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Running head: Chinese cropping systems are a net C source

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Abstract

Soil carbon sequestration is being considered as a potential pathway to mitigate climate change. Cropland soils could provide a sink for carbon that can be modified by farming practices, however, they can also act as a source of greenhouse gases (GHG), including not only nitrous oxide (N_2O) and methane (CH_4), but also the upstream carbon dioxide (CO_2) emissions associated with agronomic management. These latter emissions are also sometimes termed "hidden" or "embedded" CO_2. In this paper, we estimated the net GHG balance for Chinese cropping systems by considering the balance of soil carbon sequestration, N_2O and CH_4 emissions, and the upstream CO_2 emissions of agronomic management from a life cycle perspective during 2000–2017. Results showed that although soil organic carbon (SOC) increased by 23.2±8.6 Tg C yr^{-1}, the soil N_2O and CH_4 emissions plus upstream CO_2 emissions arising from agronomic management added 269.5±21.1 Tg C-eq yr^{-1} to the atmosphere. These findings demonstrate that Chinese cropping systems are a net source of GHG emissions, and that total GHG emissions are about 12 times larger than carbon uptake by soil sequestration. There were large variations between different cropping systems in the net GHG balance ranging from 328 to 7567 kg C-eq ha^{-1} yr^{-1}, but all systems act as a net GHG source to the atmosphere. The main sources of total GHG emissions are nitrogen fertilization (emissions during production and application), power use for irrigation, and soil N_2O and CH_4 emissions. Optimizing agronomic management practices, especially fertilization, irrigation, plastic mulching, and crop residues to reduce total GHG emissions from the whole chain is urgently required in order to develop a low carbon future for Chinese crop production.

Keywords: Agronomic management; Upstream CO_2 emissions; Life cycle analysis; Net greenhouse gas balance; N_2O and CH_4 emission; Soil organic carbon.

Introduction

Soil is a large reservoir of carbon (C) in terrestrial ecosystems with a pool size of around 1500 Pg C (1 Pg = 10^{15} g) (Davidson et al., 1998; Lal, 2004). Cropland soil accounts for 8–10% of this C pool (Eswaran et al., 1993), which plays a significant role in the global C budget (Mahecha et al., 2010). There is an estimated technical potential to sequester 1.6 Pg C equivalents (C-eq) yr^{-1} into agricultural soils globally (Smith et al., 2007; 2008). Hence, an initiative that aims to increase global agricultural SOC stocks by 0.4% (four per thousand) was launched (http://4p1000.org), in order to slow down rising levels of atmospheric CO_2 (Minasny et al., 2017), though doubts have been expressed as to whether this rate of increase is generally achievable (van Groenigen et al., 2017; Poulton et al, 2018).

The SOC stock changes represent the net exchange of CO_2 between soil and atmosphere (Mosier et al., 2006; Robertson and Grace, 2004; Shang et al., 2011). Many agricultural practices, such as optimized fertilization, reduced tillage, and straw return to the fields have been advocated to mitigate GHG emissions by enhancing removals of CO_2 from the atmosphere (Smith et al., 2008; Snyder et al., 2009). Many field experimental studies have suggested that fertilizer application and straw return can increase soil C and sequester C from the atmosphere (Mosier et al., 2006; Shang et al., 2011). However, these practices may also stimulate nitrous oxide (N_2O) emissions that offset the SOC sequestration benefits (Pathak et al., 2005; Huang et al., 2013). Further, methane (CH_4) emissions can be increased after adding organic materials, especially in rice grown under flooded conditions (Zou et al., 2005; Shang et al., 2011). In addition to SOC sequestration, N_2O and CH_4 emissions from cropland soils are two additional crucial components of the GHG balance because of their high global warming potential (Robertson et al., 2000; Cubasch et al., 2013). The manufacture and transport of fertilizers and pesticides, power use for irrigation and field operations all require fossil fuels; the combustion of which results in GHG emissions (Snyder et al., 2009; Grassini and Cassman, 2012). Fertilization, irrigation, tillage and other management practices in the different cropping systems will affect the upstream C-eq, which is defined as C-eq released from such agricultural inputs (Robertson and Grace, 2004; Lal, 2007; Snyder et al., 2009; Schlesinger, 2010). The climate benefits of SOC sequestration in croplands might be offset by N_2O and CH_4 emissions (Tian et al., 2012; Norse and Ju, 2015), and upstream CO_2 released from the life cycle of agricultural inputs (Lal, 2007; Snyder et al., 2009; Schlesinger, 2010). The C lifecycle approach considers the full C cycle as it includes upstream C-eq release, but often does not consider soil GHG emissions and SOC sequestration (West and Marland, 2002; Robertson and Grace, 2004; Mosier et al., 2006). Elevated SOC storage in croplands will only mitigate climate change if the combined GHG emissions from agronomic practices are lower than the SOC sequestration from a life cycle perspective (Powlson et al., 2011). The climate benefits of a given cropping system should not only focus on the sequestration of SOC, but also on soil GHG emissions and the associated C-eq released from agricultural inputs and management practices (Mosier et al., 2006; Lal, 2007; Schlesinger, 2010). To measure the overall climate effect, the concept of net a GHG balance (kg C-eq ha^{-1} yr^{-1}) has been proposed, based on the cumulative radiative forcing from all GHGs considered together (Robertson and Grace, 2004; Powlson et al., 2011). Greenhouse gas intensity (GHGI, kg C-eq kg^{-1} grain) is used to compare the magnitude of GHG emissions to produce the same crop yield (Mosier et al., 2006; Grassini and Cassman, 2012). This concept can assist in solving the global challenges of increasing food production and concomitantly identifying the main targets for mitigation in different cropping systems and regions, which is important when seeking ways to decrease total GHG emissions associated with agricultural production, especially in China.

China covers a broad range of soil-climatic regimes, and corresponding cropping systems. The net GHG balance of different cropping systems will vary with soil-climatic conditions, crops and management practices. Large numbers of studies have investigated soil carbon sequestration, N_2O and CH_4 emissions in different cropping systems in China (Table S1). However, few have reported the net GHG balance associated with all sinks and sources in Chinese cropping systems (Huang et al., 2013; Gao et al., 2015). Furthermore, large uncertainties exist in these previous estimates, e.g. the GHG balance ranges over one order of magnitude due to uncertainties in upstream CO_2 emissions of agronomic management practices (Zhang and Zhang, 2016). There is an urgent need, therefore, to synthesize literature for calculating the net GHG balance associated with soil N_2O and CH_4 emissions, the upstream CO_2 emissions from agronomic management and the change of SOC storage in Chinese cropping systems.

The aim of this paper is to obtain a national estimate of the net GHG balance in the main Chinese cropping systems. A database was compiled from relevant research published between 2000–2017, totaling 634 results for SOC changes and 233 results for N_2O and CH_4 emissions. Then, a meta-analysis was performed to explore the net GHG balance of each cropping system. We focused on five key questions in our analysis: (*i*) was there a change in topsoil (0–20 cm) SOC storage? (*ii*) what were the total GHG emissions associated with soil N_2O and CH_4 emissions and upstream CO_2 emissions from agronomic management? (*iii*) what were the main sources of total GHG emissions? *(iv)* what was the net GHG balance? and (*v*) what are the most effective measures to improve the net GHG balance of Chinese crop production?

Materials and Methods

Description of Chinese cropping systems

Rice (*Oryza sativa*), wheat (*Triticum aestivum*), maize (*Zea mays L.*), potato (*Solanum tuberosum*), soybean (*Glycine max*), rapeseed (*Brassica rapa*), cotton (*Gossypium spp*), and vegetables (*Herbas*) are the main crops in China, accounting for 18.2%, 14.5%, 22.9%, 3.3%, 3.9%, 4.5%, 2.3% and 13.2% of the national total crop area sown, respectively (NBSC, 2016). Vegetables mainly included leafy vegetables, cabbages, fruit vegetables, melons, root and stem vegetables, etc. In 2015, China had 12.8 Mha orchard (*Hortus*), including apple, citrus, pear, peach and grape, etc., equivalent to 53.1% of the sown area of wheat (24.1 Mha) (NBSC, 2016). China has multiple crops each year, e.g. winter wheat–summer maize double cropping system, i.e. two harvests in one year, so we provide summary data on a yearly basis. Double cropping is different to annual rotations such as maize–soybean rotations that have a single crop in each year in places such as the US and elsewhere (Mosier et al., 2006). The main principle for classification of cropping systems in this study is based on different agricultural zones and the management practices used by farmers, because soil type, climate and fertilization practices of a given cropping system are similar within an agricultural zone. We collected data for greenhouse vegetables, open-field vegetables, potato and orchard systems across China, because these crops are widely distributed in all agricultural zones and there is insufficient available data on soil GHG emissions and SOC change for these systems within a single agricultural zone This allows 15 distinct cropping systems to be defined (Table 1). We evaluated the mean GHG emissions and SOC change of a given cropping system in its dominant agricultural zone. The proportions of these cropping systems in relation to total crop sowing area are given in Appendix S1. The spatial patterns of the different cropping systems at a county-level (Fig. 1) were developed from a 30 × 30 m resolution land use map at a county-scale for China in 2010 provided by Wu et al. (2014). The specific combinations and distributions of cropping systems at county-level are shown in Appendix S2.

