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

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Christian de Perthuis², Xiaotang Ju⁴, Erda Lin^{1,6}, Dominic Moran⁵

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 CH₄ 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.

Keywords : food security, low-carbon agriculture, greenhouse gas intensity, China.

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1 **Title**

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

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16

17 **Abstract:**

18 China faces significant challenges in reconciling food security goals with the objective of becoming a
19 low-carbon economy. Agriculture accounts for approximately 11% of China's national greenhouse gas
20 (GHG) emissions with cereal production representing a large proportion (about 32%) of agricultural
21 emissions. Minimizing emissions per unit of product is a policy objective and we estimated the GHG
22 intensities (GHGI) of rice, wheat and maize production in China from 1985 to 2010. Results show
23 significant variations of GHGIs among Chinese provinces and regions. Relative to wheat and maize,
24 GHGI of rice production is much higher owing to CH₄ emissions, and is more closely related to yield
25 levels. In general, the south and central has been the most carbon intensive region in rice production
26 while the GHGI of wheat production is highest in north and northwest provinces. The southwest has
27 been characterized by the highest maize GHGI but the lowest rice GHGI. Compared to the baseline
28 scenario, a 2% annual reduction in N inputs, combined with improved water management in rice
29 paddies, will mitigate 17% of total GHG emissions from cereal production in 2020 while sustaining
30 the required yield increase to ensure food security. Better management practices will entail additional
31 gains in soil organic carbon further decreasing GHGI. To realize the full mitigation potential while
32 maximizing agriculture development, the design of appropriate policies should accommodate local
33 conditions.

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

35 **Research highlights**

36 ► Greenhouse gas intensity (GHGI) of rice, wheat and maize production are estimated on provincial,
37 regional and national scales in China

38 ► Substantial variation in GHGI of cereal production exists among provinces and regions

39 ► Reducing GHG emissions, ensuring food security and improving soil fertility can be achieved
40 simultaneously

41 ► GHGI of cereal production stabilized or decreased after 2005 and should further decline to ensure the
42 successful transition towards low-carbon agriculture

43

44 **1. Introduction**

45 China has made substantial efforts to increase crop production to feed about 20% of the global
46 population with only 8% of the world's arable land (World Bank 2013). From 1961 to 2010, total
47 cereal production has increased almost five-fold from 107 to 497 million tons (Mt) and crop yields
48 have improved at almost the same pace (FAO 2013). Looking towards 2020, the government set a
49 target of increasing national grain production capacity to over 545 Mt to meet growing demands for
50 higher animal protein diets and to maintain the domestic food self-sufficiency rate at 95% (NDRC
51 2009). This implies that while constrained by limited arable land, grain yield must grow by at least 0.9%
52 annually in the period 2011-2020. While facing this food security challenge China is also grappling
53 with related constraints in terms of declining water availability, an increasing opportunity cost of rural
54 labour and the challenges of climate change. The latter has emerged as a significant threat to
55 agricultural production, altering weather conditions and causing more frequent extreme weather events
56 and disasters (IPCC 2007a).

57 While vulnerable to climate change agriculture is also a significant source of anthropogenic
58 greenhouse gases (GHG) emissions (IPCC 2007b). The sector emitted approximately 820 Mt CO₂
59 equivalent (CO₂e) in 2005, or 11% of the national total (NCCC 2012). Agriculture was the largest
60 source of nitrous oxide (N₂O) and methane (CH₄) emissions, arising mainly from livestock enteric
61 fermentation, Nitrogen (N) additions to cropland, rice cultivation and animal waste management.
62 Cropland N₂O emissions produced in soils through the microbial processes of nitrification and
63 denitrification was responsible for 25% of agriculture GHG emissions in 2005 and CH₄ emissions from
64 rice cultivation contributed 20%. Cereal production (rice, wheat and maize) accounted for about 47%
65 of national N fertilizer consumption (Heffer 2009) and generated around 32% of GHG emissions from
66 agriculture.

