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Carbon sinks and sources in China's forests during 1901–2001

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Abstract

This paper reports the annual carbon (C) balance of China's forests during 1901–2001 estimated using the Integrated Terrestrial Ecosystem C-budget model (InTEC). Annual carbon source and sink distributions are simulated for the same period using various spatial datasets including land cover and leaf area index (LAI) obtained from remote sensing, soil texture, climate, forest age, and nitrogen deposition. During 1901–1949, China's forests were a source of $21.0 \pm 7.8\,\mathrm{Tg\,C\,yr^{-1}}$ due to disturbances (human activities). Its size increased to $122.3 \pm 25.3\,\mathrm{Tg\,C\,yr^{-1}}$ during 1950–1987 due to intensified human activities in the late 1950s, early 1960s, 1970s and early 1980s. The forests became large sinks of $176.7 \pm 44.8\,\mathrm{Tg\,C\,yr^{-1}}$ during 1988–2001, owing to large-scale plantation and forest regrowth in previously disturbed areas as well as growth stimulation by nondisturbance factors such as climatic warming, atmospheric CO_2 fertilization, and N deposition. From 1901 to 2001, China's forests were a small carbon source of 3.32 Pg C, about $32.9 \pm 22.3\,\mathrm{Tg\,C\,yr^{-1}}$. The overall C balance in biomass from InTEC generally agrees with previous results derived from forest inventories of China's forests. InTEC results also include C stock variation in soils and are therefore more comprehensive than previous results. The uncertainty in InTEC results is still large, but it can be reduced if a detailed forest age map becomes available.

Keywords: Carbon sink; Carbon source; Forest; Carbon cycle; China

1. Introduction

Estimation of regional or national carbon (C) sinks and sources is a core component for Intergovernmental Panel on Climate Change (IPCC) reports. Nations participating in the UN Framework Convention on Climate Change (UNFCCC) are engaged in providing national inventories of net greenhouse gas emissions including C sources and sinks associated with forests, and in meeting their obligations for emission reductions under the Kyoto Protocol (Houghton, 1999; Chen et al., 2000a–c; Cao et al., 2003). The knowledge of the distribution of C sources and sinks and their changes over time is critical for understanding the mechanisms controlling the global terrestrial C cycle and the sustainability of current C sinks (Chen et al., 2003). Such knowledge is also essential for formulating climate change policies (Houghton, 1999; Cao et al., 2003). Therefore, many researchers are currently focusing on the C budget studies at national or regional scales.

Relative to other ecosystems, forests store vast amounts of C and play a dominant role in the C cycle. Small shifts in the balance between photosynthesis and ecosystem respiration can result in a great difference in the net uptake or emission of carbon dioxide (CO_2) from forests to the atmosphere (Goodale et al., 2002; Pregitzer and Euskirchen, 2004). Hence, quantification of forest C sources and sinks is an important part of national inventories of net greenhouse gas emissions (Phillips et al., 2000). Forest and soil inventory data provide reliable ground-based estimates of C stocks and fluxes across heterogeneous regions and statistically represent the land-use change and disturbance (Brown and Lugo, 1984; Bui, 2004; Pan et al., 2004). Although existing forest and soil inventory databases offer the potential for estimating regional or national ecosystem C budgets, ecological modeling is currently the best

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approach to integrating all available data sets for large-scale applications (Holmes et al., 2004). At present, the application of inventory data to global change study has several limitations. First, national forest inventories were generally made once in several years due to cost, hence they could not give the annual forest C budget. Second, forest inventory methods could not integrate the effects of disturbance factors (i.e., forest fire, insect-induced mortality and harvest) and nondisturbance factors (i.e., CO_2 concentration, climate change and N deposition) on the C cycle (Chen et al., 2000a–c). Third, extrapolation of forest inventory data has relatively high spatial variability due to the limited number of plots in large regions. Recently, scientists have begun to integrate satellite remote sensing data and C process models in forest C cycle studies, having concluded that satellite remote sensing and ecosystem models are so far the most effective tools to describe detailed spatial distribution patterns of C sources and sinks in forests at regional scales (Chen et al., 2003).

Although the northern hemisphere could occupy a large percentage of the missing sink of the atmosphere CO_2, the magnitude, geographical distribution and causes of a northern mid-latitude terrestrial C sink are uncertain (Houghton and Hackler, 2003). Because China is the world's third largest country and has diverse climates and biomes mostly within northern mid-latitudes (Cao et al., 2003; Houghton and Hackler, 2003), quantification of the spatial distribution of C sinks and sources in China's forests is of great significance for understanding the global C dynamics. Many studies have been conducted to estimate the net primary productivity (NPP) and C budget in China's forests and terrestrial ecosystems in the last decade (Fang et al., 1998; Wang et al., 2001a, b, 2003, 2004; Liu et al., 2004). For example, based on forest inventory data, Fang et al. (2001) found substantial increases in C storage of China's forests during the 1980s and 1990s due to national afforestation and reforestation programs. However, estimates of the magnitude of China's terrestrial ecosystem C balance are rather inconsistent, ranging from a net sink of $0.048\,\mathrm{Pg\,C\,yr^{-1}}$ (Zhou et al., 2000), $0.1\,\mathrm{Pg\,C\,yr^{-1}}$ (Streets et al., 2001), $0.02\,\mathrm{Pg\,C\,yr^{-1}}$ (Fang et al., 2001), $0.04\,\mathrm{Pg\,C\,yr^{-1}}$ (Goodale et al., 2002) and $0.07\,\mathrm{Pg\,C\,yr^{-1}}$ during 1981–1998 (Cao et al., 2003) to a net source of $0.06\,\mathrm{Pg\,C\,yr^{-1}}$ (Houghton, 1999) and mean loss of $0.09\,\mathrm{Pg\,C\,yr^{-1}}$ due to land use change from 1700 to 2000 (Houghton and Hackler, 2003). Although several such large-scale C budget projects have been carried out in China (e.g., Fang et al., 1998, 2001; Pan et al., 2004; Liu et al., 2004), few of them have attempted to estimate interannual variations and trends in the terrestrial C budget in China in response to climate variability (Cao et al., 2003). Furthermore, no effort has been made to analyze the spatiotemporal patterns of C balance in China's forests at centennial scale and detailed spatial resolution.

China has a total landmass of 960×10^6 ha and a rich variety of ecosystems. The country has a north–south gradient in temperature and an east–west gradient in precipitation driven by the summer monsoon (Hou, 1983; Fullen and Mitchell, 1994; Menzies, 1996). The natural forest ecosystems in the east vary from boreal forests in the northeastern provinces; through cold-, then warm-temperate deciduous forests; to mixed evergreen and deciduous subtropical forests; to evergreen tropical forests in the south (Hou, 1983; Houghton and Hackler, 2003). Most of China's forests are located on hills, mountains and high plateaus. This spatial distribution of forests has made the simulation of spatial variability of C sinks and sources in China's forest difficult.

Since changes in land use and management of the past play a major role in the current C balance (Houghton, 1999), we attempted to analyze temporal changes in C balance of China's forest at 1 km resolution using the Integrated Terrestrial Ecosystem C-budget (InTEC) model that was developed and validated for Canadian forests. The purpose of the study was to calculate the annual C balance for China's forests during 1901–2001. Changes in forest stand age represent the integrated effects of natural and anthropogenic factors, such as insect infestation, fire and harvest. Due to the lack of temporally and spatially explicit data of these disturbances, we did not attempt to separate these natural and anthropogenic influences on the C balance of China-wide forests in this study, and treat all disturbances as fire disturbance.

