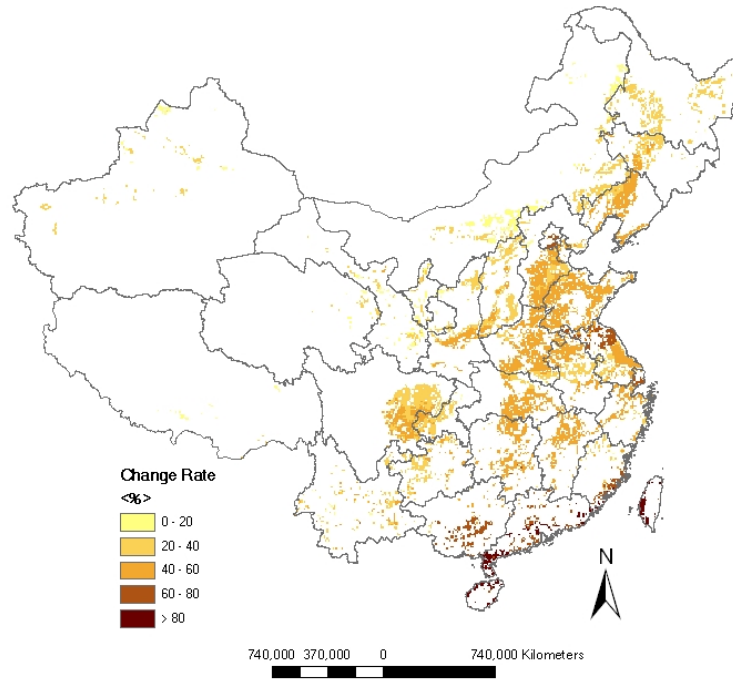
Percentage of NPP Change between 1990s and1960s
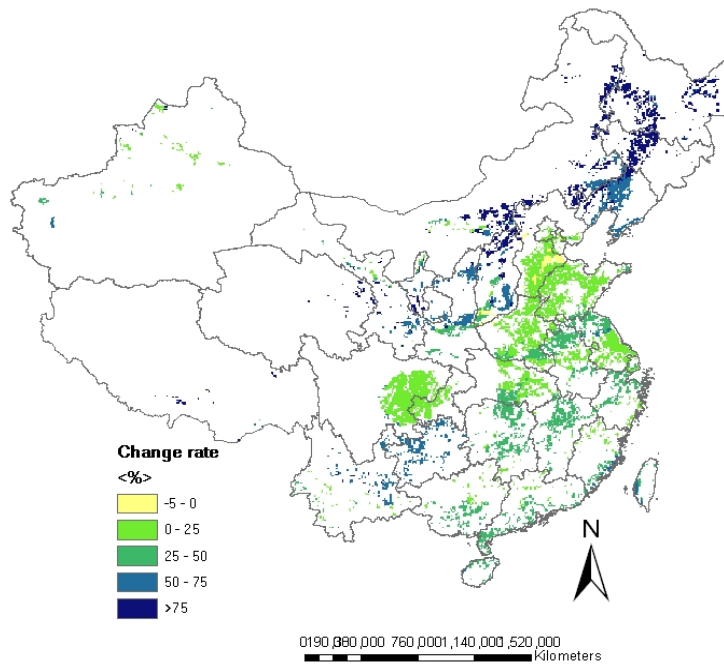
Figure 6-5 Change rates in net primary productivity (NPP) and soil organic carbon (SOC) between 1990s and 1960s

Table 6-4　Decadal mean carbon flux and carbon pools in five regions

	NPP				
	NW	MN	NE	SW	SE
1960s	0.034	0.092	0.064	0.056	0.160
1970s	0.044	0.139	0.092	0.086	0.220
1980s	0.064	0.220	0.112	0.147	0.307
1990s	0.077	0.242	0.133	0.164	0.335
2000-2005	0.079	0.242	0.140	0.156	0.319
Average(%)	0.06(8.0)	0.18(25.0)	0.10 (14.5)	0.12(16.3)	0.26 (36.3)
(90s-60s)%	126.5	163.0	107.8	192.9	109.4
	Vegetation Carbon				
	NW	MN	NE	SW	SE
1960s	0.030	0.052	0.062	0.039	0.121
1970s	0.039	0.086	0.087	0.073	0.166
1980s	0.056	0.138	0.103	0.122	0.230
1990s	0.068	0.152	0.135	0.130	0.252
2000-2005	0.069	0.150	0.127	0.119	0.236
Average(%)	0.05(9.2%)	0.12(20.3)	0.10(18.0)	0.10(17.0)	0.20(35.4)
(90s-60s)%	126.7	192.3	117.7	233.3	108.3
	Soil Organic Carbon				
	NW	MN	NE	SW	SE
1960s	0.66	1.00	1.67	0.62	1.13
1970s	0.66	0.99	1.92	0.67	1.25
1980s	0.68	1.06	2.08	0.85	1.33
1990s	0.72	1.18	2.41	0.94	1.47
2000-2005	0.69	1.22	2.61	0.97	1.54
Average(%)	0.68(11.3)	1.09(18.0)	2.14(35.2)	0.81(13.4)	1.35(22.2)
(90s-60s)%	9.1	18.0	44.3	51.6	30.1

The SE and MN accounted for 22.2% and 18.0% of total soil carbon storage, respectively. During the past 45 years, carbon storage in soils continued to increase in four regions in each decade, ranging from 0.99 - 1.22 Pg C in the MN, 1.67-2.61Pg C in the NE, 0.62-0.97 Pg C in the SW, and 1.13-1.54 Pg C in the NE, respectively. However, the total soil carbon storage in the NW has slightly decreased since 2000.

3.2 Environmental controls on NPP

The simulated total crop NPP under multiple environmental changes significantly increased from 1961 to 2005. Among the five environmental factors investigated in this study, land cover and land use change (LCLUC) accounted for 80.4% of the total NPP increase over a 45 year period (Table 6-4). CO_2 fertilization was the second important factor that contributed an additional 16.3% of the total NPP increase, while nitrogen deposition (NDEP) accounted for 14.3%.   Both tropospheric O_3 pollution and climate variability/change decreased NPP by ~9.1% and ~2.0%, respectively. Change in annual NPP was related to all these environmental factors. LCLUC, CO_2 and NDEP had uniformly positive effects on NPP increase while O_3 pollution had an increasingly negative effect on NPP. Climate variability/change caused annual NPP variation due largely to the change in annual precipitation. LCLUC had smaller contribution to NPP increase than NDEP in the 1960s, but a much higher contribution since 1970s (Figure 6-6).

3.3. Environmental controls on soil carbon storage

Similar to total crop NPP, the simulated total SOC was influenced by the combined effects of multiple stresses. LCLUC, CO_2 fertilization, NDEP accounted for 84.3%, 17.5%, and 12.1% of the total soil carbon storage increases over the 45 years, respectively. The tropospheric O_3 pollution and climate variability/change decreased carbon storage by ~9.3% and ~4.7%, respectively (Table 6-5). CO_2 fertilization and NDEP had positive effects on annual total SOC increase while O3 pollution and climate variability/change had negative effects on it. Unlike the effects of LCLUC on NPP which enhanced NPP continuously, LCLUC decreased soil carbon storage until the late 1980s

124

Table 6-5 The relative contribution of each factor to the total increase of net primary production (NPP) and soil carbon storage (SOC) estimated by DLEM-Ag

		Climate	CO2	LCLUC	Ndep	O3
NPP (%)	1960s	-3.6	12.0	36.0	57.8	-2.1
	1990s	-3.0	20.5	89.9	4.2	-11.6
	45-average	-2.0	16.3	80.4	14.3	-9.1
SOC	1960s	-20.2	4.1	-127.0	43.4	-0.2
	1990s	-17.9	26.1	46.3	55.5	-10.0
	45-average	-4.7	17.5	84.3	12.1	-9.3

Note: LCLUC - land cover and land use change; Ndep - nitrogen deposition.
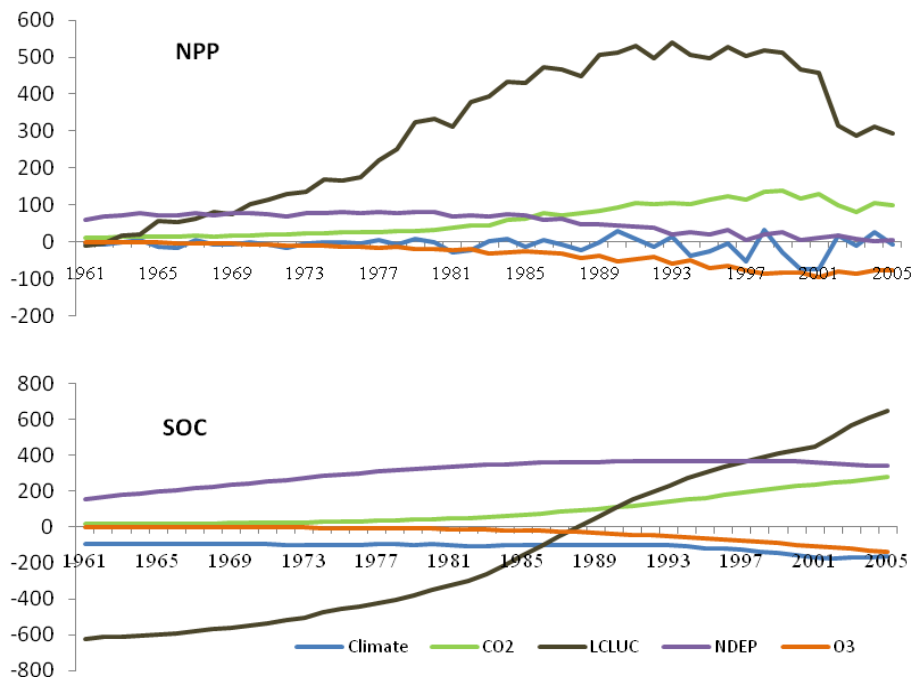
Figure 6-6 Changes in (A) net primary productivity and (B) soil carbon storage (Tg C) resulted from multiple factors including climate, CO_2, O_3, Nitrogen deposition, Land-cover and land-use change (LCLUC)

and enhanced more soil carbon sequestration since then (Figure 6-6). Among the effects of LCLUC which included land conversion and land management (e.g. nitrogen fertilizer and irrigation), nitrogen fertilizer increased soil carbon storage in China's croplands about 1.5 Pg C between 1961 and 2005, while land conversion resulted in 0.2 Pg C loss mainly due to the decreasing cropland area at the national level (Liu et al., 2006; Yang et al., 2007; Ge et al., 2008; Wang et al., 2008).

4. Discussion

4.1 Estimation of NPP and SOC in China's croplands

Carbon in agricultural ecosystems is the most active of global carbon pools due to the tremendous influences of human activities. It was reported that carbon sequestration in global agricultural soils could reach 40-80 P g C in the coming 50-100 years (Cole, et al., 1996; Paustaian et al.1998). The simulated SOC in China's croplands was estimated as 7.08 P g C in recent five years 2001-2005, which accounted for 8%-17.7% of the global carbon sequestration in agricultural soils. Furthermore, the simulated continually increasing SOC since 1961 indicates that China's croplands have large potential for soil carbon sequestration. The simulated NPP in China's croplands also increased since 1961, indicating increased vegetation carbon storage and hence enhanced soil carbon storage (Table 6-2).

4.2 Contributions of multiple environmental changes to NPP and SOC

Land cover and land use change. LCLUC investigated in this study included land conversion and land management (e.g. irrigation, fertilizer application). We found that LCLUC was the dominant factor controlling the temporal and spatial variations of NPP
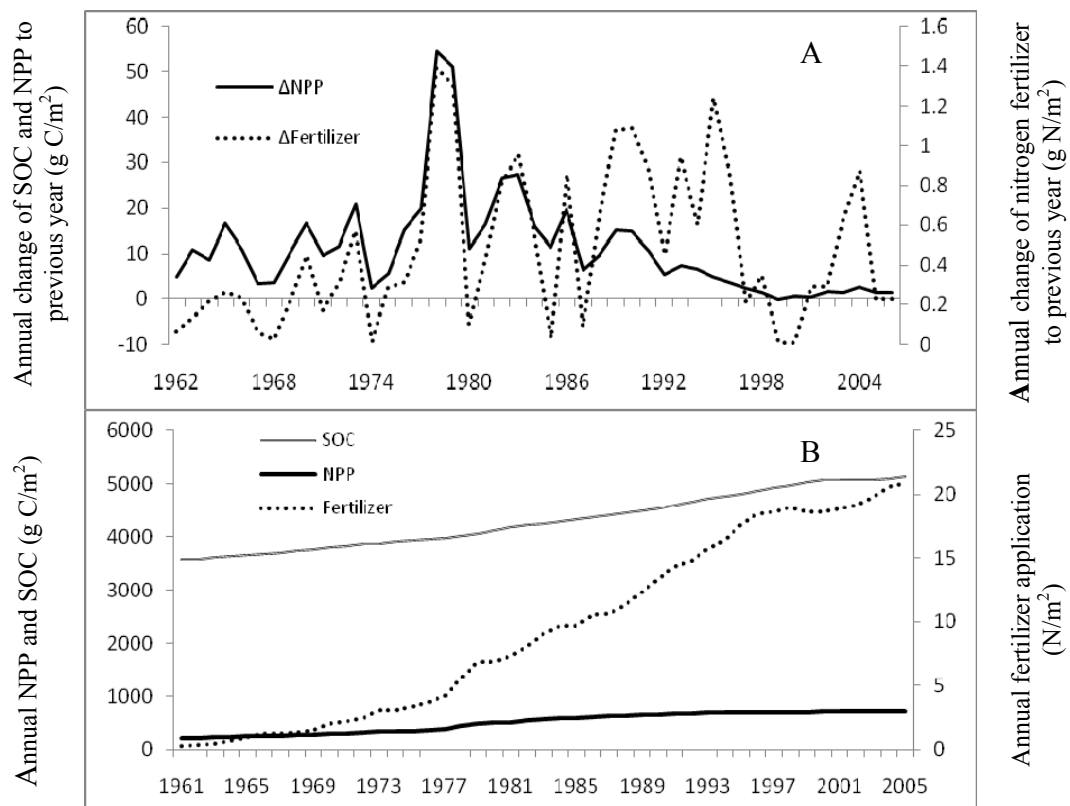
Figure 6-7 (A) Annual changes to previous year in net primary productivity (NPP) and nitrogen fertilizer application; (B) Annual soil carbon storage (SOC) and NPP (gC/m^2) in China's croplands from 1961 to 2005.

and SOC in China's croplands during 1961-2005, and accounting for more than 80% of the total changes in both NPP and SOC. Similar to previous studies, we found that land conversion could lead to an increase in total NPP and SOC when land cover changed from natural vegetation into cropland (e.g. Davidson and Ackerman, 1993; Murty et al.,2002; Guo and Gifford, 2002; Houghton and Goodale, 2004), and also could result in a deduction in total NPP and SOC when cropland area decreased. Further analysis indicated that small changes in interannual variation of crop NPP were caused by fertilizer application since the late 1980s (Figure 6-7A), which indicates that agricultural

ecosystems in China may have possibly reached nitrogen satuation (Tian et al., 2009a,b), even though both the unit NPP and soil carbon density continually increased (Figure 6-7B) thanks to optimized land management (Cole et al., 1996; Yan et al., 2009; Huang et al., 2006, 2007).

4.3 Other environmental factors

Also in this study, we considered climate variability/change, atmospheric CO_2 and nitrogen deposition, and tropospheric O_3 pollution. The combined contributions of these changes to total NPP and SOC were equivalent to about 20% during 1961-2005. Both NPP and SOC show substantial interannual variation in response to climate variability, similar to the study by Tian et al. (1999). The direct positive effects of increasing CO_2 / nitrogen deposition (e.g. Tian et al., 1998; Melillo and Gosz, 1983; Schindler and Bayley, 1993; Holland et al., 1997; Neff et al., 2000) and the negative effects of elevated O_3 (Heagle, 1989; Felzer et al., 2004; Ren et al., 2007a) on NPP and SOC were consistent with previous studies. Their influences on NPP and SOC in the future could also be accelerated with increasing temperature (e.g. Tao et al., 2009), elevated CO_2, and increasing O_3 pollution (e.g. Felzer et al., 2005).

4.4. Interactive effects of multiple factors

We conducted sensitivity experiments on NPP and SOC and found that the effect of single factor could be enhanced or weakened and even the direction of the response could be reversed when it interacted with other environmental factors (Figure 6-8). For example, the effects of elevated CO_2 and nitrogen input, well recongnized as fertilizers to increase carbon storage capacity, were enhanced by each other (e.g. Reich et al., 2006).
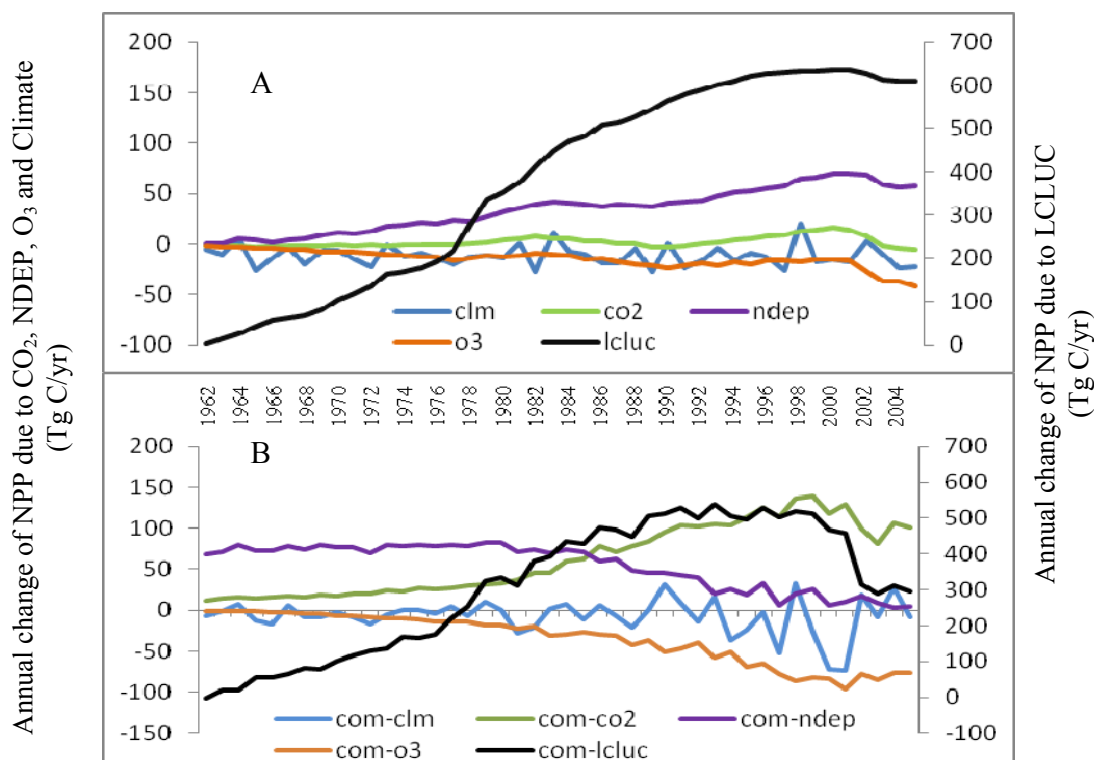
Figure 6-8 Annual changes in net primary productivity (NPP) (Tg C/yr) due to the effects of single factor and the effects of single factor plus the interactive effects with other factors.