Table 1 Abbreviations for different Chinese cropping systems in this paper.

Abbreviation	Cropping systems
WMN	winter wheat – summer maize double-cropping in Northern China
WMSW	winter wheat – summer maize double-cropping in Southwestern China
RW	rice – winter wheat double-cropping in Central and Eastern China
RR	rice – rapeseed double-cropping in Central and Southwestern China
DR	double rice per year in Central and Southern China
SRNE	single rice in Northeastern China
SRENE	single rice across China except for Northeastern China
SB	soybeans in Northeastern China
MNE	single spring maize per year in Northeastern China
MNW	single spring maize in Northern and Northwestern China
PS	potato system across China
CS	cotton system in Northern and Northwestern China
GV	greenhouse vegetables across China
OV	open field vegetables across China
OS	orchard system across China

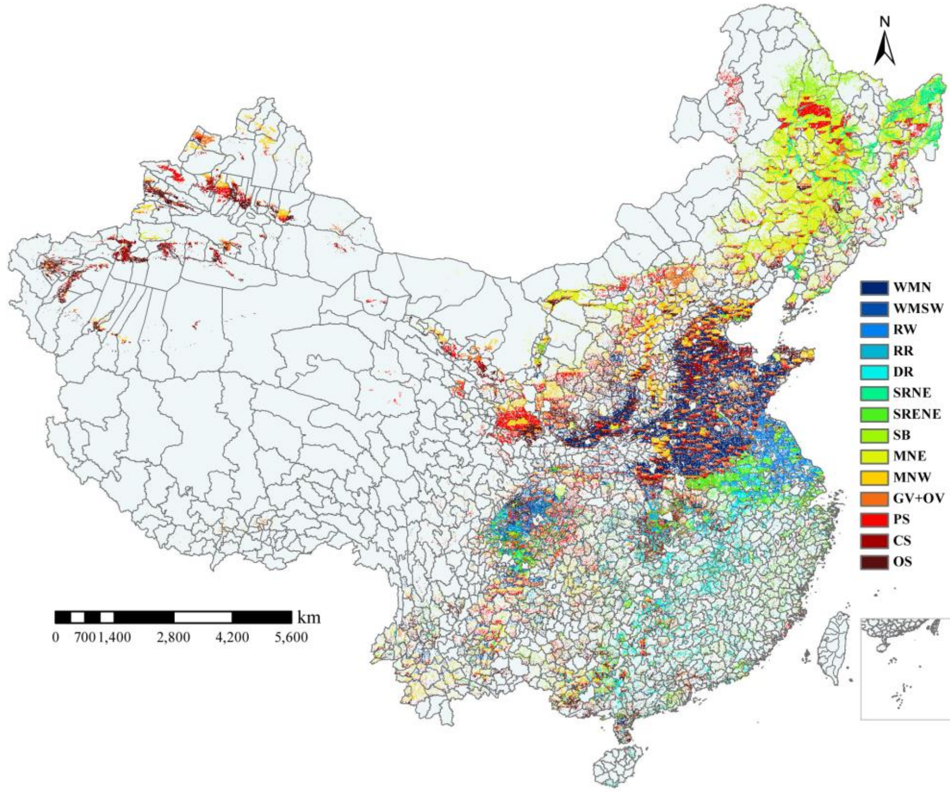

Fig. 1. Spatial pattern of cropping systems in China. Abbreviations are shown in Table 1, GV+OV.represents total vegetable production. These abbreviations are also used in subsequent figures.

Data sources

Basal data used in this study, such as crop sowing area and yields, were taken from official statistical yearbooks and bulletins (NBSC, 2016). The data used for calculating the net GHG balance of the cropping systems, including the change of topsoil SOC, soil N_2O and CH_4 emissions, the upstream CO_2 from the manufacture and transportation of the chemical fertilizer (N, P_2O_5 and K_2O), power used for irrigation, fuel combustion in farm operations, application of pesticides, and film used for mulching crops or as a cover for greenhouse or fruits (Robertson et al., 2000; Mosier et al., 2006; Grassini and Cassman, 2012), were collected from the published literature, dissertations or books with data records for the period between 2000–2017.

The SOC data were categorized into three groups: (I) SOC data with monitoring of < 5 years; (II) SOC data with monitoring of ≥ 5 years; (III) data describing the mean changes of SOC for entire Chinese croplands. As most studies reported SOC change over at least a 5 year period in a given cropping system

(Mosier et al., 2006; Lemke and Janzen, 2007; Huang et al., 2013); we mainly considered the effect of the duration of the experiment on SOC changes for this group classification. The detailed criteria for collecting SOC data are described in Appendix S3. In total, 647 results were collected, which were divided between each of the cropping systems as follows; WMN (41), WMSW(13), RW (47), RR (32), DR (100), SRNE (28), SRENE(46), SB (21), MNE (63), MNW (34), GV (42), OV (25), PS (30), CS (49), OS (76) and entire Chinese croplands (CC 14). We also obtained information for 582 locations of the collected SOC data under categories I and II (Fig. 2).

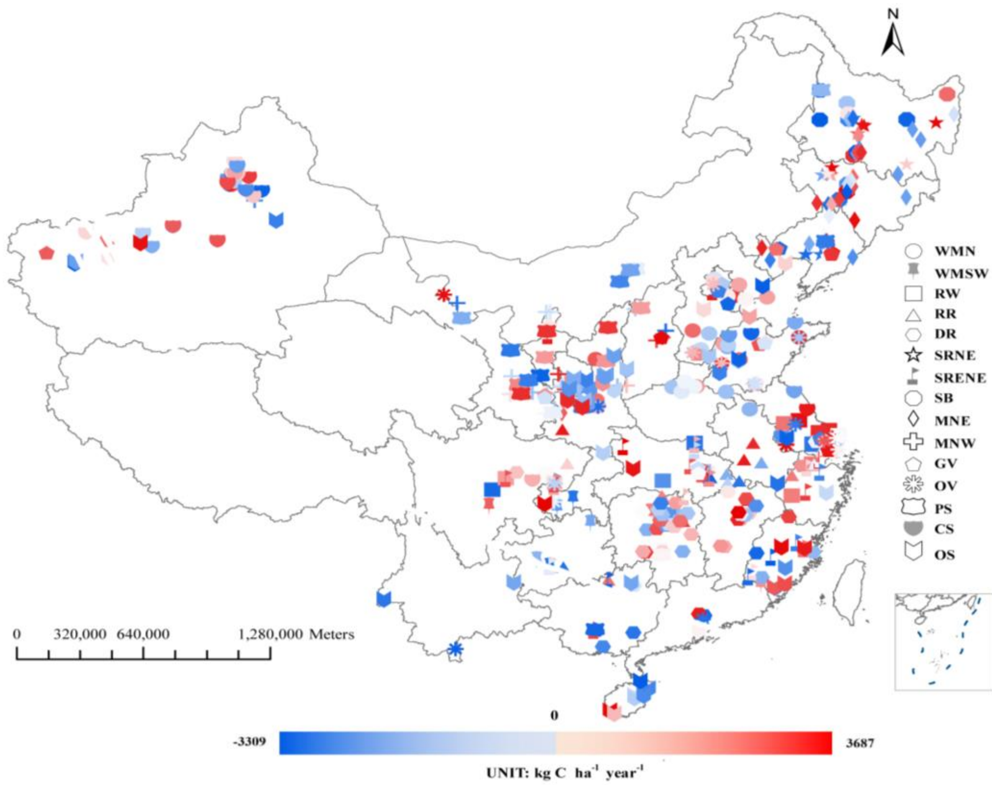

Fig. 2. The locations and mean changes of SOC density in Chinese cropping systems from 2000 to 2017.

The criteria for collecting soil N_2O and CH_4 emissions, fertilizer input, the power used for irrigation, pesticides, fuel, and plastic film are described in Appendix S3. Results from 233 studies reporting N_2O and CH_4 emissions were collected from all cropping systems (Table S1). The detailed criteria for collecting the consumption of power use for irrigation, fuel and plastic film are described in Appendix S4. The adopted carbon emission factors are presented in Table 2.