2. Materials and methods

2.1. InTEC model

The InTEC model, established by Chen et al. (2000b), simulates carbon dynamics in a forest ecosystem. The core of InTEC is the mechanistic integration of a canopy level photosynthesis model scaled up from Farquhar's leaf biochemical model of photosynthesis (Farquhar et al., 1980) with a soil C/N cycling model of Century (Parton et al., 1987; Schimel et al., 1994) and empirical NPP–age relationships (Chen et al., 2002). The model is driven by spatially distributed data that include climate, soil texture, nitrogen (N) deposition, forest stand age and vegetation parameters derived from remote sensing (leaf area index (LAI) and land cover).

The overall methodology of satellite-based carbon balance mapping is summarized in Fig. 1. The information on land cover and LAI, in combination with soil texture and meteorological data, allows us to estimate the spatial distribution of NPP (Liu et al., 1999, 2002). An annual NPP value in each pixel estimated for 2001 in daily steps (Feng et al., 2006) was used to compute NPP for the same pixel retrospectively in yearly steps for the period 1901–2001 using climate and forest stand age information. In addition, C pools of biomass and soil are initialized assuming that the C and N exchanges between the terrestrial ecosystems and the atmosphere were in equilibrium for stands at equilibrium age under the mean conditions of climate, CO_2 concentration and N deposition in the pre-industrialization period. In InTEC, the historical annual NPP values are progressively simulated according to the initial NPP, integrated effects of nondisturbance factors on NPP and stand age (Chen et al., 2003). The NPP in a reference year (2001) output from the Boreal Ecosystems Productivity Simulator (BEPS) model (Feng et al., 2006), which operated at a daily time step, is used as a benchmark to tune the initial NPP. The tuning continues until the absolute value of the difference between NPP from InTEC and that from BEPS is smaller than a threshold, 1% of the NPP value from BEPS. NPP and stand age in the initial year (1900) along with mean climatic conditions prior to industrialization are employed to initialize C pools in biomass and soil. Summation of changes in these C pools gives the area-averaged annual C balance or NEP. The cumulative change in each of these C pools is obtained by integrating the change since industrialization. Meanwhile, parameters of the InTEC model have been calibrated using experimental data obtained from Canada's forest ecosystems (Chen et al., 2000a–c; Table 1) and further calibrated in China (Shao et al., 2006, this issue). Detailed descriptions about the InTEC model, state variables, parameters and data pre-processing methods are reported elsewhere (Chen et al., 2000a–c, 2005).

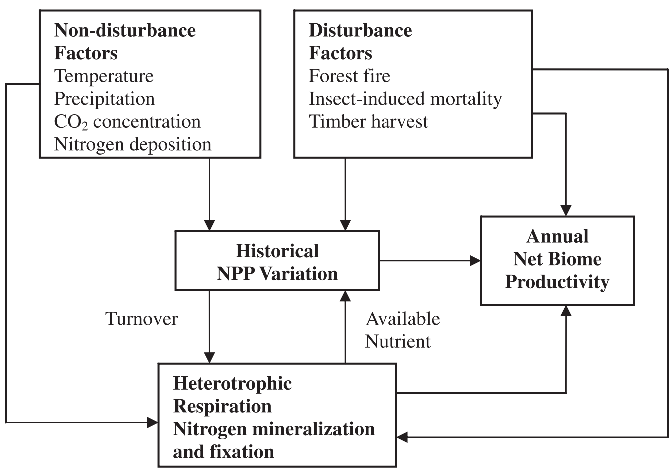

Fig. 1. The major components of the Integrated Terrestrial Ecosystem Carbon Cycle (InTEC) model applied to each 1 km pixel. The historical variation in NPP is central for estimating the amounts of dead organic matter in nine pools (Table 1) (cite from Chen et al., 2003).

The C balance of a forested region is the sum of the changes in all biomass, soil and forest product C pools. In addition, the C in soil and product pools originates in biomass pools, which are in turn supplied from NPP (Chen et al., 2000c). NPP changes with climate, atmospheric conditions and stand age, the latter in turn resulting from disturbances. Therefore, the C balance of a forested region is a function of these external forcing factors. InTEC considers both direct and indirect effects of disturbances on C balance. Fires release a fraction of biomass and soil C into the atmosphere and transfer the remaining biomass C into soil pools. Harvest transfers a fraction of biomass into forest product C pools and the remainder to soil C pools, while

Table 1
Carbon allocation coefficients, turnover rates and decomposition rates of the vegetation and soil carbon components[a] (cite from Chen et al., 2000a, b, 2003)

Symbol	Description	Mean cover type				
		Conifer	Deciduous	Mixed forest	Open land	Unit
Vegetation carbon						
f_w	NPP allocation coefficient to wood	0.301	0.4624	0.3817	0.3817	Unitless
f_{cr}	NPP allocation coefficient to coarse root	0.1483	0.1190	0.1536	0.1536	Unitless
f_l	NPP allocation coefficient to leaf	0.2128	0.2226	0.2077	0.2077	Unitless
f_{fr}	NPP allocation coefficient to fine root	0.3479	0.1960	0.2570	0.2570	Unitless
K_w	Wood turn over rate	0.0279	0.0288	0.0279	0.0139	yr^{-1}
K_{cr}	Coarse root turn over rate	0.0269	0.0448	0.0268	0.0268	yr^{-1}
K_l	Leaf turn over rate	0.1925	1.0000	0.3945	0.3945	yr^{-1}
K_{fr}	Fine root turn over rate	0.5948	0.5948	0.5948	0.3000	yr^{-1}
Soil carbon						
K_{ssl}	Surface structural leaf litter decomposition rate			$3.9L_cA$		yr^{-1}
K_{sml}	Surface metabolic leaf litter decomposition rate			$14.8A$		yr^{-1}
K_{fsl}	Soil structural litter decomposition rate			$4.8L_cA$		yr^{-1}
K_{fml}	Soil metabolic litter decomposition rate			$18.5A$		yr^{-1}
K_w	Woody litter decomposition rate			$2.88L_cA$		yr^{-1}
K_{sm}	Surface microbe decomposition rate			$6.0A$		yr^{-1}
K_m	Soil microbe decomposition rate			$7.3AT_m$		yr^{-1}
K_s	Slow carbon decomposition rate			$0.2A$		yr^{-1}
K_p	Passive carbon decomposition rate			$0.0045A$		yr^{-1}

[a] A is the combined abiotic impact of soil moisture and soil temperature on the decomposition rate, based on Century model and its adaptation to annual step calculations (Chen et al., 2000b). L_c quantifies the impact of lignin content of structural materials on decomposition. T_m is the effect of soil texture on soil microbe carbon decomposition.

insect-induced mortality transfers all biomass to soil C pools. Due to lack of historical data on fire, insect-induced mortality and harvest, this paper did not separate the effects of disturbance factors on C sinks and sources in China's forests during 1901–2001. For a given pixel, stand age increases with time and NPP changes correspondingly with age using a generalized, temperature-dependent empirical NPP–age relationship (Chen et al., 2002). Methods for driving InTEC using remote sensing, climate and forest inventory data are also described in Chen et al. (2000a–c; 2005).

2.2. Data sources

To consider all the major effects of nondisturbance factors on carbon balance in each pixel, the following data sets were produced or compiled from various sources. All spatial data were made compatible with remote sensing imagery at a 1 km resolution grid of 5300×4300 pixels. All grids employed Albers conformal conic projection with 25° and 47° N standard parallels and 110° E meridian. All input data were processed into this resolution and projection before or during model execution.

The year 2001 was selected as the reference year to calibrate the initial value of NPP. The China-wide LAI map, provided by Feng et al. (2006), was produced from 250 m resolution cloud-free composite images of the Moderate Resolution Imaging Spectroradiometer (MODIS) in 2001. Detailed descriptions of the 2001 LAI map and data processing methods are reported elsewhere (Feng et al., 2006, this issue).

