Are soils of Iowa USA currently a carbon sink or source? Simulated changes in SOC stock from 1972 to 2007

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Shuguang Liu
Central South University of Forestry and Technology

Zhengxi Tan
ASRC at USGS EROS

Zhengpeng Li
University of Maryland, College Park

Shuqing Zhao
Peking University

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Are soils of Iowa USA currently a carbon sink or source? Simulated changes in SOC stock from 1972 to 2007

Shuguang Liu[a], Zhengxi Tan[b,*], Zhengpeng Li[b], Shuqing Zhao[c], Wenping Yuan[d]

[a] U.S. Geological Survey (USGS) Earth Resources Observation and Science (EROS) Center, Sioux Falls, SD 57198, USA
[b] ARTS at U.S. Geological Survey (USGS) Earth Resources Observation and Science (EROS) Center, Sioux Falls, SD 57198, USA
[c] College of Urban and Environmental Sciences, Peking University, Beijing 100871, China
[d] College of Global Change and Earth System Science, Beijing Normal University, Beijing 100875, China

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ABSTRACT

Upscaling the spatial and temporal changes in carbon (C) stocks and fluxes from sites to regions is a critical and challenging step toward improving our understanding of the dynamics of C sources and sinks over large areas. This study simulated soil organic C (SOC) dynamics within 0–100 cm depth of soils across the state of Iowa in the USA from 1972 to 2007 using the General Ensemble biogeochemical Modeling System (GEMS). The model outputs with variation coefficient were analyzed and assembled from simulation unit to the state scale based upon major land use types at annual step. Results from this study indicate that soils (within a depth of 0–100 cm) in Iowa had been a SOC source at a rate of 190 ± 380 kg C ha^{-1} yr^{-1}. This was likely caused by the installation of a massive drainage system which led to the release of SOC from deep soil layers previously protected under poor drainage conditions. The annual crop rotation was another major force driving SOC variation and resulted in spatial variability of annual budgets in all croplands. Annual rate of change of SOC stocks in all land types depended significantly on the baseline SOC levels; soils with higher SOC levels tended to be C sources, and those with lower levels tended to be C sinks. Management practices (e.g., conservation tillage and residue management practices) slowed down the C emissions from Iowa soils, but could not reverse the general trend of net SOC loss in view of the entire state due mainly to a high level of baseline SOC stocks.

1. Introduction

Due to the importance of agricultural land in terrestrial carbon (C) accounting many studies on agricultural ecosystems since the mid-1990s have focused on C dynamics and their associated driving forces (Paustian et al., 1997; Lal, 2004). Carbon accounting at some long-term agricultural experimental sites suggests that land use changes caused substantial soil organic C (SOC) loss from North American terrestrial ecosystems between the 1850s and the 1950s; since the 1970s, improved farming practices (e.g., conservation tillage and residue management, crop rotation, and elevated fertilization rates) in many areas stabilized or increased SOC stock (Paustian et al., 1997; Roose et al., 2006). However, most such studies have some drawbacks: (1) sites were specific and under experimental control; (2) the studies were limited to the top soil layer (usually less than 30 cm in depth); and (3) the results from those studies were derived from either a static land use scenario or land use change statistics with decadal time intervals. The magnitude and spatial variations of C sources or sinks in crop-dominated ecosystems over large areas are still uncertain because of the difficulty in quantifying the spatial variability of site conditions (such as antecedent SOC stock and contemporary land use change) and the diversity of land management. For the U.S. Corn Belt, few regional C estimates are available. Brenner et al. (2001) reported that conventional farming systems (including reduced tillage) in Iowa made the soil a C sink of 80 kg C ha^{-1} in 1996. Evrendilek and Wali (2004) reported that Ohio croplands (except for the corn-for-grain cropland) acted as a C source of 56 kg C ha^{-1} in 1996, while the continuous corn cropland turned out to be a C sink of 26 g C m^{-2} in the same year. Those estimates depended highly on their specific natural and management variables.

Land use and land cover change (LUCC) information is critical for estimating regional C budget. Changes in cropland area, crop composition, fertilization rate, tillage, and other management practices influence C fluxes over cropping systems (Kern and Johnson, 1993). Unfortunately, many previous C modeling studies could not include temporal land use change because of the limited availability of temporal LUCC data. The General Ensemble biogeochemical Modeling System (GEMS) is a new type of LUCC-oriented, regional level, biogeochemical simulation system designed for assimilating dynamic

* Corresponding author. Tel.: +1 605594 6903; fax: +1 605594 6529.
E-mail address: ztan@usgs.gov (Z. Tan).

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LUCC data (Liu, 2009). GEMS can spatially simulate the impacts of dynamic land use and climate, as well as the effects of management practices. GEMS can also estimate uncertainty through ensemble simulations, which transfer uncertainties from inputs to outputs and capture the spatial and temporal variability of net primary productivity (NPP) and crop yield. For agricultural ecosystems, GEMS can generate crop rotations/combinations based upon agricultural census data and produce various soil input data for biogeochemical simulations. This model has been applied in diverse ecosystems (Liu et al., 2004a,b; Tan et al., 2005, 2006a, 2009a,b, 2010; Zhao et al., 2010). GEMS can also function as a platform to encapsulate other biogeochemical models (e.g., CENTURY, EDCM (Liu et al., 2003; Zhao et al., 2010)); additionally, it can drive and automatically parameterize them with the same input data (with or without minimum modifications) to the encapsulated models; this is useful for reusing models that are difficult to modify (Liu, 2009; Zhao et al., 2010).

To simulate the dynamics of SOC under nonequilibrium conditions, models with a multi-soil-layer structure are necessary (Sharpley and Williams, 1990). Because of soil erosion or deposition, characteristics of a soil profile and SOC stocks in all soil layers may change dynamically. If the thickness of the top layer is fixed, its SOC content must experience a dynamic replacement (Harden et al., 1999; Liu et al., 2003). Therefore, soil decomposition processes must be changed as well due to the increase (under erosion) or decrease (under deposition) of exposure of SOC in deep layers, and the biomass, growth, and death of plant roots must also change dynamically (Liu et al., 2003).