Table 2 Emission factors of soil GHG emission and agriculture inputs

Emission source	Abbreviation	Unit	Emission factor (kg C-eq unit^{-1})	Literature
N_2O	CE_{N2O}	kg	81.3	Cubasch et al., 2013
CH_4	CE_{CH4}	kg	9.3	Cubasch et al., 2013
	CE_N	kg	2.3	Zhang et al., 2013
Fertilizer	CE_P	kg (P_2O_5)	0.4	Huang et al., 2013
	CE_K	kg (K_2O)	0.3	Huang et al., 2013
Power for irrigation	CF_I	kwh	0.4	Zhang et al., 2013
Fuel in farm operations	CF_F	L	0.7	Cheng et al., 2011
Pesticides	CF_{PE}	kg	4.9	West and Marland, 2002
Plastic film	CF_{PF}	kg	5.2	Cheng et al., 2011
Paper bags	CF_{PB}	kg	0.3	Yan et al., 2016

Calculation of C-eq

An annual topsoil SOC sequestration rate (δSOC, kg C ha^{-1} yr^{-1}) was estimated on the basis of an increased rate of change of the topsoil SOC density (dSOC/dt, g C kg^{-1} yr^{-1}) [Eqn (1)] (Robertson et al., 2000; Shang et al., 2011).

$$\delta SOC = dSOC/dt \times \gamma \times 20/10 \quad (1)$$

where γ is the bulk density (g cm^{-3}) of 0–20 cm depth topsoil, which was directly reported in the soil physical-chemical properties together with SOC concentrations in most of the collected literature. For the few sites with no reported bulk density, we firstly used bulk density of the same cropping system at a nearby site. If this was not available we assumed the mean bulk density values of 1.3 and 1.2 g cm^{-3} reported for upland and rice paddy fields in China, respectively (Pan et al., 2010). Since pedo-transfer functions also introduce uncertainty. The use of mean values for final gap filling was considered adequate for the objectives of our study, so no pedo-transfer functions were used to estimate missing bulk density values. The values of 20 and 10 in the equation (1) are the topsoil depth and the area conversion coefficient, respectively. Most of the studies in Chinese cropping systems define the 0–20 cm soil depth as the plough layer under long-term conventional tillage practices, and the soil samples were taken to a depth of 20 cm to determine the SOC content, so we adopt a soil depth of 0–20 cm for calculating the SOC stock change.

Values of C-eq from soil N_2O and CH_4 emissions were estimated by multiplying the annual emissions in different cropping systems with the global warming potential (GWP) values over a 100-yr time horizon, which are 34 kg CO_2-eq kg^{-1} or 9.3 kg C-eq kg^{-1} for CH_4 and 298 kg CO_2-eq kg^{-1} or 81.3 kg C-eq kg^{-1} for N_2O (Cubasch et al., 2013). The C-eq emissions from certain agricultural inputs, i.e. the applied fertilizers, pesticides, plastic film and paper bags (kg ha^{-1} yr^{-1}), power use for irrigation (kwh ha^{-1} yr^{-1}), and fossil fuel use in farm operations (L ha^{-1} yr^{-1}) were estimated by agricultural input, multiplying with the individual carbon intensity (in kg C per unit volume or mass) of manufacture and transportation of synthetic fertilizers, and/or applied for individual agricultural inputs (Table 2).

Calculation of net GHG balance and GHGI

We defined the boundary of the soil-crop system as the carbon gains and emissions per hectare per year between the soil and atmosphere, and calculated the main carbon fluxes from crop sowing to harvest using a life cycle approach (Robertson et al., 2000; Mosier et al., 2006; Smith et al., 2010). The "hidden" or "embedded" CO_2 from upstream production of fertilizers etc. or farmers' operations are important components to be included in these comparison. The GHG emissions from non-agricultural IPCC sectors (e.g. energy) were included only where they are used specifically for agricultural use.

All the main fates of GHGs including upstream CO_2 emission, soil GHG emissions, CO_2 fixed by crops (photosynthesis) and emitted by crops (respiration) and soil SOC change, have to be considered when assessing a system's capacity to act as a GHG sink or source. But the actual calculations of net GHG flux depend on the characteristics and nature of different ecosystems, in which only the actual carbon fluxes are included. For example, in forest systems, CO_2 fixed by plant photosynthesis must be included in addition to the change of soil SOC, because the aboveground biomass of forest is cumulative (Tang et al., 2018). However, for cropland ecosystems, the carbon in crops is not included, since these crops are harvested and consumed within a year, so there is no carbon sink; the carbon is simply recycled to the atmosphere within a year, so (very temporary) carbon stocks in crops should not be included in the GHG balance (Smith et al., 2010). The SOC change is the net balance between carbon inputs and outputs of the returned crop residues and soil respiration, and it also represents the net exchange of CO_2 between soil and atmosphere (Mosier et al., 2006; Smith et al., 2010).

Organic fertilizers (manure) accounted for 14.5% of the total N fertilization in China in 2010 (Gu et al., 2015). Our study set the boundary as Chinese cropping systems, which includes SOC stock changes and N_2O and CH_4 emissions induced by manure being applied to cropping systems, but does not include N_2O and CH_4 during storage, treatments (e.g. compost) and transportation of manure, which are regarded as emissions from animal production (Smith et al., 2010). To evaluate a net GHG balance and yield basis of GHG emissions (GHGI), we used the following equation [Eqn (2) and (3)], which has widely been used in the calculations of net GHG balance and GHGI of different cropping systems (Robertson and Grace, 2004; Mosier et al., 2006; Shang et al., 2011, Grassini and Cassman, 2012; Gao et al., 2015; Zhou et al., 2017). A positive net GHG balance represents a source of C-eq to atmosphere, while a negative value represents a sink of C-eq from the atmosphere (Robertson et al., 2000; Mosier et al., 2006).

$$\text{Net GHG balance (kg C-eq ha}^{-1}\text{ yr}^{-1}) = a\times CE_{N2O} + b\times CE_{CH4} + c\times CE_{N} + d\times CE_{P} + e\times CE_{K} + f\times CE_{I} + g\times CE_{F} + h\times CE_{PE} + i\times CE_{PF} + j\times CE_{PB} - \delta SOC \quad (2)$$

$$\text{GHGI (kg C-eq Mg}^{-1}) = \text{Net GHG balance/Yield} \quad (3)$$

where the different small letters represent the amounts of soil GHG emissions and different agricultural inputs. CE_{N2O}, CE_{CH4}, CE_N, CE_P, CE_K, CE_I, CE_F, CE_{PE}, CE_{PF} and CE_{PB} represent the individual C emission equivalents for soil N_2O and CH_4 emissions, inputs of synthetic N, P_2O_5 and K_2O fertilizers, power use for irrigation, fuel, pesticides, plastic films and paper bags used for crop production, respectively (Table 2). 12 and 44 are the molecular weights of C and CO_2. δSOC is the change of cropland SOC.

Variations at county scale

We converted each source and sink of the net GHG to an annual basis based on the compiled the datasets from relevant publications between 2000 and 2017. We then estimated the spatial patterns of topsoil SOC stock, and GHG emissions (kg C-eq ha^{-1}) from N_2O and CH_4 emissions, N fertilizer input, power use for irrigation and other sources including P_2O_5 and K_2O application, fuel in farm operations, pesticides and plastic film use and total GHG emissions (kg C-eq ha^{-1}) in Chinese cropping systems from the above sources (Fig. S1), and net GHG balance based on the available 30 m × 30 m land use map and spatial pattern of cropping systems at county-scale for China in 2010, which represents the mean pattern of the study period. The weighted mean SOC stock change and net GHG balance between greenhouse vegetables (342 kg C ha^{-1} yr^{-1}, 7567 kg C-eq ha^{-1} yr^{-1}) and open-field vegetables (296 kg C ha^{-1} yr^{-1}, 4617 kg C-eq ha^{-1} yr^{-1}) was calculated based on the average SOC stock change and net GHG balance, and the area of the two types vegetables and total vegetable production, because of lack of the proportions of two types of vegetable production at county scale.

Statistical and uncertainty analysis

The significance of the differences in SOC stock change of different cropping systems were tested with an analysis of variance (ANOVA) using the SPSS16.0 statistical package. Statistical significance was determined at the 95% confidence level at $p < 0.05$. In addition, to minimize the uncertainty of our analysis, we first set up uniform criteria for collecting topsoil SOC change, and other emission sources from different cropping systems. Then the mean and variation of these data were calculated with the 90th percentile confidence interval. The uncertainty of the net GHG balance was analyzed using the error propagation equation of mathematical statistics (IPCC, 2001) by the uncertainties of the collected data on SOC change, soil GHG emissions and upstream CO_2 emissions from agronomic managements. A detailed description of the error of propagation equation from a mathematical statistical analysis is shown in Appendix S5. The same error propagation equation was used for calculating the uncertainties for total SOC stock change and total GHG emissions.