The land use map was produced from 560 TM images with 30-m resolution acquired in 2000 and 2001 for the whole of China (Liu et al., 2003, 2005a). There are 25 land use types in the map legend, of which 4 are forest types. We generated a new land cover map by integrating this land use map with a 1:4 million vegetation map of China. All forest and woodland types were divided into 4 main cover types (coniferous, deciduous and mixed forests; shrub lands) that are included in this study. In the long-term simulation of net biome productivity (NBP), the cover types are assumed to be unchanged, but disturbances would reset the stand age to zero while the cover types remained the same. According to the land cover map, the total area of China's forests was about 229.01×10^6 ha in 2001. The areas of coniferous, broadleaf, mixed forests and shrubland were 47.53×10^6, 51.15×10^6, 68.54×10^6 ha and 61.75×10^6 ha in 2001, respectively. Using the LAI and land cover maps of China described above, daily NPP and evapotranspiration (ET) in 2001 were estimated at 1 km resolution using BEPS (Liu et al., 2002). The mean NPP of China's forests simulated by BEPS was $551.7\,\mathrm{g\,m^{-2}\,yr^{-1}}$ in 2001 (Feng et al., 2006).

Monthly mean air temperature and total precipitation for China in the period 1901–1998 were obtained from the 0.5° global data set interpolated by the UK Climate Research Unit (New et al., 2000). Measurements at 743 stations of the National Meteorological Administration in China were processed and spatially interpolated to produce monthly mean temperature and precipitation data sets for the period of 1999–2001. From these monthly data, we calculated climatic inputs to the InTEC model, including the annual mean air temperature, precipitation, growing season length (5-day running mean air temperature $T_a \geqslant 10\,^\circ\mathrm{C}$) and mean growing season temperature (May–September). Atmospheric CO_2 concentration data were taken from Chen et al. (2000a–c) for 1900–1995 and from Zhou et al. (2005) for 1996–2001.

Maps of soil C and N contents and of silt and clay fraction to a depth of 30 cm were produced using 5405 soil profiles of the second national soil survey in 1980s. We generated these data layers with the same projection and resolution as for other data layers by using the ARC/INFO geographic information system. Methods for processing soil inventory data for soil C and N density are given in Wang et al. (2001a, 2003, 2004). These compiled soil data were then entered into the soil module of InTEC according to requirements of the Century model.

Wet nitrogen deposition data were obtained from the N cycle database established by Zhou (2000), which consists of observations of 7 different forest stations of the Chinese Ecosystem Research Network and is supported by 43 published papers since the 1980s. Zhou (2000) calculated the mean wet nitrogen deposition value in late 1990s for cold temperate coniferous forest, coniferous and broadleaf mixed forest, deciduous broadleaf forest, temperate coniferous forest, warm temperate coniferous forest, evergreen broadleaf, and tropical rain forest and seasonal rain forest. Due to lack of historical records and dry nitrogen deposition observation for forests, these measurements and mean calculated values were assumed to be the nitrogen deposition average in 2001 and extrapolated to all forested areas in China in accordance with spatial distribution and wet deposition data of forest types.

A map of forest stand age in 2001 was produced from the fourth national forest inventory data recorded between 1989 and 1993 in 32 provinces (FRSC, 1994). The mean stand age in 2001 was calculated according to different forest survey times in individual provinces. The age of stands in a forest polygon was assumed to be normally distributed. The age of each forested pixel was determined as the sum of mean age and the standard deviation of stand age in a polygon multiplied by a random number between −1 and 1. The standard deviation was determined according to sample polygon data in each province.

Relationships between NPP and stand age were established from analysis of stand yield data for black spruce in Ontario (Chen et al., 2002). These relationships vary with site conditions quantified using a site index in terms of tree height at the age of 50 years (Chen et al., 2003). For application of the relationship to large areas, a general semi-empirical mathematical function was developed:

$$\mathrm{NPP(age)} = A\left(\frac{1 + b(\mathrm{age}/c)^d - 1}{\exp(\mathrm{age}/c)}\right), \tag{1}$$

where the parameters A, b, c and d are dependent on the site index (Chen et al., 2003). Since long-term observations of NPP and site index are not available for most cases, we adopted the method described by Chen et al. (2003) to apply the relationship to all China's forests through making the coefficients A, b, c and d empirical functions of annual mean temperature.

3. Results

To compute the spatial distribution of NEP, we first produced the stand age map from the forest inventory data as the essential model input for spatially explicit calculations of NEP and NBP (Chen et al., 2003). The 2001 map of forest stand age was created using the combined information from the fourth forest inventory (FRSC, 1994). Since the inventory did not include data on fire and insect disturbance, their effects on forest age were not separated in InTEC. The China-wide distribution of forest stand age in 2001 (Fig. 2) indicates that the (10–20) age class had the largest area. Through a statistic analysis of the 1 km resolution grid map of forest stand age, we found that the age class (0–10, 10–20, 20–40, 40–60, 60–80, 80–100, 100–120, 120–160 and older) occupied, respectively, about 20.17%, 39.35%, 19.39%, 8.75%, 6.33%, 3.21%, 1.67%, 0.72% and 0.42% of the total of China's forest area. The map suggests that young forests with stand age below 40 are the main part of China's forest at present. However, it is assumed in Fig. 2 that the distribution of forest stand age was even among age classes. According to the forest inventory data, there was significant increase from 0–10 to 10–20 age class and decrease from 10–20 to 20–40 age class; these variations are believed to have influenced the carbon balance of China-wide forests around 1980.

Because ChinaFLUX with eight eddy covariance stations was established only in the summer of 2002 (www.chinaFLUX.org), model simulations of the historical changes in the carbon balance for individual pixels were not compared with measured annual total NEE at the tower flux sites. However, the computed mean biomass C storage and NEP values during the 1970s–1990s for China's forests are compared with previous studies.

3.1. NPP

The average NPP of China's forests increased by $89\,g\,C\,m^{-2}\,yr^{-1}$ in 2001, a 21.0% increase from 1901

