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Greenhouse gas intensity of three main crops and implications for low-carbon agriculture in China

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Abstract:

China faces significant challenges in reconciling food security goals with the objective of becoming a low-carbon economy. Agriculture accounts for approximately 11% of China's national greenhouse gas (GHG) emissions with cereal production representing a large proportion (about 32%) of agricultural emissions. Minimizing emissions per unit of product is a policy objective and we estimated the GHG intensities (GHGI) of rice, wheat and maize production in China from 1985 to 2010. Results show significant variations of GHGIs among Chinese provinces and regions. Relative to wheat and maize, GHGI of rice production is much higher owing to CH_4 emissions, and is more closely related to yield levels. In general, the south and central has been the most carbon intensive region in rice production while the GHGI of wheat production is highest in north and northwest provinces. The southwest has been characterized by the highest maize GHGI but the lowest rice GHGI. Compared to the baseline scenario, a 2% annual reduction in N inputs, combined with improved water management in rice paddies, will mitigate 17% of total GHG emissions from cereal production in 2020 while sustaining the required yield increase to ensure food security. Better management practices will entail additional gains in soil organic carbon further decreasing GHGI. To realize the full mitigation potential while maximizing agriculture development, the design of appropriate policies should accommodate local conditions.

Key words: food security, low-carbon agriculture, greenhouse gas intensity, China

1. Introduction

China has made substantial efforts to increase crop production to feed about 20% of the global population with only 8% of the world's arable land (World Bank 2013). Looking towards 2020, the government has set a target of increasing the national grain production capacity to over 545 Mt from 497 Mt in 2010 to meet the growing demand for higher animal protein diets and to maintain the domestic food self-sufficiency rate at 95%. This implies that average grain yield must grow by at least 0.9% annually in the period 2011-2020. However, Chinese agriculture is grappling with related constraints in terms of limited arable land, declining water availability, an increasing opportunity cost of rural labour and increasing vulnerability to climate change (Fan et al. 2011). The sector is also a significant source of anthropogenic greenhouse gases (GHG) emissions emitting approximately 820 Mt CO_2 equivalent (CO_2e) in 2005, or 11% of the national total (NCCC 2012). Cropland N_2O emissions produced in soils through the microbial processes of nitrification and denitrification was responsible for 25.3% of agriculture GHG emissions in 2005 and CH_4 emissions from rice cultivation contributed 20%. Cereal production (rice, wheat and maize) accounted for about 47% of national N fertilizer consumption (Heffer 2009) and generated around 32% of GHG emissions from agriculture.

Agriculture is also under increasing scrutiny to mitigate climate change through both emissions reduction and carbon sequestration. The Ministry of Agriculture (MOA) has initiated programs to improve fertilizer use efficiency by 3% and enhance irrigation water use efficiency by 6% by 2015 from 2010. The government also plans to bring an additional 11.3 Mha of croplands under conservation tillage between 2009-2015 in north China (MOA 2009). The aim of integrating mitigation into agriculture translates into a reduction in GHG intensity (GHGI), expressed as the overall GHG emissions per unit of product (Chen et al. 2011; Venterea et al. 2011; Tubiello et al. 2012). Applying this indicator can encourage better practices resulting in higher crop yields and reduced N losses and GHG emissions, which is vital to pursue low carbon development in agriculture (Norse 2012).

FAO (Tubiello et al. 2014) reported that over the period 1961-2010 the world average GHGI of rice decreased by 49% while that of wheat and maize increased by 45%, suggesting that effective mitigation strategies are needed to achieve sustainable intensification; i.e. ensuring that efficiency improvements can lead to reduced absolute emissions. Bonesmo et al. (2012) investigated the GHGI of

95 arable farms in Norway, showing that increased gross margins in grain and oilseed production could be achieved with decreasing GHGI. The GHGIs of cereal production on experimental sites were also quantified in China indicating that economic and climate benefits can be simultaneously achieved by some improved management practices (Shang et al. 2011; Huang et al. 2013; Ma et al. 2013). But to date there is no synthetic estimate of current and historical GHGI of cereal production on a national, regional or provincial level in China. Such information is crucial for identifying efficient regional mitigation strategies and actions tailored to local agricultural production systems and management practices.

This paper estimates GHGI for rice, wheat and maize production using data for the national, regional and provincial scale for 2006. Illustrating the trends and evolution of intensity we quantity national and regional GHGI from 1985 to 2010 at 5-year intervals and analyze emission reduction and carbon sequestration potentials from cereal production. The analysis informs potential national or regional policies to foster sustainable intensification in rural China.

2. Materials and methods

2.1. Methodology

GHGI is calculated by dividing total Global Warming Potential (GWP)-weighted emissions from cereal production by crop yield (Eqn (1)). N_2O emissions are accounted for quantifying GHGI of wheat and maize production while both CH_4 and N_2O are considered for rice paddies. Carbon sequestration is not directly included in the estimate due to large uncertainties in soil organic carbon (SOC) content and limited data availability. Despite consensus on the average SOC increment in China's cropland, discrepancies in annual intensity change rates have been reported using various methods (Pan et al. 2010; Sun et al. 2010; Yan et al. 2011; Yu et al. 2012). Nevertheless, SOC change patterns and interactions with GHGI will be analyzed in the discussion section. The analysis focuses on emissions within the farm gate, i.e. they are not full life-cycle assessment (e.g. emissions related to energy use and fertilizer manufacture and transportation).

We followed IPCC Guidelines (IPCC, 2006) to estimate direct N_2O emissions from the three major N input sources - synthetic fertilizers, organic manure and crop residues. Although indirect N_2O emissions via N deposition and nitrate leaching and runoff could be significant depending on the local conditions (e.g. Venterea et al. 2011; Maharjan et al. 2014), especially in cases where there is a high rate of N application, they were not taken into account into this study due to high uncertainty. Quantification of CH_4 emissions from rice paddies was based on regional CH_4 flux.

$$
\begin{aligned}
&GHGI = I_{N2O} + I_{CH4} = \frac{Emissions_{N2O} + Emissions_{CH4}}{Yield} \\
&Emissions_{N2O} = N_2O\text{-}N_{input} \times EF_1 \times 44/28 \times \mathrm{GWP}_{N_2O} \\
&Emissions_{CH4} = Flux_{CH4} \times \mathrm{GWP}_{CH_4} \\
&N_2O\text{-}N_{input} = F_{SN} + F_{AW} + F_{CR}
\end{aligned} \tag{1}
$$

Where: GHGI ($kgCO_2e/t$); I_{CH4} is the intensity of CH_4 emissions in rice paddies and I_{N2O} is the intensity of N_2O emissions ($kgCO_2e/t$); $Emissions_{N2O}$ is the N_2O emissions from rice, wheat or maize fields ($kgCO_2e/ha$); $Emissions_{CH4}$ represent the CH_4 emissions from rice paddies($kgCO_2e/ha$); Yield denotes the per hectare average production (t/ha); $N_2O\text{-}N_{input}$ represents the total N inputs (kgN/ha); EF_1 is the emission factor for N_2O emissions from N inputs (kg N_2O–N/kg N input); 44/28 is to convert emissions from kg N_2O-N to kg N_2O; $Flux_{CH4}$ represents the CH_4 flux from rice paddies ($kgCH_4/ha$); GWP_{N2O} and GWP_{CH4} denote the direct GWP of N_2O and CH_4 at the 100yr horizon, 298 and 25; F_{SN}, F_{AW}, F_{CR} represent N inputs from synthetic fertilizers, animal manure and crop residues, respectively (kgN/ha).