Note: clm, co2, lcluc, ndep and o3 mean the effects of climate, CO_2, land use cover and land use change, nitrogen deposition and O_3, respectively; com- stands for the combined effects of single factor plus the interactive effects with other factors.

On everage, increasing nitrogen fertilizer enhanced the CO_2 fertilization effect on carbon storage by 10 times in China's croplands. However, on average, the negative effects of elevated O_3 on crop growth were doubled when combined with drought and nitrogen fertilizer (Felzer et al., 2005; Ren et al., 2007b). The overall effects of elevated O_3, increasing CO_2 and nitrogen fertilizer on ecosystem production and carbon storage are complex effects that are related to the photosynthesis process, stomatal conductance and are dependent on nutrient and water conditions. The current version of DLEM-Ag

integrates all these processes, but does not include the direct effects of O_3 pollution on stomata conductance and uses the AOT40 index rather than O_3 flux data, which might underestimate the effects of O_3 on NPP and SOC. In spite of uncertainties in this study ude to current limited conditions, the primary results highlight that it is crucial to consider the combined effects of multiple stresses and explore the underlying mechanistic links in order to better understand the role of agriculture ecosystems in the global carbon budget.

5, Conclusion

We quantified the spatial and temporal variations of NPP and SOC in China's croplands in response to multiple environmental stresses during 1961-2005 using the DLEM-Ag model. A newly developed historical information database of land cover and land use change (LCLUC), atmosphere CO_2, nitrogen deposition, tropospheric O_3 and climate variability in China was used to drive the model run. The simulation results indicated that both the total NPP and SOC in China's croplands increased by 130% and 26%, respectively, from 1961-2005. The relative contributions of main environmental factors to the total increase in NPP and SOC were very consistent. LCLUC was the dominant factor to sequestrate carbon in cropland soils and enhance the crop production, accounting for more than 80% of total contributions to the changes in NPP and SOC. Fertilizer application led to a 1.5 Pg carbon storage increase in soils. Contribution of nitrogen fertilizer application to NPP and SOC gradually decreased, suggesting a saturation condition where fertilizer application was beyond crop demands in croplands after the 1970s. This implies that redundant nitrogen fertilizer will lead to negative consequences such as water pollution, soil acidification, increasing N_2O emission and air

pollution. Both elevated O_3 and climate change/variability reduced total NPP and SOC in the last half 20th century In general, DLEM-Ag has the ability to simulate the variations of agricultural carbon fluxes and pools in response to multiple historical global change factors. The estimations of crop NPP and C storage in China during 1961-2005 and their attribution analysis are the most groundbreaking part of this study, at this required applying the agricultural ecosystem model to a historical study considering multifactor interactions on the country level. To improve assessment accuracy, key processes in response to main environmental changes need to be combined into the DLEM-Ag model, such as increasing aerosol/O_3 and their effects on photosynthesis and stomata conductance. Additionally, well-established database, such as GHGs emissions monitoring and O_3 only/multiple-response derived from field controlled experiments in China, are very important for model calibration and evaluation.

Chapter 7

Model Validation and Uncertainty Analysis –

Model Comparisons with Field Measurements, Remote Sensing

Observations, and Other Models

Abstract

To examine model behavior and analyze the potential uncertainties in this study, in this chapter we present the following: 1) sensitivity analysis that tests model behavior in response to changing environmental factors; 2) model validation at site and regional levels; 3) a comparison of model results with other methods. Preliminary results indicate that our DLEM model has the ability to capture the response of carbon fluxes and pools to environmental changes such as elevated ozone (O_3) concentration and climate variability (precipitation and temperature). The simulated key variables (net primary productivity-NPP and net carbon exchange-NCE) are close to site level observations and the regional estimations are comparable to other studies. We found that the uncertainties in this study mainly concerned the use of different methods; parameters and input data also contributed to errors in the context of a limited database. While this analysis focuses on agricultural ecosystems and their response to O_3 exposure, future work will provide an assessment of grassland and forest ecosystems in China and model responses to other environmental factors.

Keywords: Agricultural ecosystems, CO_2 flux, Flux tower, Model validation, Ozone (O_3), Remote Sensing, Survey data, Terrestrial Ecosystem Model (TEM), Uncertainty analysis

1. Introduction

As this study demonstrates, it is important to investigate regional-scale ecosystem responses to changing environmental conditions, such as elevated O_3 concentration and dynamic climate change, both as a scientific question and as the basis for making policy decisions. Another critical question that could be raised at the same time, is, how much confidence we should have in regional model results? In general, uncertainty in modeling is caused by hypotheses and mathematical formulation, parameterization process (e.g. photosynthesis, respiration) and environmental model driving variables (e.g. vegetation type, climate, soil texture) (Haefner, 2005). Th sources of uncertainty which relate to DLEM model have been evaluated in previous work (Tian et al., 2005; Chen et al., 2006; Ren et al.,2007a,b; Zhang et al.,2007; Liu et al.,2008). For regional ecosystem modeling, however, it is important to evaluate how well the predictions agree with observed data for the region. Several studies indicate that it is possible to conduct research to assess the accuracy of regional model forecasts for terrestrial carbon cycling, with the contributions of multidisciplinary development such as remote sensing observation and eddy flux tower monitoring (e.g. Nemani et al.2003; Running et al., 2004; Heinsch et al., 2006; Scurlock et al., 1999; Rahman et al., 2001; Zheng et al., 2003; Xiao et al., 2004; Turner et al., 2005; Sims et al., 2006). The improvement of point-scale measurements and regional-scale model outputs derived from high-resolution satellite data suggest that the outputs from regional and global models agreed well with measurements undertaken at flux tower sites. However, further study is needed to validate modeled results at different scales due to the

133

gap in spatial scale between point-scale flux measurements and regional-scale model analyses (Sasai et al., 2007).

This analysis aims to compare model estimations of carbon fluxes among multi-scale independent model analyses with ground measurements at the site level, satellite-observation data at the regional level, and inventory data at the biome level. After investigating model sensitivities in response to main environmental factors (more specifically on dynamic O_3 concentration variations), the study conducted multi-scale comparisons of modeled carbon fluxes (CO_2, CH_4, net primary productivity-NPP) and pools (soil organic carbon-SOC), focusing on agricultural ecosystems.

2. Materials and methods

To conduct model validation and uncertainty analysis, this study used all available databases, including field measurements, survey records, remote sensing observations, ecosystem modeling, and published results in scientific journals.

2.1 The Dynamic Land Ecosystem Model (DLEM) and input data

The same method was used as described in detail in Chapter 3 and Chapter 6.

2.2 Field observation

Observation data from more than thirty agrometeorology observation stations and agroecosystem synthetic observation stations in China was collected for model calibration and phenology parameterization (Table 1, appendix). An independent dataset was collected for model validation including two sites: one dry farmland (rotation of

134

winter wheat and summer maize in Yucheng) and one rice paddy field (three harvested

rice in Qingyuan) for the model validation. Site data was retrieved from the regional

dataset for model run if the inputs at these sites were unavailable.

2.3 Survey-based data

Survey data on crop yield from 1961 to 2003 from the National Agriculture

Database (China Agriculture Yearbook, 2003) was collected to estimate the crop yield

carbon at the province and biome levels. Datasets of crop NPP and soil carbon organic at

the regional and biome level were derived from other literature reviews (Huang et al.,

2006, 2007; Zhang et al., Wu et al., 2007).

2.4 Remote sensing

We compared our simulated crop NPP with a production efficiency model,

GLO-PEM (Prince and Goward, 1995; Goetz et al. 2000; Cao et al. 2004), which has a

spatial 8km resolution and runs at a 10-day time step. GLO-PEM was driven almost

entirely by satellite-derived variables, including both the Normalized Difference

Vegetation index (NDVI) and meteorological variables. We overlaid the GLO-PEM NPP

images with the yearly cropland cover data that we developed and extracted from the

GLO-PEM crop NPP in ArcInfo 9.2. Similarly, we derived the MOD17 MODIS NPP in

China's croplands from 2002 to 2005 (Running et al. 2004; Heinsch et al. 2003).

2.5 The biogeochemistry model TEM

The Terrestrial Ecosystem Model (TEM) is a process-based biogeochemical

model that uses spatially referenced information on climate, elevation, soils and

vegetation to make monthly estimates of important carbon and nitrogen fluxes and pool sizes. In TEM, the net carbon exchange between the terrestrial biosphere and the atmosphere is represented by net ecosystem production (NEP), which is calculated as the difference between net primary production (NPP) and heterotrophic respiration (RH). Net primary production is calculated as the difference between gross primary production (GPP) and plant respiration (RA). Gross primary production represents the uptake of atmospheric CO_2 during photosynthesis and is influenced by light availability, atmospheric CO_2 concentration, temperature and the availability of water and nitrogen. Plant respiration includes both maintenance and construction respiration, and is calculated as a function of temperature and vegetation carbon. The flux RH represents microbially mediated decomposition of organic matter in an ecosystem and is influenced by the amount of reactive soil organic carbon, temperature and soil moisture. The annual NEP of an ecosystem is equivalent to its net carbon storage for the year. The TEM can be used either in equilibrium mode (McGuire et al., 1992, 1995, 1997; Melillo et al., 1993; VEMAP Members, 1995) or transient mode (Melillo et al., 1996; Tian et al., 1998a, 1999; Xiao et al., 1998; Kicklighter et al., 1999; Felzer et al., 2004).

3. Experimental design

To perform the general responses of the DLEM model to multiple stresses, several simulation experiments were first designed to simulate the annual crop yield under the scenarios of O_3 only, climate only, and fertilizer only effects. To examine the sensitivities of the model to O_3 pollution in particular, sensitivity simulations were conducted using different levels of the O_3 (AOT40: accumulated O_3 exposure over a threshold of 40 parts per billion). To validate the model, the model was run to compare

CO_2 flux in dry farmland and CH_4 flux in rice paddy fields at the site level, and also to compare the regional simulations in chapter 6 using survey-based and remote sensing databases. To identify the responses of different models to O_3 exposure, the simulations were conducted and analyzed using the DLEM and TEM models driven by the same input data.

4. Results and discussion

4.1 Sensitivity analysis
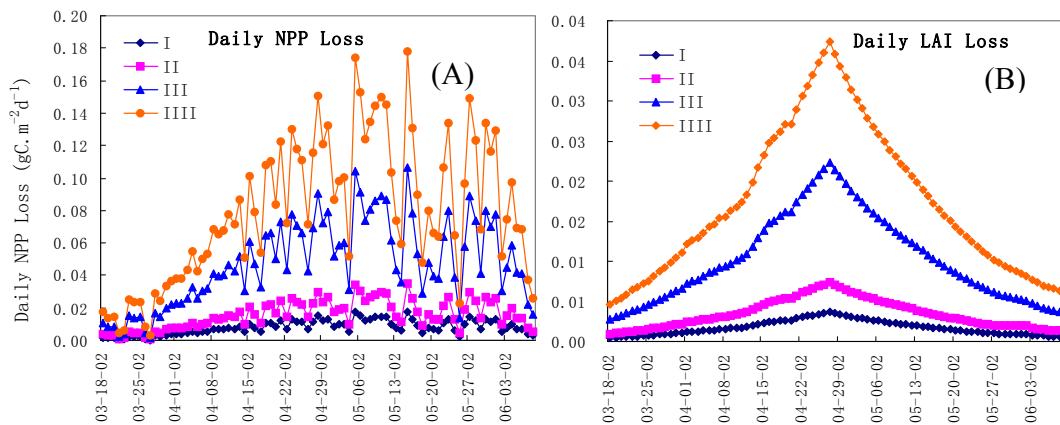
Figure 7-1 Daily reductions of (A) net primary productivity (NPP) and (B) leaf area index (LAI) in response to four different ozone treatments (AOT40: I500, II1000, III3000, IV5000), background data in Yucheng Integrated Agricultural Experimental Station (116.6º, 36.7º)

The sensitivity experiments were conducted under three scenarios in which only O_3, climate and fertilizer changed and all other factors were kept constant. Then the annual mean crop yields of dry farmland (one-harvest wheat in Gansu province) and rice paddy fields (three-harvest rice in Zhejiang province) were calculated.
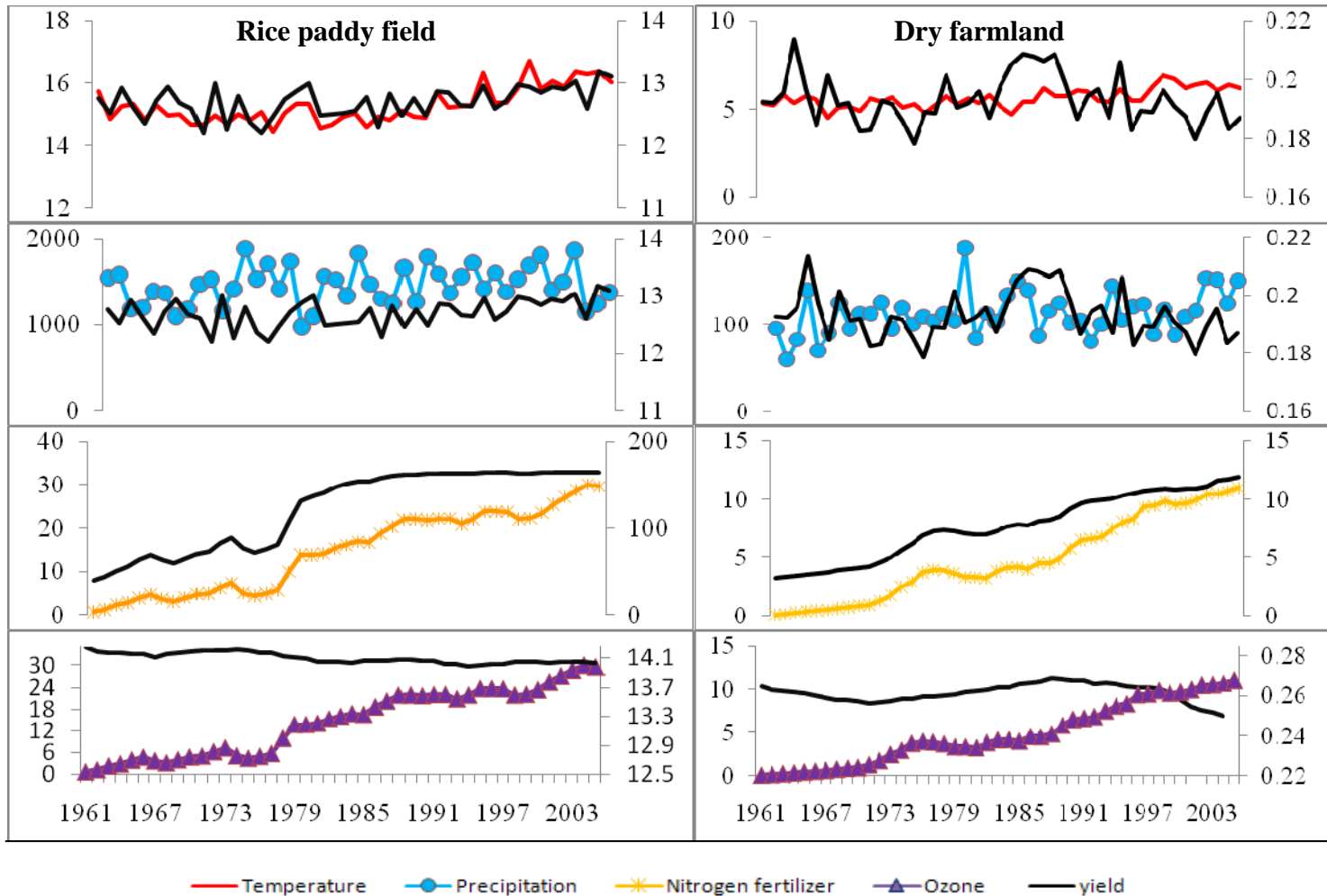
Figure 7-2 Responses of annual crop yield to the changes in annual temperature (^{o}C), precipitation (mm), fertilizer (g N/m^{2}) and O$_3$ (ppm-hr) in dry farmland and rice paddy field. (Note: left axis represents environmental factors and the right axis is the changes in crop yield carbon, g C/m^{2}/yr).
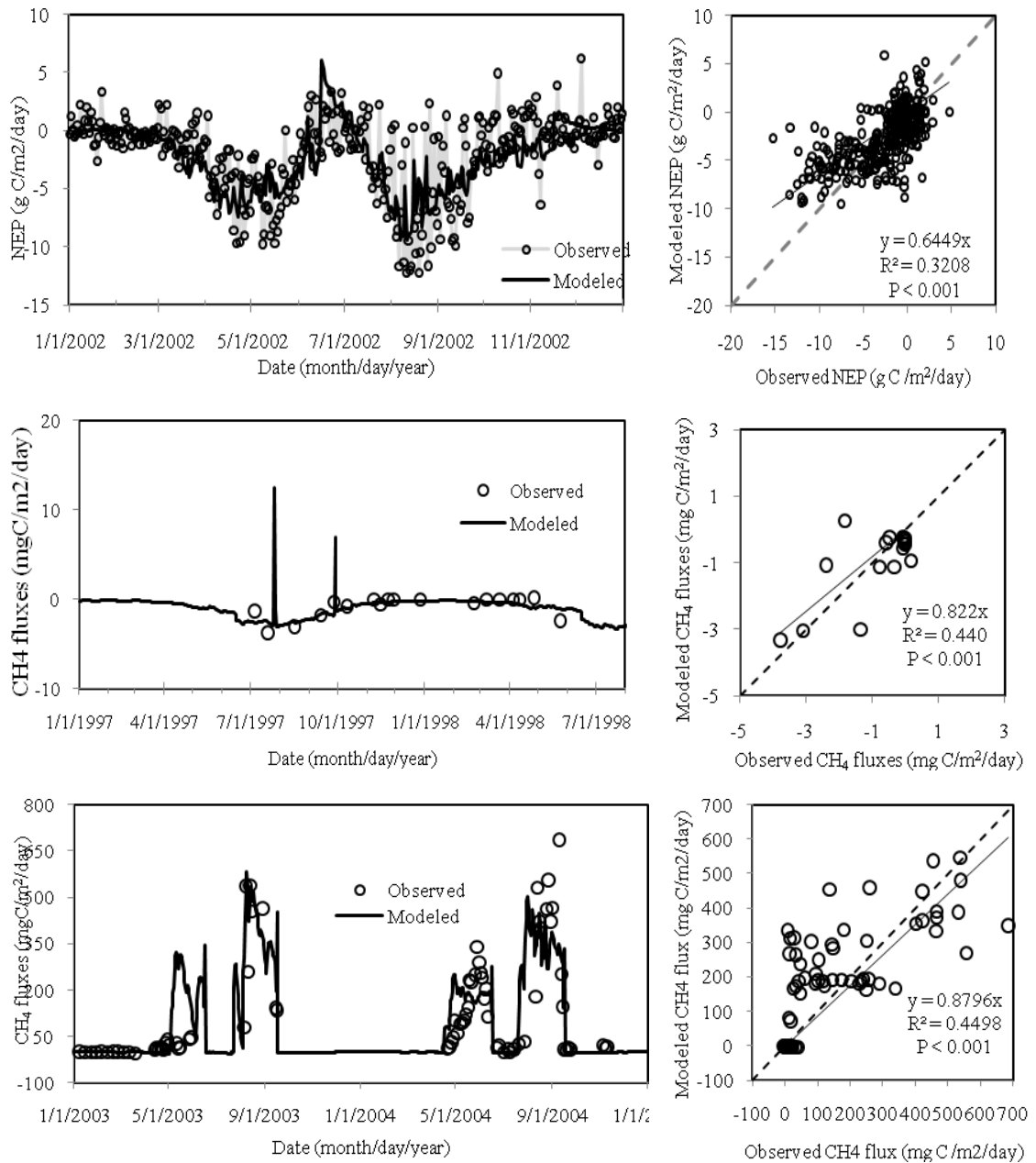
Figure 7-3 Comparison of DLEM-estimated CO_2 and CH_4 fluxes with field observations (CO_2 flux (a) and CH_4 flux (b) in Yucheng Station (116° E, 36° N); (c) CH_4 flux in Qingyuan (112° E, 23° N); the regression models validations are: (d) Observed = 1.551 * modeled; r = 0.476; P < 0.001 for CO_2 flux; (e) Observed = 1.216 * modeled; r = 0.440 in Yucheng; P < 0.001 for CH_4 flux in Yucheng)

Note: in (a) positive values indicate CO_2 emission and negative values indicate CO_2 uptake.