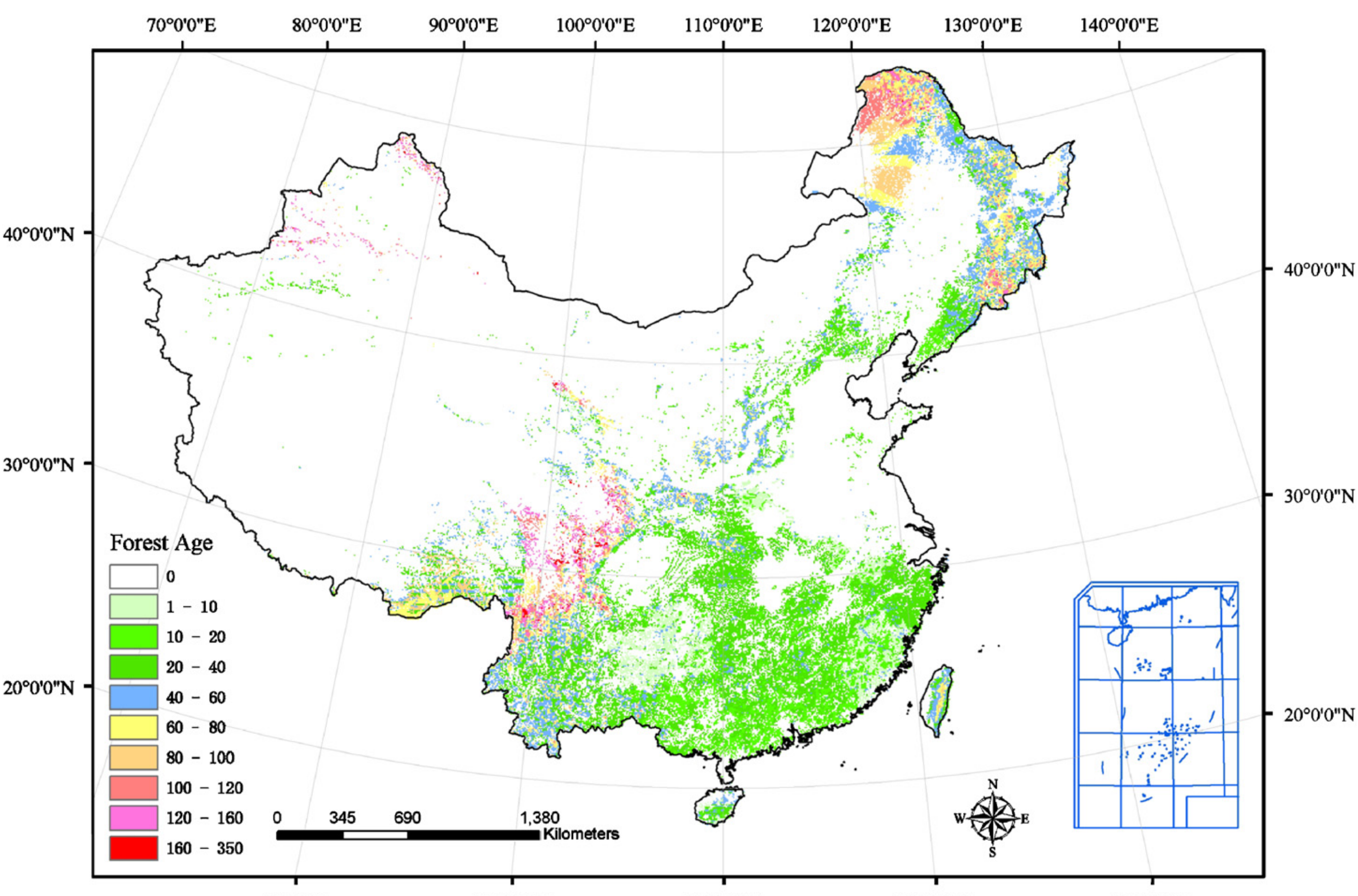

Fig. 2. Spatial pattern of China-wide forest stand age in 2001.

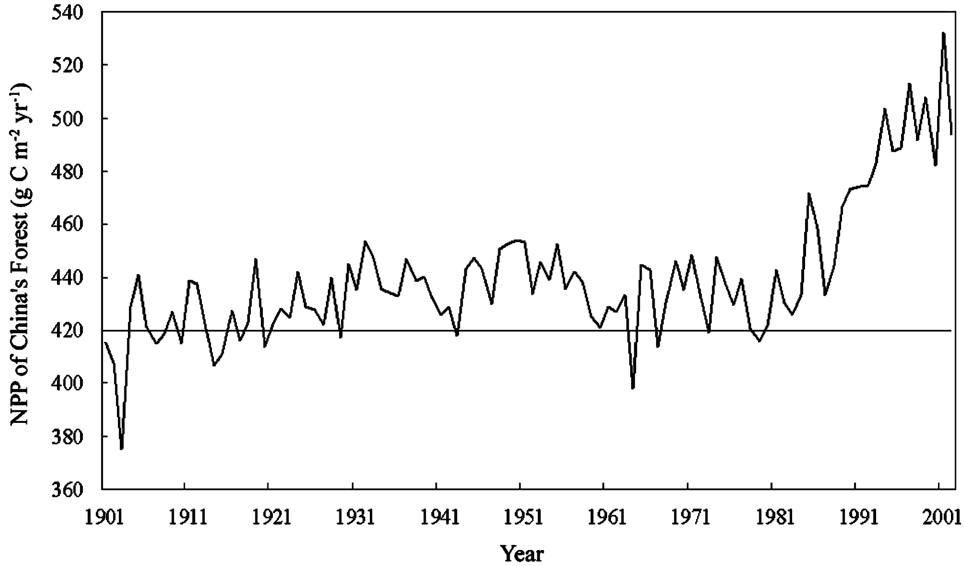

Fig. 3. Annual average NPP of China's forests during 1901–2001.

(Fig. 3). During 1901–1949, the average NPP in China's forests was $0.96 \pm 0.01\,\mathrm{Pg\,C\,yr^{-1}}$, or $421 \pm 4\,\mathrm{g\,C\,m^{-2}\,yr^{-1}}$. During 1950–1987, the average NPP was $0.97 \pm 0.01\,\mathrm{Pg\,C\,yr^{-1}}$ or $424 \pm 5\,\mathrm{g\,C\,m^{-2}\,yr^{-1}}$. The annual NPP reached $1.12 \pm 0.03\,\mathrm{Pg\,C\,yr^{-1}}$ during 1988–2001, or $487 \pm 13\,\mathrm{g\,C\,m^{-2}\,yr^{-1}}$. NPP began to increase after the late 1970s because of large-scale plantations, forest regrowth in previously disturbed areas, and additional growth stimulated by nondisturbance factors such as climate warming, atmospheric CO_2 fertilization and N deposition.

Although NPP fluctuated around $431 \pm 7\,\mathrm{g\,C\,m^{-2}\,yr^{-1}}$ from 1901 to 1978 (Fig. 3), the average NPP increased by $72\,\mathrm{g\,m^{-2}\,yr^{-1}}$ from 1979 to 2001. The large-scale plantations produced a higher total NPP in the 1980s and 1990s than in previous decades. We found a rapid increase in the annual mean NPP of China's forests during the period 1979–2001 (Fig. 3). Bunce (1989) indicated that planting eliminates the regeneration delay, and so shifts the productive stage forward by 5–9 years. However, Chen et al. (2000c) argued that planting increases NPP in the first 50 years, and reduces it thereafter because naturally regenerated forest stands will be younger and more productive than planted stands at the later stages. Because the Chinese government carried out several huge planting projects in the 1980s and 1990s (Fang et al., 1998, 2001), NPP in China's forest should therefore rise until the 2030s, and may decrease thereafter; however, the trends of NPP dynamics need to be confirmed through measurements in the future.

The effects of harvest and planting activities on NPP were minimal and maximal in the 1990s, respectively. Hence, there were more young and unproductive stands (i.e., <20 years old) and productive stands (20–100 years old) after the late 1970s. NPP reached the maximum point in 2000, about $532.27\,\mathrm{g\,C\,m^{-2}\,yr^{-1}}$ (Fig. 3). The long-term effects of planting and harvesting on NPP are achieved through altering age, C:N ratios of biomass and soil C pools, leaf area and leaf N content (Chen et al., 2000c). Due to the large interannual variations in many of the external forcing factors, it is very difficult to separate short- and long-term effects on NPP.

3.2. Carbon stocks

We estimated C storage for soil organic carbon (SOC), litter, aboveground biomass, roots and the total ecosystem (ECO) of China's forests from 1900 to 2001 by using InTEC. The total storage in China's forest ecosystems varied between 28 and 35 Pg C during 1900–2001. The C stocks in SOC, vegetation and litter stocks ranged, respectively from 13.3 to 14.3 (0–30 cm), from 11.5 to 17.4 and from 2.4 to 3.5 Pg C during 1900–2001. The C storage in aboveground biomass and roots was 8.4–13.5 and 3.0–4.2 Pg C during 1901–2001, respectively. Fig. 4 shows the changes in C stocks for China's forest during 1901–2001.

Changes in C stocks of China's forests occurred in three successive phases. The total C stocks slightly decreased in the first phase (1901–1949), but it decreased rapidly in the second phase from 1950 to 1987 (see Fig. 4). Fang et al. (2001) also found that the minimum forest area was in the period of 1977–1981 inventory rather than in 1949 when the civil war ceased. Their finding is consistent with our result showing the lowest total C stock in China's forests of 28.80 Pg. This minimum occurred in 1987 and was caused by the time lag of C loss from land use changes in the 1960s, 1970s and early 1980s.