The role of conventional tillage practices in SOC losses from cropland and the potential to sequester atmospheric C into cropped soils by adopting conservation tillage measures have been intensively investigated and widely realized (West and Post, 2002; Lal, 2004). Beside tillage disturbances, drainage condition (especially internal drainage) has been recognized as the second major force driving SOC dynamic in cropland (Baker et al., 2007). Generally, poorly drained environments favor SOC accumulation and well-drained environment enhance the soil organic matter decomposition and C emissions (Tan et al., 2004). Improvement of drainage conditions through an internal tile drainage system within poorly drained soils can promote crop root development and lead to an increase in crop biomass (both above- and below-ground) and yield (Kanwar et al., 1988). For instance, internal (tile) drainage was reported to lead to yield increases of 630–2820 kg ha^{-1} for corn and 100–400 kg ha^{-1} for soybean in Iowa (Wright and Sands, 2001; Drury et al., 2009) even though tile drainage could also promote the NO_3-N leaching (Randall et al., 1997) due to soil aeration improvement for microbial activities to increase N mineralization (Updegraff et al., 1995) and subsequent nitrification (Regina et al., 1996). However, there are no systematic observation data available for demonstrating the effects of internal drainage system on SOC dynamics over large areas yet, especially for illustrating the vertical variability through the soil body above the tile system.

In this study, we integrated existing research capabilities in biogeochemical cycling, remote sensing, and ecosystem science to respond to the need for information by policy makers and landowners regarding how annual crop rotation, tillage practices, and soil internal drainage affect the SOC budgets within the 0–100 cm soils in the Western Corn Belt of the United States.

2. Materials and methods

2.1. Study area

The study area, the state of Iowa, is located in the north-central part of the United States and covers an area of 144,066 km^2; it is a core part of the Western Corn Belt Plains Ecoregion (Omernik, 1987). Average annual precipitation varies from 710 mm in the northeast to 965 mm in the southwest, and average annual minimum temperature is only 0.5 °C in the northeast and 6.1 °C in the southwest, with monthly average temperatures ranging from −14.3 to 30.1 °C. Soils in the central and northern regions have relatively lower clay content, higher bulk density, and higher SOC stock levels than other parts of the state. Cropland accounts for more than 85% of the state, and much of the remaining land is used for feed grain to support livestock production. Major crops include corn, soybean, alfalfa, and grain sorghum.

2.2. Modeling system—GEMS-EDCM

2.2.1. The encapsulated plot-scale model EDCM

The Erosion Deposition Carbon Model (EDCM) (Liu et al., 2003; Zhao et al., 2010) was used in this study to quantify SOC stocks, while GEMS (see Liu, 2009 for detail) was used as a platform to encapsulate EDCM, drive the encapsulated model with the same input data and automatically parameterize the EDCM according to biophysical conditions of any land parcel, and deploy it across space without considering the interactions among land pixels from plot scale to regional scale (Liu et al., 2003).

EDCM, similar to CENTURY (Parton et al., 1987), is a process-based biogeochemical model (Liu et al., 2003; Zhao et al., 2010) and was developed to characterize the SOC dynamics in a soil profile and to be capable of evaluating the impacts of soil erosion and deposition. CENTURY has a one-topsoil-layer structure for simulating C cycle, but EDCM adopts a multiple soil-layer structure to account for the stratification of a soil profile and SOC stock in each soil layer. EDCM dynamically tracks the evolution of the soil profile and C storage as influenced by soil erosion and deposition. It was selected in this study as the underlying ecosystem biogeochemical model in GEMS to simulate C and N cycles.

EDCM, like the CENTURY, can simulate C and N cycles in diverse ecosystems at a monthly time step and model the impacts of management practices including LUCC, fertilization, and cultivation (e.g., Liu et al., 2003; Zhao et al., 2010). The major inputs for EDCM include land cover and land use type, monthly average maximum and minimum air temperature, monthly precipitation, soil texture, initial SOC level, atmospheric N deposition, and various management practices. The major output variables relevant to the proposed project include NPP, grain yield, C decomposition, C exchange rates between ecosystems and the atmosphere, biomass removal by harvesting, and C stocks in vegetation and soils.

Stochastic simulation ensembles have been used in GEMS to (1) incorporate variances and covariance of input data and (2) transfer input data uncertainty into model outputs. Through ensemble simulations, the nonlinearity of the ecosystem models are adequately addressed, and the uncertainty of the model outputs quantified by performing 20 ensemble stochastic model simulations were performed for each simulation unit to capture the heterogeneity and uncertainty of the data that define the simulation unit.

2.2.2. Computing and partitioning SOC pools in a soil profile for modeling

The STATSGO soil database (USDA, 1994) was used in this study to provide the initial soil information for GEMS (e.g., soil layer depth, soil organic matter content, bulk density, soil texture fractions). The STATSGO database for the state of Iowa contains 76 map units where each map unit consists of numerical components. The numbers of soil layers for a soil map unit may vary from 1 to 6 and the depth of each layer ranges from 10 cm to 160 cm. EDCM model can simulate up to 10 soil layers with an equal layer depth of either 10 cm or 20 cm. In this study, the values of soil variables from the database were recalculated at an equal interval of 20 cm based on the depth weight of each soil layer.