139

The results (Figure 7-2) indicate that harvested crop carbon was more sensitive to precipitation in dry farmland while more sensitive to temperature in rice paddy; crop yields increased with increasing fertilizer inputs but leveled off since the 1980s; crop yields reduced with elevated O_3 concentrations. These results are consistent with observations and other studies (e.g. Heagle et al., 1989; Tian et al., 1998; Tao et al., 2005; Huang et al., 2007). Further sensitivity analysis (Figure 7-1) showed that the continuous reductions of daily NPP and LAI were due to ideally designed increasing AOT40 levels, the underlying mechanisms of which were derived from and similar to other studies (e.g. Martin et al., 2000; Ollinger et al., 2002; Felzer et al., 2004). However, the potential uncertainties or even errors could be due to a lack of knowledge on quantifying relationships between continuously increasing O_3 concentration and plant adaptation to its exposure.

4.2 Model validations and comparisons

Two sites, including one dry farmland (rotation of winter wheat and summer maize in Yucheng) and one rice paddy field (two crops of rice Qingyuan) were selected for model validations (Figure 7-3a-f). We retrieved the site data from our regional dataset for the model run because the input data for these sites were unavailable to us. The simulated daily CO_2 flux (NEP) and CH_4 fluxes were consistent with the observational data for dry cropland in Yucheng (Figure 7-3a-d) and the rice paddy in Qingyun (Figure 7-3e, f). For CO_2 flux in the dry cropland in Yucheng, DLEM captured seasonal patterns of daily flux, but missed some pulses. Overall, the modeled annual CO_2 flux was quite close to observed NEP, -827 g C m^{-2} vs. -722 g C m^{-2}. A comparison of modeled CH_4 fluxes with observed CH_4 fluxes in dry cropland (Figure 7-3c, d) and rice paddy (Figure

7-3e, f) demonstrated the DLEM's ability to capture not only seasonal patterns, but also the absolute values of CH_4 fluxes. However, two pulses of CH_4 flux were simulated in DLEM because of extremely high precipitation in two different time periods. Further investigation showed that the first peak in CH_4 emission was caused by a two-day strong precipitation event with a total rainfall of 69.3mm, and the second peak of CH_4 emission was associated with a strong precipitation event of 60.4 mm per day. It should be noted that the annual precipitation for Yucheng station in 1997 was 574 mm. DLEM also simulated the seasonal pattern of CH_4 fluxes from a rice paddy field in the Qingyun, Southern China (Khalil et al., 2007).

NPP and SOC simulated by DLEM-Ag were comparable with survey data and satellite products (Table 7-1 & Figure 7-4). The estimations of soil C storage by the DLEM-AG were also comparable to other studies (Table 7-1), although few of these studies were conducted at the national level or for a long historical period. It was found that C storage in the soils across China's croplands increased from 1961 to 2005. The estimations of 16 Tg Cyr^{-1} for the top 20 centimeters across China's croplands and 11.5 Tg C yr^{-1} for the rice paddy field were comparable to Huang and Sun's survey estimation of 18 - 22 Pg Cyr^{-1} (Huang and Sun, 2006) and Zhang et al.'s simulated estimation of 4.0-11.0 Tg C yr^{-1} (Zhang et al., 2007) between 1980 and 2000, respectively.

Table 7-1 Comparisons of soil carbon change and net primary productivity (NPP) at the national level and different cropping systems between 1961 and 2000 among ecosystem modeling, inventory estimate

	Method	Period	Other study	This study
Huang et al., 2006	Inventory	1980-2000	Increased soil organic carbon (SOC) on national scale on the top soil (0.2 m)	
			0.018-0.022 Pg C/yr	0.016 Pg C/yr
Zhang et al., 2007	Modeled	1980-2000	Increased soil organic carbon (SOC) in rice paddy field	
			0.15±0.07 Pg C	0.23Pg C
Wu et al., 2007	Inventory	1979-1985	Soil organic carbon storage (SOC) in top soil layer (1m)	
			4.4 Pg C at national level 1.6 Pg C in rice paddy land 2.8 Pg C in dry farm land	4.8 Pg C at national level 1.2 Pg C in rice paddy land 3.6 Pg C in dry farm land
Huang et al., 2007	Inventory	1950-1999	Increased decadal mean annual Crop NPP in China (87% of total nation) between 1960s and 1990s	
			354±77 Tg C/yr	374 Tg C/yr
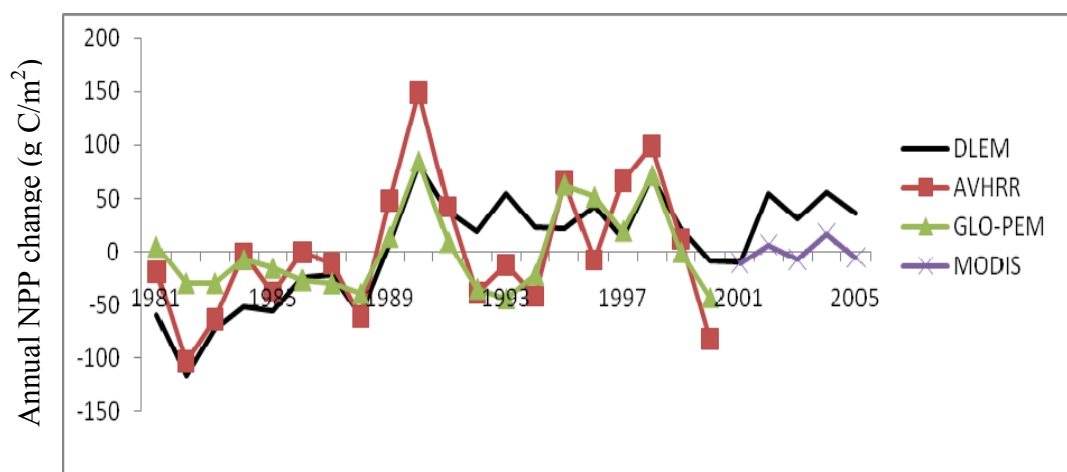
Figure 7-4 Changes in annual net primary production (NPP) (relative to the average for 1981-2005) of China's croplands estimated by DLEM-Ag model, GLO-PEM model, AVHRR, and MODIS database during 1981-2005.

The simulated crop NPP by DLEM was also compared with a production efficiency model, GLO-PEM. GLO-PEM has a spatial 8km resolution and runs at a 10-day time step). GLO-PEM was driven almost entirely from satellite-derived variables, including both the Normalized Difference Vegetation index (NDVI) and meteorological variables (Prince and Goward 1995; Goetz et al. 2000; Cao et al. 2004). The GLO-PEM NPP images were overlaid with the yearly cropland cover data that we developed and then the GLO-PEM crop NPP was extracted in ArcInfo 9.2. Similarly, the MOD17 NPP and AVHRR NPP in China's croplands from 2002 to 2005 was also derived (Running et al. 2004; Heinsch et al. 2003). The results (Figure 7-3) of annual NPP change showed that the simulated NPP by DLEM-Ag had a similar temporal pattern to that estimated by AVHRR, GLO-PEM and MODIS17, possibly because both model-based and remote sensing-based results were influenced or driven by the major environmental factor of climate change (e.g. Nemani et al., 2003).

4.3 Comparisons of two models

4.3.1 Differences between two models

The main discrepancies between those two models include (detail information in appendix I, Table 2,3): 1) temporal scale of O_3 effects (DLEM-daily; TEM-monthly); 2) effects of O_3 on ecosystem processes; 3) plant functional types (similar natural PFTs including forest and grass in three models; however, managed C_3 grass is simulated as crop in TEM, while there are managed main food crop types in DLEM ; 4) processes of land use and land cover change (one irrigation map was used in both TEM and DLEM; the date of C_3 harvest in TEM is based on degree-day simulation and prescribed no

143

fertilization limitation to occurs once the N fertilization was more than 100 kg N/ha, while in DLEM the harvest date and fertilization application are prescribed according to observation at agrometeorology stations (Table 1, appendix) and the Agricultural Almanac of China.

4.3.2 Simulated results comparisons

The simulated results showed that O_3 had uniform negative effects on NPP in China's terrestrial ecosystem in the past decades, as simulated by DLEM and TEM (Figure 7-5A). Both the TEM and DLEM results showed that the highest NPP reduction occurred in north China (Figure 7-5B), where most of the croplands had experienced high O_3 concentrations during growth and harvest seasons. Both models simulated a larger total NPP loss in dry farmland and deciduous forest ecosystems, which accounts for the increasing O_3 concentration in these areas (Table 7-2). Dry farmland lost more NPP due to a higher AOT40 increase and it was the most sensitive to O_3 pollution (mean -4.2Tg C / ppm-hr). Broadleaf forest was more sensitive (mean -2.4Tg C / ppm-hr) to O_3 pollution than needle leaf forest (mean -1.6Tg C / ppm-hr), despite less AOT40 increase and total NPP loss. Two models were able to capture the same temporal variations and spatial patterns of annual NPP in response to O_3 exposure, though the absolute values, which possibly were derived from the model parameterization and the inner model structures, varied greatly. For example, without calibration based on observations in China, TEM simulated a higher than actual biomass. Also, croplands in TEM were designated as C_3 grassland, which resulted in large discrepancies in estimations for croplands in both models.

Table 7-2 Changes of ozone pollution (AOT40) and the responses of total net primary productivity (NPP) loss and unit NPP loss to increased ozone (AOT40) between the 1960s and the1990s at biome level

Biome	ΔAOT40 (ppm-hr)	DLEM		TEM	
		Total loss (TgC)	Unit loss (TgC/AOT40)	Total loss (TgC)	Unit loss (TgC/AOT40)
NEF	1.6	0.0	0.0	-0.4	-0.2
BEF	1.9	-0.4	-0.2	-1.6	-1.2
NDF	1.4	-2.8	-1.3	-4.3	-1.9
BDF	1.0	-1.3	-2.0	-2.3	-2.8
Grass	2.1	-0.1	0.0	0.0	0.0
Dry farmland	1.8	-4.9	-2.6	-15.0	-5.8
Paddy fields	1.4	-1.9	-1.5	-6.0	-3.6
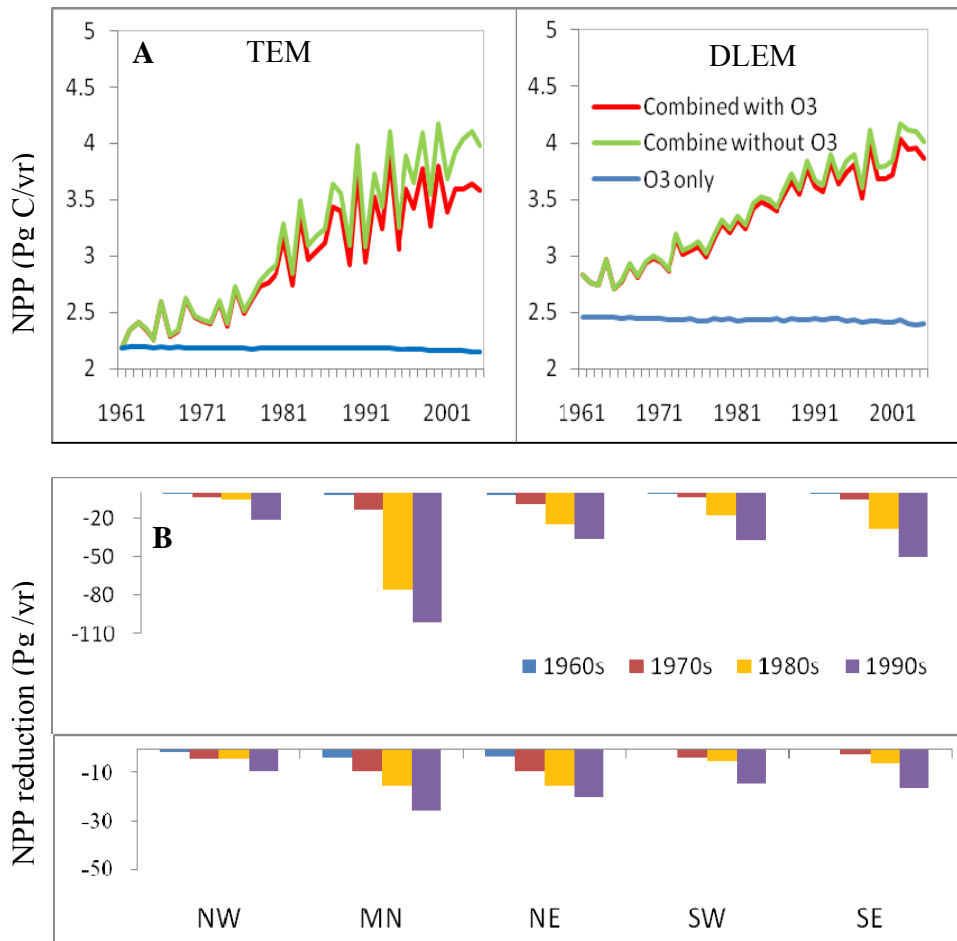
Figure 7-5 National estimations of annual NPP under three scenarios (A) and regional estimations of decadal mean annual NPP (B) simulated from the TEM and DLEM models.

145

5. Conclusions

In this study, sensitivity analysis, model results validations and comparisons all indicate that the DLEM model has the ability to capture variations in carbon fluxes and pools in response to multiple environmental changes, especially O_3 concentration changes. The simulated carbon fluxes (NPP, CO_2 flux, CH_4 flux) and pools (soil carbon storage) were comparable to observations at site level and surveys at the regional level. Discrepancies in the results of various studies are mainly due to input data (e.g. study area) and the gaps in different methods (e.g. model inner discrepancy). In the future, similar work needs to be conducted in grassland and forest ecosystems, and the uncertainty of absolute percentage changes in carbon flux and pools in response to O_3 pollution should be examined as the necessary database information becomes available. Also the model results need to be compared against results from other models (e.g. PnET).

Chapter 8

General Conclusion and Future Research Recommendation

It was suggested that China's terrestrial ecosystems have large potential capacity of carbon sequestration, however, over past decades China has been affected by a complex set of changes in climate, atmosphere chemical compositions (e.g. O_3 pollution), and land-use and land-cover (e.g. Chen et al.,2006; Liu et al., 2005a,b; Ren and Tian, 2007). To accurately assess the variations and the trends of carbon storage in response to the global change, both major environmental (e.g. climate change, land use/cover change) and the future factors (e.g. increasing tropospheric O_3, elevated nitrogen deposition) are necessary to be taken into account to reduce the uncertainties. Using the Dynamic Land Ecosystem Model (DLEM), the carbon storage of China's terrestrial ecosystems in the past half 20th century was assessed, focusing on its responses to elevated tropospheric O_3 concentration and historical climate change in the context of global change.

First, an overall assessment was conducted to investigate the influences of multiple stresses with or without O_3 pollution on the net primary productivity and carbon storage in terrestrial ecosystems of China: the general simulation showed that that elevated O_3 could result in a mean 4.5% in NPP and 0.9% reduction in total carbon

storage nationwide from 1961 to 2000. The reduction of carbon storage varied from 0.1 Tg C to 312 Tg C (a decreasing rate ranging from 0.2% to 6.9%) among plant functional types. Significant reductions in NPP occurred in northeastern and central China, where a large proportion of cropland is distributed. These simulation results suggest that 1) the adverse effects of elevated O_3 on carbon fluxes and pools are significant and cannot be ignored; 2) the responses to elevated O_3 varied among different ecosystems; 3) the O_3 effects on carbon storage are dependent upon other environmental factors, therefore, the effects of O_3 only and its interactive effects with other environmental factors should be considered to accurately assess the regional carbon budget in China.

In the following studies, specific analysis on the assessments of NPP and carbon storage in responses to O_3 pollution, climate change and other environmental factors among grassland, agricultural and forest ecosystems with the newly improved model and refined database based on the different features of each ecosystem were conducted. For grassland ecosystems, the results show that the combined effects including O_3 could potentially lead to an decrease of 14 Tg C in annual NPP and 0.11Pg C in total carbon storage in China's grasslands during 1961-2000 although it acted as a weak carbon sink in the same period. China's grassland ecosystems, mostly distributed in arid and semi arid regions with high O_3 concentrations, are more sensitive and vulnerable to O_3 pollution and climate change than other regions of grassland ecosystem, due to lack of the intensive human management in croplands and long-term adaptation to environmental stresses in forests. For forest ecosystems, the simulated results show that elevated O_3 could result in about a 0.2-1.6% reduction in total NPP and 3.5-12.6% reduction in

148

carbon storage nationwide from- 1961 to 2005. Changes in annual NPP and carbon storage across China's forestlands exhibited substantial spatial variability, and the reduction rates of NPP and carbon storage ranging from 0.1% to 2.6% and from 0.4% to 43.1% indicated varied sensitivity and vulnerability to elevated O_3 pollution among different forest types. For the agricultural ecosystems, the simulations of the multiple stresses' influences including O_3 pollution, climate change, and other stresses on crop NPP and soil carbon storage was conducted. The results suggested that both NPP and SOC in China's croplands increased from 1961 to 2005 with rates of 0.036 Pg C a^{-1} and 0.045 Pg C a^{-1}, respectively. However, the influences of increasing O_3 pollution and climate change both caused about ~9% reduction in NPP, and ~2% and ~5% soil carbon loss, respectively. And the sensitivity experiments showed that the single negative effects of O_3 pollution and climate change were exacerbated when combining the interactive effects with other factors.