67 The sector is now under increasing scrutiny for its ability to mitigate climate change through both
68 emissions reduction and carbon (C) sequestration. A range of abatement measures have been identified
69 as applicable in the arable sector (e.g. IPCC 2007b; Oenema et al 2001; Smith et al 2008), which can
70 be broadly grouped into increased nitrogen use efficiency (NUE), improving water regimes in rice
71 paddies and sequestering C into cultivated soils. Many mitigation measures could actually be cost
72 saving, simultaneously reducing input costs and/or enhancing productivity (Wreford et al. 2010).
73 Further, in recent decades soil organic carbon (SOC) content of cropland has increased along with

74 improved crop yields in most regions of China (Huang and Sun 2006; Pan et al. 2010; Yan et al. 2011;
75 Yu et al. 2012) and is predicted to continue to increase in the next 40 years (Yu et al. 2013). These
76 findings highlight the important role of cropland in achieving emission reduction, safeguarding food
77 security and enhancing carbon sequestration.

78 The Chinese government has recently put more effort into combating climate change and national
79 mitigation aspirations have been outlined in the 12th Five-Year Plan (FYP) to cut the carbon intensity
80 of the economy by 17% in 2015 compared with 2010 levels. The 12th FYP also called for controls on
81 agricultural GHG emissions. In response, the Ministry of Agriculture (MOA) has initiated programs to
82 improve fertilizer use efficiency by 3% and enhance irrigation water use efficiency by 6% by 2015
83 from 2010. In addition, the government has planned to bring an additional 11.3 Mha of croplands
84 under conservation tillage between 2009-2015 in north China (MOA 2009). The growing desire to
85 integrate climate change dimensions into agricultural policies reflects the government's willingness to
86 pursue low carbon development in agriculture, characterized by higher productivity, more efficient use
87 of resources and low GHG emissions intensity (Norse 2012).

88 The concept of GHG intensity (GHGI), expressed as the overall GHG emissions per unit of
89 product, is suggested as a useful metric to evaluate NUE and to help identify mitigation strategies
90 (Chen et al. 2011; Venterea et al. 2011; Tubiello et al. 2012). Applying such an indicator can encourage
91 better management practices resulting in higher crop production per area and reduced N losses and
92 GHG emissions (van Groenigen et al. 2010).

93 In this context, the Global Research Alliance on Agricultural Greenhouse Gases was launched in
94 December 2009 to help reduce the GHGI of agricultural production. FAO (Tubiello et al. 2014)
95 reported that over the period 1961-2010 the world average GHGI of rice decreased by 49% while that
96 of main cereals (wheat and maize) increased by 45%, and suggested that effective mitigation strategies
97 are needed to achieve sustainable intensification, ensuring that further efficiency improvements can
98 lead to reduced absolute emissions. Bonesmo et al. (2012) investigated the GHGI of 95 arable farms in
99 Norway and suggested that increased gross margins in grain and oilseed crop production could be
100 achieved with decreasing GHGI. The GHGI of cereal production on experimental sites was also
101 quantified to compare the overall mitigation effects of different abatement measures. Findings (e.g.
102 Mosier et al. 2006; Shang et al. 2011; Huang et al. 2013; Ma et al. 2013) indicated that economic and
103 climate benefits can be simultaneously achieved by improved management practices. But to date there

104 is no synthetic estimate of current and historical GHGI of cereal production on a national, regional or
105 provincial level in China. Such information is crucial for identifying efficient regional mitigation
106 strategies and actions tailored to local agricultural production systems and management practices.

107 This paper provides estimates of GHGI for rice, wheat and maize production using agro-statistics
108 data for the national, regional and provincial scale for 2006. To illustrate the trends and the evolution
109 of intensity we quantify national and regional GHGI from 1985 to 2010 at 5-year intervals and analyze
110 emission reduction and carbon sequestration potentials from cereal production. The aim is to provide
111 suggestions on possible national or regional policies to foster sustainable intensification in rural China.
112 The paper is structured as follows. Section two describes the derivation of GHGI and outlines data
113 sources for projecting intensities. Section three discusses the results before a conclusion in section
114 four.