Detailed equations for calculating N inputs from animal manure (F_{AW}) and crop residues (F_{CR}) are presented in Annex A.

2.2. Data sources

Agriculture activity data were collected at the provincial level while emission factors and other parameters (e.g. IPCC default factors) were average national values. In other words, data for $N_2O\text{-}N_{input}$ in Eqn (1) are province-specific and $Flux_{CH4}$ are region-specific, while other factors were held identical among provinces. Regions in China refer to northeast, north, northwest, east, south and central, and southwest China, each of which includes 3-7 provinces/municipalities. Farming activity data (cropping area, production, yield and total N fertilizer consumption) were extracted from the China Rural Statistical Yearbooks (MOA 1986-2013). Per hectare N application rates for individual crops were collected from the China Agricultural Products Cost-Benefit Yearbooks (NDRC 2001-2011), which are the sum of N fertilizer (pure nutrient) and 30% N fraction in compound and mixed fertilizers. To calculate the national and provincial GHGI in 2006, we used the three-year average of 2005-2007 to represent 2006 conditions to avoid large inter-annual variations. China-specific emission factors for direct N_2O emissions from croplands were obtained from Gao et al. (2011), which are 0.0105 and 0.0041 for upland and rice paddies, respectively. CH_4 fluxes of rice paddies in each region were direct CH4MOD modeled results from studies by Zhang et al. (2011a), which were employed for compiling National GHG Emission Inventories.

For estimating N inputs from animal manure, livestock numbers and the fraction of grazing animals were derived from the China Livestock Yearbooks (MOA 2001-2011) while other information required was selected from relevant literature and IPCC default values corresponding to conditions in China as displayed in Table S1. Estimation of N inputs from crop residues were mainly based on values reported by Gao et al. (2011) summarized in Table S2. Detailed information for data selection is provided in Annex B.

Regional level SOC data in 2010 were derived from Yu et al. (2013) to represent 2006 levels, and historic SOC contents were derived from similar research by Yu et al. (2012).

2.3. Design of emission scenarios for future cereal production

To project total GHG emissions and investigate mitigation potential from cereal production in China to 2020, we designed four agricultural management scenarios based on historical trends and the increase in expected future productivity. Total GHG emissions shall be affected by the GHGI and grain production, or N input and CH_4 flux levels, yield and cultivated area of each crop.

To focus on the impacts of GHGI change on overall emissions, cultivated area of each crop were assumed constant from 2010 to 2020. In all scenarios, 0.5%, 1% and 1.5% annual increase in yield were assigned for rice, wheat and maize respectively, based on 2005-2013 yield data released by the MOA (2006-2013). S0 is a conservative scenario that prescribes the same proportion of increase in N input relative to yield improvement. Scenario S1 assumes that no further N input is required to sustain equal productivity as in S0, while the N rate decreases by 1% per year under S2. Scenario S3 is an optimal scenario incorporating best management practices to cut the overall N rates and improve the irrigation regimes in rice paddies while achieving the yield requirements for safeguarding national food self-sufficiency (annual N input decrease at 2% and CH_4 flux decrease at 1%). The annual rates of change for these factors over 2010-2020 are summarized in Table 1.

3. Results and discussions

3.1. GHGI of rice production in 2006

GHGI of rice production in 2006 ranged from 730 kgCO_2e/t in Ningxia Province to 1,549 kgCO_2e/t in Hainan Province, with a national average of 947 kgCO_2e/t (Fig. 1a). In general, CH_4 made up about 90% of the total GHG emissions and was therefore the dominant gas in determining the carbon footprint of rice cultivation. Consequently, there was no obvious relationship between GHGI levels and N application rates, the latter being the major source of N_2O emissions. For example, the Jiangsu Province in east China received 51% higher N application than national average in rice production but was moderate in GHGI (16% lower than national average). It is, however, evident that the estimated GHGI for rice production was negatively correlated with yield levels. There was a large provincial variation in GHGI (Fig. 2a) with the most carbon intensive provinces located in the southeast coastal areas due to the highest regional CH_4 flux (250 kg/ha) because of higher temperature and greater level of organic matter input (Zhang et al. 2011a). The low GHGI in the southwestern provinces (Sichuan, Chongqing, Guizhou and Yunnan) can be attributed to lower CH_4 flux (200 kg/ha) relative to other places (215-250 kg/ha) because of low levels of organic matter application and rice biomass productivity. Among the six major rice producing provinces, which accounted for 55% of the national production, Hunan and Jiangxi had higher GHGIs than the national average, while Hubei, Jiangsu, Sichuan and Heilongjiang were below the national mean.

3.2. GHGI of wheat and maize production

The national average GHGI of wheat (Fig. 1b) and maize (Fig. 1b) for 2006 production were 265 and 230 kgCO_2e/t, respectively. Large spatial variability can be observed among provinces. For example, producing one ton of wheat in Ningxia emitted 3 times more N_2O than in Heilongjiang, attributable to significant differences in synthetic N input and wheat and maize yields between Chinese provinces. In general, synthetic N fertilizer made up at least 70% of total emissions and was therefore the primary emission contributor. Fig. 1(b, c) also shows that the trends of GHGI, which are affected by place-specific yield levels, were not necessarily consistent with those of per hectare N application rates. For instance, although the N application rate for maize in Ningxia (280 kgN/ha) was 30% higher than in Guangxi (215 kgN/ha), a much higher yield in Ningxia (6.97t/ha) than in Guangxi (3.88 t/ha) results in a lower maize GHGI in Ningxia. In contrast, a high N rate and low wheat productivity made Ningxia the most carbon intensive province for wheat cultivation.

The geographic variations of GHGIs of wheat (Fig. 2b) and maize (Fig. 2c) show both similarities and differences. In general, similar levels of GHGI can be observed for wheat and maize production (except for Ningxia); e.g. Yunnan was one of the most carbon intensive areas for both wheat and maize production in 2006. More N fertilizers were added to croplands in the northwest provinces to compensate poor soil fertility, resulting in elevated regional GHGI of wheat and maize production. The levels of maize GHGI converged to the range of 200-300 kgCO_2e/t, with obvious correlation with N rates and yields. Provincial discrepancies were more evident for wheat GHGI, implying that farmers were potentially more rational in determining the fertilizer amount for maize than for wheat. Among the five major wheat producing areas - Henan, Shandong, Hebei, Anhui and Jiangsu, which contributed about 73% of the national production, GHGI levels in Hebei and Jiangsu were superior to the national average. Among the major maize producing areas, only Hebei had a higher GHGI than the national mean, while Jilin, Shandong, Henan and Heilongjiang were lower.