Results

Change of SOC in Chinese cropping systems

The frequency distribution of the annual SOC change showed an overall increase of the SOC stocks in the topsoil of Chinese cropping systems (Fig. 3) and the SOC changes were significantly ($p < 0.05$) different among the 15 cropping systems (Table S4). Overall, 78.1% of the observations showed an increase in SOC stocks, with decreases in SOC mainly occurring in MNE, OS, CS, SB and PS systems (Table S2). The annual change values of SOC stock were in the range -3,309 to 3,687 kg C ha^{-1} yr^{-1}, and followed a normal distribution, with the minimum values occurring in the MNE system in the Northeast and maximum values in the DR system in Central China.

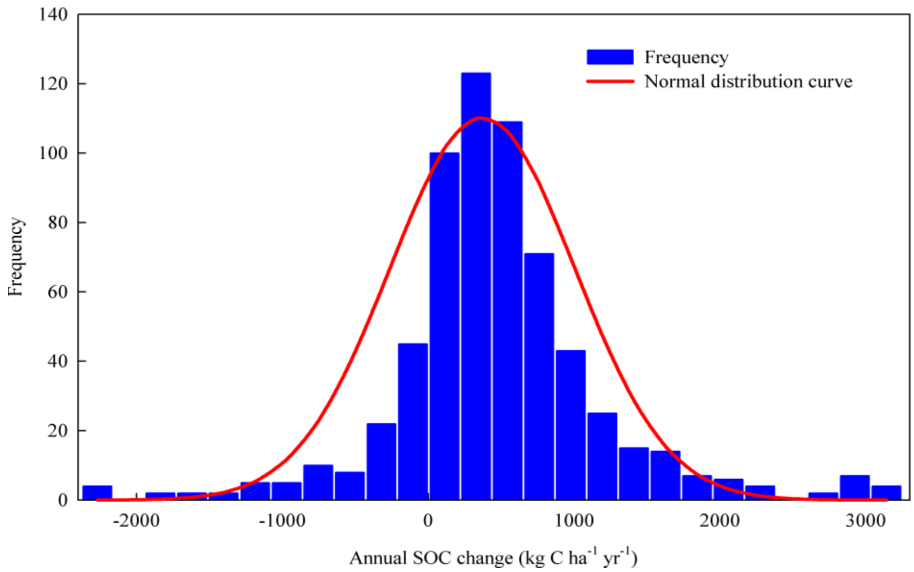

Fig. 3. The frequency distribution of annual SOC changes in Chinese cropping systems.

All cropping systems except SB showed an increase in SOC stock during the 2000-2017 period. The magnitudes were 170–825 and 50–432 kg C ha^{-1} yr^{-1} in groups I and II, respectively (Fig. 4). In group III, the mean change of SOC across all Chinese croplands was 178 ± 27 kg C ha^{-1} yr^{-1} with a range of 27–538 kg C ha^{-1} yr^{-1}. The standard deviations of the observations in group I were much larger than those in groups II and III (Fig. 4), highlighting greater uncertainties of SOC stock changes when the study duration is less than 5 years. Therefore, the changes of SOC from method II were used to estimate the net GHG balance of Chinese cropping systems.

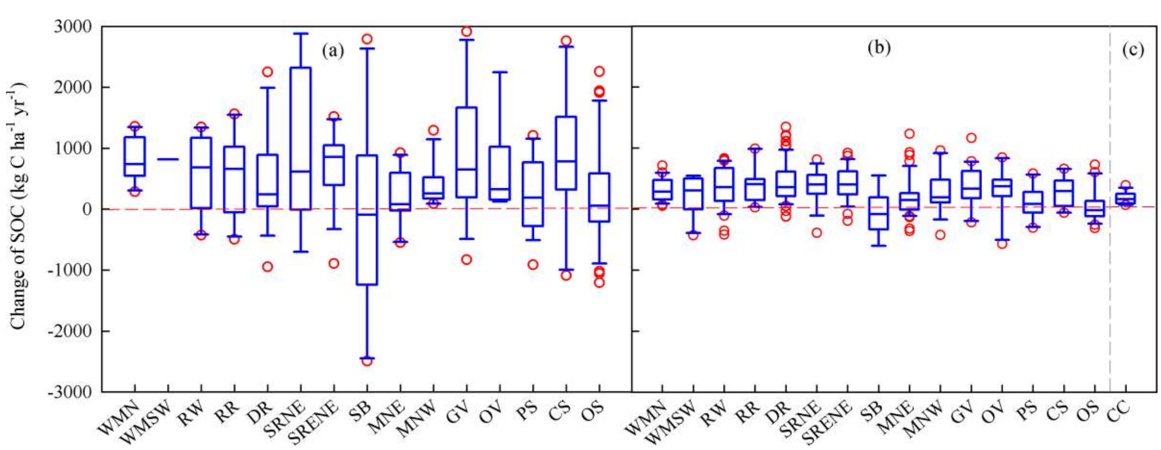

Fig. 4. Changes of SOC in Chinese cropping systems. CC represents Chinese total croplands. Different letters in parentheses represent the changes of SOC under methods I, II and III, respectively. Box-and-whisker diagrams show the median, 5th, 25th, 75th and 95th percentiles for relative change in SOC stocks.

The distribution of SOC stock changes in different cropping systems at county-scale is shown in Fig. 5. The regions with the highest SOC increase have a high proportion of paddy fields (Fig. 1), located in the Middle and Lower Yangtze River, Sichuan Basin, eastern Heilongjiang province, as well as some scattered regions in Southern and Eastern China. The regions with the second-highest SOC increase were dominated by winter wheat, summer maize, cotton, and vegetable production (Fig. 1). These regions are mainly concentrated on the North China Plain, north of Sichuan Basin, southern Shanxi, Shaanxi and Gansu provinces and northwest Xinjiang. The regions with relatively low SOC increases were dominated by single spring maize, potato and orchard planting, located in Northeastern China, southern Inner Mongolia, northern Shanxi and Shaanxi provinces, Sichuan Basin and Southwestern China. A decrease of SOC storage occurred in northeastern Inner Mongolia and western Heilongjiang due to the cultivation of soybean in soils with an initially high SOC stock (Fig. 1).

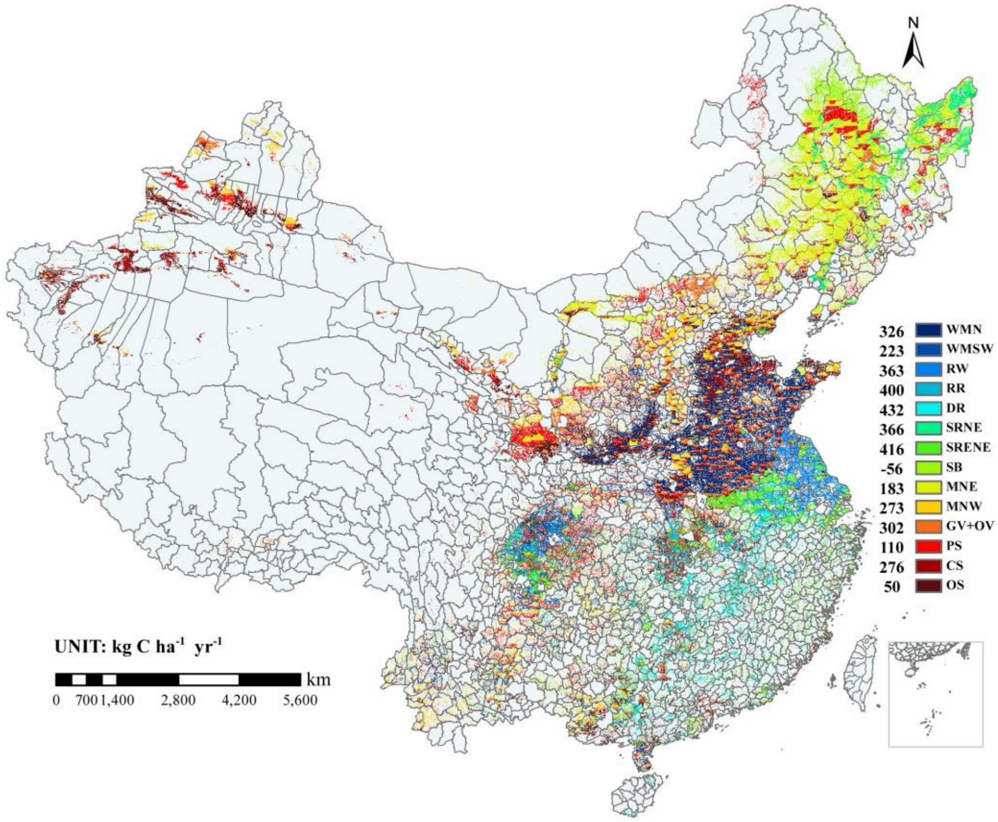

Fig. 5. Spatial patterns of the SOC stock change in different cropping systems.