115 2. Materials and methods

116 2.1. Methodology

117 GHGI refers to the climatic impacts of agriculture practices in terms of per unit of product and is
118 calculated by dividing total Global Warming Potential (GWP)-weighted emissions of cereal production
119 by crop yield. N₂O emissions are accounted for quantifying GHGI of wheat and maize production
120 while both CH₄ and N₂O are considered for rice paddies. Carbon sequestration is not directly included
121 in the estimate of GWP-weighted emissions due to large uncertainties in SOC content and limited data
122 availability. Despite consensus on the average SOC increment in China's cropland, discrepancies in
123 annual intensity change rates have been reported using various methods (Huang and Sun 2006; Sun et
124 al. 2010 ; Pan et al. 2010; Yan et al. 2011; Yu et al. 2012). In addition, SOC density change data at the
125 provincial level is unavailable. Nevertheless, SOC change patterns and interactions with GHGI will be
126 analyzed in the discussion section. The analysis focuses on emissions within the farm gate, i.e. they are
127 not full life-cycle assessment (e.g. emissions related to energy use and fertilizer manufacture and
128 transportation).

129 We followed the 2006 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006) to
130 estimate N₂O emissions from rice, wheat and maize production (Eqn (S1)). We considered direct N₂O
131 emissions from the three major N input sources - synthetic fertilizers, organic manure and crop
132 residues. Due to high uncertainty and relatively minor contribution, indirect N₂O emissions via N
133 deposition (associated with ammonia volatilization) and nitrate leaching and runoff were not taken into
134 account. Quantification of CH₄ emissions from rice paddies was based on regional CH₄ flux from
135 comprehensive studies conducted by Zhang et al. (2011a).

$$GHGI = \frac{Emissions_{N_2O} + Emissions_{CH_4(FR)}}{Yield}$$

$$136 \quad Emissions_{N_2O} = N_2O - N_{input} \times EF_{1(FR)} \times \frac{44}{28} \times GWP_{N_2O} \quad (S1)$$

$$Emissions_{CH_4(FR)} = Flux_{CH_4(FR)} \times GWP_{CH_4}$$

$$N_2O - N_{input} = F_{SN} + F_{AW} + F_{CR}$$

137 GHGI is the GHG intensity of crop production (kgCO₂e/t). Emissions_{N₂O} is the per hectare N₂O
138 emissions from rice, wheat or maize fields (kgCO₂e/ha). Emissions_{CH₄(FR)} is the per hectare CH₄
139 emissions from rice paddies(kgCO₂e/ha). Yield denotes the per hectare average production (t/ha).
140 N₂O-N_{input} represents the per hectare total N inputs (kgN/ha). EF₁ and EF_{1(FR)} are the emission factors

141 for N₂O emissions from N input for uplands and rice paddies, respectively (kg/kg). 44/28 is to convert
 142 emissions from kg N₂O-N to kg N₂O. Flux_{CH₄(FR)} represents the CH₄ flux from rice paddies (kgCH₄/ha).
 143 GWP_{N₂O} and GWP_{CH₄} denote the direct GWP of N₂O and CH₄ respectively at the 100yr horizon, 298
 144 and 25. F_{SN}, F_{AW}, F_{CR} represent per hectare N input from synthetic fertilizers, animal manure and crop
 145 residues, respectively (kgN/ha).

146 F_{AW} was estimated following Eqn (S2).

$$F_{AW} = \frac{\sum_T N_T [(1 - \text{Frac}_{\text{Grazing}(T)}) \text{Nex}_T (1 - \text{Frac}_{\text{Loss}(T)})]}{CA_{eqv}} \quad (S2)$$

$$\text{Nex}_T = N_{\text{rate}(T)} \left[\frac{TAM_T}{1000} \right] 365$$

$$N_T = \text{Days_alive}_T \left[\frac{N_{S(T)}}{365} \right] \quad \text{if } \text{Days_alive}_T < 365$$

$$CA_{eqv} = a [CA_{veg} + b [CA_{fruit} + CA_{other}]$$

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

157 F_{CR} was estimated following Eqn (S3).

$$F_{CR} = \frac{\sum_i F_{CR-AG(i)} + F_{CR-BG(i)}}{\sum_i CA_i} \quad (S3)$$

$$= \frac{\sum_i Pdt_i [R_{ST-GR(i)} N_i (R_{SR(i)} + R_{BG-AG(i)})]}{\sum_i CA_i}$$