Soil organic C stock is usually partitioned into three pools based upon resistance to microbial degradation as defined in the CENTURY model (Parton et al., 1987, 1994): labile (or active), slow, and passive. Labile C pool has a short turnover time (less than 5 years) and consists of rapidly decomposable SOC fractions. Passive C pool has a turnover time of several hundred years or even longer, and consists of microbial-resistant components such as humic substances (Nicolardot et al., 1994). The dynamics of each SOC pool and their fractions in the total SOC stock vary with soil type and horizon (depth). Generally, the passive SOC fraction is about 0.5 and the labile fraction is about 0.1 for the top layer. GEMS uses the following algorithms to partition the total SOC stock into the three C fractions within a soil profile (see Liu et al., 2003):

$$\text{Passive SOC pool} = 0.86 - 0.69 * \text{Passive\_deep} + (1.69 * \text{Passive\_deep} - 0.86) * (1 - \text{EXP}(-0.05 * \text{Soil\_depth}) \tag{1}$$

$$\text{Active SOC pool} = 0.06 * \text{EXP}(-0.018 * \text{Soil\_depth}) \tag{2}$$

$$\text{Slow SOC pool} = 1 - \text{passive–active} \tag{3}$$

where Passive_deep is the passive SOC fraction in deep layers (can be 0–1).

Soil depth in cm is the middle depth of each layer. For example, it could be 10 cm for the top 20 cm layer of soil.

2.2.3. Supporting data for automated model parameterization

Below are the essential geospatial datasets used in this study and most of them are required by GEMS:

(1) Annual cropland grid maps from 2000 to 2007. They were derived from the Cropland Data Layer (CDL) that were generated by USDA National Agricultural Statistics Service and downloaded from http://www.nass.usda.gov/research/Cropland/SARS1a.htm.
(2) Climatic variables grids covering the years from 1972 to 2007 (consisting of mean monthly precipitation and mean monthly minimum and maximum temperatures). These grid layers were derived from PRISM Group of Oregon State University (http://www.prism.oregonstate.edu/).
(3) Iowa STATSGO soil database (USDA, 1994). This soil map consists of 2028 polygons that are associated with 76 STASGO map units across the state of Iowa.
(4) Nitrogen deposition map. This map spatially depicts the total atmospheric N deposition from wet and dry sources. It was gathered from the National Atmospheric Deposition Program (http://nadp.sws.uiuc.edu/).
(5) Drainage class grid layer. It was derived from the Compound Topographical Index (CTI) map using an empirical approach (see Eq. (4) in Section 2.2.4). The original CTI map was obtained from the U.S. Geological Survey (USGS) Earth Resources Observation and Science (EROS).
(6) Irrigation distribution map. It was extracted from the 2001 U.S. national irrigation layer generated by USGS EROS.

A GIS layer of the county codes (FIPS) of Iowa.

Other attribute data include the following:

(1) Crop composition and crop rotation probabilities, which were derived from the National Resources Inventory (NRI) database (http://www.nrcs.usda.gov/technical/NRI/).
(2) Forest Inventory and Analysis (FIA) data, including forest species and age distribution, cutting or logging records, etc. They were obtained from USDA Forest Service.
(3) Tillage practices and residue management statistics from 1989 to 2008. They were collected from the Conservation Technology Information Center (2008).

All geospatial data layers were processed to be an identical projection and coordinate system, and then overlain to form one grid layer with a common cell size of 2 km by 2 km (spatial resolution). This grid layer is called the “Joint Frequency Distribution” (JFD) layer and its attribute table is called the JFD table, which was used in GEMS. There were 35,370 different JFD cases in this study, and each JFD case was the simulation unit in GEMS.

2.2.4. Impact of tile drainage

It is critical to represent the dramatic change in drainage conditions throughout Iowa. A massive tile drainage system was developed in Iowa to convert native prairies and wetlands to highly productive croplands. An empirical model was developed to define drainage conditions at any depth in a soil profile and is described as follows:

$$\text{Drainage} = \text{drain0} * \frac{(1 - 1/(1 + \text{EXP}(-d0 * (\text{depth} - \text{tile\_depth} + 40))))}{(1 - 1/(1 + \text{EXP}(d0 * (\text{tile\_depth} - 40))))} \tag{4}$$

where Drainage is the drainage change coefficient ranging from 0 to 1 representing classes from poorly drained to well-drained; drain0 is the drainage at the top of the soil surface; *d*0 is the drainage change coefficient or drainage curve flatness transitioning from well-drained to poorly drained; tile_depth is the burial depth of tile drainage system, and depth is the middle point of each layer. The unit for all depths is in cm.

Tile_depth is an important parameter in the estimation of drainage in a soil profile. Most tile drainage systems were buried between 75 cm and 120 cm below the soil surface (Singh et al., 2007). For the first run, we set tile_depth = 100 cm, then investigated the sensitivity of SOC stock to the tile depth (between 75 cm and 120 cm).

Following Eq. (4), it can be seen that a smaller *d*0 value indicates gradual and long-tail vertical distribution of drainage, and a larger *d*0 value indicates a sudden change. The values greater than 0.2 do not create any significant differences. So the value for *d*0 should be from 0 to 0.2. With *d*0 = 0, the value of drainage does not change vertically, and the EDCM returns to the original EDCM values. For Eq. (4), we set *d*0 = 0.05, and drain0 = 1.0 which represents the well-drained class in this study. The uncertainty of tile drainage on SOC dynamics was quantified by EDCM based on the built-in algorithms (see Liu et al., 2003).

2.2.5. Automation of model parameterization

Because most information in spatial databases is aggregated to the map unit level, direct injection of such information into modeling processes is often problematic and subject to potential biases (Kimball et al., 1999; Reiners et al., 2002). Consequently, scaling methods are usually needed to incorporate field-scale spatial heterogeneities of state and driving variables into the simulations. An automated model parameterization system (AMPS) in GEMS generally consists of two major interdependent parts: (1) data search and retrieval algorithms and (2) data processing mechanisms. The first part searches and retrieves relevant information from various databases according to the keys provided by the JFD table that was associated with the JFD layer as defined in Section 2.2.3. The data processing mechanisms downscale the aggregated information at the map unit level to the field scale using a Monte Carlo approach.

Table 1
Areal percentages of major land use types in Iowa from 2000 through 2007.

