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MODELING SOIL ORGANIC CARBON CHANGE IN CROPLANDS OF CHINA

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Abstract. Using 1990 conditions, we modeled carbon (C) and nitrogen (N) biogeochemical cycles in croplands of China (and, for comparison, the United States) to estimate the annual soil organic-carbon (SOC) balance for all cropland. Overall, we estimate that China's croplands lost 1.6% of their SOC (to a depth of 0.3 m) in 1990, and that U.S. cropland lost 0.1%. A key element in this difference was that ~25% of aboveground crop residue in China was returned to the soil, compared to ~90% in the United States. In China, SOC losses were greatest in the northeast (~10^3 kg C·ha^{-1}·yr^{-1}), and were generally smaller (<0.5 × 10^3 kg C·ha^{-1}·yr^{-1}) in regions with a longer cultivation history. Some regions showed SOC gains, generally <10^3 kg C·ha^{-1}·yr^{-1}. Reduced organic-matter input to China's cropland soils, and lower overall SOC levels in those soils, led to lower levels of N mineralization in the simulations, consistent with higher rates of synthetic-fertilizer application in China. C and N cycles are closely linked to soil fertility, crop yield, and non-point-source environmental pollution.

Key words: agroecosystem; biogeochemistry; carbon sequestration; China, farming management; Chinese agriculture, sustainability; crop residue; global warming and carbon sequestration; modeling C and N biogeochemical cycles in cropland; organic C in cropland soil.

INTRODUCTION

Conversion of naturally vegetated land to agriculture generally results in loss of soil organic carbon (SOC), and this will continue as long as carbon loss, mainly from decomposition, exceeds carbon inputs, mainly as plant litter and amendments (e.g., manure). Cai (1996) quantified the effects of land use on SOC in eastern China (Fujian, Jiangxi, Anhui, Jiangsu, and Zhejiang Provinces), based on mean SOC values reported in The Second Soil Survey of China (Office for Soil Survey of China 1993). He found that mean paddy-field SOC was about half that of soils with natural vegetation, and mean upland crop field SOC was about half that of paddy fields. He estimated total SOC loss for the cropland soils (0–0.62 m) in the region at 850 Tg C, based on the assumption that soils supporting natural vegetation at the time of the Second Soil Survey were representative of all pre-agricultural soils. Loss of SOC with cultivation is a common phenomenon (e.g., Davidson and Ackerman 1993), and efforts to recoup some of these losses are receiving attention in current analyses of the global carbon cycle (e.g., Buyanovsky and Wagner 1998, Follett 2001).

Crop residue management is an important component of the carbon and nitrogen budget of agroecosystems. After grain harvest, crop residues can be removed from the field (to be used for livestock feed or bedding, for fuel, or for other off-field purposes), burned in place (primarily for pest control or for simple residue removal), plowed or tilled into the soil, or left on the surface as mulch (conservation tillage or no-till) (Sandretto 1997). If returned to the soil, crop residues are a direct C and N input to the soil, and an indirect source of mineral N through net N mineralization as the crop residues decompose. Over a period of decades, soil organic-matter stocks will respond measurably to changes in crop residue management (e.g., Li et al. 1994, Buyanovsky and Wagner 1998).

Over the past half century crop residue management in China has changed significantly. There was previously a long-term tradition among the Chinese farmers to return all available organic matter (animal dung, night soil, pond muck, and crop litter) to their soil (Buck 1937). During the profound social and economic evolution in China over the past five decades, traditional agricultural management has been modified (Liu and Mu 1993). Changes in land tenure led to changes in farmer priorities toward production and sustainability. With the development of the rural economy during the last two decades, fuel availability has improved and