159 F_{CR-AG(i)} and F_{CR-BG(i)} represent the N input from aboveground and belowground crop residues,
 160 respectively (kgN/ha). i denotes crop type (rice, wheat, maize). CA_i is the annual cropping area (kha).
 161 Pdt_i is the annual harvested product (kt). R_{ST-GR(i)} is the ratio of straw to grain in terms of dry matter. N_i

162 is the N content of crop *i* residue (g/kg). $R_{SR(i)}$ is the proportion of above-ground residue returned to
 163 land (%). $R_{BG-AG(i)}$ is the ratio of below-ground residue weight to above-ground plant weight.

164 Since N application rates for the three main cereals are only available for 2005 and 2010 at 5-year
 165 intervals, Eqn (S4) was formulated to estimate the N application rate in a given year.

$$166 \quad F_{SN(i)j} = F_{SN(i)2005} \cdot \frac{F_{SNj}}{F_{SN2005}} = F_{SN(i)2005} \cdot \frac{TN_j}{TCA_j} \cdot \frac{TCA_{2005}}{TN_{2005}} \quad (S4)$$

167 $F_{SN(i)j}$ is the N application rate in year *j* in a province (kgN/ha). *i* denotes crop type (rice, wheat,
 168 maize) 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
 169 crop-wide average N rate in year *j* and 2005, respectively (kgN/ha). TN_j and TN_{2005} are the provincial
 170 total synthetic N consumption in year *j* and 2005(kg). TCA_j and TCA_{2005} represent the total cropping
 171 area in year *j* and 2005(kha).

172 2.2. Data sources and treatment

173 We used the three-year average of 2005-2007 to represent 2006 conditions to avoid large interannual
 174 variations in the dataset. Agriculture activity data (cropping area, production, yield, total N fertilizer
 175 consumption and livestock number) were extracted from the China Rural Statistical Yearbooks (MOA
 176 1986-2013) and the China Livestock Yearbooks (MOA 2001-2011). Per hectare N application rates for
 177 individual crops were collected from the China Agricultural Products Cost-Benefit Yearbooks (NDRC
 178 1998-2011), and we adopted N fraction of 30% in the reported compound and mixed fertilizers (Sun
 179 and Huang, 2012). China-specific emission factors for direct N_2O emissions from croplands were
 180 obtained from studies by Gao et al. (2011), which are 0.0105 and 0.0041 for upland fields and rice
 181 paddies, respectively. CH_4 fluxes of rice paddies were direct CH_4MOD modeled results from studies
 182 by Zhang et al. (2011a), which were employed for compiling National GHG Emission Inventories.

183 The annual number of livestock slaughtered was collected for pigs, hens, broiler chicken and
 184 rabbits with the average breeding days standing at 158, 65, 352 and 105, respectively (MOA
 185 2001-2011). For other types of animals, annual stock numbers were used. The fraction of grazing cattle
 186 or sheep was the ratio of total grazing animals (the sum of livestock numbers in grazing areas and
 187 half-grazing areas) to the total stock number (MOA 2001-2011). *a* and *b* in Eqn (S2) were assigned 4
 188 and 5 since survey results (Huang and Tang 2010; Zhang et al. 2013) reported that vegetable and fruit
 189 fields generally received respectively 4 and 5 times more organic manure than cereal cropping lands in

190 the 2000s.