Finally, model sensitivity analysis, validations and results comparisons were conducted at site and regional levels using field measurements, remote sensing and other ecosystem models, which indicated that DLEM model was able to capture the response of carbon cycle to O_3 pollution and climate variability/change; and the simulated results of DLEM are comparable to measurements and results from other studies. In the future work, to improve the accuracy of assessment and further understand the mechanisms of carbon cycle responses to O_3 pollution, climate change and other environmental factors, it is necessary to refine or develop new input data such as O_3 flux data instead of AOT40, and address the mechanisms of O_3 flux effects on key processes such as photosynthesis

and stomatal conductance rather than using dose-response relationship. In addition, re-calibration and validation of the DLEM model are necessary in order to reduce the prediction errors and to extrapolate the model to a broad domain.

This is the first reported study investigating historical (1960-2005) temporal and spatial changes of terrestrial carbon budgets across China in response to historical O_3 pollution, climate variability/change and other multiple environmental stresses. Though there is still uncertainty due to limited conditions such as database and field experiments, the main findings in this study suggest that improvement in air quality could significantly enhance the potential of carbon sequestration in China in addition to benefits to natural resource conservation and human health. Regarding future research, several aspects are important: 1) Air quality monitoring is needed, especially in rural or remote locations of grassland and forest; emissions data such as VOCs, CO and NO_x would be helpful. 2) Ecosystem models (e.g. DLEM-Ag) need to be improved through coupling the mechanisms of O_3 pollution and its interactive effects with other environmental factors on ecosystem functioning and processes dynamically among different crop types as season changes, such as O_3 effects on stomatal conductance and carbon allocation in different growth stage. 3) Further model validation should be conducted in collaboration with O_3-response field experiments. 4) In addition, it is meaningful to select O_3-resistant crops for adaption to global environmental change and to supply needed food.

References

Aber, J.S., Spellman, E.E., Webster, M.P. (1993). Landsat remote sensing of glacial terrain. In: Aber, J.S. (eds), Glaciotectonics and mapping of glacial deposits. Canadian Plains Research Center, Canadian Plains Proceedings, 25(1), pp. 215–225, Regina, Canada.

Adams, R.M., Glyer, D.J., Johnson, S.L., McCarl, B.A. (1989). A reassessment of the economic effects of ozone on United States agriculture. *Journal of Air Pollution Control Association*, 39, 960–968.

Adams, R.M., Hamilton, S.A., McCarl, B.A. (1986). The benefits of pollution control: the case of ozone and U.S. agriculture. *American Journal of Agricultural Economics*, 68, 886–893.

Agrawal, M. and Deepak, S.S. (2003). Physiological and biochemical responses of two cultivars of wheat to elevated levels of CO2 and SO2, singly and in combination. *Environmental Pollution*, 121, 189–197.

Akimoto, H. (2003). Global air quality and pollution. *Science*, 302, 1716–1719.

Allen, L.H., Albrecht, S.L., Colon–Guasp, W., Covell, S.A., Baker, J.T., Pan, D., Boote, K.J. (2003). Methane emissions of rice increased by elevated carbon dioxide and temperature. *Journal of Environmental Quality*, 32, 1978–1991.

Alvarado, R., de Bauer, L.I., Galindo, A.J. (1993). Decline of sacred fir (Abies religiosa) in a forest park south of México City. *Environmental Pollution*, 80,115–121.

Andersen C.P., Wilson R., Plocher M., Hogsett W.E. (1997). Carry-over effects of ozone on root growth and carbohydrate concentrations of ponderosa pine seedlings. *Tree Physiology*, 17, 805–811.

Ashmore, M.R. (2005). Assessing the future global impacts of ozone on vegetation. *Plant, Cell and Environment*, 28, 949–964.

Aunan, K., Berntsen, T.K., Seip, H.M. (2000). Surface ozone in China and its possible impact on agriculture crop yields. *Ambio*, 29, 294–301.

Bai Y.M., Wang, C.Y., Wen, M., Huang, H. (2003). Experimental study of single and interactive effects of double CO_2, O_3 concentrations on soybean. *Journal of Applied IED Meteorological Science*, 14(2), 245–251 (In Chinese with English Abstract).

Ball, J.T., Woodrow, I.E., Berry, J.A. (1987). A model predicting stomatal conductance and its contribution to the control of photosynthesis under different environmental conditions. In: Biggins, I. (Eds), Progress in Photosynthesis Research, Vol. IV. Proceedings of the International Congress on Photosynthesis. Martinus Nihjoff, Dordrecht, pp221–224.

Barnes, J.D., Wellburn, A.R. (1998). Air pollutant combinations. In: Kok de, L.J., Stuhlen, I. (Eds.), Responses of plant metabolism to air pollution and global change. Leiden: The Netherlands: Backhuys Publishing, pp147–164.

Bassin, S., Wolk, M., Fuhrer, J. (2006). Factors affecting the ozone sensitivity of temperate European grasslands: an overview. *Environmental Pollution,* 146(3), 678–691.

Bergin, M. H., Greenwald, R., Xu, J., Berta, Y., Chameides, W. L. (2001). Influence of aerosol dry deposition on photosynthetically active radiation available to plants: A case study in the Yangtze delta region of China. *Geophysical Research Letter,* 28(18), 3605– 3608.

Bobbink, R. (1998). Impacts of tropospheric ozone and airborne nitrogenous pollutants on natural and seminatural ecosystems: a commentary. *New Phytologist*, 139, 161–168.

Boisvenue, C., Running, S.W. (2006). Impacts of climate change on natural forest productivity - evidence since the middle of the 20th century. *Global Change Biology,* 12, 862–882.

Bonan, G. B., Pollard, D., Thompson, S. L. (1992). Effects of boreal forest vegetation on global climate. *Nature*, 359, 716–718.

Bondeau, A., Smith, P., Zaehle, S., Schaphoff, S., Lucht, W., Cramer, W., Gerten, D., Lotze–Campen, H., Müller, C., Reichstein, M., Smith, B. (2007). Modelling the role of agriculture for the 20th century global terrestrial carbon balance. *Global Change Biology*, 13, 679–706.

Braswell, B. H., Schimel, D. S., Linder, E., Moore, B. III. (1997). The response of global terrestrial ecosystems to interannual temperature variability. *Science,* 278, 870–872.

Brohan, P., Kennedy, J. J., Harris, I., Tett, S.F.B., Jones, P.D. (2006). Uncertainty estimates in regional and global observed temperature changes: a new dataset from 1850. *Journal of Geophysical Research*, 111, D12106, doi:10.1029/2005JD006548.

153

Brown, S.L., Schroeder, P.E., Kern, J.S. (1999). Spatial distribution of biomass in forests of the eastern USA. *Forest Ecology and Management,* 123: 81–90.

Caemmerer, S.V., Farquhar, G.D. (1981). Some relationships between the biochemistry of photosynthesis and the gas exchange of leaves. *Planta*, 153, 376–387.

Cao, M. K., Prince, S.D., Li, K.R., Tao, B., Small, J., Shao, X.M. (2003). Response of terrestrial carbon uptake to climate interannual variability in China. *Global Change Biology*, 9, 536–546.

Cao, M.K. and Woodward, F.I. (1998). Dynamic responses of terrestrial ecosystem carbon cycling to global climate change. *Nature*, 393, 249–252.

Cao, M.K., Prince, S. D., Small, J., Goetz, S. (2004). Satellite remotely sensed interannual variability in terrestrial net primary productivity from 1980 to 2000. *Ecosystems*, 7, 233–242, 2004.

Chameides, W., Li, X., Zhou, X., Luo, C., Kiang, C.S., John, J. St., Saylor, R.D., Liu, S.C., Lam, K.S. (1999). Is ozone pollution effecting crop yield in China? *Geophysical Research Letters*, 26, 867–870.

Chameides, W.L., Kasibhatla,P., Yienger, J., Levy, H. II. (1994). Growth of continental–scale metro–agro–plexes, regional ozone pollution and world good production. *Science*, 264, 74–77.

Chapin, F.S., Matson, P.A.III., Mooney, H.A. (2002). Principles of Terrestrial Ecosystem Ecology. Springer, New York.

Chappelka, A.H. (2002). Reproductive development of blackberry (Rubus cuneifolius) as influenced by ozone. *New Phytologist*, 155, 249–255.

Chappelka, A.H., Neufeld, H.S., Davison, A.W., Somers, G.L., Renfro, J.R. (2003).

Ozone injury on cutleaf coneflower (Rudbeckia laciniata) and crown–beard
(Verbesina occidentalis) in Great Smoky Mountains National Park.
Environmental Pollution, 125, 53–60.

Chappelka, A.H., Samuelson, L. (1998). Ambient ozone effects on forest trees of the
eastern United States: a review. *New Phytologist,* 139, 91-108.

Chen, G., Tian, H., Liu, M., Ren, W., Zhang, C., Pan, S. (2006). Climate Impacts on
China's Terrestrial Carbon Cycle: An Assessment with the Dynamic Land
Ecosystem Model. In: Tian, H.Q. (Eds.), Environmental Modeling and Simulation,
ACTA Press, Anaheim, Calgary, Zurich, pp56–70.

Chen, L. Y., Liu, H. Y., Lam, K. S., Wang, T. (1998). Analysis of the seasonal behavior
of   tropospheric ozone at Hong Kong. *Atmosphere Environment*, 32(2),
159–168. (In Chinese with English Abstract)

China Agriculture Yearbook. (2003). China Agriculture Press, Beijing.

China's National Climate Change Programme. (2007). National Development and
Reform Commission People's Republic of China.

Ciais, P., Tans, P.P., Trolier, M., White, J.W.C., Francey, R.J. (1995). A large northern
hemisphere   terrestrial CO_2 sink indicated by the $^{13}C/^{12}C$ ratio of atmospheric CO_2.
Science, 269, 1098– 1102.

Cleland, E.E. (2005). The influence of multiple interacting global changes on the
structure and function of a California annual grassland ecosystem. Ph.D.
Dissertation, Department of Biological Sciences, Stanford University.

Cole, V. C., Cerri, K.M., Mosier, A., Rosenberg, N., Sauerbeck, P. (1996). Agricultural
options for mitigation of greenhouse gas emissions. pp 745–771. In R.T. Watson et

al. (eds.) Climate Change 1995. Impacts, adaptations and mitigation of climate change: Scientific–technical analyses. IPCC Working Group II. Cambridge Univ. Press, Cambridge.

Cole, V., Cerri, C., Minami, K., Mosier, A., Rosenberg, N., Sauerbeck, D., Dumanski, J., Duxbury, J., Freney, J., Gupta, R., Heinemeyer, O., Kolchugina, T., Lee, J., Paustian, K., Powlson, D., Sampson, N., Tiessen, H., van Noordwijk, M., Zhao, Q. (1996). Agricultural options for mitigation of greenhouse gas emissions. pp. 745–771. In R.T. Watson et al. (ed.) Climate Change 1995. Impacts, adaptations and mitigation of climate change: Scientific–technical analyses. IPCC Working Group II. Cambridge University Press, Cambridge.

Cooley, D.R. and Manning, W.J. (1987). The impact of ozone on assimilation partitioning in plants–a review. *Environmental Pollution*, 47, 95–113.

Cramer, W., Bondeau, A., Woodward, F. I., Prentice, I. C., Betts, R.A., Brovkin, V., Cox, P.M., Fisher, V., Foley, J., Friend, A.D., Kucharik, C., Lomas, M.R., Ramankutty, N., Sitch, S., Smith, B., White, A., Young-Molling, C. (2001). Global response of terrestrial ecosystem structure and function to CO_2 and climate change: results from six dynamic global vegetation models. *Global Change Biology*, 7, 357–373.

Curtis, P.S., Hanson, P.J., Bolstad, P., Barford, C., Schmid, H.P., Wilson, K.B. (2002) Biometric and eddy covariance based estimates of annual carbon storage in five eastern North American deciduous forests. *Agricultural and Forest Meteorology*, 113, 3–19.

Davidson, E.A. and Ackerman, I.L. (1993). Changes in soil carbon inventories following cultivation of previously untilled soils, *Biogeochemistry,* 20, 161–193.

Davison, A.W. and Barnes, J.D. (1998). Effects of ozone on wild plants. *New Phytologist*, 139, 135–151.

DeLucia, E. H., Moore, D.J., Norby, R.J.   (2005). Contrasting responses of forest ecosystems to rising atmospheric CO_2: implication for the global C cycle. Global Biogeochemical Cycles, 19, GB3006.

Denman, K. et al. (2007). Couplings Between Changes in the Climate System and Biogeochemistry. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K., Tignor, M., Miller, H. (eds.), IPCC, Climate Change 2007: *The Physical Science Basis, Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, Cambridge University Press, Cambridge, United Kingdom and United States.

Dewar, R.C. (1996). The correlation between plant growth and intercepted radiation: an interpretation in terms of optimal plant nitrogen content. *Annals of Botany*, 78: 125–136.

Dixon, R. K., Brown, S., Houghton, R.A., Solomon, A.M., Trexler, M.C., Wisniewski, J. (1994). Carbon pools and flux of global forest ecosystems. *Science*, 263, 185–190.

Elliot, S., Blake, D.R., Duce, R., Lai, A., McCreary, I., McNair, L.A., Rowland, F.S., Russell, A.G., Streit, G.E., Turco, R.P. (1997). Motorization of China implies changes in Pacific air chemistry and primary production. *Geophysical Research Letters*, 24, 2671–2674.

Emberson, L.D., Ashmore, M.R., Murray, F., Kuylenstierna, J., Percy, K.E., Izuta, T., Zheng, Y., , Shimizu, H., Sheu, B.H., Liu, C.P, Agrawal, M., Wahid, A.,

Abdel-Latif, N.M., van Tienhoven, M., de Bauer, L.I., Domingos, M. (2001). Impacts of air pollutants on vegetation in developing countries. *Water, Air, and Soil Pollution*. 130, 107–118.

Emmett, B.A., Stevens, P. A., Reynolds, B. (1995). Factors influencing nitrogen saturation in Sitka spruce stands in Wales, UK. *Water, Air and Soil Pollution,* 85, 1629–1634.

Enting, I. E., Wigley, T. M. L., Heimann, M. (1994). Future Emissions and Concentrations of Carbon Dioxide: Key Ocean/Atmosphere/Land Analyses. CSIRO Division of Atmospheric Research Technical Paper No. 31.

Falkowski, P., Scholes, R. J., Boyle, E., Canadell, J., Canfield, D., Elser, J., Gruber, N., Hibbard, K., Hogbeg, P., Linder, S., MacKenzie, A. F., Moore, B. III, Pedersen, T. F., Rosenthal, Y., Seitzinger, S., Smetacek, V., Steffen, W. (2000). The global carbon cycle: a test of our knowledge of Earth as a system. *Science,* 290: 291-296.

Fang, J. Y. (2000). Forest biomass carbon pool of middle and high latitudes in the north hemisphereis probably much smaller than present estimates. *Acta Phytoecological Sinica*, 24(5), 635–638.

Fang, J. Y., Guo, Z. D., Piao, S. L., Chen, A. P. (2007). Terrestrial vegetation carbon sinks in China, 1981-2000. *Science in China (Series D)*, 1341-1350.

Fang, J. Y., Liu, G. H., Xu, S. L. (1996). Biomass and net production of forest vegetation in China. *Acta Ecology Sinica*, 16(5), 497–508.

Fang, J.Y., Brown, S., Tang, Y.H., Nabuurs, G.J., Wang, X.P., Shen, H.H. (2006). Overestimated biomass carbon pools of the northern mid- and high latitude forests. *Climatic Change*, 74(1–3), 355–368.

Fang, J.Y., Chen, A.P., Peng, C.H., Zhao, S.Q., Chi, L.J. (2001). Changes in forest biomass carbon storage in China between 1949 and 1998. *Science*, 292, 2320–2322.

Fang, J.Y., Liu, G.H., Xu, S.L. (1996) Carbon pools in terrestrial ecosystems in China. In: Wang, R.S., Fang, J.Y., Gao, L., Feng, Z.W. (Eds.), Hot Spots in Modern Ecology. Beijing: China Science and Technology Press, pp251–277.

Farage, P. K., Long, S. P., Lechner, E. G., Baker, N.R. (1991). The sequence of change within the photosynthetic apparatus of wheat following shout–term exposure to ozone. *Plant Physiology*, 95, 529–535.

Farquahar, G. D., Caemmerer, S., Berry, J. A. (1980) A biochemical model of photosynthetic CO_2 assimilation in leaves of C_3 species. *Planta*, 149, 78–90.

Felzer, B. S., Cronin, T. W., Melillo, J. M., Kicklighter, D. W., Schlosser, C. A. (2009). Importance of carbon-nitrogen interactions and ozone on ecosystem hydrology during the 21st century. Journal of Geophysical Research, 114, doi:10.1029/2008JG000826.

Felzer, B., Kicklighter, D. W., Melillo, J. M., Wang, C., Zhuang, Q., Prinn, R. (2004). Effects of Ozone on Net Primary Production and Carbon Sequestration in the Conterminous United States using a Biogeochemistry Model. *Tellus*, 56(B), 230–248.

Felzer, B., Reilly, J. Melillo, J., Kicklighter, D. W., Sarofim, M., Wang, C., Prinn, R. G., and Q. Zhuang, Q. (2005). Future effects of ozone on carbon sequestration and climate change policy using a global biochemistry model. *Climatic Change*, 73 (3), 345–373.

Feng X F. (2004). Simulating net primary productivity and evaportranspiration of terrestrial ecosystem in China using a process-based model driven by remote sensing. Doctoral dissertation, Institute of Geographical Sciences and Natural Resources Research (In Chinese with English Abstract).

Feng, Z. W., Zhang, J. W., Deng, S. J. (1980). Studying in the phytomass of planted *Cunninghamia lanceolata* forest. In: Report of Synthesis inventory in Taoyuan. Henan Science and Technology press, Henan (In Chinese).

Feng, Z., Jin, M., Zhang, F. (2003). Effects of ground–level ozone (O_3) pollution on the yields of rice and winter wheat in the Yangtze River Delta. *Journal of Environmental Science*, 15, 360–362.

Field, C.B., Mortsch, L.D., Brklacich, M., Forbes, D.L., Kovacs, P., Patz, J.A., Running, S.W., Scott, M.J. (2007). North America. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, M.L. Parry, et.al., Eds., Cambridge University Press, Cambridge, UK, 617–652.

Fiscus, E.L., Booker, F.L., Burkey, K.O. (2005). Crop responses to ozone: uptake, modes of action, carbon assimilation and partitioning. *Plant Cell and Environment*, 28, 997–1011.

Fu, C.B., Wen, G. (1999). Variation of ecosystems over East Asia in association with seasonal interannual and decadal monsoon climate variability. *Climate Change*, 43, 477–494.

Fuhrer, J., Booker, F.L. (2003). Ecological issues related to ozone: agricultural issues. *Environment International*, 29, 141–154.