GHGIs at the provincial level were further integrated to the regional scale for 2006 and compared with yields and SOC contents (Fig. S1) to indicate regional GHGI reduction strategies (Annex C).

3.3. Historical trends of regional GHGI of cereal production

Fig. 3a shows that national GHGI of rice production evolved at a different way to those of wheat and maize production, and the latter has always been the least carbon intensive of the three crops. Rice GHGI saw little variation beween 1985 and 2000, which can be explained by nearly the same rate of growth in the CH_4 flux, yield (Fig. 3b) as well as the N rate over this period. However, when rice yield reached a periodic peak in 1998 the CH_4 flux continued to climb, resulting in a sharp rise in GHGI in the first decade of the 21st century. Wheat and maize GHGIs had been steadily increasing from 1985 to 2000 since the growth rate of N application exceeded the rate of yield improvement. The GHGI began to stablize or even decrease after 2000 as the combined effects of increasing yields, abeit at a lower rate, and a stabilized synthetic N rate promoted by the national “Soil testing and fertilizer recommendation program” (MOA 2005) initiated in 2005. At the national level, there was also a positive correlation between SOC improvement and cereal productivity increase (Fig. 3b) (Pan et al. 2009).

Fig.4 illustrates that nearly all regional GHGI of rice, wheat and maize production reached a higher level in 2010 relative to 1985. For rice production (Fig. 4a), south and central and east regions have consistently been the most carbon intensive areas due to favorable climate conditions and greater level of organic matter application (Zhang et al. 2011a). In parallel, rice paddies in eastern, southern and central China are found to have experienced the greatest SOC increase (Zhang et al. 2007; Pan et al. 2010). In contrast, a lower level of crop residues, farm manure and green manure application enabled the southwest to emit least GHG in producing same amount of rice.

For wheat production (Fig. 4b), all regions except north China exhibited the same trend as the national average. Consequently, reducing N rates should be advocated in northern provinces, confirming the findings of other experimental and theoretical studies (Ju et al. 2009, 2011). Maize GHGI evolution patterns (Fig. 4c) were more diverse between geographic regions, with northeast China having the lowest GHGI . The northwest has been characterized with the highest GHGI in both wheat and maize production.

3.4. Ways to improve GHGI of cereal production while safeguarding food security

Integrated soil-crop management systems and better nutrient management techniques are advocated to address the key constraints to yield improvement and alleviate environmental impacts (Fan et al. 2012; Zhang et al. 2012). Extensive overuse of synthetic N fertilizers is well documented in China (Cui et al. 2010; Chen et al. 2011), resulting in significant losses and serious environmental externalities (Guo et al. 2010). Zhang et al. (2013) suggest a possible 42% cut in nationwide N fertilizer use applying the balance concept to equalize N input and above ground N removal. In parallel to optimal quantity, application time, right placement and appropriate product are also essential to better nutrient management. Adjusting basal/topdressing ratio of N fertilizers and popularizing fertilizer deep placement could improve crop N uptake and minimize N losses compared with conventional practices of applying largest proportion of N fertilizers on the surface before seeding (Cui et al. 2008; Zhang et al. 2011b). Replacing a proportion of ammonium-based fertilizers with nitrate-based fertilizers can also help minimize N_2O emissions and ammonia losses (Zhang et al. 2013). Nitrogen use efficiency (NUE) can also be improved by applying fertilizers added with nitrification and/or urease inhibitors and slow- and controlled-released fertilizers (Akiyama et al. 2010).

Better recycling of organic manures including animal excreta, crop residues and green manure enables further improvement in NUE, SOC content and land productivity. Adopting conservation tillage is found to be conducive to accumulate SOC density, improve water availability and reduce water and wind erosion, especially on land of poor productivity (Xu et al. 2007; He et al. 2010). Such practices shall be extended to wider areas supported by the MOA. Finally, biochar addition can be beneficial to soil quality and yield increase therefore offering substantial mitigation potential when it becomes economically available (Wang et al. 2014). As to CH_4 emissions from rice paddies, upgrading irrigation regimes from mid-season drainage, currently being practiced in most rice cultivation regions, to intermittent irrigation or controlled irrigation, could avoid as much as 1.256 CO_2e per hectare according to nationwide meta-analysis results (Wang et al. 2014).

In addition to mitigating climate change, some of these measures could actually be cost saving, simultaneously reducing input costs and/or enhancing productivity (Wreford et al. 2010; Wang et al. 2014). Further, in recent decades SOC content of cropland has increased along with improved crop yields in most regions of China (Pan et al. 2010; Yan et al. 2011; Yu et al. 2012). These findings highlight the important role of cropland in achieving emission reduction, safeguarding food security and enhancing carbon sequestration.

3.5. Implication for mitigation potential from cereal production

Fig. 5 illustrates that total GHG emissions from rice, wheat and maize production have grown by 12% from 2005 to 2010 caused by an 11% increase in cropping area and a 5% increase in average yield (Fig. 3b). In the S0 baseline scenario, although yields improve at the same rate of increase in N inputs, resulting in constant GHGI, total GHG emissions will still go up because of higher production levels. However, if no more N input is needed to enhance yields, emissions will stop increasing (scenario S1) and GHGIs will decrease. In contrast, if better fertilization practices are promoted to suppress the overuse of N fertilizers, total emissions will decline (scenario S2) by 8% compared to S0. Scenario S3 assumes substantial efforts are dedicated to minimizing the GHGI of cereal production by eradicating N over-application, adopting better water management in rice paddies and improving yield levels. In this case, I_{N2O} of rice, wheat and maize shall decline by 2.5%, 3% and 3.4% respectively, and I_{CH4} by 1.5% annually. Under this scenario, total GHG emissions are estimated to be 224MtCO_2e, a 17% decrease relative to S0 enabled by an 18% decrease in N input, 0.5-1.5% improvement in yields and 1% cut in average CH_4 flux. Such a mitigation scenario is feasible since the 18% cut in N use falls under the lower range of suggested 30-60% reduction (Ju et al. 2012; Zhang et al. 2013) and the 546 Mt cereal production meets the target for ensuring national food security.

Apart from the emission reduction potential, SOC density is projected to continue to increase at a rate of 0.4-0.48 tC/ha/yr in paddy soils and 0.16-0.22 tC/ha/yr in upland soils in the 2010s (Yu et al. 2013). This implies that even the C inputs (including manure and crop residue) to Chinese croplands remain unchanged with no improvement in tillage practices, aggregate national SOC stocks will still increase over the period 2010-2020. If improved agricultural management practices are widely adopted, as much as 70MtCO_2 could be sequestrated in the cropland soils. Carbon sequestration is therefore able to compensate 31% of GHG emissions under scenarios S3.