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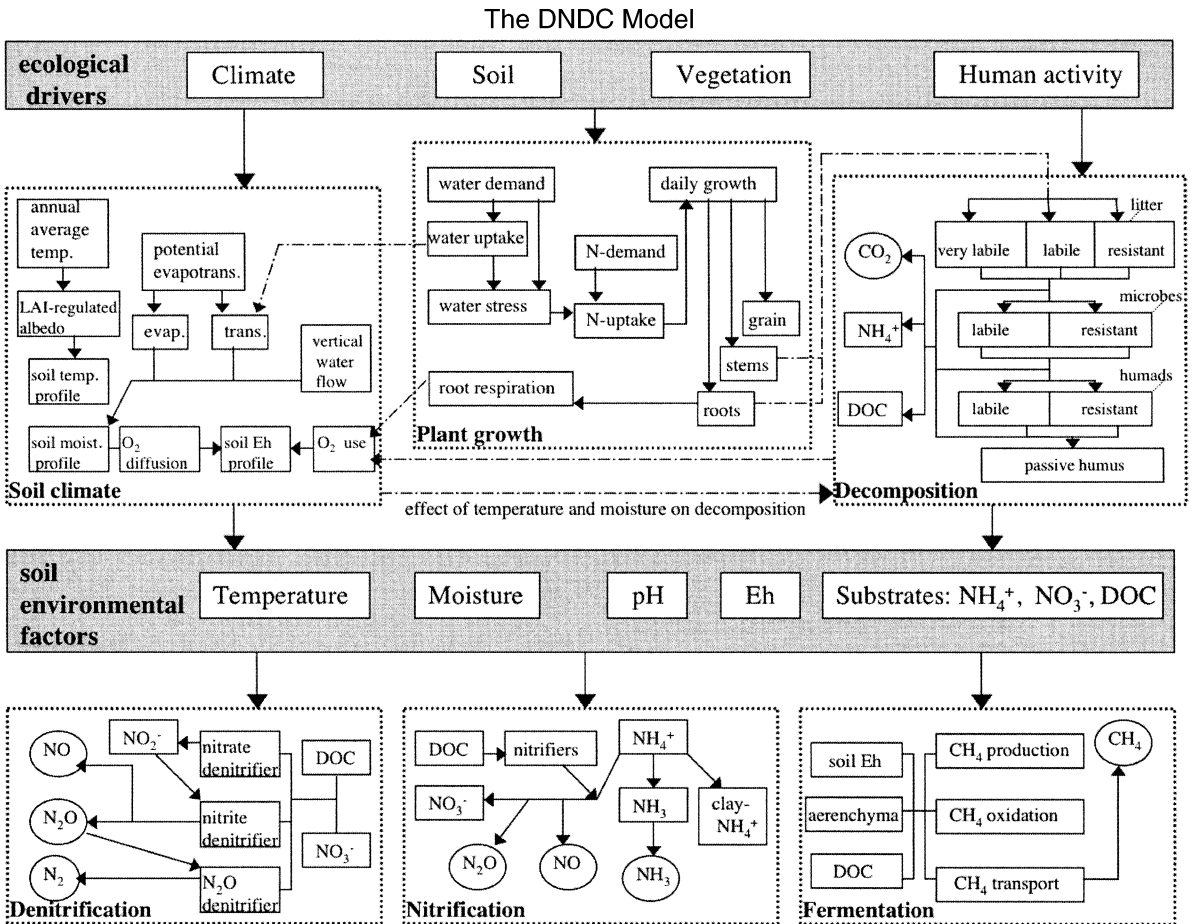

FIG. 1. The DNDC (denitrification–decomposition) model structure. DOC = dissolved organic carbon.

rural households are becoming more dependent on fossil fuels and less dependent on crop residues as a resource. Crop residue was widely used as a fuel in rural areas before the 1980s, but is no longer a major fuel source in many areas (Lin 1998). Small tractors have been replacing draft animals for cultivation, so there is less demand for crop residue as animal fodder. However, in most rural areas, farmers lack machinery to adequately chop crop residues for incorporation into the soil, so they must burn their crop residues in their fields to prepare their land for the next planting season (Zhang 2000). In the United States fossil fuels have been a primary resource for many decades. The practice of removing aboveground residues from the field, particularly for small grain crops, was common until the mid-20th century, but was gradually abandoned (Buyanovsky and Wagner 1998).

Using 1990 conditions, we modeled carbon (C) and nitrogen (N) biogeochemical cycles in croplands of China, and, for comparison, the United States. We investigated how tillage, crop residue management, and other practices in that year affected annual soil C and N budgets, including net changes in SOC and other C and N pools and fluxes for the croplands of the two countries.

METHODS

We conducted agroecosystem biogeochemical simulations tracking the cycles of C and N in all cropland of China and the United States. The denitrification–decomposition model (DNDC) was used to predict the interacting effects of various environmental factors (e.g., climate, soil properties, and land use and management) on crop yield, soil fertility, nitrogen leaching, and trace gas emissions to the atmosphere. The core of DNDC is a soil biogeochemistry model describing C and N transport and transformation driven by a series of soil environmental factors such as temperature, moisture, redox potential (i.e., Eh), pH, and substrate concentration gradients (Fig. 1). Submodels for soil climate, decomposition, nitrification, denitrification, and fermentation have been incorporated in DNDC to track the coupled C and N biochemical or geochemical reactions in the soil (Li et al. 1992, 1994, Li 2000). In addition, a crop growth submodel has been integrated with the biogeochemical submodels to simulate plant

TABLE 1. Characteristics of field sites in China.