From 1988 to 2001, the total C storage in China's forests started to increase, primarily because forest cover increased by about 20×10^6 ha due to plantations after 1980 (Fang et al., 2001). However, young forests under 5 years old could not sequester enough C to offset the loss of carbon

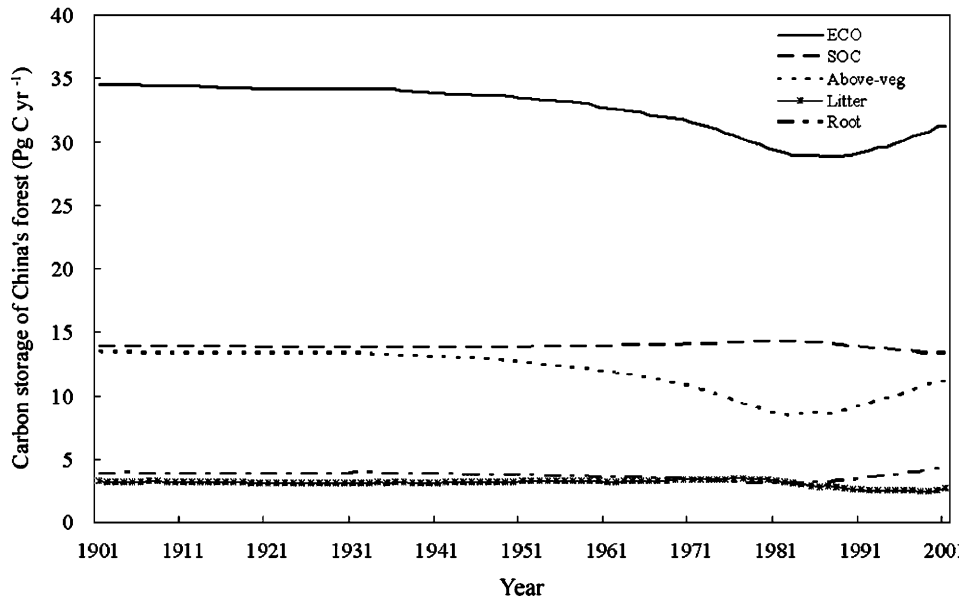

Fig. 4. Biomass and soil C pools storage in China's forests during 1901–2001.

Table 2
Cumulative stock changes of China's forest aboveground biomass, soil organic carbon, litter and roots C pools for periods during 1901–1949, 1950–1987 and 1987–2001

	1901-1949 (Pg C)	1950-1987 (Pg C)	1988-2001 (Pg C)	Total
SOC	−0.10	0.29	−0.80	−0.61
Aboveground	−0.70	−4.22	2.48	−2.44
Litter	−0.10	−0.27	−0.19	−0.56
Roots	−0.12	−0.57	0.98	0.29
Total	−1.02	−4.77	2.47	−3.32

from soils after deforestation in previous years. This led to the net C loss from China's forest during 1980–1987. Table 2 shows the cumulative changes in aboveground biomass, soil organic carbon, litter and roots C pools for three periods (1901–1949, 1950–1987, 1988–2001). From 1901 to 2001, net C loss from aboveground biomass was the largest among the four C pools in China's forest, which accounted for 73.5% of the total net loss (Table 2). The net C loss in litter C pools was close to the net loss from the SOC stock, but the root C stock increased by 0.29 Pg during 1901–2001. The net C loss was largest during 1950–1987, about 4.77 Pg C. Subsequently, China's forest ecosystems sequestered 2.47 Pg C during 1988–2001.

For biomass, the largest C stock change occurred in the aboveground biomass pool, since it was the largest among biomass C pools. Averaged over the entire 1901–2001 period, the SOC and litter C pools accounted for 52.14% and 47.86% of the C stock change in soil, respectively (Table 2). Overall, the soil C stock decreased 0.61 Pg from 1901 to 2001, or 4.58% of the current soil C stock of 13.31 Pg C to the depth of 30 cm, while the aboveground biomass decreased 2.44 Pg C, or 22.06% of current total biomass C of 11.06 Pg. During the 1987–2001 period, the aboveground biomass C stock increased by 22.42% of its current total (Table 2).

In general, the cumulative stock change of forest C pools was a result of changes in climatic and atmospheric conditions (Chen et al., 2000a–c). However, land use changes or other disturbances could also result in large variations in C pools. The aboveground biomass C stock of China's forests started to increase gradually around 1984, whereas that of soil started to decrease around 1982. In 1977, the aboveground biomass and roots C stocks decreased quickly by 256 and 50 Tg, while SOC and litter stocks increased, respectively, by 29.8 and 1.6 Tg. We presume that the huge C loss in 1977 came from the cumulative effects of the "Great Leap Forward" political activities and of serious natural disasters on forests during 1958–1960 and "Cultural Revolution" political activities during 1966–1976. Although the stand age map does indicate that a large forested area grew after the 1970s (Fig. 2), there were no spatially explicit historical records to verify our assumption. In addition, much of the C loss in biomass C stock in this period (1958–1977) was transferred to litter and soil, with the remaining 2.87 Pg C was emitted to the atmosphere or transferred to wood products. A time lag between the biomass and soil C stock changes was clearly evident here (Fig. 4 and 5). The primary reason for this time lag was the difference in the ways biomass and soil C pools respond to disturbances and climatic and atmospheric changes (Chen et al., 2000c). For example, fire, insect-induced mortality, and harvest reduce biomass C stock, but increase soil C stock in the disturbance year by transferring biomass C to soil. During decades after the disturbance, biomass is restored gradually, yet the soil C

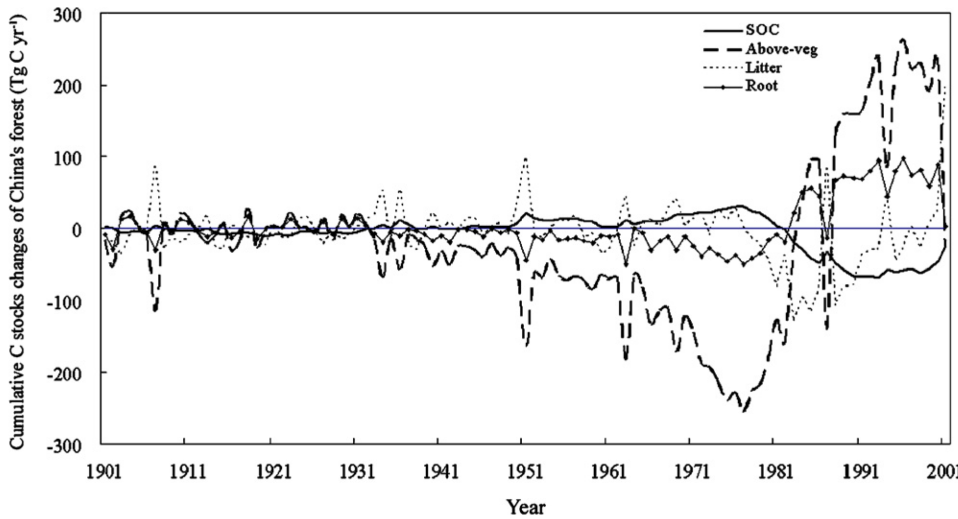

Fig. 5. The annual C stocks changes of China's forest during 1901–2001.