191 Other information required in Eqn (S2) was selected from relevant literature and IPCC default
 192 values corresponding to conditions in China as displayed in Table 1a. Values for parameters in Eqn
 193 (S3) were mainly obtained from the research by Gao et al. (2011) and are summarized in Table 1b. The
 194 proportion of above-ground straw residues returned to land in 2006 was derived from results report by
 195 Gao et al. (2009). The nationwide ratio of straw returned to land was reported at 15.2% in 1999 (Han et
 196 al. 2002) and rose to 24.3% in 2006 (Gao et al. 2009), implying an annual rate of increase of 6.93%.

197 This rate was employed to estimate the percentage of straw recycled to farmland in target years.

198 Table 1a 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 | |
| N _{ex} | 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 |

199 ^aData in this table represents the national average.

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

201 ^c Days_{alive} of chicken is the weighted number of broiler chicken (65 days) and hens (352 days), which account
 202 for 60% and 40% of chicken population, respectively.

203 Table 1b Selected values for estimating N input to croplands from crop residues

| | | Rice | Wheat | Maize |
|-----------------------|------------------|-------|-------|-------|
| R _{ST-GR} | | 0.9 | 1.1 | 1.2 |
| N | g/kg | 9.1 | 6.5 | 9.2 |
| R _{BG-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% |

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

209 Regional level SOC data for rice paddy and upland in 2010 were derived from Yu et al. (2013) to
 210 represent 2006 levels, and historic SOC contents were derived from similar research by Yu et al.
 211 (2012).

212 2.3. Design of emission scenarios for future cereal production

213 To project total GHG emissions and investigate mitigation potential from cereal production in China to
 214 2020, we designed four agricultural management scenarios based on historical trends and the increase
 215 in expected future productivity. Total GHG emissions shall be affected by the GHGI and grain
 216 production, or N input and CH₄ flux levels, yield and cultivated area of each crop. The annual rates of
 217 change for these factors over 2010-2020 are summarized in Table 2.

218 Table 2 Emission scenarios (annual rates of change) for cereal production

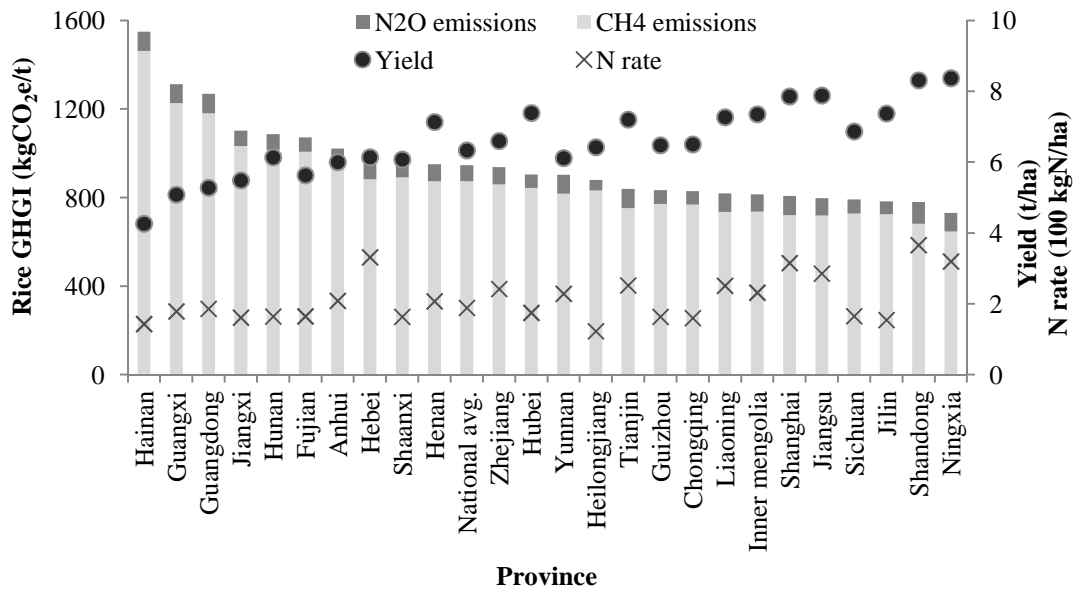
| Scenario | S0 | S1 | S2 | S3 |
|----------------------|--|--|--|--|
| I _{N2O} | Constant | rice -0.5% wheat -1% maize -1.5% | rice -1.5% wheat -2.0% maize -2.5% | rice -2.5% wheat -3.0% maize -3.4% |
| N _{input} | rice +0.5% wheat +1% maize +1.5% | Constant | rice -1% wheat -1% maize -1% | rice -2% wheat -2% maize -2% |
| Yield | rice +0.5% wheat +1% maize +1.5% | Same as S10 | Same as S10 | Same as S10 |
| I _{CH4} | -0.5% | -0.5% | -0.5% | -1.5% |
| CH ₄ flux | Constant | Constant | Constant | -1% |
| Cropping area | Constant | Constant | Constant | Constant |