Net GHG balance, their main sources and GHGI

All cropping systems acted as a net GHG source when considering the combined soil N_2O and CH_4 emissions and upstream CO_2 calculated from the life cycle emissions of agricultural inputs. The net GHG balance was in the range of 328–7567 kg C-eq ha^{-1} yr^{-1}, with a rank of SB < MNE < PS < WMSW ≈ CS < MNW < OS ≈ WMN < SRNE < SRENE < RR < RW < OV < DR < GV (Table 3). The spatial pattern of the net GHG balance at county-level showed that the regions with the highest net GHG emissions were mainly found in North China due to vegetable planting with very high fertilizer N applications, over-irrigation, and plastic used for greenhouse covering (Fig. 6). The regions with the second-highest net GHG balance were concentrated in Central, Southern and Eastern China, because of high CH_4 emissions from cultivation of rice in those regions. The regions with the lowest net GHG balance were mainly located in Northeastern China, central and southern Inner Mongolia, Gansu and Southwestern China in SB, MNE and PS systems (Fig. 1).

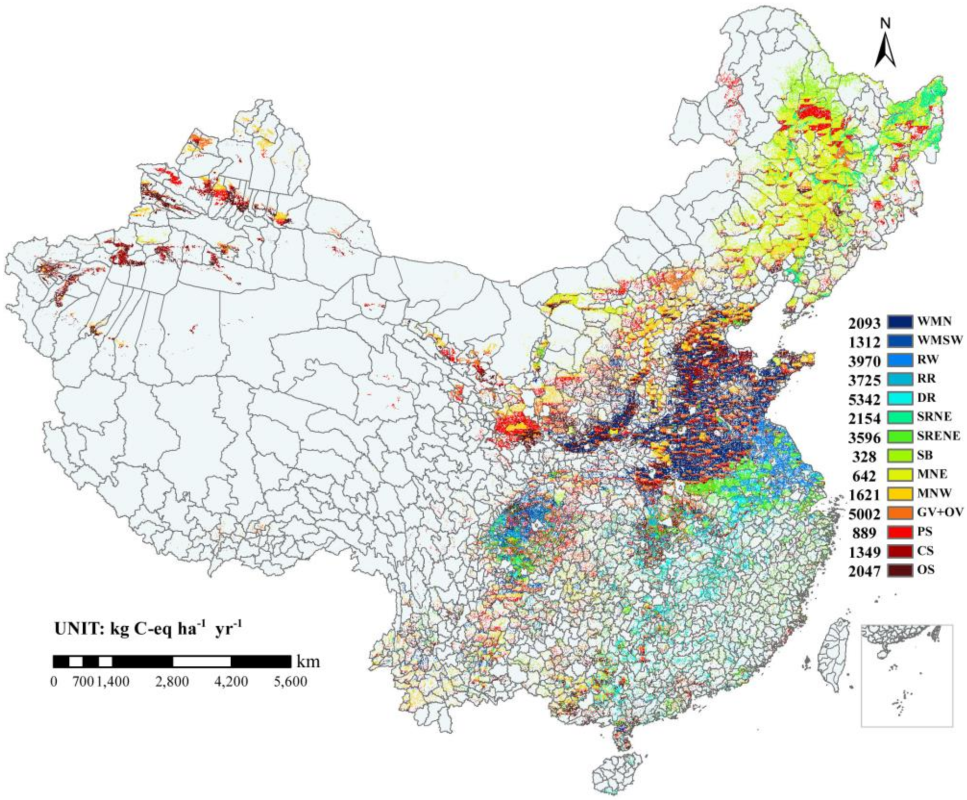

Fig. 6. Spatial pattern of net GHG balance in different cropping systems.

Table 3. The total GHG balance and sources of GHGs in different Chinese cropping systems.

Cropping systems	N_2O	CH_4	N	P_2O_5	K_2O	Irrigation	Fuel	Pesticide	Plastic film	SOC change	Net GHG balance[*]	Yield[#]	GHGI[*]	Plant area[†]	Total SOC change[‡]	Total GHGs emission[¶]	Total GHG emission/total SOC change[§]
	kg C-eq ha^{-1} yr^{-1}									kg C ha^{-1} yr^{-1}	kg C-eq ha^{-1} yr^{-1}	t ha^{-1}	kg C-eq Mg^{-1}	$\times 10^4$ ha	Tg C yr^{-1}	Tg C-eq yr^{-1}	
WMN	439±33	-17±3	1148±45	59±6	35±5	606±38	121±12	35±5	-	326±34	2093±294	14.3±0.4	146±28	1371	4.5±2.5	33.2±5.0	7.4±4.3
WMSW	496±122	-34±59	758±32	65±4	32±3	0±0	136±47	20±5	-	223±101	1312±362	9.8±0.8	134±49	205	0.5±0.7	3.1±0.8	6.9±10.1
RW	780±81	1564±23	978±29	49±4	39±4	758±107	121±27	44±10	-	363±60	3970±856	13.3±0.6	299±82	475	1.7±1.6	20.6±4.7	11.9±11.3
RR	712±81	1561±254	758±45	63±5	40±5	824±106	102±39	66±15	-	400±72	3725±776	10.0±0.4	289±71	532	2.1±1.6	22.0±4.9	10.3±7.9
DR	190±33	3935±397	616±36	47±3	52±4	780±56	103±19	51±10	-	432±38	5342±1566	12.9±0.3	535±162	564	2.4±1.9	32.6±10.2	13.4±11.0
SRNE	98±16	1091±228	333±32	18±5	19±3	855±105	67±5	15±5	-	366±74	2154±713	8.5±0.3	253±93	431	1.6±1.3	10.9±4.0	6.9±6.1
SRENE	366±81	2300±262	453±57	43±6	22±6	731±4	74±16	25±5	-	416±45	3596±817	7.7±0.2	466±122	463	1.9±1.2	18.6±4.6	9.7±6.8
SB	86±33	-8±8	68±9	26±4	6±1	0±0	76±11	10±5	-	-56±120	328±371	2.6±0.1	129±246	356	-0.2±1.3	1.0±0.7	/
MNE	211±33	-8±3	457±27	26±4	17±2	30±14	67±13	20±5	2±2	183±47	642±352	9.8±0.5	65±27	1637	3.0±5.2	13.5±3.1	4.5±7.4
MNW	382±154	-25±8	638±59	29±4	9±3	351±103	75±13	25±5	393±108	273±80	1621±824	10.3±0.7	155±70	90	0.2±0.3	1.7±0.7	6.9±9.0
GV	2064±260	-17±8	2214±226	149±24	112±15	1178±140	125±39	370±44	1719±182	342±73	7567±1738	14.5±1.2	523±222	108	0.4±0.4	8.5±2.1	23.1±24.2
OV	1699±236	17±17	1754±161	71±18	47±11	776±147	21±21	104±10	409±32	296±118	4617±1308	11.7±1.7	394±204	719	2.1±2.9	35.3±10.3	16.5±23.5
PS	106±24	-8±2	407±20	48±3	40±3	129±24	24±9	59±10	182±44	110±85	889±391	5.3±0.3	170±77	875	1.0±2.3	8.9±2.4	9.2±22.3
CS	263±60	-15±15	559±68	52±11	9±3	462±66	25±11	89±10	178±42	276±50	1349±472	3.4±0.4	399±250	485	1.4±1.2	7.9±2.3	6.0±5.6
OS	319±114	-23±5	1060±106	118±24	84±17	355±109	55±19	101±39	29±10	50±51	2047±802	3.2±0.3	648±371	1237	0.6±3.3	25.9±9.9	41.6±223.5
Mean/Total										243±92[††]					23.2±8.6	269.5±21.1	11.6±4.4

[*] Net GHG balance calculated by equation 2, values are means and uncertainty ranges.

[#] Yields of vegetable and fruit = Fresh yields in the literature × 0.1.

† Plant areas cited and calculated from China Agriculture Yearbook (MOA, 2014) and Department of Agricultural Machinery Management (DAMM, 2017).