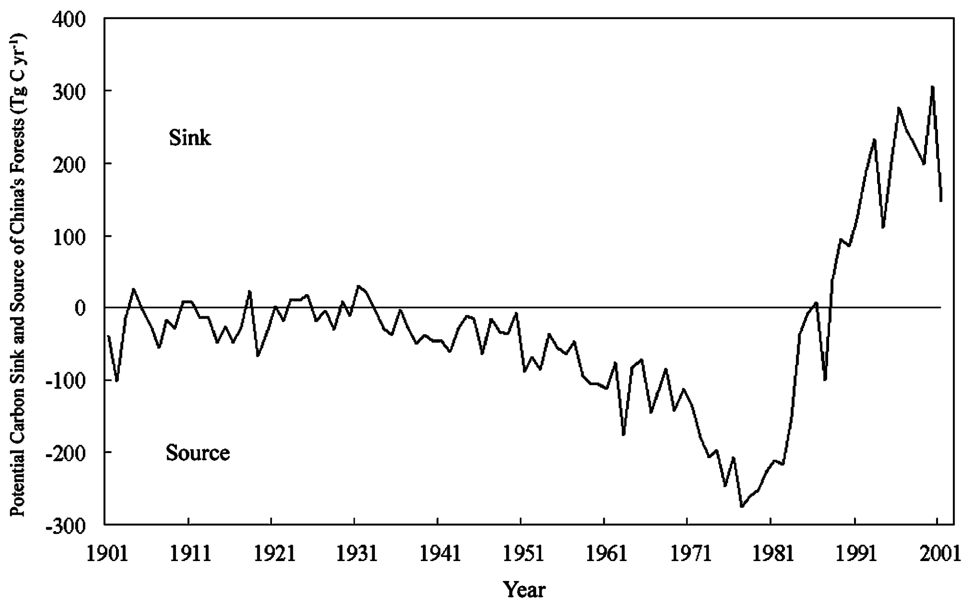

Fig. 6. Annual NEP changes of China's forest during 1901–2001.

stock continues to decline for up to ~50 years before it starts to recover again (Chen et al., 2000c). Increasing abiotic decomposition results in immediate C loss from soil, but enhances NPP and biomass C stock. It would take years before the increased biomass C stock would be transferred to soil. All other climatic and atmospheric factors increased first the biomass C stock, which would be transferred to soil C pools many years later. Such a time lag was also found between litter and root C pools (Fig. 5).

3.3. Net ecosystem production (NEP)

Table 2 and Fig. 6 show the total net ecosystem production (NEP) in China's forest during 1901–2001. As the difference between NPP and heterotrophic respiration minus the total C loss due to disturbance, the NEP of China's forests was -21.0 ± 7.8 Tg C per year (a C source) during 1901–1949, and increased to -122.3 ± 25.3 Tg C per year (a C source) during 1950–1987. It is often reported that the forest area of China reached its minimum in 1949 and that, since the start of the People's Republic of China, forest area increased as a result of massive afforestation campaigns (Houghton and Hackler, 2003). In fact, new China suffered from civil wars, the Korean War in the early 1950s, the Great Leap Forward and heavy natural disasters in the late 1950s and early 1960s, and the Cultural Revolution from 1966 to 1976, so large areas of forests were cut, while large-scale plantations were not common during 1950–1970. Moreover, the earliest forest inventories were incomplete and inappropriate for determining changes in forest area (Houghton and Hackler, 2003). These factors could cause some uncertainty in the NEP estimation around 1977.

NEP increased to 176.7 ± 44.8 Tg C per year during 1988–2001, in response to younger age structure and its greater C sequestration potential in the 1990s compared to earlier decades (Pan et al., 2004). Houghton and Hackler (2003) reported that the accelerated clearing and logging of forests in northeastern and southwestern China led to emissions of carbon that peaked at 0.2–0.5 Pg C yr^{-1} from the late 1950s through the 1970s. Lower rates of deforestation since then, as well as expanding areas of tree plantations, reversed the net C flux from a source to a sink during the 1990s (Houghton and Hackler, 2003). Furthermore, Pan et al. (2004) argued that because of the relatively young age structure in China's current forests (only 29% of the forests have reached mature stages), the age impact on C sequestration estimates will be more prominent in the near future.

The uncertainties of 25–40% in the above values are attributed to both errors from NPP calculation and those in the original data of disturbances, climate, and atmospheric variables. Averaged over the entire 1901–2001 period, NEP was around -32.9 ± 22.3 Tg C per year, with an uncertainty approaching 100% due to very high temporal variability and accumulated errors over 101 years. Fig. 7 displays the spatial pattern of average NEP over 1992–2001 in China's forests. It indicates that the largest forest C sequestration took place in northeastern and southern China.

4. Discussion

4.1. Biomass C stock

We compared our results with C stock estimates for China that were based on forest inventory data from different periods. Various methods produced different results in C storage. For example, Pan et al. (2004) developed a volume-to-biomass method based on age groups representative of forest development stages to estimate live-tree biomass C and C accumulation rates of China's forests between 1973 and 1993. Their results indicate that biomass carbon storage in China's forests was 4.34 Pg C in the early 1990s, an increase of 13% since the early 1970s. In our paper, the total area of shrubland was 61.75×10^6 ha, about 26.96% of total forested land area in 2001. In order to compare our results with other

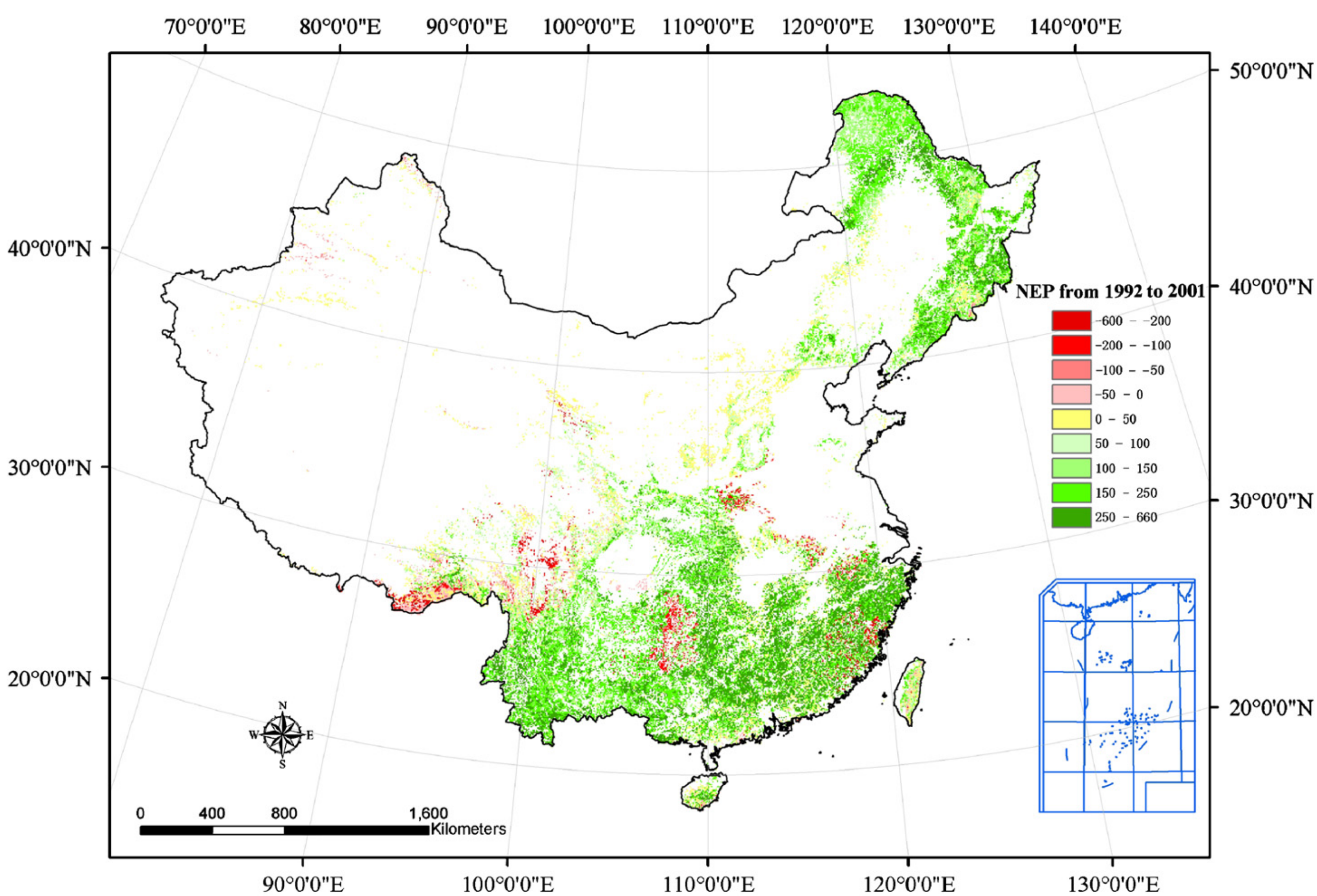

Fig. 7. The spatial distribution of China's forest NEP during 1992–2001 (g C m^{-2} yr^{-1}). Positive NEP values indicate sinks, while negative NEP values indicate sources of carbon, i.e., release to the atmosphere.

findings, Table 3 lists various estimates of forest ecosystem C stocks in China as well as other countries excluding the shrubland. Goodale et al. (2002) estimated that China's forests had 4.6 Pg C in live vegetation, 0.9 Pg C in the forest floor, 21.0 Pg C in SOC and 0.6 Pg C in woodland live vegetation.