Land use type	2000	2001	2002	2003	2004	2005	2006	2007	Mean
Water	0.8	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9
Wetlands	1.1	1.2	1.4	1.4	1.4	1.4	1.5	1.6	1.4
Urban/developed	4.2	3.7	4.9	3.8	4.5	4.5	4.0	6.1	4.5
Mixed forests	4.8	4.9	5.5	5.3	5.7	5.7	6.4	6.8	5.7
Shrub/scrub	0.0	0.1	0.1	0.1	0.1	0.1	0.1	0.2	0.1
Grass	1.9	1.8	2.6	2.6	2.5	2.5	2.6	3.8	2.5
Agricultural land									
Pasture/hay	5.1	6.0	7.6	7.2	7.7	7.7	9.0	11.3	7.7
Corn	33.2	32.7	32.4	33.6	31.7	31.7	33.7	35.0	33.0
Soybean	29.7	30.0	26.1	29.2	28.1	28.1	28.2	21.7	27.6
Other crop	10.7	9.9	11.4	9.0	11.8	11.8	8.7	12.1	10.7
Forage (alfalfa)	2.6	2.6	2.2	3.0	1.7	1.7	2.2	0.4	2.1
Fallow/idle crop	5.9	6.2	4.8	4.1	3.9	3.9	2.6	0.0	3.9
Sum	87.1	87.4	84.6	86.0	84.9	84.9	84.5	80.5	85.0

Total land surface area of Iowa is about 145,720 km^2. *Data source*: Cropland Data Layers provided by USDA National Agricultural Statistics Service and available at http://www.nass.usda.gov/research/Cropland/SARS1a.htm.

3. Results

3.1. Annual changes in land use

Historically, Iowa has been dominated by cropping systems where about 85% of all land area are cropped. Our analyses of the NASS Cropland Data Layers (CDL) from 2000 to 2007 show some change in the proportion of dominant crop types, fallow, and grassland (Table 1). Grassland area increased from 1.9% to 3.8%, and mixed forestlands increased from 4.8% to 6.8%, but the total cropland area declined from 87.1% to 80.5%, despite some interannual variation. The decrease in the total cropland area was mainly attributed to the reduction in the areas of soybeans (by 8%) and fallow (by ~6%), even though there was an increase in corn (1.8%). County-level statistics from the Conservation Technology Information Center (CTIC, 2008) indicate that the loss of cropland could be attributed to increased participation in the Conservation Reserve Program (CRP) (whose total area increased from 709,500 ha in 2000 to 918,036 ha in 2006), resulting in a corresponding increase in grassland area during this period, but the fallow area shrank from 5.9% to 2.6% between 2000 and 2006, and almost disappeared in 2007.

3.2. Annual biomass production and harvest

The data presented in Fig. 1 show that the average grain yield from 1972 to 2007 increased by 108%; specifically, the corn yield increased 138% while the soybean yield increased 36%. In the meantime, the biomass removed by harvesting from croplands increased 65%, which might be a main contributor to the deficit of net C flux in cropping systems. The continuous increment in both crop yield and total biomass over time could also be attributed to the improvement of soil drainage conditions, improved crop species, elevated fertilization rate, and weed and pest control. These profitability-oriented management measures could, to a great extent, offset the adverse impacts caused by natural disturbances such as flooding and drought. That may be why no explicit correlation could be found between crop yields and weather variables in general. However, historical low yields are found to associate with the years having either an extreme high annual precipitation (e.g., 1993) or an extreme low one (e.g., 1994, 1997, 2005, as shown in Fig. 1).

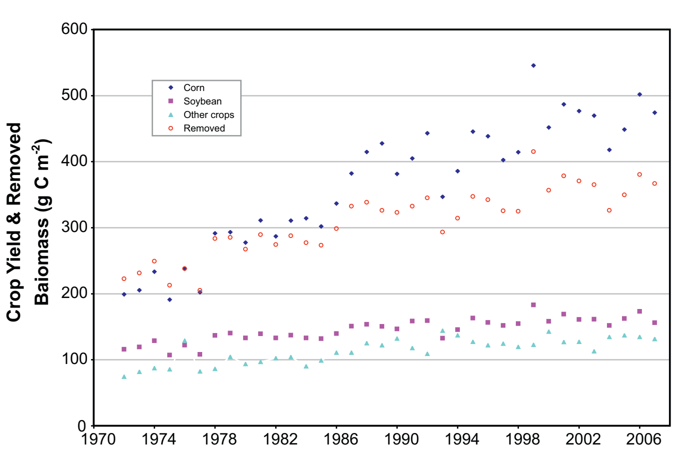

Fig. 1. Historical annual crop yields and biomass removed from cropping systems by harvest in Iowa.

3.3. Historical SOC stocks and their vertical distribution

Fig. 2 indicates that the total SOC within the 0–100 cm soil profile decreased from 185.3 Mg C ha^{-1} in 1972 to 168.8 Mg C ha^{-1} in 2007 (about 10%). This decrease was principally attributed to the reduction within the upper 60 cm of soil. The reduction rate of SOC stock prior to the mid-1980s was greater than that after the mid-1980s, especially the SOC loss from the top 20 cm of soil at an annual reduction rate of 0.453% (±0.168%). Since the mid-1980s, the SOC reduction rate from the topsoil has become smaller and tended to stabilize until 2007, with an annual reduction rate of 0.092% (±0.136%). However, the SOC stocks beneath the 20 cm depth showed a continuous decline by 13.4%, 10.9%, 3.0%, and 1.4% for each 20 cm level of soil depth, respectively, until the end of 2007. The annual reduction rate of the SOC stock within the 0–100 cm

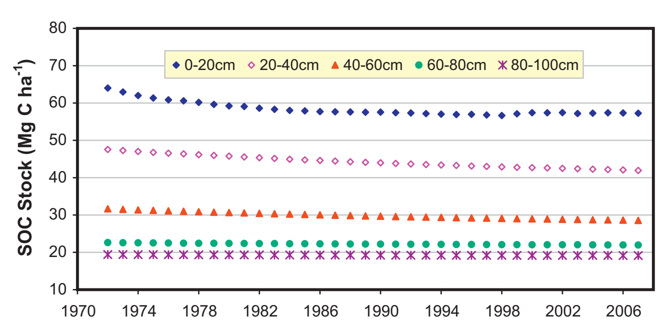

Fig. 2. Historical spectrum of SOC stocks within each 20 cm depth averaged for all land use types across the state of Iowa.