Site (province)	Temp.† (°C)	Precipitation (mm/yr)	Crop rotation	Soil properties: Loam type	Soil properties: SOC (kg C/kg)	Soil properties: pH	Farming management: Tillage (yr^{-1})	Farming management: Fert.‡	Farming management: Manure§	Time span	Data source
Lianshui (Jiangsu)	15.5	950	rice and wheat	sandy	0.005	8.0	2	160	1	1984–1993	Luo et al. (1994)
Yucheng (Shandong)	14.5	780	soybean and wheat	sandy	0.004	9.2	2	260	2	1986–1994	Sui (1996)
Pingliang (Gansu)	10.0	660	corn, soybean, hay, and wheat	silty	0.004	8.0	2	45	1.35	1979–1986	Zhou and Ding (1991)
Yueyang (Hunan)	18.0	1440	rice, rice, and clover	sandy clay	0.017	6.8	4	260	none	1986–1990	Zeng and Liu (1993)

† Mean annual air temperature.
‡ Fertilizer, reported as kilograms of nitrogen per hectare per year.
§ Reported as thousands of kilograms of carbon per hectare per year.

photosynthesis, respiration, C allocation, litter production, and water and N uptake from the soil (Zhang et al. 2002). Basic physical, chemical, and biological laws governing the relevant reactions, as well as empirical equations developed from field and laboratory observations, were utilized to construct the model framework. DNDC is driven by daily meteorological data, soil properties, vegetation status, and anthropogenic activities including farming management. DNDC simulates the daily dynamics of soil climate profiles; plant development and growth; soil C and N pools, fluxes of CO_2, CH_4, N_2O, NO, N_2, and NH_3; and N leaching (Fig. 1). DNDC has been tested against numerous field observations regarding SOC dynamics and trace gas emissions in agroecosystems worldwide (Li et al. 1992, 1994, 1997, Smith et al. 1997, 1999, Wang et al. 1997, Frolking et al. 1998, Plant et al. 1998, Xiu et al. 1999, Li 2000, Brown et al. 2002, Zhang et al. 2002). These results indicate that DNDC is able to produce reasonable predictions for SOC dynamics and trace gas emissions from croplands across climate zones, soil types, and agricultural management regimes. The DNDC model is available free to download via the internet.[7]

[7] URL: ⟨www.dndc.sr.unh.edu⟩

We first conducted site-level, multi-year simulations at four agricultural field sites in China that had multi-year measurements of SOC: a rice and winter wheat rotation in Jiangsu Province, a soybean and winter wheat rotation in Shandong Province, a corn, soybean, hay, and winter wheat rotation in Gansu Province, and a rice, rice, and clover rotation in Hunan Province. For these simulations we used site-specific information on soil properties, crop rotation, fertilization and manure amendments, and tillage patterns (Table 1). All other management factors (planting and harvest dates, tillage depths for conventional tillage, and crop residue management, for which we had no site-specific information, were handled as for the national simulations described in the Appendix.

We then conducted county-scale simulations for all counties with cropland in both China and the United States. The database compiled to run the DNDC simulations contained daily weather data, soil properties, crop acreage, crop types and rotations, and farming practices (e.g., tillage, fertilization, manure amendment, and irrigation) at county or county-cluster scale for both countries (Li et al. 2001), based on 1990 information (Table 2; also see Appendix for more detail). We conducted two, daily-time-step, annual simulations

TABLE 2. Denitrification–decomposition (DNDC) model input and output variables.

Model input data required
Soil properties (county ranges): SOC, bulk density, pH, texture
Daily weather: precipitation, maximum and minimum air temperature
Cropland areas (hectare per county): all single-, double-, and triple-crop rotations
Farming management: fertilization, tillage, planting and harvest dates manure inputs, crop residue management
Model output
Crop productivity: grain, stem and root yield (kg C and N/ha), N-uptake, N-fixation by legumes
Trace gas fluxes: CH_4, CO_2, NH_3, NO, N_2O, N_2
Soil organic C and N pools
Soil inorganic N content