The amount of carbon stored in forest vegetation in the 1990s (9.91 $\pm$ 0.09 Pg according to our analysis, Table 3) is higher than estimates based on recent forest inventories (4.34 Pg for the 1990s (Pan et al., 2004) and 4.75 Pg for 1998; Fang et al., 2001). One reason for the difference in these estimates is the difference in the forest area used by these researchers. Another reason is that our estimate is based on the changes in biomass and soil in currently forested areas, and previous forested land that was converted before 1990 to other land uses is not included in our estimation.

The aboveground biomass C in China's forests (9.91 $\pm$ 0.09 Pg in this study) is 83% of that in Canada, 72% of that in the conterminous USA and 39% of that in Russia. Pan et al. (2004) listed forest areas, stocking, biomass and carbon in four periods corresponding to China's four forest inventories. Table 4 shows the comparison between our results and those of Pan et al. (2004). The simulated aboveground biomass C pool was 110–160% larger than that of Pan et al. (2004), but the forest area in our paper was also 28–44% larger. For mean biomass C density in China's forests, our result was about 60–93% higher than Pan et al.'s estimation (Table 4). This comparison indicates that forest area alone could cause a 30–50% difference in C stock estimation.

The area decreased by approximately 6.15% from the first to second inventory (1973–1976 to 1977–1981) and increased by 12.11% afterwards because of intensive national plantation campaigns (Pan et al., 2004; Table 4). The total biomass C in forests increased from 3.836 to 4.339 Pg over the entire period, and increased by 1.93%, 2.25% and 8.53% between sequential inventories (Pan et al., 2004; Table 4). Although forest area decreased from 1973–1976 to 1977–1981, likely because of the harvesting of mature forests, an increase of biomass in the medium-aged forests compensated for the biomass loss due to harvesting (Pan et al., 2004). Simulation results from InTEC partly confirmed their analysis. Moreover, a remarkable increase in forest biomass for the periods 1984–1988 to1989–1993 was due to an increase in the productivity of large areas of forest established after plantation campaigns as these medium-age forests reached their most productive stages (Pan et al., 2004). InTEC results indicate that biomass C stock changed −9.98%, −4.21% and 6.83%, showing our results to be consistent with Pan et al. (2004).

4.2. SOC stock and change

Wang et al. (2004) estimated the total SOC stocks contained in the top 10, 20, 30, 50 and 100 cm of soil to be, respectively18.12 $\pm$ 3.35, 33.74 $\pm$ 5.93, 44.48 $\pm$ 7.80, 61.30 $\pm$ 11.40 and 82.47 $\pm$ 19.46 Pg C in the 1980s, based on 2473 soil profiles grouped by biome. The SOC storage to a depth of 30 cm was 10.89 $\pm$ 2.42 Pg C for forests (area of 121.64 $\times 10^6$ ha) and 4.23 $\pm$ 0.87 Pg C for shrubland

Table 3
Biomass C storage in China's et al., regions' forest

Author	Area ($\times 10^6$ ha)	Biomass (Pg C)	Roots (Pg C)	Period	Region
This paper	167.26	9.91 $\pm$ 0.09	3.77 $\pm$ 0.04	1990s	China
Pan et al. (2004)	130.5	4.34		1989–1993	China
Ni (2003)	112	9.11		1990s	China
Goodale et al. (2002)	195.0	4.6		1990s	China
Fang et al. (1998)	102.2	4.3		1984-1988	China
Fang et al. (2001)	133.7	4.75		1998	China
Alexeyev et al. (1995)	771.1	25.60		1990s	Russia
Liski and Kauppi (2000)	244.6	11.89		1990s	Canada
Birdsey and Heath (1995)	245.9	13.78		1990s	USA

Table 4
C storage in aboveground biomass in China's forest (excluding shrubland)

	Pan et al. (2004)			InTEC model		
	Areas ($\times 10^6$ ha)	C Pool (Pg C)	C density (kg m^{-2})	Areas ($\times 10^6$ ha)	C Pool (Pg C)	C density (kg m^{-2})
1973–1976	123.6	3.836	3.10	167.26	10.02 $\pm$ 0.41	5.99 $\pm$ 0.24
1977–1981	116.0	3.91	3.37	167.26	9.02 $\pm$ 0.34	5.39 $\pm$ 0.21
1984–1988	125.4	3.998	3.19	167.26	8.64 $\pm$ 0.09	5.16 $\pm$ 0.06
1989–1993	130.5	4.339	3.32	167.26	9.23 $\pm$ 0.35	5.52 $\pm$ 0.21

(216.54×10^6 ha), or about 72.41% and 40.87% of the soil C sequestered in the top 100 cm during the 1980s (Wang et al., 2004). According to InTEC simulations, mean SOC stock for China was 10.36 ± 1.56 and 3.82 ± 0.57 Pg C for forests with 167.26×10^6 ha and shrubland with 61.75×10^6 ha in 1980s, respectively. Because the definitions of shrubland by Wang et al. (2004) and this paper are different, it was difficult to compare the two results. However, if we consider the total SOC storage of forests and shrubland, our modeling results are consistent with the results based on soil inventory.

Liu et al. (2004) analyzed the impacts of China's land use changes on SOC from 1990 to 2000 between forest, grassland and cropland based on 2473 soil profiles and Landsat TM images. They found that China's forest lost about 55.9 ± 17.0 Tg C to a depth of 30 cm in the 1990s due to land use changes. Our simulation shows that the SOC decreased by 487 Tg C from 1990 to 2000 in China's forest (excluding shrubland). Our estimate of the amount of C loss from soils is higher than those by Wang et al. (2003) and Liu et al. (2004). The reason for the difference is unclear, but we postulate that our new results considered not only the direct disturbance effect (reduction in forested area) but also the subsequent indirect effect on carbon loss from soil after disturbance.

4.3. NEP changes in China

Pan et al. (2004) reported that the annual forest C sequestration rate from the late 1980s to the early 1990s was 0.068 Pg C yr^{-1} and approximately four to five times higher than in the 1970s and 1980s. According to Pan et al. (2004), the large C sink in China's forests in the early 1990s was likely related to age structure changes that had developed into productive stages, a consequence of reforestation and afforestation programs from the 1960s. Our results are similar to those of Pan et al. (2004) between the 1970s and 1990s.

Houghton and Hackler (2003) found that changes in land use contributed to the current terrestrial C sink in most regions of the northern mid-latitudes but were poorly documented for China. They reconstructed the last 300 years of changes in forest area over time, and, using data from ecological literature, they estimated C stocks of the major natural ecosystems (vegetation and soil). According to the data and assumptions, 180 (range: 80–200) $\times 10^6$ ha of forest were lost from 1700 to 2000; and 17–33 Pg C were released to the atmosphere between 1700 and 2000, of which 75–80% of the C lost was from vegetation and 20–25% from soils in all three scenarios (Houghton and Hackler, 2003). Their findings are consistent with our simulation results for China's forest during 1901–2001, as our study shows that 65–73% of the C loss was from vegetation and 27–35% from soils. Our research indicates that the C sink was in plantations during the 1980s and 1990s, which is in agreement with Fang et al. (2001), Houghton and Hackler (2003), Pan et al. (2004) and other studies. However, because spatially explicit records of changes in forested area are not yet available, the estimated land use change was also a source of uncertainty in China's forest NEP estimation. Accurate forested area data at different times are critical in improving China's NEP estimates.