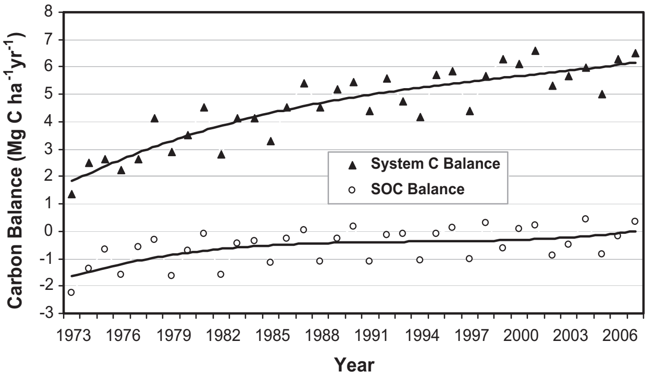

Fig. 3. Annual total C and SOC balances in cropping system (SOC balance means the difference in soil organic C stock from the previous year, and system C balance refers to the sum of SOC balance and removed C by harvesting grain and straw in cropping systems).

soil depth was 0.93% (±0.62%) between 1972 and 1985 and 0.33% (±0.27%) between 1985 and 2007.

3.4. Annual SOC budgets and spatial distribution

Counting the biomass removed by harvesting, all cropping systems were a C sink with increasing strength over time (Fig. 3) due mainly to the increase in above-biomass production (see Fig. 1). In the meantime, the SOC stock declined even though the SOC source strength tended to weaken from 1972 to 2007.

As illustrated in Fig. 4, the spatial patterns of SOC stock varied over time, and the balance as of 2007 (especially for the top 20 cm layer) appeared to be closely related to the SOC levels in 1972. The soils with higher initial SOC stocks tended to lose more SOC. After about the mid-1980s, there were no significant changes in SOC stocks (see Fig. 2).

Similarly, the spatial patterns of SOC budgets within either the top 20 cm or 100 cm depth depended heavily on the SOC magnitudes in 1972 regardless of the effects of improved drainage and cropping systems.

4. Discussion

4.1. Uncertainty control of ensemble simulations and validation

Uncertainties are manifested at the simulation unit (JFD) level in this study because all the ensemble simulations are performed at this level in GEMS. Input uncertainty (both initialization and driving forces) and model uncertainty (i.e., stochastic simulations) are also transferred to outputs at the JFD level. GEMS simulations for each JFD case were executed to incorporate the variability of inputs. Values for the selected output variables were written to a set of output files after each model execution and then aggregated for the study area using the SAS Macros program (SAS Institute Inc.,

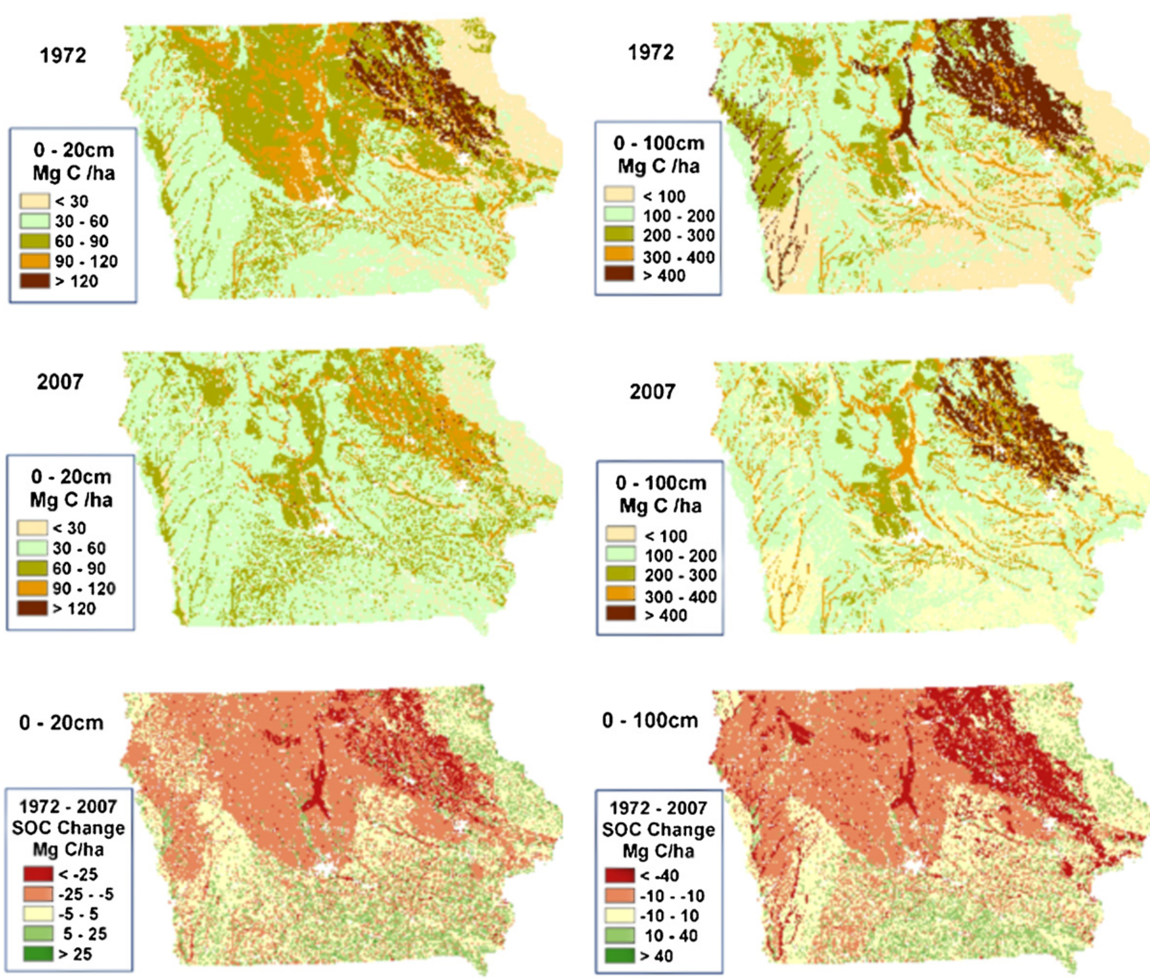

Fig. 4. Comparison of spatial distribution of simulated SOC stocks within the top 0–20 cm of soil (left) and 0–100 cm profile (right) across Iowa, and their differences between 1972 and 2007.