4. Conclusions

A low carbon development pathway implies minimization of emissions while increasing food production and GHGI is an indicator combining both objectives. As such it is a central element of any definition of sustainable intensification (Godfray and Garnett 2104). Our results on the GHGI of rice, wheat and maize production show that the southeast was the most carbon- intensive region in rice production in terms of CH_4 emissions, while GHGI of wheat and maize were both high in most north and northwest provinces due to the typical farming practices of farming systems in China. GHGI was low for all the three crops in the northeast area. The substantial heterogeneities of GHGI among provinces/regions and the inconsistency between trends of GHGIs and N application rates indicate considerable scope for improving carbon performance of cereal production and that actions and policies aiming to promote sustainable food production should be tailored to local conditions. Under the BAU scenarios where food production must grow to meet the demand of about 1.45 billion population, total GHG emissions will continue to increase albeit with constant GHGIs. Controlling GHG emissions from arable land thus requires additional mitigation efforts. Most abatement practices that improve crop yields will not only enable emission reductions but also improve soil fertility via carbon sequestration, therefore providing a triple win. Such findings can inform a broad range of policy, practitioner and investment discussions on GHG mitigation strategies, and can also serve as benchmark values for allocating quotas or as the baseline for generating carbon credits for any market-based mechanism.

Despite positive synergies with yield and soil fertility, abatement measures have not been widely adopted by farmers due to economic, political and social factors. Required capacity and infrastructure must be improved and agricultural extension service upgraded to lower GHGI and realize the mitigation potential and land productivity and fertility improvement potential that agricultural production offers.

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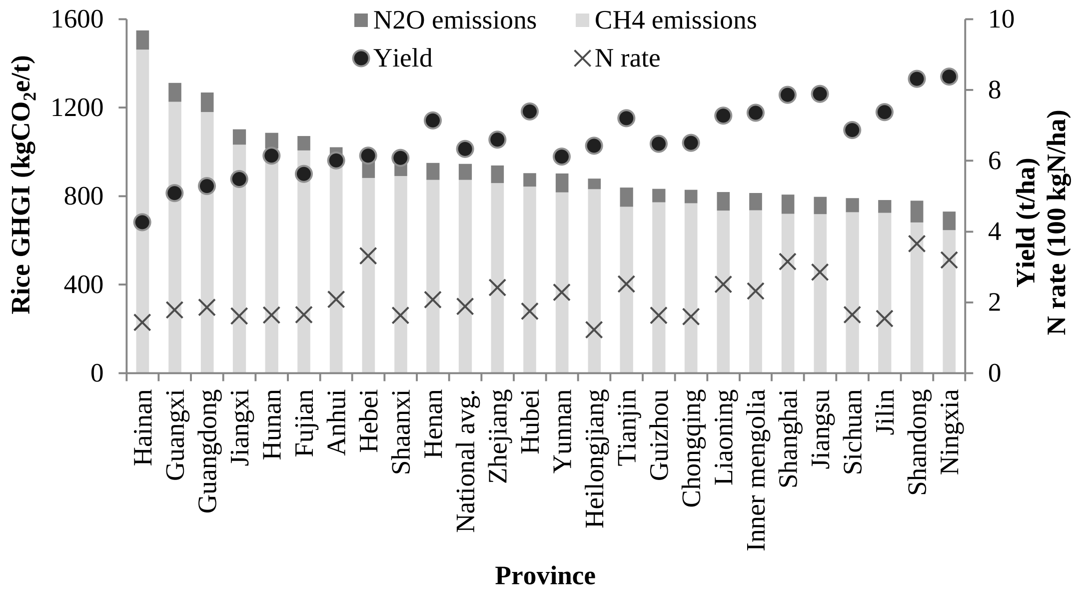
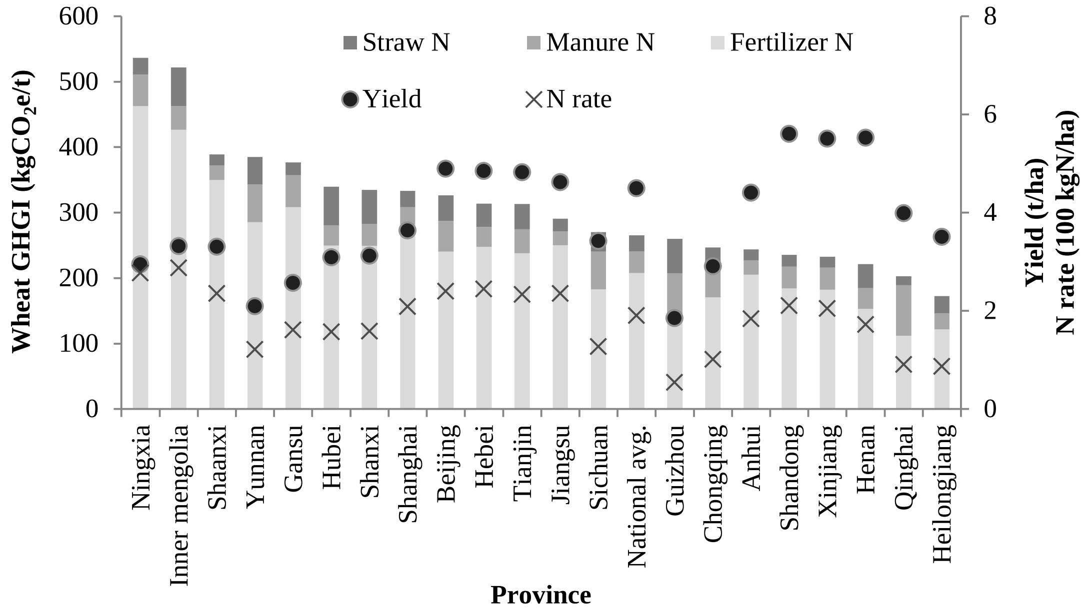
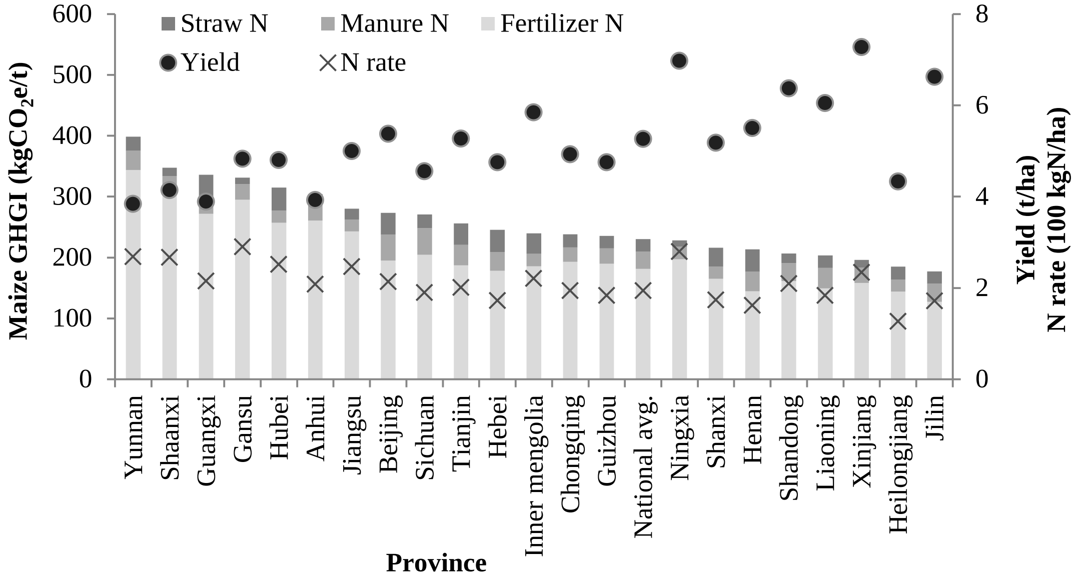
Figures:


N2O emissions
CH4 emissions
Yield
N rate
Rice GHGI (kgCO2e/t)
Yield (t/ha)
N rate (100 kgN/ha)
1600
1200
800
400
0
10
8
6
4
2
0
Hainan
Guangxi
Guangdong
Jiangxi
Hunan
Fujian
Anhui
Hebei
Shaanxi
Henan
National avg.
Zhejiang
Hubei
Yunnan
Heilongjiang
Tianjin
Guizhou
Chongqing
Liaoning
Inner mengolia
Shanghai
Jiangsu
Sichuan
Jilin
Shandong
Ningxia
Province

(a)

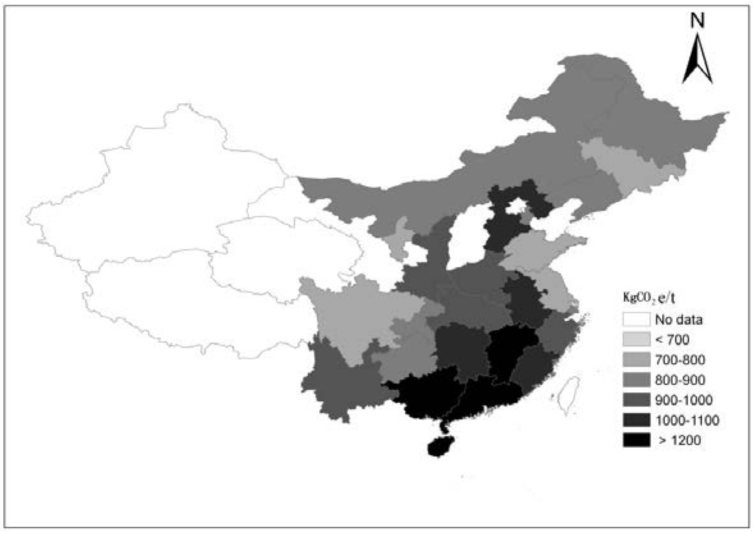

Straw N
Manure N
Fertilizer N
Yield
N rate
Wheat GHGI (kgCO2e/t)
Yield (t/ha)
N rate (100 kgN/ha)
600
500
400
300
200
100
0
8
6
4
2
0
Ningxia
Inner mengolia
Shaanxi
Yunnan
Gansu
Hubei
Shanxi
Shanghai
Beijing
Hebei
Tianjin
Jiangsu
Sichuan
National avg.
Guizhou
Chongqing
Anhui
Shandong
Xinjiang
Henan
Qinghai
Heilongjiang
Province

(b)

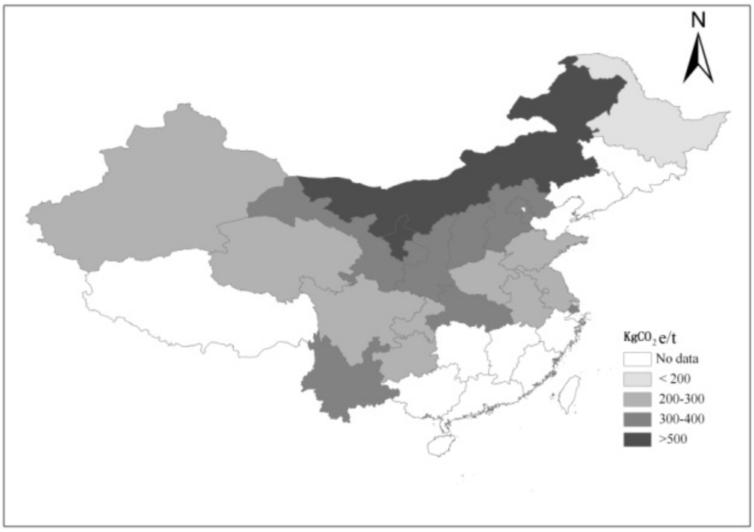


(c)

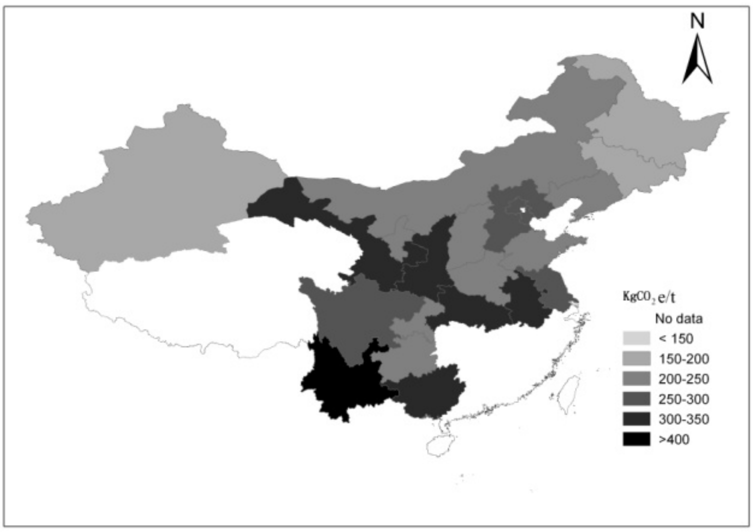
Fig.1 GHGI of rice (a), wheat (b) and maize (c) production in different provinces in 2006. The bars in (a) represent the contributions of N_2O and CH_4 to total GHGI; the bars in (b) and (c) represent the contributions of different N inputs to N_2O-derived GHGI.