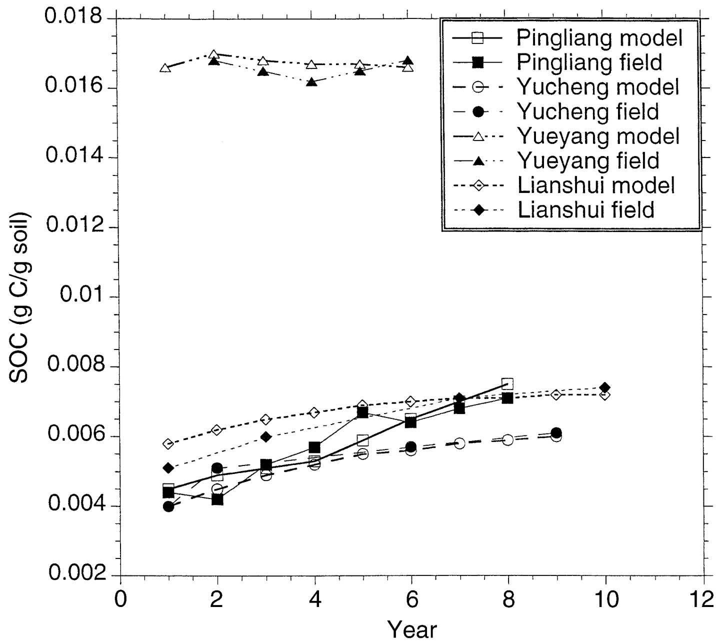

FIG. 2. Measured (solid symbols) and DNDC model simulated (open symbols) trends in soil organic C (SOC) for four multi-cropped fields in China (see Table 1 for site details and field data references).

for each crop rotation in each county. Each county had a range of SOC (based on databases listed in the Appendix), and paired simulations—one with low SOC and the other with high SOC—were conducted for all county and crop-rotation combinations. All other input variables were unchanged in the paired simulations. These paired simulations generated a range in model results (high SOC and low SOC) for each output variable for each county and crop-rotation combination. For the U.S. simulations, cropland was partitioned into three tillage classes, conventional till (tilled at planting to 0.1 m and after harvest to 0.2 m), reduced till (tilled at planting to 0.05 m and after harvest to 0.1 m), and no till (crop residue mulched on soil surface after harvest). Each crop in each county was run for all three tillage classes. County totals were calculated by averaging the simulated C and N pools and fluxes based on fractional area in each tillage class taken from state aggregate data (Kellogg et al. 2000). The DNDC model did not estimate soil and SOC loss due to physical erosion, so the results represent a conservative estimate of SOC loss rates.

Simulation outputs included a range of soil, plant, and trace gas C and N pools and fluxes (Table 2). The total C or N fluxes from each county were calculated by summing up crop-area-weighted fluxes from each crop rotation in each county, and county totals were aggregated to national totals. Reported changes in SOC thus represent the loss or gain in SOC over a one-year crop rotation, based on initialization of SOC pools to 1990 levels from available soil data.

RESULTS

Model comparison to field SOC data

The three sites with low soil organic carbon (SOC) (<0.007g C/g soil) all had manure inputs (green manure for Lianshui and Yucheng, farmyard manure for Pingliang) and showed increasing SOC over the ~10-yr period, while the moderate SOC site (~0.017 g C/g soil) had no manure inputs and showed little overall change in SOC (Fig. 2). Model trends were similar to observations, with the largest gains (~60%) at Pingliang, the site receiving farmyard manure and the only site growing maize, which produces more biomass than the other crops grown at these sites.

National analyses

Because of the relatively high sensitivity of C and N biogeochemical cycling to SOC content (e.g., Li et al. 1996), in any one county these high-SOC and low-SOC results are likely to generate a reasonable estimate of uncertainty, but when aggregated to national totals, all high-SOC or all low-SOC results will probably produce an unrealistically broad range. In the discussion below and in Fig. 3, we present mean results (average of high SOC and low SOC), while in Table 3 we report nationally aggregated high, low, and mean results.