Fuhrer, J., Skarby, L., Ashmore, M. R. (1997). Critical levels for ozone effects on
vegetation in Europe. *Environment Pollution*, 97(1–2), 91–106.

Fumagalli, I., Gimeno, B.S., Velissariou, D., de Temmerman, L., Mills, G. (2001).
Evidence of ozone-induced adverse effects on crops in the Mediterranean region.
Atmospheric Environment, 35, 2583–2587.

Galloway J.N., Aber J.D., Erisman J.W., Seitzinger S.P., Howarth R.H., Cowling E.B.,
Cosby B.J. (2003). The nitrogen cascade. *BioScience,* 53: 341–356.

Gates, D. M. (1990). Climate change and forest. *Tree Physiology*, 7, 1–5.

Ge, Q. S., D, J. H., He, F. N., Pan, Y., Wang, M. M. (2008). Land use changes and their
relations with carbon cycles over the past 300 years in China. *Science in China
(Series D)*, 51, 871–884.

Ge, Q. S., Dai, J. H., He, F.N., Zheng, J. Y., Man, Z. M., Zhao, Y. (2003). Spatiotemporal
dynamics of reclamation and cultivation and its driving factors in parts of China
during the last three centuries. *Progress in Natural Science*, 14(7), 605–613.

Gifford, R.M., Lutze, J.L. and Barrett, D. (1996). Global atmospheric change effects on
terrestrial carbon sequestration: Exploration with a global C and N cycle model
(CQUESTN). *Plant and Soil*, 187, 369–387.

Goetz, S.J., Prince, S.D., Small, J., Gleason, A.C.R., Thawley, M.M. (2000). Interannual
variability of global terrestrial primary production: reduction of a model driven with
satellite observations. *Journal of Geophysical Research*, 105: 20,007–20,091.

Gong, P., Xu, M., Chen, J. (2002). A preliminary study on the carbon dynamics of
China's terrestrial ecosystems in the past 20 years. *Earth Science Frontiers*, 2002,
9(1): 55–61.

Goodale, C. L., Apps, M. J., Birdsey, R. A., Field, C. B., Heath, L. S., Houghton, R.A., Jenkins, J. C., Kohlmaier, G. H., Kurz, W., Liu, S., Nabuurs, G., Nilsson, S., Shvidenko, A. Z. (2002). Forest carbon sink in the Northern Hemisphere. *Ecological Applications,* 12(3), 891–899.

Grace, J. (2004). Understanding and managing the global carbon cycle. *Journal of Ecology*, 92, 189-202.

Grantz, D. A., Silva,V., Toyota, M., Ott, N. (2003) Ozone increases root respiration but decreases leaf CO_2 assimilation in cotton and melon. *Journal of Experimental Botany*, 54, 1–10.

Grantz, D.A. & Yang, S. (2000). Ozone impacts on allometry and root hydraulic conductance are not mediated by source limitation nor developmental age. *Journal of Experimental Botany*, 51, 919–927.

Gregg, J. S., Andres, R. J., Marland, G. (2008), China: Emissions pattern of the world leader in CO2 emissions from fossil fuel consumption and cement production, *Geophysical Research Letter,* 35, L08806, doi:10.1029/2007GL032887.

Grulke, N. E, Neufeld, H.S., Davison, A. W., Roberts, M., Chappelka, A. H. (2007). Stomatal behavior of ozone–sensitive and –insensitive coneflowers (Rudbeckia laciniata var. digitata) in Great Smoky Mountains National Park. *New Phytologist*, 173, 100–109.

Guo, J. P., Wang, C. Y., Wen, M., Bai, Y. M., Guo, H. Z. (2001). The Experimental Study on the Impact of Atmospheric O_3 Variation on Rice. *Acta Agronomica Sinica*, 27, 822–826 (In Chinese with English Abstract).

Guo, L. B., and Gifford, R.M. (2002). Soil carbon stocks and land use change: a meta

analysis. *Global Change Biology*, 8:345–360.

Gurney, K.R., Law, R.W., Denning, A. S., Rayner, P.J., Baker, D., Bousquet, P.,
Bruhwiler, L., Chen, Y.H., Ciais, P., Fan, S., Fung, I.Y., Gloor, M., Heimann, M.,
Higuchi, K., John, J., Maki, T., Maksyutov, S., Masarie, K., Peylin, P., Prather, M.,
Pak, B.C., Randerson, J., Sarmiento, J., Taguchi, S., Takahashi, T., Yuen, C.W.
(2002), Towards robust regional estimates of CO2 sources and sinks using
atmospheric transport models, *Nature*, 415, 626–630.

Haefner, J.W. (2005). Modeling Biological Systems: Principles and Applications.
Springer: New York. ISBN: 0387250115.

Hansen, A. J., Neilson, R. P.,   Dale, V. H., Flather, C. H., Iverson, L. R., Currie, D. J.,
Shafer, S., Cook, R., Bartlein, P. J. (2001). Global Change in Forests: Responses
of Species, Communities, and Biomes. BioScience, 51(9), 765-779.

Hansen, J., Sato, Mki., Ruedy, R., Lo, K., Lea, D.W., Medina-Elizade, M. (2006). Global
temperature change. *Proceedings of the National Academy of Sciences*, 103,
14288–14293, doi:10.1073/pnas.0606291103.

Hanson, P.J., Wullschleger, S.D., Norby, R.J., Tschaplinski, T.J., Gunderson, C.A. (2005)
Importance of changing CO_2, temperature, precipitation, and ozone on carbon and
water cycles of an upland oak forest: incorporating experimental results into
model simulationd. *Global Change Biology,* 11,1402-1423

Heagle, A. S., Mille, J. S., Booker, F. L., Pursley, W. A. (1999). Ozone stress, carbon
dioxide enrichment, and nitrogen fertility interactions in cotton. *Crop Sciences*, 39,
731–41.

Heagle, A.S. (1989). Ozone and crop yield. *Annual Review of Phytopathology*, 27,

397–423.

Heal, O.W., Anderson, J.M., Swift, M.J. (1997). Plant litter quality and decomposition: An historical overview. In: Cadisch, G. & Giller, K.E.(eds.). Driven by Nature. Plant Litter Quality and Decomposition, CAB International, Wallingford, UK.

Heck, W. W., Cure, W. W., Rawlings, J. O., Zaragoza, L. J., Heagle, A. S., Heggestad, H. E., Kohut, R. J., Kress, L. W., Temple, P. J. (1984). Assessing impacts of ozone on agricultural crops. I. Overview. *Journal of Air Pollution Control Association*, 34, 729–735.

Heck, W.W. (19889). Assessment of crop losses from air pollutants in the United States. In: MacKenzie, J. J., El–Ashry, M. (Eds.), Air pollution's toll on forests and crops. Yale University Press, New Haven, pp 235–315.

Heggestad, H. E., Lesser, V. M. (1990). Effects of ozone, sulfur dioxide, soil water deficit, and cultivar on yields of soybean. *Journal of Environmental Quality*, 19, 488–95.

Heimann, M. and Reichstein, M. (2008). Terrestrial ecosystem carbon dynamics and climate feedbacks*. Nature*, 451, 289–292.

Heinsch, F. A., and Coauthors. (2003). User's guide, GPP and NPP (MOD17A2/A3) products NASA MODIS land algorithm.

Heinsch, F.A., Zhao, M., Running, S.W., Kimball, J.S., Nemani, R.R.,Davis, K.J., Bolstad, P.V., Cook, B.D., Desai, A.R., Ricciuto, D.M.,Law, B.E., Oechel, W.C., Hyojung, K., Hongyan, L., Wofsy, S.C.,Dunn, A.L., Munger, J.W., Baldocchi, D.D., Liukang, X., Hollinger, D.Y., Richardson, A.D., Stoy, P.C., Siqueira, M.B.S.,Monson, R.K., Burns, S.P., Flanagan, L.B. (2006). Evaluation of remote

sensing based terrestrial productivity from MODIS using regional tower eddy flux
network observations. IEEE Transions. *Geoscience and Remote Sensing*, 44,
1908–1925.

Hogg, E.H., Brandt, J.P., Kochtubajda, B. (2005). Factors affecting interannual variation
in growth of western Canadian aspen forests during 1951–2000. *Canadian Journal
of Forest Research*, 35, 610–622.

Holland, E. A., Braswell, B.H., Lamarque, J.F. et al. (1997). Variations in the predicted
spatial distribution of atmospheric nitrogen deposition and their impact on carbon
uptake by terrestrial ecosystems. Journal of Geophysical Research-Atmospheres
102:15849–15866.

Hollinger, D.Y., Aber, J., Dail, B., Davidson, E.A., Goltz, S.M., Hughes, H., Leclerc, M.
Y., Lee, J. T., Richardson, A.D., Rodrigues, C., Scott, N.A., Achuatavarier, D.,
Walsh, J. (2004). Spatial and temporal variability in forest-atmosphere CO2
exchange. *Global Change Biology*, 10, 1689–1706.

Houghton, R. A. and , Goodale, C. L. (2004). Effects of land–use change on the carbon
balance of terrestrial ecosystems, in Ecosystems and land use change, DeFries, R.
S., Asner, G. P., Houghton, R. A. (Eds.), American Geophysical Union,
Washington, DC, pp85–98.

Houghton, R. A., Hackler, J. L. (2003). Sources and sinks of carbon from land–use
change in China. *Global Biogeochemical Cycles*, 17(2), 1029–1034.

Houghton, R.A. (2001). Counting terrestrial sources and sinks of carbon. *Climatic
Change*, 48, 525–534.

Houghton, R.A. (2003). Revised estimates of the annual net flux of carbon to the

atmosphere from changes in land use and land management 1850-2000. *Tellus,* 55B, 378-390.

Houghton, R.A. (2007). Balancing the global carbon budget. *Annual Review of Earth and Planetary Sciences*, 35:313-347.

Houghton, R.A., Hackler, J.L., Lawrence, K.T. (1999). The U.S. carbon budget: contributions from land-use change. *Science*, 285, 574–578.

Houghton, R.A.1995. Balancing the global carbon cycle with terrestrial ecosystems. In: Zepp RG, Sonntag C (eds.) The role of nonliving organic matter in the earth's carbon cycle. Wiley, Chichester, pp133–152.

Huang, H., Wang, C., Bai, Y., Wen, M. (2004). A diagnostic experimental study of the composite influence of increasing O_3 and CO_2 concentration on soybean. *Chinese Journal of Atmospheric Sciences*, 28, 601–612 (In Chinese with English Abstract).

Huang, M., Ji, J. J., Cao, M. K., Li, K. R. (2006). Modeling study on aboveground and underground biomass in China. *Acta Ecologica Sinica*, 26(12), 4156–4163(In Chinese with English Abstract).

Huang, Y. and Sun, W. J. (2006). Changes in topsoil organic carbon of croplands in mainland China over the last two decades. *Chinese Science Bulletin*, 51, 1785–1803.

Huang, Y., Dickinson, R. E., Chameides W. L. (2006). Impact of aerosol indirect effect on surface temperature over East Asia. *Proceedings of the National Academy of Sciences*, 103(12), 4371–4376.

Huang, Y., Sass, R. L., Fisher, F. M. (1998). Model estimates of methane emission from

irrigated rice cultivation of China. *Global Change Biology*, 4, 809–822.

Huang, Y., Yu, Y. Q., Zhang, W., Sun, W. J., Liu, S. L., Jiang, J., Wu, J. S., Yu, W. T., Wang, Y., Yang, Z. F. (2009). Agro–C: A biogeophysical model for simulating the carbon budget of agroecosystems. *Agricultural and forest meteorology*, 149, 106–129.

Huang, Y., Zhang, W., Sun, W. J., Zheng, X. H. (2007). Net primary production of Chinese croplands from 1950 to 1999. *Ecological Applications*, 17(3), 692–701.

IISD (2001). COP–6. bis Final summary. International institute for sustainable development. http://www.iisd.ca/linkages/download/asc/enb12176e.txt

IPCC (2007). Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, eds. Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, Tignor M, Miller HL. Cambridge University Press, Cambridge, UK.

Islam, K.R. and Weil, R.R. (2000). Soil quality indicator properties in mid-Atlantic soils as inuenced by conservation management. *Journal of Soil and Water Conservation*, 55:69–78.

Jacob, D. J., Logan, J. A., Murti, P. P. (1999). Effect of rising Asian emissions on surface ozone in the United States. *Geophysical Research Letters*, 26(14), 2175–2178.

Jaffe, D., McKendry, I., Anderson, T., Price, H. (2003). Six 'new' episodes of trans–Pacific transport of air pollutants. *Atmospheric Environment*, 37 (3), 391–404.

Janzen, H.H., Campbell, C.A., Isaurralde, R.C., Ellert, B.H., Juma, N., McGill, W.B., Zentner, R.P. (1998). Management effects on soil C storage on the Canadian prairies. *Soil &Tillage Research*, 47,181–195.

Jiang, H., Apps, M. J., Zhang, Y. L., Peng, C. H., Woodard, P. M. (1999). Modelling the spatial pattern of net primary productivity in Chinese forests. *Ecological Modelling*, 122(3), 275–288.

Jin, L.Y., Fu, J. L., Chen, F. H. (2005). Spatial differences of precipitation over northwest china during the last 44 years and its response to global warming. *Scientia Geographica Snica*, 25, 567–572.

Jones, C. A., Kiniry, J. R. (1986). CERES–Maize: A Simulation Model of Maize Growth and Development. Texas A&M University Press, College Station, TX, pp194.

Joyce, L. A., Mills, J. R., Heath, L. S., McGuire, A. D., Haynes, R. W., Birdsey, R. A. (1995). Forest sector impacts from changes in forest productivity under climate change. *Journal of Biogeography*, 1995, 22, 703–713.

Kang, H. N., Ma, Q. Y., Yuan, J. Z. (1996). Estimation of the Chinese forest function on carbon sink. *Chinese Journal of Applied Ecology*, 7(3), 230–234 (In Chinese with English Abstract).

Karnosky, D.F., Pregitzer, K.S., Zak, D.R., Kubiske, M.E., Hendrey, G.R., Weinstein, D., Nosal, M., Percy, K.E. (2005). Scaling ozone responses of forest trees to the ecosystem level in a changing climate. Plant, Cell and Environment. 28: 965–981.

Khalil, M.A.K., Butenhoff, C.L., Rasmussen, R.A. (2007). Atmospheric methane: Trends and cycles of sources and sinks. *Environmental Science & Technology*, 10.1021/es061791t.

Kicklighter, D.W., Bruno, M., Dönges, S., Esser, G., Heimann, M., Helfrich, J., Ift, F., Joos, F., Kaduk, J., Kohlmaier, G.H., McGuire, A.D., Melillo, J.M., Meyer, R., Moore III, B., Nadler, A., Prentice, I.C., Sauf, W., Schloss, A.L., Sitch, S., Wittenberg, U., Würth, G. (1999). A first-order analysis of the potential role of CO2 fertilization to affect the global carbon budget: A comparison study of four terrestrial biosphere models. *Tellus 51B*, 343–366.

Kim, H.J., Helfand, G.E., Howitt, R.E. (1998). An economic analysis of ozone control in California's San Joaquin Valley. *Journal of Agricultural and Resource Economics,* 1998, 23–70.

Kuik, O. J., Helming, J. F. M., Dorland, C., Spaninks, F. A. (2000). The economic benefits to agriculture of a reduction of low–level ozone pollution in the Netherlands. *European Review of Agricultural Economics*, 27, 76–90.

Lal, R. (2004). Carbon emission from farm operations. *Environmental International*, 981–990.

Leff, B., Ramankutty, N., Foley, J.A. (2004). Geographic distribution of major crops across the world. *Global Biogeochemical Cycles*, 18, doi:10.1029/2003GB002108.

Li, C. S., Frolking, S., Xiao, X. M., Moore, III B., Boles, S., Qiu, J. J., Huang, Y., Salas, W., Sass, R. (2005). Modeling impacts of farming management alternatives on CO_2, CH_4, and N_2O emissions: A case study for water management of rice agriculture of China. *Global Biogeochemical Cycles*, doi:10.1029/2004GB002341.

Li, C. S., Qiu, J. J., Frolking, S., Xiao, X. M., Salas, W., Moore, III B., Boles, S., Huang,

Y., Sass, R. (2002). Reduced methane emissions from large–scale changes in water management of China's rice paddies during 1980–2000. *Geophysical Research Letters*, doi:10.1029/2002GL015370.

Li, C. S., Salas, W., DeAngelo, B., Rose, S. (2006). Assessing alternatives for mitigating net greenhouse gas emissions and increasing yields from rice production in China over the next 20 years. *Journal of Environmental Quality*, 35, 1554–1565.

Li, C., Aber, J., Stange, F., Butterbach–Bahl, K., Papen, H. (2000). A process–oriented model of N_2O and NO emissions from forest soils: 1, Model development. *Journal of Geophysical Research*, 105 (4), 44369–4384.

Li, K.R., Wang, S.Q., Cao, M.K. (2003). Plant and soil carbon storage in China. Science in China (D) 33(1), 72–80, 2003.

Li, W. H., Li, F. (1981). Research of forest in China. China forestry press, Beijing (In Chinese).

Li, X. Z., Liu, X. D., Ma, Z. G. (2004). Analysis on the drought characteristics in the main arid regions in the world since recent hundred–odd years. *Arid Zone Research*, 21, 97–103.

Li, Y.P. and Ji, J.J. (2001). Simulations of carbon exchange between global terrestrial ecosystem and the atmosphere. *Acta Geographica Sinica*. 2001, 56(4), 379-389.

Lindroth, R. L., Kopper, B. J., Parsons, W. F., Bockheim, J. G., Karnosky, D. F., Hendrey, G. R., Pregitzer, K. S., Isebrands, J. G., Sober, J. (2001). Consequences of elevated carbon dioxide and ozone for foliar chemical composition and dynamics in trembling aspen (Populus tremuloides) and paper birch (Betula papyrifera). *Environment Pollution*, 115 (3), 395–404.

170

Liu, D.W., Wang, Z. M., Zhang, B., Song, K. S., Li, X. Y., Li, J. P., Li, F., Duan, H. T. (2006). Spatial distribution of soil organic carbon and analysis of related factors in croplands of the black soil region, Northeast China. *Agriculture Ecosystems & Environment*, 113, 73–81.

Liu, G. H., Fu, B. J., Fang, J. Y. (2000). Carbon dynamics of Chinese forests and its contribution to global carbon balance. *Acta Ecologica Sinica*, 20(5), 733–740.

Liu, J. D., Zhou, X. J., Yu, Q., Yan, P., Guo, J. P., Ding, G. A. (2004). A Numerical simulation of the impacts of ozone in the ground layer atmosphere on crop photosynthesis. *Chinese Journal of Atmospheric Sciences*, 28(1), 59–68.

Liu, J. G., Diamond, J. (2005) China's Environment in a Globalizing World – How China and the Rest of the World Affect Each Other. *Nature*, 435, 1179–1186.

Liu, J., Liu, M., Tian, H. Q., Zhuang, D., Zhang, Z., Zhang, W., Tang, X., Deng, X. (2005a). Current status and recent changes of cropland in China: an analysis based on Landsat TM data. *Remote Sensing of Environment*, 98, 442–456.