N
$KgCO_2$ e/t
No data
< 700
700-800
800-900
900-1000
1000-1100
> 1200

(a)

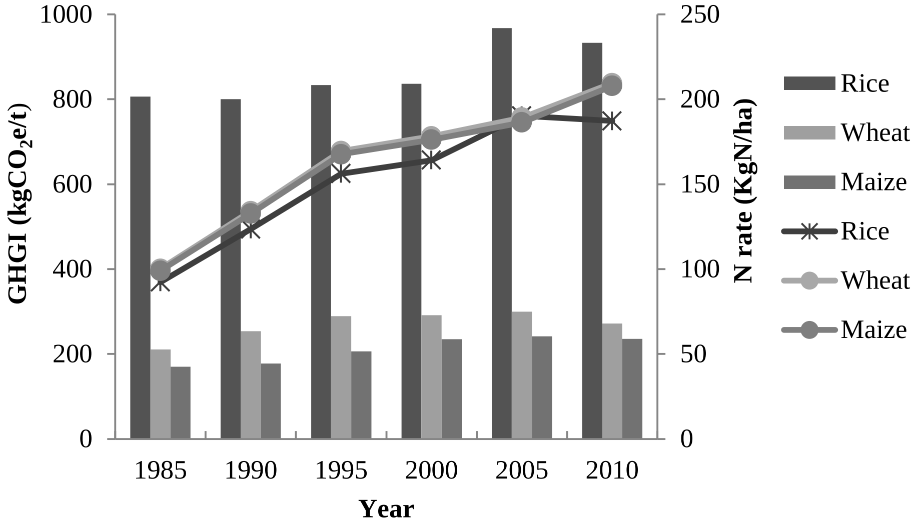

N
$KgCO_2$ e/t
No data
< 200
200-300
300-400
>500

(b)

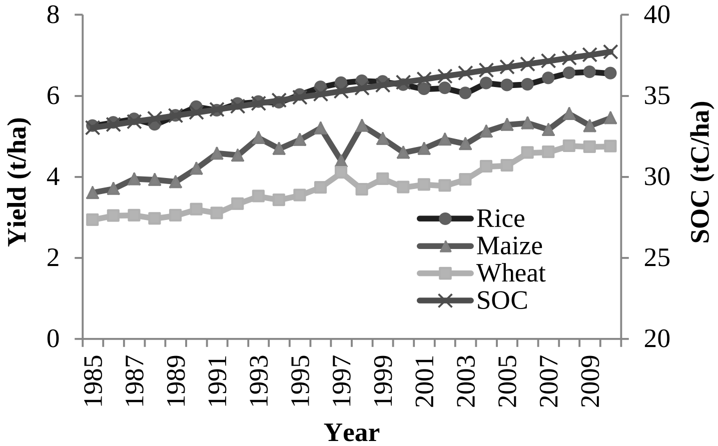

N
$KgCO_2$ e/t
No data
< 150
150-200
200-250
250-300
300-350
>400

(c)

Fig.2 The provincial GHGI levels of rice (a), wheat (b) and maize (c) production for 2006

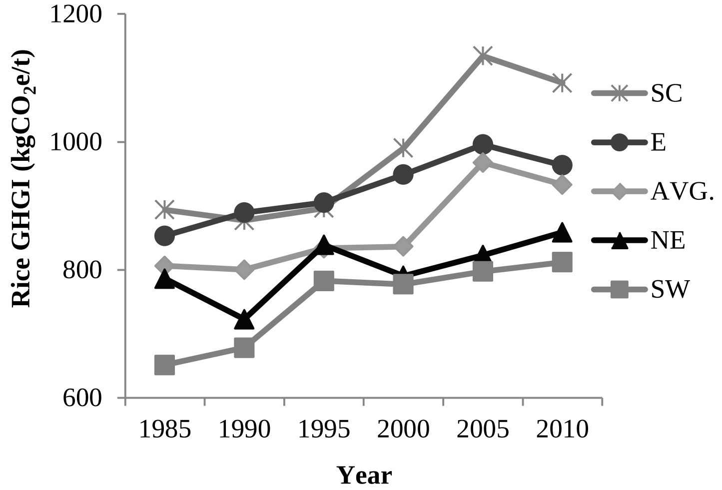


(a)

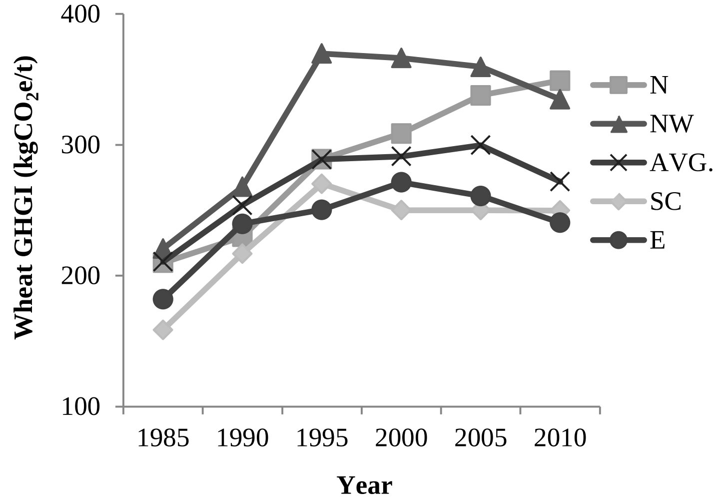


(b)

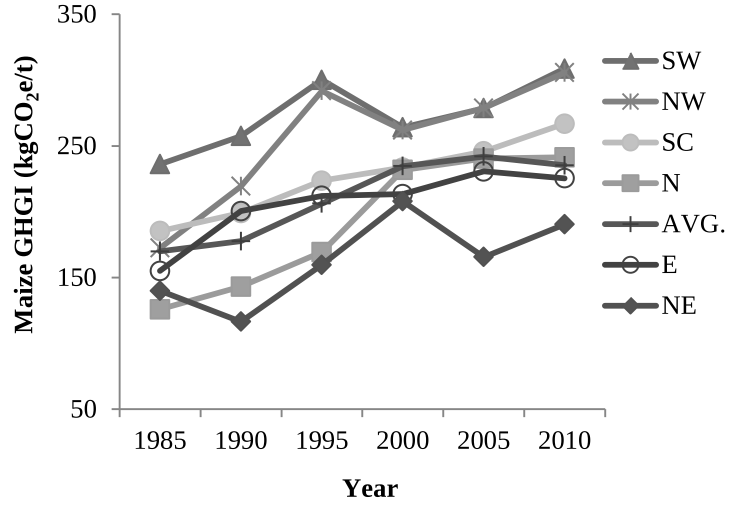
Fig.3 Historical trends of national average GHGI (a) and yield (b) of rice, wheat and maize production


Rice GHGI (kgCO2e/t)
1200
1000
800
600
1985 1990 1995 2000 2005 2010
Year
SC
E
AVG.
NE
SW

(a)


Wheat GHGI (kgCO2e/t)
400
300
200
100
1985 1990 1995 2000 2005 2010
Year
N
NW
AVG.
SC
E

(b)


Maize GHGI (kgCO2e/t)
350
250
150
50
1985 1990 1995 2000 2005 2010
Year
SW
NW
SC
N
AVG.
E
NE

(c)

Fig.4 Historic evolution of regional GHGI of rice (a), wheat (b) and maize(c) production. NE, N, NW, E, SC, SW and AVG refer to northeast, north, northwest, east, south and central, southwest China, and national average, respectively.