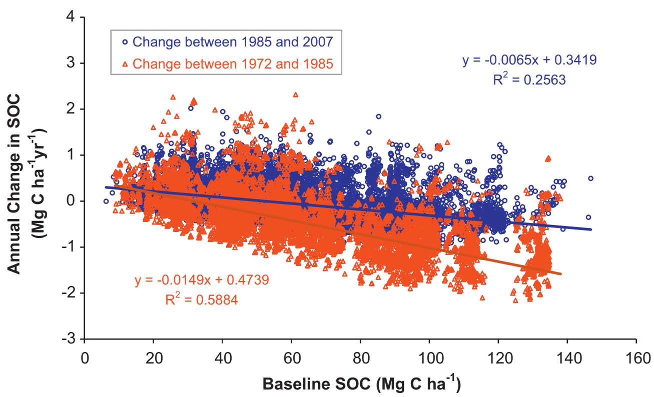

Fig. 5. Relation between annual SOC changes and the antecedent SOC stock levels in the top 20 cm depth of soils across Iowa croplands for two time periods: 1972 to 1985 (red line) and 1985 to 2007 (blue line).

2004). Meanwhile, the uncertainty of the simulations was evaluated in terms of the coefficient of variation (CV) with all model outputs.

The estimation of NPP needs to set up a maximum potential NPP parameter for each land cover and land use type. We used grain yields of major crops from USDA county statistics and field-observed biomass data from literature as references to verify corresponding outputs, and repeatedly ran simulations by adjusting parameters after each run until the outputs matched the references as closely as possible. Our validation analysis indicates that about 90% of variance in model outputs can be explained, ensuring our confidence in simulation results.

4.2. Historical trends and temporal variability

As illustrated in Fig. 2, prior to the mid-1980s, the SOC stock (especially in the top 20 cm depth) showed a rapid decline; afterward, the trend became gentle and leveled off until 2007. This trend could be attributed to the following facts. First, as indicated by the CTIC statistics, prior to the mid-1980s, the tillage methods were dominated by conventional tillage, and then by conservational tillage (such as no-till); and reduced-tillage expanded over time. Second, an extension of the internal tile drainage system installation not only facilitated dissolved organic C draining out from the soil body, but also enhanced the decomposition of soil organic matter from the soil layers above the tile system due to the improved soil aeration (for microbial activities). While various tillage practices have little effect on soil C budgets in the soils deeper than 40 cm even a significant difference could be made to the top soil layer (Blanco-Canqui and Lal, 2008; Poirier et al., 2009). However, the internal drainage-induced SOC loss could be offset to some extent by increased biomass input due to new crop varieties, elevated fertilization rate, and improved internal drainage conditions (Kanwar et al., 1988; Wright and Sands, 2001; Drury et al., 2009). Third, there were the effects of antecedent SOC levels that soils with higher C contents tended to lose more C following land surface disturbances (Tan et al., 2006a,b) and climate warming (Lark et al., 2006). In other words, the loss rate becomes smaller with a decrease in the antecedent SOC level. In fact, the change in SOC stock could vary either increasingly or decreasingly. Fig. 5 illustrates that the magnitude and direction of the annual SOC change depended significantly upon the levels of antecedent SOC stocks. Clearly, the annual SOC stock change (or decline) rate was much greater in the period from 1972 to 1985 (0.93%) than in the period from 1985 to 2007 (0.33%). Because (1) the baseline SOC stocks were greater in the former period than in the latter period due to continuous SOC losses and (2) more intensive implementation of conservation tillage and residue management was in the former period than in the latter period. The average change rate between 1972 and 2007 was about -190 ± 0.38 kg C ha^{-1} yr^{-1}. The annual change rate during the period from 1972 to 2007 was identified as a function of the baseline SOC stock:

$$y = -0.0096x + 0.4122, \quad R^2 = 0.5443 \tag{5}$$

where y is the annual change rate of SOC stock (Mg C ha^{-1} yr^{-1}) and x refers to the baseline SOC stock (Mg C ha^{-1}) in 1972.

Generally the soils with higher C stocks (>50 Mg C ha^{-1}) tended to lose more SOC.

4.3. Spatial variability and changes in SOC stock

Fig. 4 demonstrates the spatially explicit association of the magnitude of SOC stock change with the antecedent SOC magnitudes. Soils with higher antecedent C contents show greater losses while soils with lower antecedent SOC contents demonstrate smaller losses during the conventional tillage-dominated period. Such general relationships for two different periods were illustrated in Fig. 5. The relationships between the change rate and baseline SOC contents have been reported for other regions (Bellamy et al., 2005; Tan et al., 2006a, 2007). By integrating dynamic change information of land use and land management practices (e.g., no-till, reduced-tillage, and conventional tillage) and historical records of climate variables during the period from 1970 to 2000 into a biogeochemical model (i.e., GEMS) for the northwest Great Plains of the USA, Tan et al. (2007) concluded that the SOC loss from cultivated croplands with high SOC contents is unavoidable even though conservation management practices could, to some extent, slow down C emission from such soils. This conclusion is similar to that made by Tan et al. (2006a) which used meta-analysis and empirical modeling for the east central USA (including the Eastern Corn Belt) where tillage management was proven to play a critical role in SOC stock budget in croplands. The conversion from conventional tillage to no-till could reduce SOC emission by 16.8% across the Eastern Corn Belt. Bellamy et al. (2005) analyzed the data from the National Soil Inventory of England and Wales obtained between 1978 and 2003. They reported that mean annual loss rates of SOC from soils across England and Wales over the survey period ranged from 0.6% to 2.0% and found that the relative rate of SOC reduction increased linearly with antecedent SOC content levels. They also believed that SOC reduction over time and its relationship to antecedent SOC level was attributed to climate change and had no relationship to land use type. However, the soil C loss they observed in England and Wales could not be totally attributed to climate change (Smith et al., 2007). Our results show that although annual SOC changes in Iowa had a declining trend over the period from 1972 to 2007, the change rate varied with land use type and also varied temporally in cropland; those changes matched with the implementation of conservation tillage history.