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

228 **3. Results and discussions**

229 3.1. GHGI of rice production in 2006

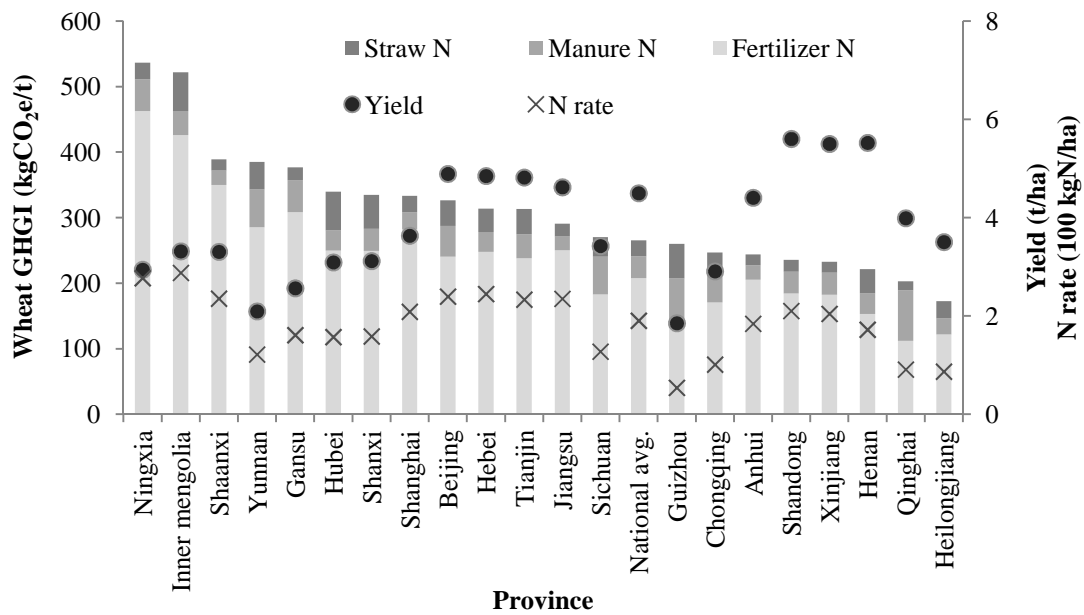
230 GHGI of rice production in 2006 ranged from 730 kgCO₂e/t in Ningxia Province to 1,549 kgCO₂e/t in
 231 Hainan Province, with a national average of 947 kgCO₂e/t (Fig. 1a). In general, CH₄ made up about 90%
 232 of the total GHG emissions and was therefore the dominant gas in determining the carbon footprint of
 233 rice cultivation. Consequently, there was no obvious relationship between GHGI levels and N
 234 application rates, the latter being the major source of N₂O emissions. It is, however, evident that the
 235 estimated GHGI for rice production was negatively correlated with yield levels. There was a large
 236 provincial variation in GHGI (Fig. 2a) with the most carbon intensive provinces located in the
 237 southeast coastal areas due to the highest regional CH₄ flux (250 kg/ha). The low GHGI of rice
 238 production in the southwestern provinces (Sichuan, Chongqing, Guizhou and Yunnan) can be
 239 attributed to lower CH₄ flux (200 kg/ha) relative to other places (215-250 kg/ha). Among the six major
 240 rice producing provinces, which accounted for 55% of the national production, Hunan and Jiangxi had
 241 higher GHGIs than the national average, while Hubei, Jiangsu, Sichuan and Heilongjiang were below
 242 the national mean.