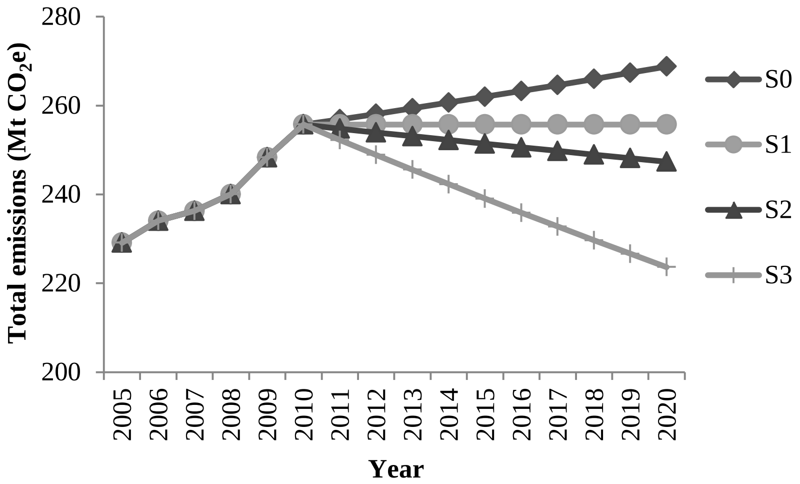


Fig.5 Total GHG emission scenarios from rice, wheat and maize production to 2020 in China

Tables:

Table 1 Emission scenarios for cereal production (annual rates of change)

Scenario	S0	S1	S2	S3
N_2O-N_{input}	rice +0.5%	Constant	rice -1%	rice -2%
	wheat +1%		wheat -1%	wheat -2%
	maize +1.5%		maize -1%	maize -2%
Yield	rice +0.5%	Same as S0	Same as S0	Same as S0
	wheat +1%			
	maize +1.5%			
I_{N2O}	Constant	rice -0.5%	rice -1.5%	rice -2.5%
		wheat -1%	wheat -2.0%	wheat -3.0%
		maize -1.5%	maize -2.5%	maize -3.4%
$Flux_{CH4}$	Constant	Constant	Constant	-1%
I_{CH4}	-0.5%	-0.5%	-0.5%	-1.5%
Cropping area	Constant	Constant	Constant	Constant

Note: changes in I_{N2O} and I_{CH4} are deduced from alterations in N_2O-N_{input}, $Flux_{CH4}$ and Yield. According to Eqn(1), I_{N2O} and I_{CH4} is proportional to N_2O-N_{input} and $Flux_{CH4}$ respectively, and both are inversely proportional to Yield.

Supplementary Material

Title: Greenhouse gas intensity of three main crops and implications for low-carbon agriculture in China

Journal: Climatic change

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Appendix A Estimation of N inputs from animal manure and crop residues

N inputs from animal manure (F_{AW}) was estimated following Eqn (S1).

$$
\begin{aligned}
F_{AW} &= \frac{\sum_{T} N_T \cdot (1 - Frac_{Grazing(T)}) \cdot Nex_T \cdot (1 - Frac_{Loss(T)})}{CA_{eqv}} \\
Nex_T &= N_{rate(T)} \cdot \frac{TAM_T}{1000} \cdot 365 \\
N_T &= Days\_alive_T \cdot \frac{N_{S(T)}}{365} \quad if \quad Days\_alive_T < 365 \\
CA_{eqv} &= a \cdot CA_{veg} + b \cdot CA_{fruit} + CA_{other}
\end{aligned}
\tag{S1}
$$

N_T is the annual population of livestock T. T denotes livestock category. $Frac_{Grazing(T)}$ is the fraction of grazing population (%). Nex_T represents the annual N excretion (kgN/animal/yr). $Frac_{Loss(T)}$ represents the amount of managed manure N that is lost in the manure management system (%). CA_{eqv} denotes the equivalent cropping area (kha). $N_{rate(T)}$ denotes the default N excretion rate (kgN/(1000 kg animal mass/day)). TAM_T is the typical animal mass (kg/animal). $Days\_alive_T$ is the average breeding days before slaughter. $N_{S(T)}$ is the average number slaughtered (or use stock number if average breeding days exceed a complete year). CA_{veg}, CA_{fruit} and CA_{other} are the cropping areas of vegetables, fruits and other crops (total excluding vegetable and fruits), respectively (kha). a and b is the ratio of organic manure received by respectively vegetable fields and fruits compared with other crop lands.

N inputs from animal manure crop residues (F_{CR}) was estimated following Eqn (S2).

$$F_{CR}=\frac{\sum_i F_{CR\text{-}AG(i)}+F_{CR\text{-}BG(i)}}{\sum_i CA_i}=\frac{\sum_i Pdt_i \cdot R_{ST\text{-}GR(i)} \cdot N_i \cdot (R_{SR(i)}+R_{BG\text{-}AG(i)})}{\sum_i CA_i} \qquad \text{(S2)}$$

$F_{CR\text{-}AG(i)}$ and $F_{CR\text{-}BG(i)}$ represent the N input from aboveground and belowground crop residues, respectively (kgN/ha). i denotes crop type (rice, wheat, maize). CA_i is the annual cropping area (kha). Pdt_i is the annual harvested product (kt). $R_{ST\text{-}GR(i)}$ is the ratio of straw to grain in terms of dry matter. N_i is the N content of crop i residue (g/kg). $R_{SR(i)}$ is the proportion of above-ground residue returned to land (%). $R_{BG\text{-}AG(i)}$ is the ratio of below-ground residue weight to above-ground plant weight.

Since N application rates for the three main cereals are only available for 2005 and 2010 at 5-year intervals, Eqn (S3) was formulated to estimate the N application rate in a given year.

$$F_{SN(i)j}=F_{SN(i)2005}\bullet\frac{F_{SNj}}{F_{SN2005}}=F_{SN(i)2005}\bullet\frac{TN_j}{TCA_j}\bullet\frac{TCA_{2005}}{TN_{2005}} \qquad \text{(S3)}$$

$F_{SN(i)j}$ is the N application rate in year j in a province (kgN/ha). i denotes crop type and j denotes year. $F_{SN(i)2005}$ is the N rate of crop i in 2005(kgN/ha). F_{SNj} and F_{SN2005} denote the crop-wide average N rate in year j and 2005, respectively (kgN/ha). TN_j and TN_{2005} are the provincial total synthetic N consumption in year j and 2005(kt). TCA_j and TCA_{2005} represent the total cropping area in year j and 2005(kha).

Appendix B Data sources for estimation of N inputs from animal manure and crop residues

The annual number of livestock slaughtered was collected for pigs, hens, broiler chicken and rabbits with the average breeding days standing at 158, 65, 352 and 105, respectively (MOA 2001-2011). For other types of animals, annual stock numbers were used. The fraction of grazing cattle or sheep was the ratio of total grazing animals (the sum of livestock numbers in grazing areas and half-grazing areas) to the total stock number (MOA 2001-2011). a and b in Eqn (S1)were assigned 4 and 5 since survey results (Huang and Tang 2010; Zhang et al. 2013) reported that vegetable and fruit fields generally received respectively 4 and 5 times more organic manure than cereal cropping lands in the 2000s.

Other information required in Eqn (S1) was selected from relevant literature and IPCC default values corresponding to conditions in China as displayed in Table S1.