5. Summary

The improvement of drainage conditions over croplands enhanced SOC emissions from soils above the internal drainage system owning to high baseline SOC contents even though conservation tillage and residue management implementation could, to some extent, mitigate such an emission rate. Consequently, Iowa soils had been a C source until 2007.

Annual change in cropping systems, such as crop rotation, was another major force driving C fluxes and resulted in spatial variability of annual SOC budgets in all croplands.

Annual change rate in SOC stock in all kinds of land depended significantly on the antecedent SOC levels, and soils with higher C contents tended to be C sources, while soils having lower C contents became C sinks; this should be a critical consideration for managers and policy-makers when setting up C sequestration and C trading programs.

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References

Baker, J.M., Ochsner, T.E., Venterea, R.T., Griffi, T.J., 2007. Tillage and soil carbon sequestration: what do we really know? Agric. Ecosyst. Environ. 118, 1–5.

Bellamy, P.H., Loveland, P.J., Bradley, R.I., Lark, R.M., Kirk, G.J.D., 2005. Carbon losses from all soils across England and Wales 1978–2003. Nature 437, 245–248.

Blanco-Canqui, H., Lal, R., 2008. No-tillage and soil-profile carbon sequestration: an on-Farm assessment. Soil Sci. Soc. Am. J. 72, 693–701.

Brenner, J., Paustian, K., Bluhm, G., Cipra, J., Easter, M., Elliott, E.T., Kautza, T., Killian, K., Schuler, J., Williams, S., 2001. Quantifying the Change in Greenhouse Gas Emissions Due to Natural Resource Conservation Practice Application in Iowa. Final Report to the Iowa Conservation Partnership. Colorado State University Natural Resource Ecology Laboratory and USDA Natural Resources Conservation Service, Fort Collins, Colorado, pp. 1–96.

CTIC (Conservation Technology Information Center), 2008. National Crop Residue Management Survey Data (CD).

Drury, C.F., Tan, C.S., Reynolds, W.D., Welacky, T.W., Oloya, T.O., Gaynor, J.D., 2009. Managing tile drainage, subirrigation, and nitrogen fertilization to enhance crop yields and reduce nitrate loss. J. Environ. Qual. 38, 1193–1204.

Evrendilek, F., Wali, M.K., 2004. Changing global climate: historical carbon and nitrogen budgets and projected responses of Ohio's cropland ecosystems. Ecosystems 7, 381–392.

Harden, J.W., Sharpe, J.M., Ojima, D., Parton, W., Fries, T., Huntington, T., Dabney, S., 1999. Dynamic replacement and loss of soil carbon on eroding cropland. Global Biogeochem. Cy. 13, 885–903.

Kanwar, R.S., Baker, J.L., Baker, D.G., 1988. Tillage and split N-fertilization effects on subsurface drainage water quality and crop yields. Trans. ASAE 31, 453–461.

Kern, J.S., Johnson, M.G., 1993. Conservation tillage impacts on national soil and atmospheric carbon levels. Soil Sci. Soc. Am. J. 57, 200–210.

Kimball, J.S., Running, S.W., Saatchi, S.S., 1999. Sensitivity of boreal forest regional water flux and net primary production simulations to sub-grid-scale land cover complexity. J. Geophys. Res. D: Atmos. 104, 27789–27801.

Lal, R., 2004. Soil carbon sequestration impacts on global climate change and food security. Science 304, 1623–1627.

Lark, R.M., Bellamy, P.H., Kirk, G.J.D., 2006. Baseline values and change in the soil, and implications for monitoring. Eur. J. Soil Sci. 57, 916–921.

Liu, S., 2009. Quantifying the spatial details of carbon sequestration potential and performance. In: McPherson, B., Sundquist, E. (Eds.), Carbon Sequestration and its Role in the Global Carbon Cycle. American Geophysical Union, Washington, DC, pp. 117–128.

Liu, S., Bliss, N., Sundquist, E., Huntington, T.G., 2003. Modeling carbon dynamics in vegetation and soil under the impact of soil erosion and deposition. Global Biogeochem. Cy. 17, 41–43.

Liu, S., Kairé, M., Wood, E., Diallo, O., Tieszen, L.L., 2004a. Impacts of land use and climate change on carbon dynamics in south-central Senegal. J. Arid Environ. 59, 583–604.

Liu, S., Loveland, T.R., Kurtz, R.M., 2004b. Contemporary carbon dynamics in terrestrial ecosystems in the Southeastern Plains of the United States. Environ. Manage., 33.

Nicolardot, B., Molina, J.A.E., Allard, M.R., 1994. C and N fluxes between pools of soil organic matter: model calibration with long-term incubation data. Soil Biol. Biochem. 26, 235–243.

Omernik, J.M., 1987. Ecoregions of the conterminous United States. Ann. Assoc. Am. Geogrs. 77, 118–125.

Parton, W.J., Schimel, D.S., Cole, C.V., Ojima, D.S., 1987. Analysis of factors controlling soil organic matter levels in Great Plains grasslands. Soil Sci. Soc. Am. J. 51, 1173–1179.