In 1990 the top 0.3 m of cropland soils in China and the United States contained 5900 Tg C and 8000 Tg C, respectively (Table 3), based on nationwide soil survey data and relevant publications for the two countries (see Appendix). The denitrification–decomposition (DNDC) model simulations indicated that, on average in 1990, cropland soils in China lost 95 Tg C/yr (1.6% of total SOC), while cropland soils in the United States lost 7 Tg C/yr (0.1% of total SOC) (Table 3). It is important to note that in the simulations not all regions of China lost soil carbon (Fig. 3a). Regions of Inner Mongolia in the north, Tibet in the southwest, and Guizhou, Yunnan, and Hunan Provinces in the south

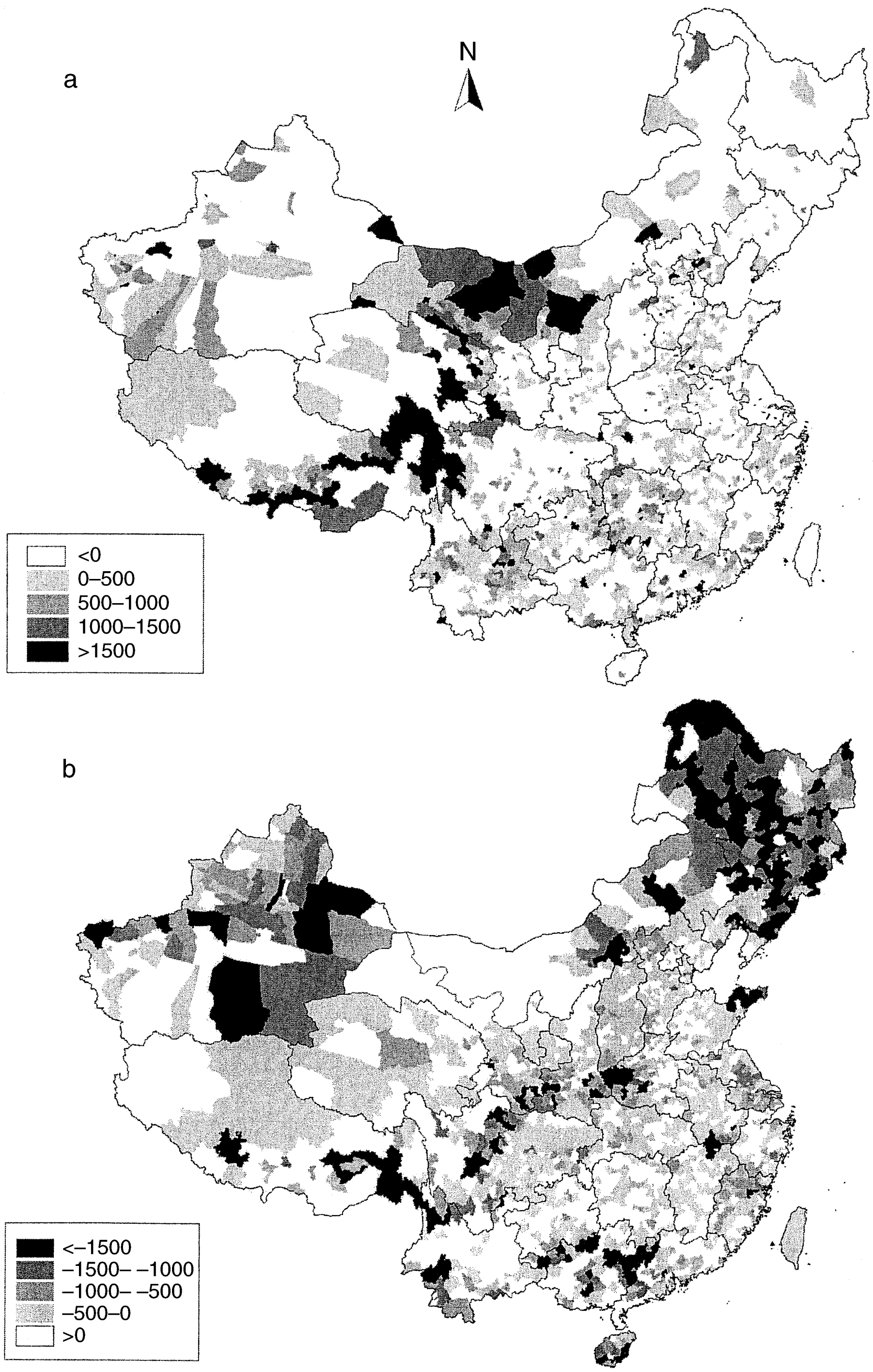

FIG. 3. Mean annual change in cropland soil organic carbon (SOC, in kilograms of C per hectare per year for the top 0–0.3 m of soil) for each county in China during the 1-yr simulation by the DNDC (denitrification–decomposition) model for (a) counties with increasing cropland SOC and (b) counties with decreasing cropland SOC. The black lines mark provincial boundaries. The maps shade an entire county based on average changes in cropland SOC, independent of cropland area per county. Large counties in the western half of China generally have small total cropland areas. SOC gains were greatest in a north–south band in central China, due to high livestock populations compared to cropland area, leading to significant manure additions. SOC losses were greatest in the northeast; loss rates were small to moderate in much of the rest of eastern China.