Liu, J., Liu, M., Zhuang, D., Zhang, Z., Deng, X. (2003). Study on spatial pattern of land–use change in China during 1995–2000. *Science in China (Series D)*. 46(4), 373–384.

Liu, J., Tian, H.Q., Liu M., Zhuang, D., Melillo, J.M., Zhang, Z. (2005b). China's changing landscape during the 1990s: Large–scale land transformation estimated with satellite data. *Geophysical Research Letters*, 32: L02405, doi: 10.1029/ 2004GL021649.

Liu, M. L., Tian, H. Q., Chen, G. S., Ren, W., Zhang, C., Liu, J. Y. (2008). Effects of land use and land cover change on evapotranspiration and water yield in China

during the 20th century. *Journal of the American Water Resources Association*, 44, 1193–1207.

Liu, M.L. (2001). Study on land use/cover change and terrestrial vegetation carbon pool and productivity in China. Doctoral dissertation, Institute of Remote Sensing Application, Beijing.

Liu, S. R., Guo, Q. S., Wang, B. (1998). Prediction of primary productivity of forests in China in respsonse to cliamte change. *Acta Ecologica Sinica*. 18(5), 478–483.

Logan, J. A., Regniere, J., Powell, J. A. (2003). Assessing the impact of global warming on forest pest dynamics. *Frontiers in Ecology and Environment,* 1(3), 130–137.

Lu, C., Tian, H. (2007). Spatial and temporal patterns of nitrogen deposition in China: synthesis of observational data. *Journal of Geophysical Research*, 112, D22S05, doi:10.1029/2006JD007990.

Luo, T. X., Li, W. H., Leng, Y. F., Yue, Y. Z. (1998). Estimation of total biomass and total biomass and potential distribution of net primary productivity in Tibetean Plateau. *Geographical research*, 17(4), 337–344 (In Chinese with English Abstract).

Luo, T. X., Li, W. H., Zhao, S. D. (1999). Productivity distribution patterns and modeling of *Pinus taulaeformis* in China. *Chinese Journal of Applied Ecology*, 10(3), 257–261(In Chinese with English Abstract).

Luo, Y.Q. (2007). Terrestrial carbon-cycle feedback to climate warming. *Annual Review of Ecology, Evolution, and Systematics,* 38, 683–712.

Ma, Q. Y., Xie, Z. M. (1996). Estimation of carbon stored in Chinese pine forest. Journal of Beijing Forestry University, 1996, 18(3), 31–34 (In Chinese with English

Abstract).

Magill, A., Aber, J., Berntson, G., McDowell,W., Nadelhoffer, K., Melillo, J.M., Steudler,
P. (2000). Long-term nitrogen additions and nitrogen saturation in two temperate
forests. *Ecosystems,* 3, 238–253.

Marland, G., Pielke Sr., R.A., Apps, M., Avissar, R., Betts, R.A., Davis, K.J., Frumhoff,
P.C., Jackson, S.T., Joyce, L., Kauppi, P., Katzenberger, J., MacDicken, K.G.,
Neilson, R., Niles, J.O., Niyogi, D.D.S., Norby, R.J., Pena, N., Sampson, N., Xue,
Y. (2003). The climatic impacts of land surface change and carbon management,
and the implications for climate-change mitigation policy. *Climate Policy*, 3:
149–157.

Martin, M. J., Farage, P. K., Humphries, S. W., Long, S. P. (2000). Can the stomatal
changes caused by acute ozone exposure be predicted by changes occurring in the
mesophyll? A simplification for models of vegetation response to the global
increase in tropospheric elevated ozone episodes. *Australian Journal of Plant
Physiology*, 27, 211–219.

Martin, M. J., Host, G. E., Lenz, K. E. (2001). Stimulating the growth response of aspen
to elevated ozone: a mechanistic approach to scaling a leaf–level model of ozone
effects on photosynthesis to a complex canopy architecture. *Environmental
Pollution*, 115, 425–436.

Martin, M.J., Farage, P.K., Humphries, S.W., Long, S.P. (2000). Can the stomatal
changes caused byacute ozone exposure be predicted by changes occurring in the
mesophyll? A simplification for models of vegetation response to the global

increase in tropospheric elevated ozone episodes. *Australian Journal of Plant Physiology,* 27, 211–219.

Matson, P., Lohse, K.A., Hall, S. (2002). The globalization of nitrogen deposition: consequences for terrestrial ecosystems. *AMBIO*, 31, 113–119.

Matyssek, R. and Sandermann, H. (2003). Impact of ozone on trees: an ecophysiological perspective. *Progress in Botany*, 64, 349–404.

Mauzerall, D. L. and Wang, X. (2001). Protecting agricultural crops from the effects of tropospheric ozone exposure: science and standard setting in the United States, Europe, and Asia. *Annual Review of Energy and the Environment*, 26, 237–268.

Mauzerall, D. L., Narita, D., Akimoto, H., Horowitz, L., Waters, S. (2000). Seasonal characteristics of tropospheric ozone production and mixing ratios of East Asia: a global three–dimensional chemical transport model analysis. *Journal of Geophysical Research*, 105, 17895–17910.

McGuire, A. D., Sitch, S., Clein, J. S., Dargaville, R., Esser, G., Meier, R. A., Melillo, J. M., Moore, B. III., Prentice, I. C., Ramankutty, N., Reichenau, T., Schloss, A., Tian, H.. Williams, L. J., Wittenberg, U. (2001). CO_2, climate and land–use effects on the terrestrial carbon balance, 1920–1992: An analysis with four process–based ecosystem models. *Global Biogeochemical Cycles*, 15, 183–260.

McGuire, A.D., Melillo, J.M., Joyce, L.A., Kicklighter, D.W., Grace, A.L., Moore III, B., Vorosmarty, C.J. (1992). Interactions between carbon and nitrogen dynamics in estimating net primary productivity for potential vegetation in North America. *Global Biogeochemical Cycles,* 6, 101–124.

McGuire, A.D., Melillo, J.M., Joyce. L.A. (1995). The role of nitrogen in the response of

    forest net primary production to elevated atmospheric carbon dioxide. *Annual*

    Review of Ecology and Systematics, 26, 473–503.

McGuire, A.D., Melillo, J.M., Kicklighter, D.W., Pan, Y., Xiao, X., Helfrich, J., Moore

    III, B., Vorosmarty, C.J., Schloss. A.L. (1997). Equilibrium responses of global net

    primary production and carbon storage to doubled atmospheric carbon dioxide:

    Sensitivity to changes in vegetation nitrogen concentration. *Global Biogeochemical*

    Cycles, 1, 173–189.

McLaughlin, S. B., Nosal M., Wullschleger, S. D., Sun, G. (2007b). Interactive effects

    of ozone and climate on Southern Appalachian Forests in the USA: Effects on

    water use, soil moisture content, and streamflow. *New Phytologist*, 174,125–136.

McLaughlin, S. B., Nosal, M., Wullshleger, S. D., Sun, G. (2007a). Interactive effects of

    ozone and climate on tree growth and water use in a southern Appalachian forest

    in the USA. *New Phytologist*, 174,109–124.

Melillo, J. M. and Gosz, J. R. (1983). Interactions of biogeochemical cycles in forest

    ecosystems, pp177–222. In: B. Bolin and R. B. Cook (eds.), The Major

    Biogeochemical Cycles and Their Interactions. John Wiley and Sons, New York.

Melillo, J. M., McGuire,   D., Kicklighter, D. W., Moore, B. III.,   Vorosmarty, C. J.,

    Schloss, A. L. (1993). Global climate change and terrestrial net primary

    production. *Nature*, 363, 234–240.

Melillo, J. M., Prentice, I. C., Farquhar, G. D., Schulze, E.-D., Sala, O. (1996). Terrestrial

    biotic responses to environmental change and feedbacks to climate. Pp. 445-482.

    In: Houghton, J. T. and others (eds.), Climate Change 1996- The Science of

175

Climate Change. Contribution of Working Group I to the Second Assessment
Report of the

Intergovernmental Panel on Climate Change. Cambridge University Press,
Cambridge, UK.

Melillo, J.M. (1995), Human influences on the global nitrogen budget and their
implications for the global carbon budget. In: Murai, S., Kimura, M.(Eds.),
Toward global planning of sustainable use of the Earth: development of global
eco–engineering. Elsevier, New York, pp117–134.

Mills, G., Buse, A., Gimeno, B., Bermejo, V., Holland, M., Emberson, L. et al. (2007). A
synthesis of AOT_{40}-based response functions and critical levels of ozone for
agricultural and horticultural crops. *Atmosphere Environment.* 41, 2630–2643.

Muntifering, R. B., Chappelka, A. H., Lin, J.C., Karnosky, D. F., Somers, G. L. (2006).
Chemical composition and digestibility of Trifolium exposed to elevated ozone
and carbon dioxide in a free–air (FACE) fumigation system. *Functional Ecology*,
20, 269–275.

Murty, D., Kirschbaum, M. F., Mcmurtrie, R. E., Mcgilvray, H. (2002), Does conversion
of forest to agricultural land change soil carbon and nitrogen? A review of the
literature, *Global Change Biology*, 8, 105–123.

National Bureau of Statistics (NBS). (2005). 1995–2006. China Statistical Yearbook.
Beijing: China Statistics Press.

Neff, J.C., S.H. Hobbie, and P.M. Vitousek . (2000). Controls over the production and
stoichiomet~ of dissolved organic carbon, nitrogen and phosphorus in tropical soils.
Biogeochemistry 51: 283–302.

Nemani, R.R., Keeling, C.D., Hasimoto, H., Jolly, W.M., Piper, S.C.,Tucker, C.J.,

      Myneni, R.B., Running, S.W. (2003). Climate-Driven Increases in Global

      Terrestrial Net Primary Production from 1982 to 1999. *Science,* 300, 1560–1563.

Neufeld, H. S., Chappelka, A. H., Somers, G. L., Burkey, K. O., Davison, A. W.,

      Finkelstein, P. L. (2006). Visible foliar injury caused by ozone alters the

      relationship between SPAD meter readings and chlorophyll concentrations in

      cutleaf coneflower. *Photosynthesis Research*, 87, 281–286.

Neufeld, H. S., Renfro, J. R., Hacker, W. D., Silsbee, D. (1992). Ozone in Great Smoky

      Mountains National Park: Dynamics and effects on plants. in: Berglund R. J.

      (Eds), Tropospheric ozone and the environment II: effects, modeling and control.

      Pittsburgh, PA, USA: Air and Waste Management Association, pp594–617.

Ni, J. (2001). Carbon storage in terrestrial ecosystems of China: Estimates at different

      spatial resolutions and their responses to climate change. *Climatic Change*, 49,

      339–358.

Ni, J. (2004). Forage yield–based carbon storage in grasslands of China. *Climate Change*,

      67, 237–246.

Nussbaum, S., Bungener, P., Geissmann, M., Fuhrer, J. (2000). Plant–plant interactions

      and soil moisture might be important in determining ozone impacts on grasslands.

      New Phytologist, 147, 327–335.

Oksanen, E. (2003). Physiological responses of birch (Betula pendula) to ozone: a

      comparison between open-soil-grown trees exposed for six growing seasons and

      potted seedlings exposed for one season. *Tree Physiology,* 23(9), 603–614.

Ollinger, S. V., Aber, J. D., Reich, P. B. (1997). Simulating ozone effects on forest

productivity: interactions among leaf–canopy and stand–level processes. *Ecological Applications*, 7(4), 1237–1251.

Ollinger, S. V., Aber, J. D., Reich, P. B., Freuder, R. J. (2002a). Interactive effects of nitrogen deposition, tropospheric ozone, elevated CO_2 and land use history on the carbon dynamics of northern hardwood forests. *Global Change Biology*, 8, 545–562.

Ollinger, S. V., Smith, M. L., Hallett, R. A., Goodale, C. L., Aber, J. D. (2002b). Regional variation in foliar chemistry and N cycling in forests of diverse history and composition. *Ecology*, 83, 339–355.

Pacala, S., R. Birdsey, S. Bridgham, R.T. Conant, K. Davis, B. Hales, R.A. Houghton, J.C. Jenkins, M. Johnston, G. Marland, and K. Paustian. (2007). The North American Carbon Budget Past and Present. p 3-1 ~ 3–21. In W.J. Brennan et al. (ed.) The First State of the Carbon Cycle Report (SOCCR) North American Carbon Budget and Implications for the Global Carbon Cycle. U.S. Climate Change Science Program. Synthesis and Assessment Product 2.2. Available online: http://cdiac.ornl.gov/SOCCR/documents.html.

Pacala, S.W., Hurtt, G.C., Houghton, R.A., Birdsey, R.A., Heath, L., Sundquist, E.T., Stallard, R.F., Baker, D., Peylin, P., Moorcroft, P., Caspersen, J., Shevliakova, E., Harmon, M.E., Fan, S.M., Sarmiento, J.L., Goodale, C., Field, C.B., Gloor, M., Schimel, D. (2001). Consistent Land- and Atmosphere-Based U.S. Carbon Sink Estimates. Science 292 , 2316–2320.

Pan, Y. D., Luo, T. X., Birdsey, R., Hom, J., Melillo, J. M. (2004). New estimates of carbon storage and sequestration in China's forests: effects of age–class and

methods on inventory–based carbon estimation. *Climatic Change*, 67: 211–236.

Pan, Y. D., Melillo, J. M., Kicklighter, D. W., Xiao, X., McGuire, A. D. (2001). Modeling structural and functional responses of terrestrial ecosystems in China to changes in climate and atmospheric CO_2. *Acta Phytoecologica Sinica*, 25(2), 175–189.

Paustian, K., C.V. Cole, Sauerbeck, D., Sampson, N. (1998). CO_2 mitigation by agriculture: An overview. Climatic Change 40: 135–162.

Paustian, K., Collins, H.P., Paul, E.A. (1997). Management controls on soil carbon. pp15–49. In: Paul, E.A. et al. (eds.) Soil organic matter in temperate agroecosystems. CRC Press, Boca Raton, FL.

Pell, E. J., Schlagnhaufer, C. D., Arteca, R. N. (1997). Ozone–induced oxidative stress: mechanisms of action and reaction. *Physiologia Plantarum*, 100, 264–273.

Peng, C. and Apps, M.J. (1999). Modeling the response of net primary productivity (NPP) of boreal forest ecosystems to changes in climate and fire disturbance regimes. *Ecological Modeling,* 20,175–193.

Percy, K.E., A.H. Legge and S.V. Krupa (2003) Tropospheric ozone: a continuing threat to global forests?. In: D.F. Karnosky, K.E. Percy, A.H. Chappelka, C. Simpson and J. Pikkarainen, Editors, *Air Pollution, Global Change and Forests in the New Millennium*, Elsevier, Amsterdam. 85–118

Percy, K.E., Legge, A.H., Krupa, S.W. (2003). Tropospheric ozone: a continuing threat to global forests? In: Karnosky, D.F., Percy, K.E., Chappelka, A.H., Simpson, C., Pikkarainen, J. (eds.), *Air Pollution and Global Change and Forests in the New Millennium, Development in Environmental Science*, 3, 85–118.

Piao, S. L., Fang, J. Y., He, J.S., Xiao, Y. (2004). Spatial distribution of grassland biomass in China. *Acta Phytoecologica Sinica*, 28, 491–498 (in Chinese with English abstract).

Piao, S.L., Fang, J.Y., Zhou, L.M., Zhu, B., Tan, K., Tao, S. (2005). Changes in vegetation net primary productivity from 1982 to 1999 in China. *Global Biogeochemical Cycles*, 19, GB2027, doi:10, 1029/2004GB002274.

Piikki K., Sellden, G., Pleijel, H. (2004). The impact of tropospheric ozone on leaf number duration and tuber yield of the potato (Solanum tuberosum L.) cultivars Bintje and Kardal. *Agriculture Ecosystems and Environment*, 104, 483–492.

Pitcairn C. E. R., Leith, I. D., Sheppard, L. J., Sutton, M. A., Fowler, D., Munro, R. C., Tang, S., Wilson, D. (1998). The relationship between nitrogen deposition, species composition and foliar nitrogen concentrations in woodland flora in the vicinity of livestock farms. *Environmental Pollution*, 102, 41–48.

Post,W. M., and Kwon, K.C. (2000). Soil carbon sequestration and land-use change: processes and potential. *Global Change Biology,* 6:317–328.

Prentice, I. C. et al . (2001). The carbon cycle and atmospheric carbon dioxide. In Climate Change 2001: the scientific basis (ed. IPCC), pp. 183–237. Cambridge University Press.

Prince, S.D., Goward, S.N. (1995). Global primary production: a remote sensing approach. *Journal of Biogeography*, 22: 815–835.

Qian, Y., Gong, D.Y., Fan, J. F., Leung, L.R., Bennartz, R., Chen, D., Wang, W.G. (2009). Heavy pollution suppresses light rain in China: Observations and modeling. *Journal of Geophysical Research*, 114, D00K02, doi: 10.1029/2008JD011575.

Rahman, A.F., Gamon, J.A., Fuentes, D.A. (2001). Modeling spatially distributed ecosystem flux of boreal forest using hyperspectral indices from AVIRIS imagery. *Journal of Geophysical Research.* 106, 33579–33591.

Ramankutty, N., Foley, J. A. (1998). Characterizing patterns of global land use: An analysis of global croplands data. *Global Biogeochemical Cycles*, 12, 667–685.

Ramankutty, N., Foley. J. A. (1999). Estimating historical changes in global land cover: croplands from 1700 to 1992. *Global Biogeochemical Cycles*, 13, 997–1027.

Reed, B. C. (2006). Trend Analysis of Time–Series Phenology of North America Derived from Satellite Data. *GIScience & Remote Sensing*, 43, 24–38.

Reich, P. B. (1987). Quantifying plant response to ozone: a unifying theory. *Tree Physiology*, 3, 63–91.

Ren, W., Tian, H., Chen, G., Liu, M., Zhang, C., Chappelka, A., Pan, S. (2007a). Influence of ozone pollution and climate variability on grassland ecosystem productivity across China. *Environmental Pollution*, 149: 327–335.

Ren, W., Tian, H., Liu, M., Zhang, C., Chen, G., Pan, S., Felzer, B., Xu, X. F. (2007b). Tropospheric ozone pollution and its influence on net primary productivity and carbon storage in terrestrial ecosystems of China. *Journal of Geophysical Research*, 112, D22S09, doi: 10.1029/2007JD008521.

Ren, W.and Tian, H. Q. (2007). Ozone pollution and terrestrial ecosystem productivity. *Journal of Plant Ecology*, 31(2), 219–230 (In Chinese with English).

Ritchie, J.T., NeSmith, D.S. (1991). Temperature and crop development. In: Hanks, R.J., Ritchie, J.T. (eds.), Modelling Plant and Soil Systems. American Society of Agronomy , Madison, WI, 5–29.

Running, S.W., Nemani, R.R., Heinsch, F.A., Zhao, M., Reeves, M., Hashimoto, H. (2004). A Continuous Satellite-Derived Measure of Global Terrestrial Primary Production. *BioScience,* 54,547–560.