In recent years, a growing number of studies have attempted to quantify C sinks and sources for all terrestrial ecosystems in China. Fang et al. (2001) reported an average C release (0.022 Pg yr^{-1}) for the years 1949–1980 and an average uptake (0.021 Pg C yr^{-1}) for the years 1980–1998. Their estimates were based on changes in forest biomass and did not consider SOC, litter, roots or changes in wood products. The same is true for the C sink estimate of 0.04 Pg C yr^{-1} for China's forests by Goodale et al. (2002) (see also Houghton and Hackler, 2003). Including soils as well as living vegetation, Zhou (2000) calculated a net forest uptake of 0.048 Pg C yr^{-1} for the period 1989–1993. Streets et al. (2001) report an annual sink of 0.098 Pg C in 1990 and of 0.112 Pg C in 2000. Houghton and Hackler (2003) reported that the annual C emissions from changes in land use increased over time, reaching a peak of 0.4 Pg C yr^{-1} (range 0.2–0.5) in the late 1950s and declining rapidly after the late 1970s, a net source averaging 0.008 Pg C yr^{-1} in the 1990s, and the last several years of the 1990s with a net annual sink ranging between 0.02 and 0.05 Pg C yr^{-1}. Cao et al. (2003) reported that China's terrestrial NPP varied between 2.86 and 3.37 Pg C yr^{-1} with a growth rate of 0.32% yr^{-1} in the period 1981–1998 and NEP with a mean value of 0.07 Pg C yr^{-1}, leading to C accumulation of 0.79 Pg in vegetation and 0.43 Pg in soils during this period. According to InTEC simulations, large sources of around 3.32 Pg C resulted from harvesting China's forest during 1901–2001, contributed by soils (1.17 Pg C) and vegetation (2.15 Pg C) as net sources. Our estimate for C sink during 1988–2001 in this analysis (0.18 ± 0.04 Tg C yr^{-1}) is somewhat higher than earlier estimates for China's forests in which C sinks were largely in biomass. The vegetation and soil stocks varied by 0.25 and −0.07 Pg C yr^{-1} between 1988 and 2001, respectively. A large uncertainty in our estimate may be contributed by the forest age map which assumes statistical mean forest age distributions rather than realistic forest age class fractions; this is an important area for improvement through further processing of the original forest inventory data.

The global carbon budget shows that the net terrestrial carbon uptake increased from 0.2 Pg C yr^{-1} in the 1980s to 1.4 Pg C yr^{-1} in the 1990s (Prentice et al., 2001; Schimel et al., 2000). The estimated potential NEP for China's forest in the present study accounts for the negative contribution (−0.08 Pg C yr^{-1} for the 1980s) and 13.5% (0.19 Pg C yr^{-1} for the 1990s) of the world's total, and it is about 2.4 times the 0.08 Pg C yr^{-1} estimated for the United States (Schimel et al., 2000). However, Cao et al. (2003) reported that NEP in China decreased from the 1980s to the 1990s because of a stronger warming than the global average. In other words, the InTEC model likely overestimated NEP in China's forest for the 1980s and 1990s. It is believed that the complex interaction among biological, physical, chemical and atmospheric processes could lead to large variability in NEP values obtained from different models for China's forest or other terrestrial ecosystems. Reasons for these differences correspond to data sources, processing methods, model structure and research objectives. It would therefore be useful to consider the integrated effects of natural and human activities on the forests carbon budget in the future.

5. Uncertainty and conclusions

This study used InTEC, a regional forest C-budget model that integrates the effects of disturbance and nondisturbance factors to analyze the dynamic changes in the C balance in China's forest during 1901–2001. In reality, the distribution of forest stand age reflects the combined effects of disturbance and nondisturbance factors considered in this paper. Because potential NEP or C pool sizes could only be evaluated with limited available data, estimates of the C balance over the 101 years have significant uncertainties. To avoid this uncertainty, Chen et al. (2000a–c) used a historical change approach, in which the C balance is determined from the changes in the disturbances and nondisturbances factors, and is assumed to be at the equilibrium state during the mean pre-industrial period. However, there are other factors to cause uncertainties in current estimates.

Many issues may cause uncertainties in C balance estimates for forests. For example, accurate quantification of the terrestrial carbon sink must account for the interannual variations associated with climate variability and change (Cao et al., 2003). For most of those uncertainties, we could not provide accurate validation of InTEC model results because of the lack of available data to evaluate errors. The age distribution for forest stands was generated based on 1266 plots and some published materials, so that the extrapolation of stand age contributed a relatively large uncertainty to the InTEC simulations, especially in 1977 (Fig. 2). In Section 3, we discussed the problem of forest stand age structure (10–40 age class), which caused abrupt changes in the C balance around the 1970s. Liu and Diamond (2005) pointed out that following the Second World War and the Chinese Civil War, the peace of 1949 brought more deforestation, overgrazing and soil erosion in China. They also reported that the Great Leap Forward in 1958–1960 saw a dramatic increase in the number of factories—a fourfold increase in 1957–1959 alone—along with pollution and more deforestation to obtain the fuel for inefficient backyard steel production. Fig. 11 shows that from 1960 to 2004, the forest cover was minimum in 1961 (Liu and Diamond, 2005). Moreover, heavy natural disasters (drought and flood) and famine that occurred between 1958 and 1961 could have greatly limited the growth of China's forests. In addition, the Cultural Revolution had a great impact on each field like economy and forestry in China during 1967–1976. Therefore, we tentatively conclude that the big fluctuation in the C balance in 1977 is the cumulative result of political activities and major natural disasters in late 1950s, 1960s and 1970s, but relevant evidence and data for 1977 need to be collected to establish the validity of our hypothesis in further studies. In Chinese studies, because the field data derived from the literature include a relatively high percentage of young and medium-age stands, a skewed distribution of sample data towards younger ages may lead to an overestimation of C sinks for forests (Pan et al., 2004). An accurate forest age map is critical for improving China's forest carbon budget estimation.

Although InTEC integrates forest and soil inventory data, carbon flux data are still needed to validate the InTEC results and to reduce uncertainties, as was done for Canada (Chen et al., 2003). Since it is difficult to validate long-term simulations, current flux measurements would provide a critical check of the model performance in recent years.

The third problem is the definition of forest in China. Our estimates were calculated for coniferous forests, broadleaf forests, mixed forests and shrubland, but other studies did not include shrubland as forest. The current area of forest in China depends on the remote sensing data in this study. Others scientists used data from different forest inventories conducted by the State Forestry Administration (SFA) of China to report a total forest area (including both planted and natural forests, Table 3). As Houghton and Hackler (2003) noted that the combined area of these categories is large, individual categories may hold different amounts of carbon, and changes in them are not well documented, especially before 1960. Based on forest inventory data and the biomass-volume relationship, Fang et al. (2001) and Pan et al. (2004) estimated that total forest aboveground biomass C in China ranged between 4.3 and 5.06 Pg C in seven forest inventories in China from 1949 to 1998. This may be mainly due to differences in biomass estimating methods and forest area calculations (Houghton and Hackler, 2003).

In addition to providing a comprehensive estimate of forest carbon balance over the last 101 years, this study also illustrates the source and sink distributions in space and time for the first time, which are valuable for understanding the C dynamics in China's forests. In order to improve past and current estimates and future predictions of China's forest C balance, we would also need to simulate the spatial distribution of direct carbon emissions in the disturbance year when spatially explicit forest fire and insects and harvest data are available. Meanwhile, further research is also required using experimental data for China to improve the NPP-stand age relationship and to further validate the InTEC model parameters.

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