TABLE 3. Simulated annual cropland C and N budgets (Tg/yr) for China and the United States.

Cropland parameters	China			United States		
	High SOC	Low SOC	Mean	High SOC	Low SOC	Mean
Crop land area (10^6 km^2)	1.3	1.3	1.3	1.4	1.4	1.4
Crop sown area (10^6 km^2)	2.0	2.0	2.0	1.4	1.4	1.4
Total SOC (Tg C)†	8900	2900	5900	11 000	5000	8000
Grain/harvested yield (Tg C/yr)	320	270	290	340	310	330
Leaf + stem residue production (Tg C/yr)	300	260	280	300	270	280
Root production (Tg C/yr)	37	31	34	58	54	56
Soil C						
Input (Tg C/yr)						
Litter incorporation	82	70	76	330	300	320
Manure amendment	110	110	110	81	81	81
Output (Tg C/yr)						
CO_2 emission	390	140	270	550	250	400
CH_4 emission	8.4	7.1	7.7	0.12	0.21	0.17
DOC leaching	8.2	2.9	5.6	11	4.6	7.9
Annual balance (0–0.3 m) (Tg C/yr)	−220	29	−95	−150	130	−7.0
Soil N						
Input (Tg N/yr)						
Fertilizer	16	16	16	8.5	8.5	8.5
N deposition	1.5	1.5	1.5	6.9	6.9	6.9
N mineralization‡	24	11	18	33	18	25
N fixation§	0.3	0.3	0.3	2.7	2.6	2.6
Output (Tg N/yr)						
N uptake by crops	18	16	17	18	17	18
N leaching‖	7.2	1.0	4.1	22.9	13.3	18
NH_3 volatilization	11	11	11	0.9	0.9	0.9
N_2O emission	2.0	0.6	1.3	2.3	0.8	1.6
NO emission	0.4	0.2	0.3	0.6	0.4	0.5
N_2 emission	2.6	0.6	1.6	3.2	1.0	2.1

Notes: The table presents results from a simulation using highest and lowest reported soil organic carbon (SOC) for each county for all cropland. These two sets of results establish a credible output range for each county. However, national aggregations of these extreme simulations may be unrealistic. National aggregations are probably well represented by the mean of the high-SOC and low-SOC simulations.

† National aggregate SOC in top 0.3 m of cropland soils, based on high and low reported values for each county.

‡ Gross N mineralization.

§ Crop areas in 1990 are as follows: United States, soybeans = 0.24×10^6 km, alfalfa = 0.09×10^6 km; China, soybeans = 0.04×10^6 km, alfalfa = 0.003×10^6 km.

‖ N leached below 0.3 m in the soil; not necessarily all leached further into regional surface and/or groundwater.

all showed SOC gains. DNDC calculated substantial losses of SOC in northeastern, northwestern, and southwestern China (Fig. 3b). Rates of soil C loss were generally lower in the North China Plain and the hilly and mountainous regions west of the Sichuan Basin, in western Henan Province, and in Guanxi and Guangdong Provinces in the south.

The simulated 14-fold greater loss of carbon from agricultural soils in China, relative to the United States, from approximately the same cropland area, can be explained primarily by differences in agricultural management practices. DNDC predicted that 280 Tg C of aboveground residue (i.e., leaves and stems) and 34 Tg C of roots were produced in China in 1990, and 280 Tg C of aboveground crop residue and 56 Tg C of roots were produced in the United States (Table 3). Lal et al. (1998) estimated total nonharvested C production on cropland soils (aboveground residue, roots, and weeds) as ~450 Tg C. In the United States ~90% of aboveground crop residue was returned to the soil (Schomberg et al. 1994, Stewart and Moldenhauer 1994). In contrast, in China it is estimated that only 25% of aboveground crop residue was returned to the soil (Ministry of Agriculture 1997). Approximately 50% of aboveground crop residue was burned, ~20% was used as animal feed, and ~2% was used as industrial material for pulp production (Liu and Mu 1993) (this latter practice is no longer permitted, and burning is officially discouraged). Manure inputs for China were 110 Tg C, and 81 Tg C for the United States; total C inputs were ~190 Tg C for China and ~400 Tg C for the United States (Table 3).