Table S1 Selected values for estimating N input to croplands from animal manure

	Non-dairy cattle	Milk cows	Sheep (goats)	Horses	Asses	Mules	Pigs	Chicken	Rabbits
$Frac_{Grazing}$[a]	17%		35%						
N_{rate}	0.34	0.47	1.27	0.46	0.46	0.46	0.50	0.82	
TAM	319	350	29	238	130	130	50[b]	2	
Nex	39.6	60.0	13.4	40.0	21.8	21.8	9.1	0.5	8.1
$Frac_{Loss}$	40%	40%	67%	50%	50%	50%	35%	50%	50%
Days_alive[c]							158	180	105

[a] Data in this table represents the national average.

[b] IPCC default value for Asia is 28. Here we adopted 50 according to Chinese conditions.

[c] Days_alive of chicken is the weighted number of broiler chicken (65 days) and hens (352 days), which account for 60% and 40% of chicken population, respectively.

Values for parameters in Eqn (S2) were mainly obtained from the research by Gao et al. (2011) and are summarized in Table S2. The proportion of above-ground straw residues returned to land in 2006 was derived from results reported by Gao et al. (2009). The nationwide ratio of straw retuned to land was reported at 15.2% in 1999 (Han et al. 2002) and rose to 24.3% in 2006 (Gao et al. 2009), implying an annual growth rate of 6.93%. This rate was employed to estimate the percentage of straw recycled to farmland in target years.

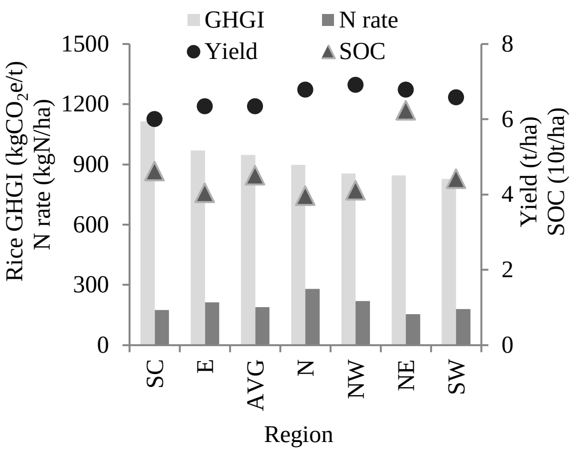
Table S2 Selected values for estimating N input to croplands from crop residues

		Rice	Wheat	Maize
$R_{ST\text{-}GR}$		0.9	1.1	1.2
N	g/kg	9.1	6.5	9.2
$R_{BG\text{-}AG}$		0.125	0.166	0.170
	North	57.7%	84.5%	51.0%
	Northeast	25.0%	36.6%	22.1%
	East	19.4%	28.5%	17.2%
$R_{SR(2006)}$	South Central	58.9%	86.3%	52.0%
	Southwest	30.1%	44.2%	26.6%
	Northwest	14.8%	21.6%	13.0%
	National average	29.9%	43.8%	26.4%

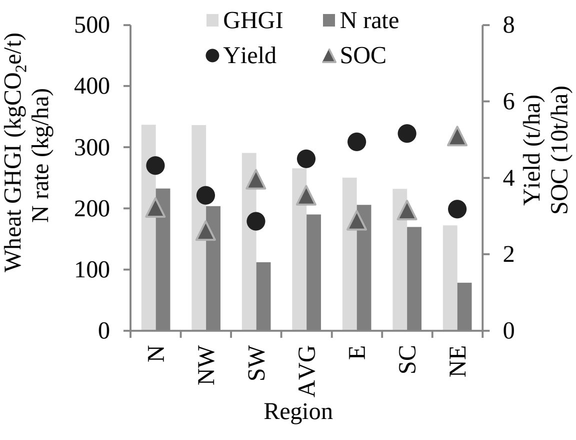
Note: North region includes Beijing, Tianjin, Hebei, Shanxi and Inner Mongolia; Northeast region includes Heilongjiang, Liaoning and Jilin; East region includes Shanghai, Anhui, Fujian, Jiangsu, Jiangxi, Shandong and Zhejiang; South Central region includes Guangdong, Hainan, Henan, Hubei, Hunan and Guangxi; Southwest region includes Chongqing, Guizhou, Sichuan, Yunnan and Tibet; Northwest region includes Gansu, Qinghai, Shaanxi, Ningxia and Xinjiang.

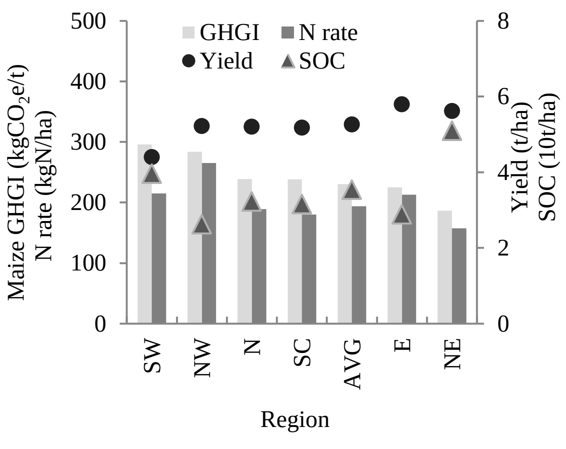
Appendix C GHGI at regional level in 2006 and implications for mitigation strategies

The GHGI, yield and synthetic N rate of rice, wheat and maize cultivation as well as the SOC content at the regional scale in 2006 are illustrated in Fig. S1. In general, the southwest had lowest cereal yields, albeit second highest SOC after the northeast. Conversely more N fertilizers were added to croplands in northwest provinces to compensate poor soil fertility, resulting in elevated regional GHGI of crop production. Fig. S1 reveals that yield levels do not necessarily correspond to local SOC status, since productivity is also influenced by climate, precipitation and other factors. In this regard, regional strategies to minimize GHGI and improve soil fertility should accommodate local climatic, soil and water conditions and management practices. For example, in the northwest measures improving SOC density (e.g. conservation tillage) should be favored to enhance soil fertility and land productivity. In intensive cropping systems in east and north China where over-fertilization is prominent, more efficient use of N fertilizer can allow N rates to be cut by 30 to 60% without sacrificing crop yields (Ju et al. 2009). Although the northeast was the least carbon intensive region in cereal production, this came at the expense of net carbon losses, especially in Heilongjiang Province (Pan et al. 2010; Yu et al. 2012), thus calling for better management practices to sustain soil fertility in this region.



(a)





(b)

(c)

Fig.S1 GHGIs of rice (a), wheat (b) and maize (c) production in different regions in 2006 and their relationship with yield, N rates and SOC content. NE, N, NW, E, SC, SW and AVG refer to northeast, north, northwest, east, south and central, southwest China, and national average, respectively.

Additional references for electronic supplemental materials not otherwise cited in paper

Gao LW, Ma L, Zhang WF et al (2009) Estimation of nutrient resource quantity of crop straw and its utilization situation in China. Transactions of the Chinese Society of Agricultural Engineering 25:173-179

Han LJ, Yan QJ, Liu XY, Hu JY (2002) Straw Resources and Their Utilization in China. Transactions of the Chinese Society of Agricultural Engineering 18(3):87-91

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