‡ Total SOC change (Tg C yr^{-1}) = SOC change (kg C ha^{-1} yr^{-1}) × Plant area (× 10^4 ha)/10^9.

¶ Total GHGs emissions (Tg C-eq yr^{-1}) = C-eq from N_2O + CH_4 + N + P_2O_5 + K_2O + Irrigation + Fuel + Pesticide + Plastic (kg C-eq ha^{-1}) × Plant area (× 10^4 ha)/10^9.

§ Times of total GHG emissions to total SOC change = Total GHG emissions/Total SOC change.

†† The weighted mean increase rate of SOC for Chinese main cropping systems.

Except for the SB system, manufactured and transported fertilizer N was the largest source of C-eq emissions in upland systems (Fig. 7), accounting for between 29.3–71.8% of the net C-eq emissions. N fertilization induced large emissions of N_2O, accounting for 3.6–37.7% of net C-eq emissions. The total C-eq emissions from power use for irrigation ranged from 129 to 1,178 kg C ha^{-1} yr^{-1}, accounting for 14.2–34.2% of the net C-eq emissions except for WMSW, SB and MNE systems. Plastic films are mainly used to cover the ground surface to promote crop germination by increasing soil surface temperature when sowing in relatively low temperature periods, and to increase water use efficiency by reducing soil surface evaporation loss, or for use to cover greenhouses. Plastic films are also an important emissions source, accounting for up to 22.7–24.5% of the net C-eq emissions in MNW, GV and PS systems. In paddy fields, CH_4 emissions were the largest contributor to emissions, accounting for 39.4%, 41.9%, 73.7%, 50.8% and 64.0% of the net C-eq emissions in RW, RR, DR, SRNE and SRENE systems, respectively, followed by emissions from irrigation (14.6–40.7%) and fertilizer N (11.5–24.6%). Nitrous oxide was an important source of emissions in RW and RR systems, accounting for 19.7% and 19.1% of the net C-eq emissions, respectively. Methane uptake from WMN, SB, MNE, MNW, GV, OV, PS, CS and OS contributed a small amount of negative emissions (< 1%). Emissions from the application of P_2O_5 and K_2O, fuel in farm operations and pesticides application contributed to 0.8–5.8%, 0.6–4.4%, 0.4–10.3% and 0.7–6.7% of net C-eq emissions, respectively, in all cropping systems except the SB system.

Changes of the SOC sink accounted for only 2.5–28.7% of the total C-eq emissions of the cropland systems (Fig. 7). The decrease in SOC stocks in the northeast SB system acted as a source of CO_2, accounting for 17.0% of the total C-eq emissions in this system.

The magnitude of GHG emissions to produce the same crop yield were in the range of 65–648 kg C-eq Mg^{-1}, with a rank of MNE < SB < WMSW < WMN < MNE < PS < SRNE < RR < RW < OV ≈ CS < SRENE < GV < DR < OS (Table 3).

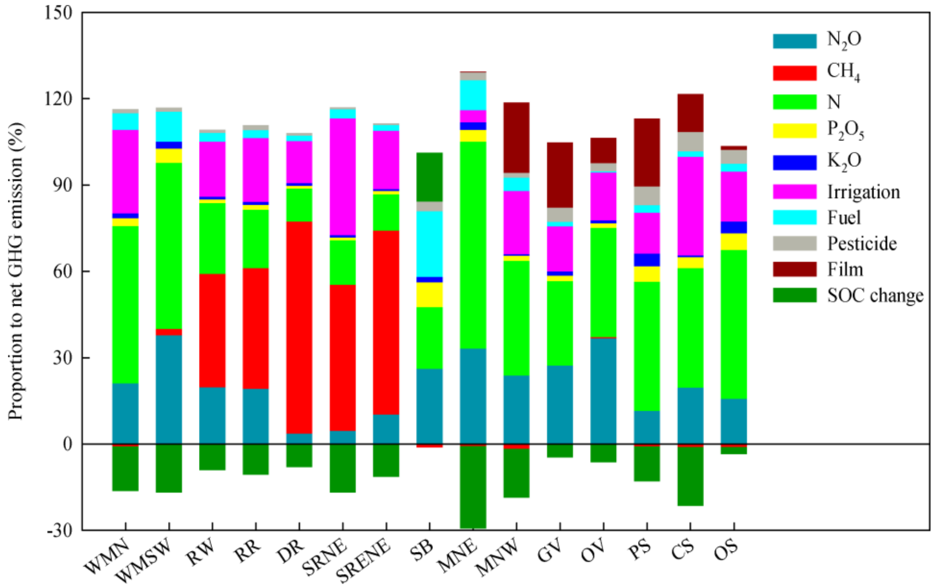

Fig. 7. Sources and allocation of greenhouse gas emissions in different cropping systems in China.

Soil C sequestration *vs* total GHG emission

We estimated the total topsoil SOC increase and total GHG emissions of Chinese cropping systems by multiplying the SOC change and the total C-eq emissions of each cropping system with their sowing area (Table 3). Total topsoil SOC changed at rates of between -0.2–4.5 Tg (1 Tg = 10^{12} g) C yr^{-1}, resulting in an accumulation of 23.2 ± 8.6 Tg C yr^{-1} across the whole China, close to the calculated mean increase in the topsoil C stock of China's croplands of 25.5 Tg C yr^{-1} between 1985 and 2006 (Pan et al., 2010). The GHG emissions from different cropping systems was in the range of 1.0–35.3 Tg C-eq yr^{-1}, with a rank of SB < MNW < WMSW < CS < GV < PS < SRNE < MNE < SRENE < RW < RR < OS < DR ≈ WMN < OV, summing to 269.5 ± 21.1 Tg C-eq yr^{-1}, accounting for 13.9–15.2% of total national GHG emissions in 2005–2007 (Yan and Yang, 2010; National Development & Reform Commission of China, 2014). We further calculated the ratio of total GHG emissions from different cropping systems to total soil carbon sequestration. These ratios were 7.4 (WMN), 6.9 (WMSW), 11.9 (RW), 10.3 (RR), 13.4 (DR), 6.9 (SRNE), 9.7 (SRENE), 4.5 (MNE), 6.9 (MNW), 23.1 (GV), 16.5 (OV), 9.2 (PS), 6.0 (CS), and 41.6 (OS), respectively, and about 12 for all Chinese croplands (Table 3).

Discussion

Changes in SOC in Chinese cropping systems

In order to achieve the objectives of the Paris Climate Change Agreement, to keep global temperature increases well below 2 °C, it is widely recognized that negative emission technologies will be needed to lower atmospheric concentrations of CO_2 (Rockstorm et al., 2017). Soil carbon sequestration by cropland soils could play a potential role in many regions (Robertson and Grace, 2004; Smith et al., 2008; Powlson et al., 2011; Wollenberg et al., 2016). In this paper, we indeed find that cropland SOC increased in Chinese cropping systems, with a range of 50 to 432 kg C ha^{-1} yr^{-1}, close to a suggested global mean rate of 300 to 500 kg C ha^{-1} yr^{-1} (Lal, 2007). The increase of cropland SOC in China in the past forty years can be mainly attributed to increased crop yields resulting from improvements in agronomic management in Chinese (i.e. new crop varieties, fertilizer inputs, improved protection from pests, diseases and weeds, irrigation), and increased yields leading to larger organic matter returns to soil from roots and stubble (Huang and Sun, 2006; Yan et al., 2011; Han et al., 2017; Zhao et al., 2018; Powlson et al., 2018). We found that the average yields per unit area showed a good correlation ($p < 0.05$) with the SOC increase of the different cropping systems (Fig. S2). The increase in SOC has also been attributed to the development of no-tillage and reduced-tillage practices in China (Huang and Sun, 2006; Yan et al., 2011). The initially low average SOC content (11.5–12.0 g kg^{-1} for 0–20 cm depth) between 1979 and 1982 is another contributory factor for the observed increase in SOC across China (Yan et al., 2011; Yang et al., 2017). Even though the average SOC content in soil had increased to 12.7–14.3 g kg^{-1} in 2005–2014, it is still lower than in many European and US cropland soils (Johnston et al., 2009; Yan et al., 2011; Fan et al., 2012; Yang et al., 2017; Zhao et al., 2018). The lower starting point was mainly caused by soil mining, together with low crop residue returns to the soil due to lower yields, in turn caused by low inputs and poor agronomic management before policy changes in 1978 (Huang and Sun, 2006; Fan et al., 2012; Zhao et al., 2018).