DISCUSSION AND CONCLUSIONS

Though we conducted only one-year simulations, relatively stable management practices for crop residues during the past 4–5 decades in China (Liu and Mu 1993) would indicate that these results represent relatively long-term trends. Our results are consistent with numerous published reports based on Chinese soil surveys made in the 1950s, 1970s, and 1990s (Loess Plateau Investigation Team 1991, Duan et al. 1996, Li 1998, Lin 1998, Guan and Zhang 1999, Feng et al. 2000). Wang and Zhou (1999) report that farmland soils in China have lost significant SOC (soil organic C) over the past 50 yr. Cai (1996) inferred SOC losses of more than 50% in cropland in eastern China. More specifically, SOC decreased by 40% in cropland in Keshan County, Heilongjiang Province, from the 1950s to the 1980s (State Soil Survey Office of China 1998). According to our database, 1990 SOC contents in this county ranged from 2.0–3.4 g C/kg soil, which is equivalent to 65 000–110 000 kg C/ha for the top 30 cm of soil. Based on the reported 40% decrease, the SOC contents would have been 110 000–190 000 kg C/ha in the 1950s. The annual average SOC loss would have been 1500–2500 kg $C \cdot ha^{-1} \cdot yr^{-1}$; modeled SOC losses in Keshan were 1400–2950 kg $C \cdot ha^{-1} \cdot yr^{-1}$. In a cropland recently converted from grassland at Mousu grassland, Inner Mongolia, SOC decreased from 4.1 g C/kg (equivalent to 137 000 kg C/ha for the top 30 cm of soil) to 2.2 g C/kg (72 000 kg C/ha) during only five years after the conversion (Inner Mongolia Soil Survey Office 1994), a loss of ~13 000 kg $C \cdot ha^{-1} \cdot yr^{-1}$. The modeled SOC loss rate for cropland east of Huhehaot, Inner Mongolia, which had an initial SOC content similar to the Mousu grassland site, was ~7500 kg $C \cdot ha^{-1} \cdot yr^{-1}$ in 1990. Higher SOC losses observed in the field would be partially due to wind and/or water erosion (which are severe in this area); these factors are not included in the model simulations.

The regions with large simulated SOC losses, particularly in the northeast, have been relatively recently cultivated (generally in the past 100 yr) and had high initial SOC contents, and thus the potential for significant and sustained SOC losses. Much of the North China Plain, the lower Yangtze River basin, and the Sichuan Basin, which have been cultivated for millennia, were losing SOC at a much lower rate, primarily because their soils are already depleted in SOC (e.g., Cai 1996). The regions with large simulated increases in SOC—Inner Mongolia, Tibet, Guizhou, Yunnan, and Hunan Provinces—generally had high animal populations relative to cropland area, so manure inputs were high in the simulations. Paddy rice-dominant regions (southern China) generally had low losses or slight gains in SOC, primarily due to seasonal flooding that reduced SOC decomposition (e.g., Cai 1996, Witt et al. 2000).

Net loss of carbon from agricultural soils contributes to the growth of atmospheric CO_2 concentration and the potential for global warming. Higher average SOC in U.S. cropland soils led to larger soil CO_2 emissions to the atmosphere (Table 3). However, net cropland soil C losses were larger for China's cropland soils, because C inputs from crop residues were much less. Currently, sequestration of atmospheric CO_2 in agricultural soils is a component of the Kyoto strategy for reducing the threat of global warming. Our results raise an important issue of how a baseline might be established for measuring credits for agricultural sequestration of C in soils. Would countries be required to achieve ''best practices'' in soil C management before receiving credit for changes in management practices that enhance sequestration and lead to net storage? Changing SOC is one component of the total agricultural CO_2 budget, which is one component of the agricultural-sector greenhouse gas budget, and only a small component of the total greenhouse gas budget of a nation (e.g., U.S. EPA 2002). Larger SOC losses from cropland soils in China than the United States do not necessarily mean that the agricultural sector in China is overall a larger source of greenhouse gases than the agricultural sector in the United States.

Declining cropland SOC levels also impact the soil nitrogen cycle. Lower SOC levels and crop residue inputs in China's croplands reduce net N mineralization to ~75% of the United States total (Table 3). Chinese farmers overcome this shortfall of in situ N availability by applying about twice as much synthetic N fertilizer as U.S. farmers (Table 3). About 40% of China's cropland was multi-cropped (two or more harvests per year from the same plot of land), so total sown and harvested areas were greater for China than the United States (Table 3). Multiple cropping generates an increased demand of N per unit area, leading to significantly higher fertilization rates in China; mean national fertilization rates were ~120 kg N/ha for China and ~60 kg N/ha for the United States. Synthetic fertilizers, typically applied once or a few times per crop cycle, are generally more prone to leaching losses than is more slowly released mineralized N (Zhang et al. 1997).