Saleem, A., Loponen, J., Pihlaja, K., Oksanen, E. (2001). Effects of long-term open-field ozone exposure on leaf phenolics of European silver birch (Betula pendula Roth). *Journal of Chemical Ecology*, 27, 1049–1062.

Sasai,T., K. Okamoto, T. Hiyama and Y. Yamaguchi (2007), Comparing terrestrial carbon fluxes from the scale of a flux tower to the global scale. *Ecological modeling,* 208,135-144

Scheller, R.M. and Mladenoff, D.J. (2005). A spatially dynamic simulation of the effects of climate change, harvesting, wind, and tree species migration on the forest composition, and biomass in northern Wisconsin, USA. *Global Change Biology,* 1,307–321.

Schimel, D., House, J., Hibbard, K., Bousquet, P., Ciais,P., Peylin, P., Apps, M., Baker, D., Bondeau, A., Brasswell, R., Canadell, J., Churkina, G., Cramer, W., Denning, S., Field, C., Friedlingstein, P., Goodale, C., Heimann, M., Houghton, R.A., Melillo, J., Moore III, B., Murdiyarso, D., Noble, I., Pacala, S., Prentice, C., Raupach, M., Rayner, P., Scholes, B., Steffen, W., Wirth, C. (2001). Recent patterns and mechanisms of carbon exchange by terrestrial ecosystems, *Nature* 414, 169–172.

Schimel, D., Melillo, J.M., Tian, H., McGuire, A.D., Kicklighter, D.W., Kittel, T., Rosenbloom, N., Running, S., Thornton, P., Ojima, D., Parton, W., Kelly, R., Sykes,

M., Neilson, R., Rizzo, B. (2000). Contribution of increasing CO2 and climate to carbon storage by ecosystems in the United States. *Science,* 287, 2004–2006.

Schindler, D. W. (2001). The cumulative effects of climate warming and other human stresses on Canadian freshwaters in the new millennium. *Canadian Journal of Fisheries and Aquatic Sciences*, 58, 18–29.

Schindler, D. W. and , Bayley, S. E. (1993). The biosphere as an increasing sink for atmospheric carbon: estimates from increased nitrogen deposition. *Global Biogeochemical Cycles*, 7, 717–733.

Scurlock, J.M.O., Cramer, W., Olson, R.J., Parton, W.J., Prince, S.D. (1999). Terrestrial NPP: Toward a consistent data set for global model evaluation, *Ecological Applications,* 9, 913–919.

Severinghaus, J.P., Beaudette, R., Headly, M.A., Taylor, K., Brook, E.J. (2009). Oxygen-18 of O_2 Records the Impact of Abrupt Climate Change on the Terrestrial Biosphere. Science, 324(5933), 1431–1434.

Sha, W.Y., Shao, X.M, Huang, M.(2002). Climate warming and its impact on natural regional boundaries in China in the 1980s. *Science in China (Series D)*, 1099-1113.

Shafer, S. L., Bartlein, P.J., Thompson, R.S. (2001). Potential changes in the distributions of western North America tree and shrub taxa in future climate scenarios. *Ecosystems*, 4:200–215.

Shi, X., Yu, D., Pan, X., Sun, W., Wang, H., Gong, Z. (2004). 1:1,000,000 soil database of China and its application. In: Proceedings of 10th National Congress of Soil Science of China. Science Press, Beijing, China, pp142–145

(In Chinese).

Sims, D., Luo, H., Hastings, S., Oechel,W.C., Rahman, A.F., Gamon, J.A.(2006). Parallel adjustments in vegetation greenness and ecosystem CO2 exchange in response to drought in a Southern California chaparral ecosystem. *Remote Sensing of Environment,* 103, 289–303.

Sims, H. (1999). One–fifth of the Sky: China's Environmental Stewardship. *World Development*, 27(7), 1227–1245.

Sitch, S., Cox, P. M., Collins, W. J., Huntingford, C. (2007). Indirect radiative forcing of climate change through ozone effects on the land–carbon sink. *Nature*, 448, 791–794.

Skelly, J.M., Innes, J.L., Savage, J.E., Snyder, K.R., VanderHeyden, D., Zhang, J., Sanz, M.J. (1999) Observation and confirmation of foliar ozone injury symptoms of native plant species of Switzerland and southern Spain. *Water, Air and Soil Pollution*, 116, 227–234.

Smith, G., Coulston, J., Jepsen, E., Prichard, T. (2003). A national ozone biomonitoring program – results from field surveys of ozone sensitive plants in northeastern forests (1994–2000). *Environmental Monitoring and Assessment*, 87, 271–291.

Smith, T.M., Halpin, P.N., Shugart, H. H. (1995). Global forest. In: Strzepek, K.M., Smith, J.B. As Climate Change: International Impact s and Implications. Cambridge: Cambridge University Press, pp59–78.

Streets, D. G., Waldhoff, S. (2000). Present and future emission of air pollutants in China: SO_2, NO_x, CO. *Atmospheric Environment*, 34, 363–374.

Sun, Ge, Noormets, A., Chen, J., McNulty, S.G.(2008) Evapotranspiration estimates from

eddy covariance towers and hydrologic modeling in managed forests in Northern Wisconsin, USA, *Agricultural and Forest Meteorology,* 148, 257-267

Tao, B. (2005). Preliminary study on spatio–temporal patterns of terrestrial carbon budgets driven by climatic change in China. Postdoctoral report of Institute of Geographical Science and Natural Resources Research (In Chinese with English Abstract).

Tao, B., Cao, M. K., Li, K. R., Gu, F. X., Ji, J. J., Huang, M., Zhang, L. M. (2007). Spatial patterns of terrestrial net ecosystem productivity in China during 1981–2000. *Science in China (Series D)*, 50(5), 745–753.

Tao, F. L., Yokozawa, M., Liu, J. Y., Zhang, Z. (2008). Climate–crop yield relationships at provincial scales in China and the impacts of recent climate trends. *Climate Research*, 38, 83–94.

Tao, F. L., Yokozawa, M., Zhang, Z., Hayashi, Y., Grassl, H., Fu, C. B. (2004). Variability in climatology and agricultural production in China in association with the East Asian summer monsoon and EL Nino Southern Oscillatin. *Climate Research*, 28, 23–30.

Tao, F. L., Zhang, Z., Liu, J. Y., Yokozawa, M. (2009). Modelling the impacts of weather and climate variability on crop productivity over a large area: A new super–ensemble–based probabilistic projection. *Agricultural and Forest Meteorology*. 149(8), 1266–1278.

Tao, F.L., Yokozawa, M., Zhang, Z., Xu, Y., Hayashi, Y. (2005). Remote Sensing of Crop Production in China by Production Efficiency Models: Models Comparisons, Estimates and Uncertainties. *Ecological Modelling*, 183, 385–396.

Tate, K.R., Scott, N.A., Parshotam, A., Brown, L., Wilde, R.H., Giltrap, D.J., Trustrum, N.A., Gomez, B., Ross, D.J. (2000). A multi-scale analysis of a terrestrial carbon budget: is New Zealand a source or sink of carbon? Agriculture, *Ecosystems and Environment*, 82, 229–246.

Thornton, P. E., Running, S.W., White, M.A. (1997). Generating surfaces of daily meteorological variables over large regions of complex terrain. *Journal of Hydrology*, 190, 241–251.

Tian, H. Q., Hall, C. A. S., Qi, Y. (1998a). Modeling primary productivity of the terrestrial biosphere in changing environments: Toward a Dynamic Biosphere Model. *Critical Review in Plant Sciences*, 17(5), 541–557.

Tian, H. Q., Liu, M. L., Zhang, C., Ren, W., Chen, G. S., Xu, X. F. (2005). DLEM – The Dynamic Land Ecosystem Model, User Manual, the ESRA Laboratory, Auburn University.

Tian, H. Q., Melillo, J. M., Kicklighter, D. W., McGuire, A. D., Helfrich, J., Moore, B. III, Vörösmarty, C. J. (1998b). Effect of interannual climate variability on carbon storage in Amazonian ecosystems. *Nature*, 3, 664–667.

Tian, H. Q., Melillo, J. M., Kicklighter, D. W., McGuire, A. D., Helfrich, J. (1999). The sensitivity of terrestrial carbon storage to historical atmospheric CO_2 and climate variability in the United States. *Tellus*, 51(B), 414–452.

Tian, H. Q., Melillo, J. M., Kicklighter, D. W., Pan, S. F., Liu, J. Y., McGuire, A. D., Moore, B.III. (2003). Regional carbon dynamics in monsoon Asia and its implications for the global carbon cycle. *Global and Planetary Change*, 37, 201–217.

Tian, H. Q., Wang, S., Liu, J., Pan, S., Chen, H., Zhang, C., Shi, X. (2006). Patterns of
soil nitrogen storage in China. *Global Biogeochemical Cycles*, 20, GB1001, doi:
10.1029/2005GB002464.

Tian, H., Chen G.S., Liu M.L., Zhang C., Sun G., Lu C.Q., Xu X.F., Ren W., Pan S.F.,
Chappelka, A.(2009a) Model estimates of net primary productivity,
evapotranspiration, and water use efficiency in the terrestrial ecosystems of the
southern United States during 1895–2007. Forest Ecology and Management (in
press).

Tian, H., Xu, X., Zhang, C., Ren, W., Chen, G., Liu, M., Lu, D., Pan, S. (2008).
Forecasting and Assessing the Large–scale and Long–term Impacts of Global
Environmental Change on Terrestrial Ecosystems in the United States and China.
In: S. Miao, S. Carstenn, and M. Nungesser (Eds), Real World Ecology:
large–scale and long–term case studies and methods. Springer–Verlag, New York,
pp235–266.

Tian, H.Q., Melillo, J. M., Kicklighter, D. W., McGuire, A. D., Helfrich, J., Moore, III
B., Vörösmarty, C. J. (2000). Climatic and biotic controls on annual carbon
storage in Amazonian ecosystems. *Global Ecology and Biogeography,* 9,
315–336.

Tian, H.Q., Ren, W., Xu, X.F., Liu, M.L., Chen, G.S., Lu, C.Q., Wang, Q. (2009b).
Sources and sinks of carbon (CO_2 and CH_4) in agricultural ecosystem of China in
response to multiple environmental changes, *Global Change Biology,* (to be
submitted)

Tilman, D., Cassman, K.G., Matson, P.A., Naylor, R. and Polasky, S. (2002).

Agricultural sustainability and intensive production practices.   Nature, 418, 671–677.

Tjoelker, M. G., Volin, J. C., Oleksyn, J., Reich, P. B. (1995). Interaction of ozone pollution and light effects on photosynthesis in a forest canopy experiment. *Plant, Cell and Environment*, 18, 895–905.

Torsethaugen, G., Pell, E. J., Assmann, S. M. (1999). Ozone inhibits guard cell K+ channels implicated in stomatal opening. *Proceedings of the National Academy of Sciences*, 96, 13577–13582.

Turner, D.P., Ritts, W.D., Cohen, W.B., Maeirsperger, T.K., Gower, S.T., Kirschbaum, A.A., Running, S.W., Zhao, M., Wofsy, S.C., Dunn, A.L., Law, B.E., Campbell, J.L., Oechel, W.C., Kwon, H., Meyers, T.P., Small, E.E., Kurc, S.A., Gamon, J.A.(2005). Site-level evaluation of satellitebased global terrestrial GPP and NPP monitoring. *Global Change Biology*. 11, 666–684

Van Aardenne, J.A., Carmichael, G.R., Levy, H. I., Streets, D., Hordijk, L. (1999). Anthropogenic NO_x emissions in Asia in the period 1990–2020. *Atmospheric environment*. 33, 633–646.

VEMAP Members (1995). Vegetation/ecosystem modeling and analysis project: Comparing biogeography and biogeochemistry models in a continental-scale study of terrestrial ecosystem responses to climate change and CO2 doubling. *Global Biogeochemical Cycles*, 9, 407–437.

Vitousek, P.M., Mooney, H.A., Lubchenco, Melillo, J.M. (1997). Human Domination of Earth's ecosystems. *Science*, 277(5325), 494–499.

Volk, M., Bungener, P., Oiscontat, F., Montani, M., Rgfuhrer, J. (2006). Grassland yield

declined by a quarter in 5 years of free–air ozone fumigation. *Global Change Biology*. 12, 74–83.

Vollenweider, P., Ottiger, M., Ginthardt-Goerg M.S. (2003) Validation of leaf ozone symptoms in natural vegetation using microscopical methods. *Environment Pollution,* 124,101-118

Vukicevic, T., Braswell, B. H., Schimel, D. (2001). Diagnostic study of temperature controls on global terrestrial carbon exchange. *Tellus*, 53(B), 150– 170.

Wang, C. Y. (1995). A Study of the Effect of Ozone on Crop. *Journal of Applied Meteorology*. 6(3), 343–349 (In Chinese with English).

Wang, C. Y., .Guo, J. P., Bai, Y. M., Wen, M. (2002). Experimental study of impacts by increasing ozone concentration on winter wheat. *Acta Meteorologica Sinica*, 60(2), 239–242 (In Chinese with English).

Wang, C., Prinn, R. G., Sokolov, A. (1998). A Global Interactive Chemistry and Climate Model: Formulation and Testing. *Journal of Geophysical Research*, 103(D3), 3399–3418.

Wang, S., Tian, H. Q., Liu, J., Pan, S. (2003). Pattern and change in soil organic carbon storage in China: 1960s–1980s. *Tellus*, 55(B), 416–427.

Wang, X. K., Feng, Z. W., Oyang, Z. Y. (2001a). Vegetation carbon storage and density of forest ecosystem in China. *Chinese Journal of Applied Ecology*. 12(1), 13–16 (In Chinese with English Abstract).

Wang, X. K., Feng, Z. W., Qyang, Z. Y. (2001b). The impact of human disturbance on vegetation carbon storage in forest ecosystem in China. *Forestry Ecology and Management*. 148, 117–123.

Wang, X. K., Guo, Q. (1990). The effects of ozone on respiration of the plants Fuchsia hybrida Voss. And Vicia faba L. *Environmental Science*. 11, 31–33 (In Chinese with English).

Wang, X. K., Manning, W., Feng, Z. W., Zhu, Y. G. (2007). Ground–level ozone in China: Distribution and effects on crop yields. *Environmental Pollution*. 147(2), 394–400.

Wang, X. P. & Mauzerall, D.L. (2004). Characterizing distributions of surface ozone and its impact on grain production in China, Japan and South Korea: 1990 and 2020. *Atmospheric Environment*, 38, 4383–4402.

Wang, Y.X., Zhao, S.D., Niu, D., Li, X.Y. (2002). Forests ecosystem carbon pools and its regional characteristic in China. In: Li, K.R., (eds). Land use change, greenhouse gases emissions and terrestrial ecosystem carbon cycle. pp: 187–203, China Meteorological Press, Beijing.

Wang, Z. P., Han, X. G., Li, L. H. (2008). Effects of grassland conversion to cropland on soil organic carbon in the temperate Inner Mongolia. *Journal of Environmental Management*, 86, 529–534.

White, A., Cannell, M.G.R., Friend, A.D. (1999). Climate change impacts on ecosystems and the terrestrial carbon sink: a new assessment. *Global Environmental Change*, 9, S21-S30.

Wild, O., Akimoto, H. (2001). Intercontinental transport of ozone and its precursors in a three–dimensional global CTM. *Journal of Geophysical Research*, 106, 27729–27744.

Woodbury, P.B., Smith, J.E., Heath, L.S. (2007). Carbon sequestration in the US Forest

sector from 1990 to 2010. *Forest Ecology and Management*,
doi:10.1016/j.foreco.2006.12.008.

Wu, L.,Cai, Z. (2007). Estimation of the change of topsoil organic carbon of croplands in China based on long–term experimental data. *Ecology and Environment*, 16, 1768–1774.

Xiao, X. M., Jiang S., Wang Y. F., Ojima, D. S., and Bonham, C. D. (1995). Temporal variation in aboveground biomass of Leymus chinense steppe from species to community levels in the Xilin River Basin, Inner Mongolia, China. *Plant Ecology*, 123, 1–12.

Xiao, X. M., Melillo, J. M., Pan, Y. D., McGuire, A. D., Helfrich, J. (1998). Net primary productivity of terrestrial ecosystem in China and its equilibrium reponses to changes in climate change and atmopheric CO_2 concentration. *Acta Phytoecologica Sinica*, 1998, 22, 97–118.

Xiao, X. W. (2005). Atlas of forest resources of China. China forestry press, Beijing.

Xiao, X.M., Zhang, Q.Y., Braswell, B., Urbanski, S., Boles, S., Wofsy, S., Moore III, B. (2004). Modeling gross primary production of temperate deciduous broadleaf forest using satellite images and climate data. *Remote Sensing of Environment*, 91: 256–270.

Xu, Z. J., Zheng, X. H., Wang, Y. S., Han, S. H., Huang, Y., Zhu, J. G., Butterbach–Bahl, K. (2004). Effects of elevated CO_2 and N fertilization on CH_4 emissions from paddy rice fields. *Global Biogeochemical Cycles*, 18, GB3009 doi:10.1029/2004GB002233.

Xu, D. F. (1983). Statistical Data of Agricultural Production and Trade in Modern

China. Shanghai People's Press, Shanghai.

Yan, H. M., Cao, M. K., Liu, J. Y., Tao, B. (2007). Potential and sustainability for carbon sequestration with improved soil management in agricultural soils of China. Agriculture *Ecosystems and Environment*, 121, 325–335.

Yan, H. M., Liu, J. Y., Huang, H. Q., Tao, B., Cao, M. K. (2009). Assessing the consequence of land use change on agricultural productivity in China. *Global and Planetary Change*, 67(1), 13–19.

Yan, H.M., Liu, J.Y., Huang, H.Q., Tao, B., Cao, M.K. (2009). Assessing the consequence of land use change on agricultural productivity in China. *Global and Planetary Change,* doi:10.1016/j.gloplacha.2008.12.012.

Yang, J. P., Ding, Y. J., Chen R. S., Liu L. Y (2002). The interdecadal fluctuation of dry and wet climate boundaries in China in recent 50 years. *Acta Geographica Sinica*, 57, 655–661.

Yang, Y. H., Mohammat, A., Feng, J. M., Zhou, R., Fang, J. Y. (2007). Storage, patterns and environmental controls of soil organic carbon in China. *Biogeochemistry*, 84, 131–141.

Younglove, T., McCool, P. M., Musselmann, R. C., Kahl, M. E. (1994). Growth–stage dependent crop yield response to ozone exposure. *Environmental Population*, 86, 287–295.

Yu, X .F., Zhuang, D. F., Hou, X. Y., Chen, H. (2005). Forest phenological patterns of Northeast China inferred from MODIS data. *Journal of Geographical Sciences*, 15, 239–246.

Yun, S.C., Park, E.W., Laurence, J.A. (2001) Simulation of 1-year-old Populus

tremuloides response to ozone stress at Ithaca, USA, and Suwon, Republic of
Korea. *Environmental Pollution,* 112, 253-260

Zhang, C., Tian, H. Q., Liu, J., Wang, S., Liu, M., Pan S., Shi, X. (2005). Pools and
distributions of soil phosphorus in China. *Global Biogeochemical Cycles*, 19,
GB1020, doi:10.1029/2004GB002296.