The weighted mean rate of increase in SOC was 243 ± 92 kg C ha^{-1} yr^{-1} for the main Chinese cropping systems (Table 3), higher than the mean annual SOC increase of 178 ± 27 kg C ha^{-1} yr^{-1}, which was 27 to 538 kg C ha^{-1} yr^{-1} derived from reviewing previous studies on total cropland area in China (Table S2). This difference might be explained by the following factors. Firstly, in addition to the main crops (rice, maize, wheat, and vegetables), Chinese croplands also produce other crops such as millet, sorghum, peanut, tobacco, hemp crops, etc. which might lead to smaller SOC increases compared to the OS and PS systems across China, or cause a decrease as in the SB system in Northeastern China, because the new carbon input from crop growth in these systems is lower than in the main cropping systems for production of rice, maize, wheat and vegetables; the quantity of crop residues is a key factor in determining changes in SOC stocks (Yan et al., 2011; Yang et al., 2017). Secondly, long-term field experiments on SOC change are typically established on the main crops, including wheat, rice, maize, and vegetables, which have higher yields compared to those of soybean, millet, sorghum, peanut, tobacco, hemp crops, etc., which may lead to an overestimation of the SOC increase for some relatively low yielding crops. Thirdly, study durations of the SOC data used in this paper are mainly between 5–15 years, shorter than previous long-term studies on total cropland of China such as Yan et al. (2011), in which the study duration was around 30 years. The annual increase rates of SOC in cropland are usually larger in the initial years following a change in management, declining with the duration of study, and reaching an equilibrium after around 20–50 years (Mosier et al., 2006; Lemke and Janzen, 2007; Wang et al., 2017). The SOC change might also be underestimated a little since changes of SOC stock consider only the top 0–20 cm soil layer, and changes of SOC stock below 20 cm might also contribute to soil carbon sequestration (Yan et al., 2011; Zhao et al., 2018), but any such underestimation should not greatly affect our main results and conclusions.

There is still a potential to increase SOC stocks in Chinese croplands. Compared to the high SOC concentrations of 14.5–23.2 g kg^{-1} or SOC stocks (0–20 cm) (40.2–43.7 t C ha^{-1}) in Europe and the US (Fan et al., 2012; Zhao et al., 2018), over half of the topsoil in China's croplands have SOC concentrations lower than 11.6 g kg^{-1} (Yang et al., 2017), and the estimated 0–20 cm SOC stocks were 26.6–29.4 t C ha^{-1} in 1979–1982 and 31.4–33.5 t C ha^{-1} in 2007–2011 (Yan et al., 2011; Zhao et al., 2018). Topsoil SOC pools can increase rapidly in the early years after a change of management and then more slowly thereafter, as observed for reduced- and no-tillage (West and Post, 2002; Lal, 2007; Johnston et al., 2009). Conservation tillage practices have only begun recently in China, which means there should be potential to increase SOC in Chinese croplands in the coming years (Yang et al., 2017); however, recent evidence suggests that conservation tillage may largely redistribute carbon within the profile, though net carbon gains are often still observed (Powlson et al., 2014). Changes in SOC stock can be influenced by the availability of nutrients (van Groenigen et al., 2017) and it is possible that more appropriate management of nutrients in Chinese croplands (probably decreased N applications but increased P, K, S or micronutrients) could lead to greater increases in SOC.

Soil N_2O and CH_4 emissions and upstream CO_2 emission

Although topsoil SOC stocks increased in the main cropping systems in China, its net effect on GHG balance was more than offset by large emissions of soil N_2O and CH_4 and upstream CO_2 emissions from agronomic management (Fig. 6). This emphasizes that climate change mitigation strategies cannot rely only on SOC sequestration in croplands, and more effort is required for reducing total GHG emissions from cropland management (Fig. S1). Indeed, the SOC increase is smaller than the emissions of soil N_2O and CH_4, even without considering upstream CO_2 emissions from cropland management practices.

Emissions from the manufacture and transportation of N fertilizer are the largest contributor to total GHG emissions in China (Fig. 7). Emissions are about 2–31 times greater than that of the major cropping systems in the US, because of the higher N application rate and higher CO_2 emissions associated with the manufacture and transportation of N fertilizer in China, which is 8.3 vs 3.0 kg CO_2 kg^{-1} N applied in China *vs* US, respectively (Mosier et al., 2006; Zhang et al., 2013). In China, the cropland SOC sink is mainly caused by N fertilizer applications (Tian et al., 2012), but this also results in large N_2O emissions (Bouwman et al., 2002; Synder et al., 2009; Gao et al., 2015).

Pumping of groundwater for irrigation is one of the most energy consuming on-farm processes and it represents an important source of GHG emissions that has rapidly increased, and which at present is largely unregulated (Wang et al., 2012). The high emissions from irrigation are mainly due to a combination of excessive irrigation, low energy use efficiency for pumping, and high power generation emissions. In China, power demand for pumping per unit of water is 4.3 kwh mm^{-1} ha^{-1}, which falls into the range of 2.1–6.4 kwh mm^{-1} ha^{-1} estimated by Wang et al. (2012), compared to only 0.15 kwh mm^{-1} ha^{-1} in the US. Further, CO_2 emissions from electricity generation in China are 1.32 kg CO_2 kwh^{-1}, much higher than the 0.32 kg CO_2 kwh^{-1} in the US (Mosier et al., 2006; Zhang et al., 2013). This part of the emissions budget will increase with the decline of groundwater table and the increase in air temperature in China if no improvements in water conservation are made (Foster and Garduño, 2004; Liu et al., 2010; Powlson et al., 2018).

Plastic film usage has a large contribution to total GHG emissions in MNW, GV, OV, PS, and CS systems, even exceeding the contribution of N_2O emissions in MNW and PS systems. In the GV system, pesticides contribute to large emissions of about 3.0–33.3 times that of the same source in other cropping systems. More attention needs to be paid to this source, given that the planting area of greenhouse vegetables has been increasing by around 10% per year (Fan et al., 2014).

Except for the OV system, we found that upland soils act as a weak sink for CH_4, but it can be neglected given that it represents <1% of the net GHG balance (Cui et al., 2013; Gao et al., 2015). However, CH_4 emissions are an important contributor to total GHG emissions in Chinese rice–based rotations (Ma et al., 2013; Shang et al., 2011). CH_4 emissions from paddy rice cultivation accounts for 20.3% of total GHG emissions in Chinese agriculture (National Development & Reform Commission of China, 2014).

With population and economic growth, Chinese grain demand is expected to increase by 6.9%, 3.3% and 52.9% for rice, wheat and maize by 2030, respectively, relative to 2012 (Chen et al., 2014). Due to the limitation on arable land expansion in China (Burney et al., 2010; Cui et al., 2013), producing more food might occur at the expense of increasing nutrient inputs if no other improvements in agronomic management are made. This presents the challenge of producing more grains with fewer inputs, and with reduced environmental and climate impacts.

We showed the weighted mean SOC stock change and net GHG balance for total vegetable production (GV+OV) at county scale, because there is no data on the proportion of the two types of vegetables at this scale. This represents the mean status of SOC change and net GHG emissions of Chinese vegetables production. However, the SOC change and net GHG balance of greenhouse vegetables were about 1.2 and 1.6 times greater than open-field vegetables. Greenhouse vegetables are, mainly distributed around Bohai and Huang-Huai-Hai region, the middle and low reaches of Yangtze river, and Northwest China, and these regions accounted for 60.3%, 19.7% and 7.5% of the total sown area of greenhouse vegetables in 2010, respectively (DAMM, 2017). As a result, this study might have underestimated the SOC change and net GHG emissions of vegetable production in counties with a high proportion of greenhouse vegetables in the three main greenhouse vegetable producing areas mentioned above, and might have overestimated the SOC change and net GHG emissions of vegetable production in counties with low proportions of greenhouse vegetables. The impacts of the production on SOC change and net GHG balance of greenhouse vegetables at a county scale requires further study.