Parton, W.J., Schimel, D.S., Ojima, D.S., Cole, C.V., 1994. A general model for soil organic matter dynamics: sensitivity to litter chemistry, texture and management. In: Bryant, R.B., Arnold, R.W. (Eds.), Quantitative Modeling of Soil Farming Processes. SSSA Special Publication 39. ASA, CSSA, and SSA, Madison, WI, USA, pp. 147–167.

Paustian, K., Collins, H.P., Paul, E.A., 1997. Management controls on soil carbon. In: Paul, E.A., Paustian, K., Elliott, E.T., Cole, C.V. (Eds.), Soil Organic Matter in Temperate Agroecosystems: Long-term Experiments in North America. CRC Press, Boca Raton, FL, USA, pp. 15–46.

Poirier, V., Angers, D.A., Rochette, P., Chantigny, M.H., Ziadi, N., Tremblay, G., Fortin, J., 2009. Interactive effects of tillage and mineral fertilization on soil carbon profiles. Soil Sci. Soc. Am. J. 73, 255–261.

Randall, G.W., Huggins, D.R., Russelle, M.P., Fuchs, D.J., Nelson, W.W., Anderson, J.L., 1997. Nitrate losses through subsurface tile drainage in conservation reserve program, alfalfa, and row crop systems. J. Environ. Qual. 26, 1240–1247.

Regina, K., Nykänen, H., Silvola, J., Martikainen, P.J., 1996. Fluxes of nitrous oxide from boreal peatlands as affected by peatland type, water table level and nitrification capacity of the peat. Biogeochemistry 35, 401–418.

Reiners, W.A., Liu, S., Gerow, K.G., Keller, M., Schimel, D.S., 2002. Historical and future land use effects on N_2O and NO emissions using an ensemble modeling approach: Costa Rica's Caribbean lowlands as an example. Global Biogeochem. Cy. 16, 11–16.

Roose, E.J., Lal, R., Feller, C., Barthes, B., Stewart, B.A., 2006. Soil Erosion and Carbon Dynamics. CRC, Boca Raton, FL, USA.

SAS Institute Inc., 2004. SAS OnlineDoc® 9.1.2. SAS Institute Inc., Cary, NC, USA.

Sharpley, A.N., Williams, J.R., 1990. EPIC—Erosion/Productivity Impact Calculator. USDA, Agricultural Research Service, Washington, DC, USA, Technical Bulletin No. 1768.

Singh, R., Helmers, M.J., Crumpton, W.G., Lemke, D.W., 2007. Predicting effects of drainage water management in Iowa's subsurface drained landscapes. Agric. Water Manage. 92, 162–170.

Smith, P., Chapman, S.J., Scott, W.A., Black, H.I.J., Wattenbach, M., Milne, R., Campbell, C.D., Lilly, A., Ostle, N., Levy, P.E., Lumsdon, D.G., Millard, P., Towers, W., Zaehle, S., Smith, J.U., 2007. Climate change cannot be entirely responsible for soil carbon loss observed in England and Wales, 1978–2003. Global Change Biol. 13, 2605–2609.

Tan, Z., Lal, R., Liu, S., 2006a. Using experimental and geospatial data to estimate regional carbon sequestration potential under no-till management. Soil Sci. 171, 950–959.

Tan, Z., Lal, R., Smeck, N.E., Calhoun, F.G., 2004. Relationships between surface soil organic carbon pool and site variables. Geoderma 121, 187–195.

Tan, Z., Liu, S., Johnston, C.A., Liu, J., Tieszen, L.L., 2006b. Analysis of ecosystem controls on soil carbon source-sink relationships in the northwest Great Plains. Global Biogeochem. Cy. 20, GB4012, doi:10.1029/2005GB002610.

Tan, Z., Liu, S., Johnston, C.A., Loveland, T.R., Tieszen, L.L., Liu, J., Kurtz, R., 2005. Soil organic carbon dynamics as related to land use history in the northwestern Great Plains. Global Biogeochem. Cy. 19, 1–10.

Tan, Z., Liu, S., Li, Z., Loveland, T.R., 2007. Simulated responses of soil organic carbon stock to tillage management scenarios in the Northwest Great Plains. Carbon Balance. Manage. 2, 7, doi:10.1186/1750-0680-2-7.

Tan, Z., Liu, S., Tieszen, L.L., Tachie-Obeng, E., 2009a. Simulated dynamics of carbon stocks driven by changes in land use, management and climate in a tropical moist ecosystem of Ghana. Agric. Ecosyst. Environ. 130, 171–176.

Tan, Z., Tieszen, L.L., Liu, S., Tachie-Obeng, E., 2010. Modeling to evaluate responses of ecosystem carbon budgets and crop yields to climate change and nitrogen fertilization in a Sahel cultivated savanna. Climatic Change 100, 703–715.

Tan, Z., Tieszen, L.L., Tachie-Obeng, E., Liu, S., Dieye, A.M., 2009b. Historical and simulated ecosystem carbon dynamics in Ghana: land use, management, and climate. Biogeosciences 6, 45–58.

Updegraff, K., Pastor, J., Bridgeham, S.D., Johnston, C.A., 1995. Environmental and substrate controls over carbon and nitrogen mineralization in northern wetlands. Ecol. Appl. 5, 151–163.

USDA, 1994. State Soil Geographic (STATSGO) Database: Data Use Information. U.S. Dept. of Agriculture, Natural Resources Conservation Service, National Soil Survey Center, Washington, DC.

West, T.O., Post, W.M., 2002. Soil organic carbon sequestration by tillage and crop rotation: a global data analysis. Soil Sci. Soc. Am. J. 66, 1930–1946.

Wright, J., Sands, G., 2001. Planning an Agricultural Subsurface Drainage System. Pub. No. BU-07685. Minnesota Ext. Ser., University of Minnesota, St. Paul, MN, USA.

Zhao, S., Liu, S., Li, Z., Sohl, T.L., 2010. Federal land management, carbon sequestration, and climate change in the Southeastern U.S.: a case study with Fort Benning. Environ. Sci. Technol. 44, 992–997.