Zhang, C., Tian, H., Chappelka, A., Ren, W., Chen, H., Pan, S., Liu, M., Styers, D.
M.,Chen, G., Wang, Y. (2007). Impacts of climatic and atmospheric changes on
carbon dynamics in the Great Smoky Mountain. *Environmental Pollution*, 149,
336–347.

Zhang, N., Yu, G. R., Yu, Z. L., Zhao, S. D. (2003). Analysis on factors affecting net
primary productivity distribution in Changbai Mountain based on process model
for landscape scale. *Journal of Applied Ecology*, 14(5): 659–664 (In Chinese with
English Abstract).

Zhang, W., Yu, Y.Q., Sun, W.J., Huang, Y. (2007). Simulation of Soil Organic Carbon
Dynamics in Chinese Rice Paddies from 1980 to 2000. *Soil Science Society of
China*, 17, 1–10.

Zhao, M. and Zhou, G. S. (2004). Forest inventory data(fid)–based biomass models and
their prospects. *Chinese Journal of Applied Ecology*, 15(8),1468–1472 (In
Chinese with English Abstract).

Zheng, D., Prince, S., Wright, R. (2003). Terrestrial net primary production estimates for
0.5∘ grid cells from field observations-a contribution to global biogeochemical
modeling. *Global Change Biology.* 9, 46–64.

Zhou, G. S. And Zhang, X. S. (1995). A natural vegetation NPP model. *Acta*

Phytoecoligica Sinica, 19(3), 193–200 (In Chinese with English Abstract).

Zuo, H. C., Wu, X. M., Lv, S. (2003). The changing tendency analysis of temperature and precipitation in China during the past 50y. Scientific and technical innovation of meteorology and atmospheric sciences development in new century–Climate change and climatic system. The climatology committee of Meteorological Society of China. China Meteorological press, pp382–386 (In Chinese with English Abstract).

Appendix I

Table 1 Agricultural experiment stations in China used in the study (selected)

Name	latitude	Longitude	Elevation (m)	Soil type	Cropping system
Harbin(Heilongjiang)	45°45'	126°46'	142.3	black soil	soybean
Yanbian(Jilin)	42°53'	129°28'	176.8	sand soil	soybean
Jinzhou(Liaoning)	41°09'	121°10'	27.4	loam	soybean
Hulunbeier(Neimeng)	48°	122°44'	306.5	sand-loam	soybean
Zunhua(Hebei)	40°12'	117°57'	54.9	loam	rice-wheat
Xuzhou(Jiangshu)	40°12'	117°57'	637.5	loam	wheat-rice
Chenggu(Shanxi)	33°10'	107°20'	750	Sand	rice
Taizhoushi(Guizhou)	28°40'	121°26'	4.6	loam	rice-rice-rice
Huishui(Guizhou)	26°08'	106°38'	990.9	loam	rice-wheat
Dali(Yunnan)	25°42'	110°11'	1990.5	loam	soybean-rice
Guyang(Neimeng)	41°02'	110°03'	1360.4	sand-loam	spring wheat
Hami(Xinjiang)	42°94'	93°31'	737.2	sand-loam	spring wheat
Zhangbei(Hebei)	114°42'	41°09'	1393.3	loam	spring wheat
Ruomuhong(Hainan)	36°26'	96°25'	2790.4	sand-loam	spring wheat
Laiyang(Shandong)	36°58'	120°44'	675	sand-loam	wheat-maize
Yongni(Ningxia)	38°15'	106°15'	524.95	loam	spring wheat
Nenjiang(Heilongjiang)	49°10'	125°14'	242.2	loam	spring wheat
Tongzhou(Hebei)	39°55'	116°38'	43.3	sand-loam	maize-wheat
Yuncheng(Shandong)	35°03'	111°03'	365	sand	wheat-wheat
Xuzhou(Jiangshu)	34°17'	117°09'	41.2	loam	wheat-maize
Baicheng(Jilin)	45°38'	122°50'	169.9	sand	maize-maize
Chaoyangshi(Liaoning)	41°33'	120°27'	568	loam	maize-maize
Jiuquan(Gansu)	39°46'	98°29'	1477.2	loam	wheat-maize

Table 2 Comparisons of main input data, plant functional types, and land use and land cover change in two models (DLEM and TEM)

Input requirements

	Mean temperature	Min/Max-temperature	Precip-itation	Relative humidity	Solar radiation	Soil texture	Soil depth	Elevat-ion
DLEM	D	D	D	D	D	X	X	X
TEM	M		M		M	X		X

Considered plant functional types

	Forests	Grassland	Shrub land	Crop	Meadows	Desert	Wetland
DLEM	Yes	Yes	Yes, evergreen/ deciduous	Yes	Meadow	Open shrub	Wetland
TEM	Yes	Yes	Yes, arid, Mediterranean	Yes, grass	Wet tundra	Arid shrub	

Processes of land use and land cover change

	upland crops		paddy land	urban	crop harvest	fertilizer application	land conversion
DLEM	corn, spring wheat/ winter wheat	corn, spring wheat/ winter wheat	rice	lawn	prescribed	prescribed	yes natural-crop, crop-natural, urbanization
TEM	harvested C_3 grass	harvested C_3 grass	harvested, no water limited grass	grass	degree-day	prescribed: un-limited (>100 kg N/ha), limited <100 kg N/ha	Yes natural-crop, crop-natural, urbanization

Where, required variables are indicated with an 'X', except for climate variables where models required daily (D) or monthly (M) inputs; 'yes' means the plant functional type is included in the simulation.

Table 3 Key ecosystem processes related to ozone effects in two models

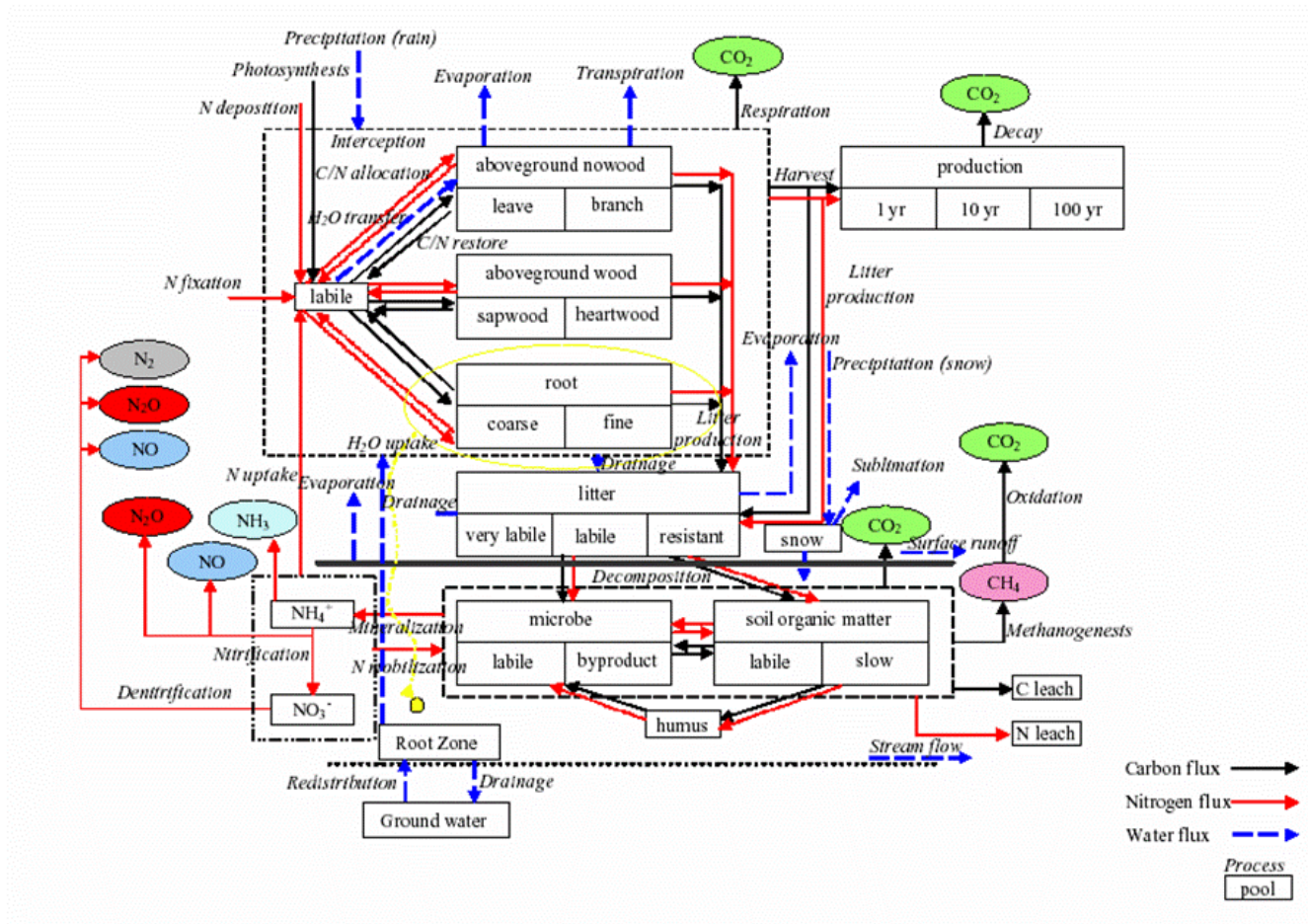|	TEM	DLEM
PET/ET	Jensen-Haise [1963]	Modified Penman-Monteith
Carbon uptake by vegetation	Multiple limitation	Farquhar
Nitrogen uptake by vegetation	Monthly	Daily
Ci = f(Ca)	yes	yes
Stomatal conductance = f(Ca)	no	yes
Vegetation C/N = f(Ca)	no	yes
Plant respiration Q10	1.5-2.5	
No. of soil water layer	1	3
No. of vegetation carbon/nitrogen pool	1	4
No. of litter/soil carbon/nitrogen pool	2	3
Equilibrium	Dynamic simulation (10 to 3000 years) until carbon and nitrogen pools come into balance(g.g., NEP = 0, NMIN = NUPTAKE, N input = N lost)	Dynamic simulation (500 to 3000 years) until carbon and nitrogen pools come into balance(e.g., NEP = 0, NMIN = NUPTAKE, N input = N lost)
Temporal scale	monthly	Daily/monthly/annual
Ozone input data	Monthly	Daily
Stomotal conductance		Yes
persistent damage	Yes	Yes

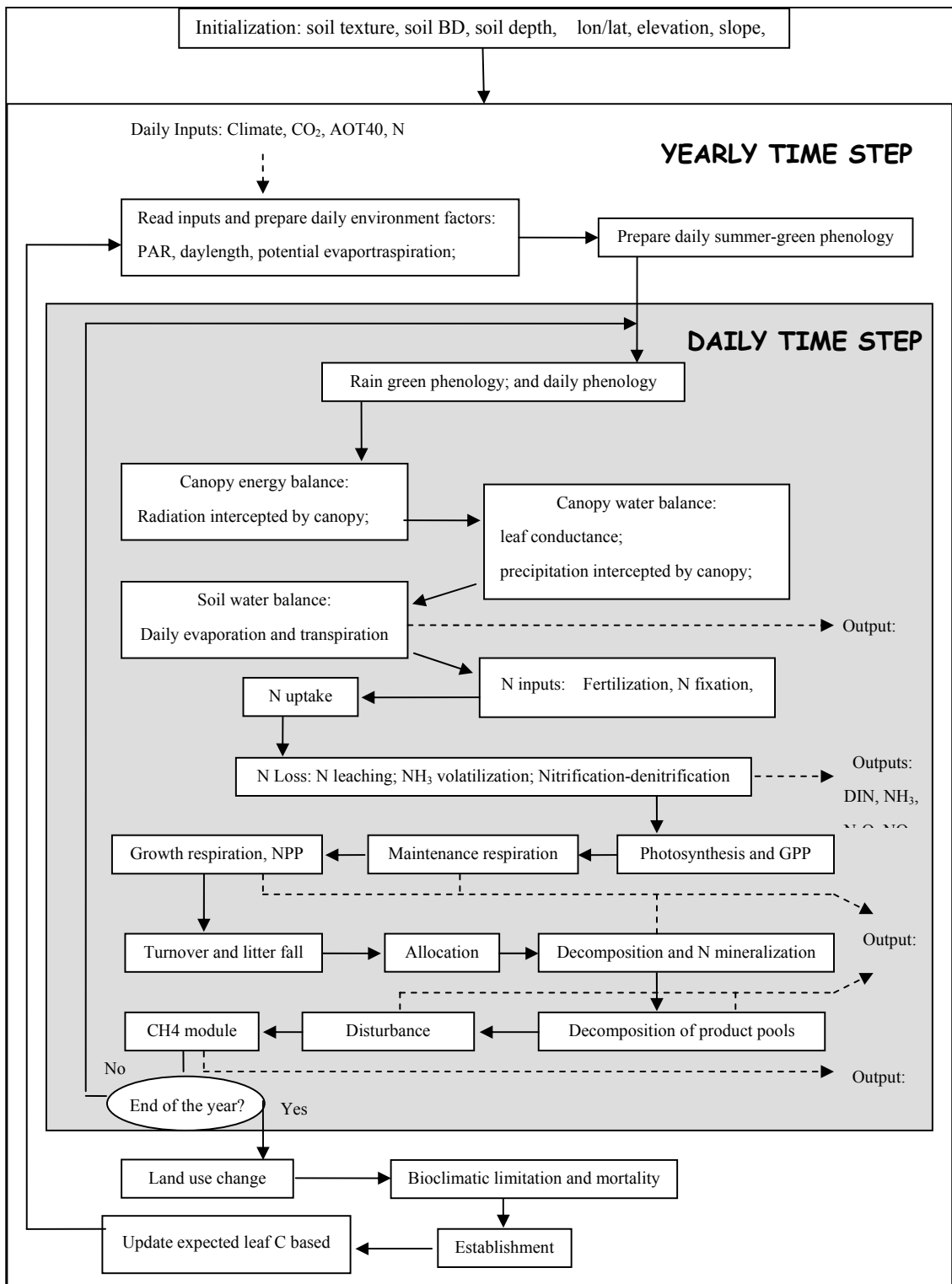Appendix II

Figure 1 Structure of DLEM

Figure 2 Flow chart of DLEM

Table 1 Some important parameters in DLEM

	Description	deciduous broadleaf	evergreen coniferous	grass	shrub	crop
k_{ext}	Light extinction coefficient	0.7	0.7	0.6	0.5	0.85
α_{sw}	Albedo	0.2	0.27	0.2	0.2	0.2
α_{O3}	Ozone coefficient	2.6×10^{-6}	0.7×10^{-6}	2.6×10^{-6}	2.6×10^{-6}	3.9×10^{-6}
$R_{year,leaf}$	Residential time of leaf	1	4	1	1	1
$R_{year,sapwood}$	Residential time of sapwood	20	15	1	20	1
$R_{year,fineroot}$	Residential time of fine root	2.5	2.5	1.5	2.5	0.5
$R_{year,coarseroot}$	Residential time of coarse root	80	80	20	80	20
MIN_CN_{leaf}	Leaf C:N ratio	21	40	30.9	23.6	21
$k_{alloc,crt}$	Fraction of carbon allocated to root	0.7	0.6	0.8	0.6	0.2
$CNmrt_{leaf}$	C:N ratio of leaf litter	49	93	40	83	49
$CNmrt_{frt}$	C:N ratio of fineroot litter	42	60	49	60	49
CN_{swd}	C:N ratio of sapwood	50	70	40	42	62
CN_{hwd}	C:N ratio of heartwood	150	150		150	82
CN_{crt}	C:N ratio of coarseroot	73	80	60	60	28.13
CN_{frt}	C:N ratio of fineroot	40	42	35	40	28.13
CN_{prd}	C:N ratio of reproduct pool	40	40	30	30	30
k_{nup}	Nitrogen uptake coefficient	0.47	0.42	0.12	0.1	0.35
mg0	maximum CH4 production rate	0.25	0.25	0.25	0.25	0.25
Omax	maximum CH4 oxidation rate	0.19	0.09	0.22	0.11	0.1
gdd_base	Leaf onset temperature	2	5	0	0	0
Sgdd	Maximum sum of heat when the leaf cease to grow	200	500	200	1000	700
Z1	Fraction of root in the upper soil layer	0.8	0.8	0.8	0.7	0.8
minT	Minimum temperature for survive	-17	-2	--	--	--

maxT	Maximum temperature for survive	15.5	22	15.5	--	--
minGDD	Minimum annual heat accumulation for survive	1200	900	--	--	--
leafRootMax	Maximum leaf:root ratio	1	1	0.75	1	0.75
G_BL	Boundary layer conductance	0.01	0.008	0.01	0.008	0.04

Table 2 Main inputs to DLEM

Category	Symbol	Units	Explanation	Timestep
Climate	TMAX	^0C	Maximum temperature	Daily
	TMIN	^0C	Minimum temperature	Daily
	PPT	mm/day	Precipitation	Daily
	VPD	Pa	Vapor pressure deficit (optional)	Daily
	SRAD	w/m^2	Short wave radiation (optional)	Daily
Atmospheric composition	CO_2	ppmv	Atmospheric CO_2 concentration	Yearly
	AOT40	ppb-hr	Accumulated dose over a threshold of 40 ppb through the past 30 days [@]	Daily
	NH4DEP	g N/m^2/day	NH_4 deposition	Daily
	NO3DEP	g N/m^2/day	NO_3 deposition	Daily
Land-use	LUCC	0 / 1	Land use type: 0 means natural land type, 1 means cropland	Yearly
	IRRIG	0 / 1	Cropland irrigation: 0 means no irrigation; 1 means irrigation applied	Yearly
	FERT	0 / 1	Cropland fertilization: 0 means no fertilization; 1 means fertilization applied (i.e. no N limitation)	Yearly
Disturbances	DISTURB	Integer numbers	Disturbances. Each number is denotes a disturbance type.	Daily

Vegetation Maps	VEG, POTVEG	Index of PFTs	Required for non-dynamic mode. VEG: Potential vegetation; POTVEG: contemporary vegetation map	Initialization
Geography and topography	LAT	Degree	Latitude	Initialization
	LON	Degree	Longitude	Initialization
	ELEV	Meter	Elevation (from sea level)	Initialization
	ASPECT	degree	Aspect	Initialization
	SLOPE	Degree	Slope	Initialization
	APET	mm/yr	Long-term average potential evapor-transpiration	Initialization
	APPT	mm/yr	Long-term average precipitation	Initialization
Soil	BD	g/cm^3	Bulk density	Initialization
	DEPTH	meter	Soil thickness	Initialization
	CLAY	%	Clay content	Initialization
	SAND	%	Sand content	Initialization
	SILT	%	Silt content	Initialization
	PH	pH	Soil pH	Initialization

@ AOT40 is the sum of the differences between the hourly mean ozone concentration (ppb) and 40 ppb for each hour when the concentration exceeds 40 ppb, accumulated during daylight times.