GHGI of different cropping systems

GHGI provides a platform for comparing the overall effects of any given cropping system on GHG emissions per unit of crop yield (Mosier et al., 2006; Grassini and Cassman, 2012). The magnitude of GHG emissions per unit of grain yield ranged from 65 to 648 kg C-eq Mg^{-1} in different Chinese cropping systems. This is significantly higher than that of 32–61 kg C-eq Mg^{-1} in conventional irrigated maize systems in Northeastern Colorado and Nebraska of the U.S., which was -35 kg C-eq Mg^{-1} in conventional no-till corn–soybean rotation in Northeastern Colorado of the U.S (Mosier et al., 2006; Grassini and Cassman, 2012), and 75 kg C-eq Mg^{-1} in conventional wheat and double-cropped soybean in mid-Atlantic region of the U.S. (Cavigelli et al., 2009), In China, the net GHG balance of rice production systems (DR, SRNE and SRENE) was 2154–5342 kg C-eq ha^{-1} yr^{-1}, close to that of 1718–5342 kg C-eq ha^{-1} yr^{-1} in Japan, USA and Italy. The high net GHG balance of rice production in different countries resulted from high baseline of

CH_4 emissions (Hokazono and Hayashi, 2012). However, the GHGI of Chinese rice production (253–535 kg C-eq Mg^{-1}) is lower relative to GHGI for rice in Japan, USA and Italy (398–753 kg C-eq Mg^{-1}), because rice yield reach 7.7–12.9 Mg ha^{-1} in China (Table 3), but only 4.4–7.3 Mg ha^{-1} in Japan, USA and Italy (Hokazono and Hayashi, 2012). The high GHGI of upland cropping systems in China was caused by over-use of fertilizer and over-irrigation, but with low yields of maize, wheat and soybean relative to the U.S. (Mosier et al., 2006; Cavigelli et al., 2009; Grassini and Cassman, 2012). Many studies have indicated that China has a large potential to produce more grain with lower GHG emissions by optimizing fertilization and irrigation (Ju et al., 2009; Cui et al., 2013; Chen et al., 2014), and this could close the gap of GHGI for grain production between China and low GHGI countries.

Reducing total cropland GHG emissions

The ratio of total GHG emissions to soil C sequestration was reported in this study, which expresses the ability of soil C sequestration to offset the total GHG emissions of the soil-crop system. This showed that total GHG emissions are about 12 times larger than soil carbon sequestration, indicating that non-CO_2 GHG emissions need to be reduced substantially if Chinese crop production is to become GHG neutral with the help of soil C sequestration. Major increases in yield have been achieved in Chinese cropping systems over the time period for which soil carbon changes were assessed, but the yield increase came at the cost of higher N_2O emissions. By evaluating the overall GHG balance, the environmental cost of these yield increases can be seen, so that improved agronomic management practices can be identified to reduce these impacts. This study shows that the main measures to reduce the net GHG balance of Chinese cropping systems should focus on reducing N_2O emissions, chemical N fertilizer use, GHG emissions from fertilizer manufacture, power use for irrigation, and CH_4 emissions in rice-based cropping systems, while at the same time, increasing soil C sequestration. Moreover, emissions from the use of plastic film and pesticides should also be considered in vegetable cropping systems.

Emissions of N_2O are normally positively correlated with N fertilizer inputs, and with N surpluses (difference between total N input and crop N uptake) in cropping systems (van Groenigen et al., 2010; Grassini and Cassman, 2012; Cui et al., 2013; Gao et al., 2015). Appropriate N fertilization not only reduces N_2O emissions, but also reduces emissions associated with manufacture and distribution of N fertilizer (Mosier et al., 2006; Huang and Tang, 2010; Gao et al., 2015). Chinese cropping systems usually receive excessive amounts of N fertilizer, about two to even tens of times greater than the actual crop demand (Ju et al., 2009; Chen et al., 2014; Nayak et al., 2015). It has been clearly demonstrated that N fertilization rates can be reduced by knowledge-based N management practices without compromising crop yields, and in some cases even causing an increase (Ju et al., 2009; Chen et al., 2014; Xia et al., 2016). In addition to decreasing the quantity of N fertilizer applied, altering the timing of N fertilizer application can increase N use efficiency and decrease N_2O emissions (Ju et al., 2009; Cui et al., 2013; Chen et al., 2014). Practices such as deep placement of N fertilizer could also reduce N_2O emissions by 5–40% (Xia et al., 2016). If a 30% reduction in N fertilization rate was achieved, a potential reduction in GHG emissions would reach 16.4 Tg C-eq from production of paddy rice, wheat, maize and soybean (Cheng et al., 2015). Although knowledge-based practices are available in China, there remain socioeconomic barriers, such as small farm size that need to be addressed to facilitate their widespread implementation (Ju et al., 2016).

The reduction of emissions from power use for irrigation should concentrate on reducing energy consumption, which depends on the efficiency of irrigation and power generation (Mosier et al., 2006; Grassini and Cassman, 2012). Measures for reducing irrigation emissions include testing of soil water content (Meng et al., 2012) and development of fertigation systems (Fan et al., 2014). These knowledge-based irrigation practices could reduce irrigation emissions by 16–43% in Chinese cropping systems (Cabangon et al., 2004; Meng et al., 2012; Fan et al., 2014). Energy use, both for the purposes of irrigation and fertilizer manufacture, is currently associated with high GHG emissions as a consequence of its dependence on fossil fuel-based energy sources. Therefore, substitution of these energy sources by low emission renewable alternatives could significantly reduce the carbon footprint of food production (Schandl et al., 2016).

Methane emissions were mainly affected by water regimes and organic amendments in paddy fields (Wassmann et al., 2000; Shang et al., 2011; Nayak et al., 2015). Strategies for reducing CH_4 emissions include mid-season drainage and intermittent irrigation (Wassmann et al., 2000; Zou et al., 2005; Ma et al., 2013; Cheng et al., 2014; Nayak et al., 2015). These measures could reduce CH_4 emissions from paddy fields by 36–65% (Zou et al., 2005; Ma et al., 2013). However, a trade-off between CH_4 and N_2O emissions appeared from mid-season drainage and intermittent irrigation (Wassmann et al., 2000; Zou et al., 2005;

Ma et al., 2013). The increase of N_2O emissions offset the reduction of CH_4 emissions by 49.2% and 67.6% for plots without and with wheat straw amendment in midseason drainage, respectively. Reductions in CH_4 emissions could be completely offset by increased N_2O emissions when the field was moist but not waterlogged by intermittent irrigation, in comparison with the treatment that was frequently waterlogged within the midseason drainage period (Zou et al., 2005). Therefore, Wassmann et al. (2000) proposed that the changes in water regime are only recommended for rice systems with high baseline emissions of CH_4 from waterlogged and midseason drainage to intermittent irrigation.

There is considerable interest in the possibility of mitigating climate change by sequestering extra C from atmosphere into soil through changes in land management (Smith et al., 2008; Powlson et al., 2011, 2018). Effective measures for increasing SOC stocks mainly include the return of crop residues to soils, the application of biochar, conservation tillage, and mulch plants (Synder et al., 2009; Zhao et al., 2014; Qian et al., 2015; Nayak et al., 2015; Zhou et al., 2017; Powlson et al., 2018). However, the direct return of straw to rice paddy fields is not recommended, because CH_4 emissions are increased by a factor of 1.6–3.7 and the net GHG emissions associated CH_4 and N_2O are greatly increased in rice paddy fields when receiving organic amendments (Zou et al., 2005). Converting straw to biochar then applying it to soils is a possible alternative to soil C sequestration, and could contribute to CH_4 mitigation, and improvements of soil and crop productivity, without increasing N_2O emissions (Zhao et al., 2014). However, there are widely divergent opinions in the scientific community about the practicalities and economics of biochar production and use (Powlson et al., 2018). The global mean rate of SOC sequestration for conversion from conventional tillage to no-till is 100–200 kg ha^{-1} yr^{-1} (Lal, 2007). Converting conventional tillage to reduced tillage in rice-based cropping systems in China could sequester 213 kg C ha^{-1} yr^{-1} (Nayak et al., 2015) but, again, the small size of most farms in China presents practical and economic barriers to adoption of reduced tillage. Mulching different living plants could significantly increase SOC sequestration in orchards soil (Qian et al., 2015).

Emissions associated with the manufacture of plastic film use in the GV system could be reduced by prolonging its service life and recycling of the abandoned film (Chen et al., 2011). Further, the high rates of pesticide application in Chinese GV systems could be reduced by controls on the occurrence of disease and insect pests through technologies such as reduced temperature and humidity in greenhouses, and physical controls (i.e. trapping and insect screens), advanced application equipment and drip irrigation (Wang and Wang, 2016) and integrated pest management (Pretty and Bharucha, 2015).

Despite the soil carbon sink found in Chinese cropland soils, emissions of N_2O and CH_4 and upstream CO_2-eq emissions associated with agronomic management are about one order of magnitude larger than the soil carbon sink under current farmers' practices. Chinese croplands are therefore a net GHG source. Over-fertilization with N and low energy use efficiency of irrigation and other agronomic management practices are largely responsible for these high GHG emissions. To feed an increasingly wealthy population, Chinese crop production is expected to continue to expand in the future, posing great challenges for reducing GHG emissions. However, there is still much room for improving the net GHG balance of Chinese croplands. Mitigation measures can focus on, but are not limited to, optimizing fertilizer applications, better irrigation practices and conservation tillage.

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