A dramatic increase in use of synthetic fertilizers has been crucial to maintaining and increasing crop yields in China. However, DNDC predicted that 4.1 Tg N/yr was leached below the crop root zone as dissolved nitrate in China (Table 3). Zheng et al. (2002) estimated that excess fertilizer application in China in the 1990s was about 3–6 Tg N/yr. According to statistics from the Ministry of Agriculture of China, the average efficiency for N fertilizer use is about 30% (Zhu 1992, Sun 1995). Excess fertilizer can be lost to the atmosphere and to surface water or groundwater bodies. Our results suggest that inefficient utilization of fertilizer N could be contributing to eutrophication of surface water and nitrate contamination of subsurface water supplies (Zhang et al. 1997). These predictions are supported by a recent report (Fu 2001) from the People's Daily, China's official newspaper, that reported that application of synthetic fertilizers increased by 90% while crop production was elevated only by 10% from 1984 to 1994. The 30–35% efficiency for fertilizer use is 10–15% lower than that in most developed countries. This inefficiency has caused severe contamination, especially eutrophication, to China's water bodies (Fu

2001). Tai Lake, a very large lake in Jiangsu Province in a major rice agriculture region, received $48\,000 \times 10^3$ kg N every year during the 1980s (Zhang 1993). The Nanjing Institute of Limnology, Chinese Academy of Sciences, reported that half of the lakes in China were severely eutrophic (Zhang 1993). In 1994–1996, the Institute of Soil and Fertilizer, Chinese Academy of Agricultural Sciences, tested 102 groundwater samples from 15 counties in Beijing, Tianjin, Hebei, and Shandong Provinces, and found 47% of the samples contained nitrate over the drinking-water standards (Zhang 1995).

During the last five decades China has largely fed ~20% of the world population using only ~10% of the world's arable land (FAOSTAT 2001). Chinese agricultural yields increased substantially from the 1960s through the 1980s due to the introduction of new plant varieties, a 30% increase in irrigation, and increases in fertilization rates averaging 18% annually; however, in the 1990s there was a slowing of both growth in yields and total agricultural productivity (World Bank 1997*a*). Changes in agricultural policies in the 1990s, inadequate protection of arable land, and environmental degradation are cited as factors that could rapidly undermine food production in China (Baiming 1999), with eroded land area increasing by ~10 000 km^2/yr and desertified land increasing by ~2500 km^2/yr (Zhang 2000). Recently reported widespread problems with water supply and quality, soil degradation, and severe erosion have a common character—they are occurring at an unprecedented scale and they are causing severe damage to both agricultural development and environmental quality (Smil 1993, World Bank 1997*b*, Qi et al. 1999, Xie 2000, Fu 2001).

The sustainability of Chinese agriculture faces the multiple stresses of soil degradation, water shortages, water pollution, and climate change (Baiming 1999). Based on numerous reports in China (Loess Plateau Investigation Team 1991, Duan et al. 1996, Li 1998, Lin 1998, Guan and Zhang 1999, Wang and Zhou 1999, Feng et al. 2000), long-term loss of soil organic matter is becoming one of the major threats to sustainable agriculture in China. By simply burning the crop residue or using it for other purposes, farmers are losing a valuable resource for maintaining or improving the quality of their soil.

The recently reported increases in soil degradation and water pollution have gotten the attention of managers and policy makers in China (e.g., Xie 2000, Zhang 2000, Fu 2001), and efforts are now underway to enhance the sustainability of agriculture in China, with, for example, increased use of slow-release fertilizers and decreased burning of crop residues. Improved soil C management will require a widespread change in the practices and economic priorities of Chinese farmers. Based on the predictions of the DNDC model (Li et al. 1994), an increase in percentage of crop residue incorporated in the soils from 15% to 90% will increase the N mineralization rates by 10–25% (depending on the climatic, soil, and crop conditions) in the following year, which is equivalent to about an additional 3 Tg N annually added to Chinese cropland; it will take a longer time (50–100 yr) to elevate the SOC contents to significantly higher levels.

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APPENDIX

Data sources used in the DNDC model for both China and the United States are available in ESA's Electronic Data Archives: *Ecological Archives* A013-006-A1.