243

244

(a)

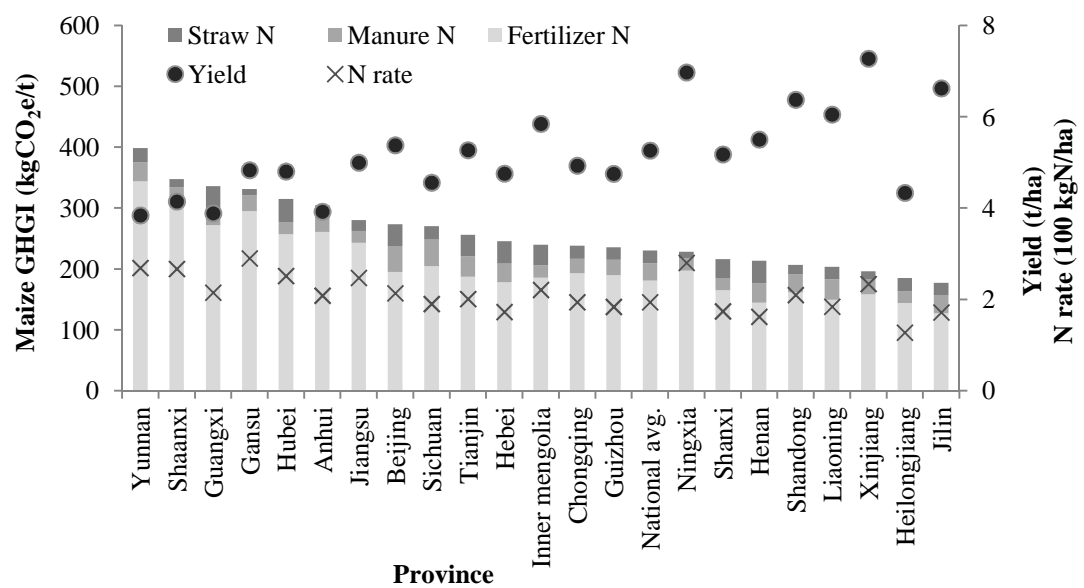


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246

247

(b)

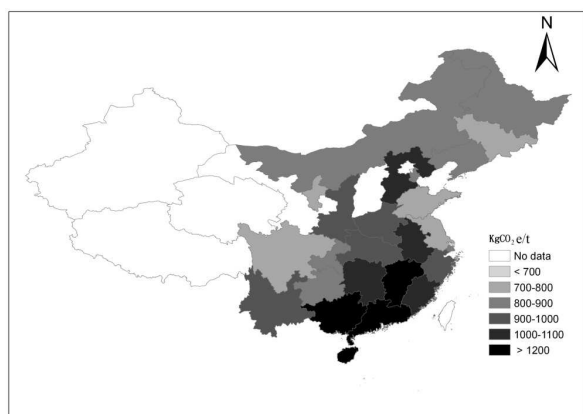


248

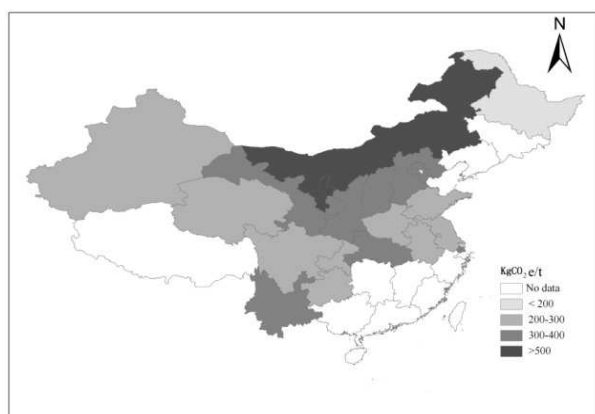
249

(c)

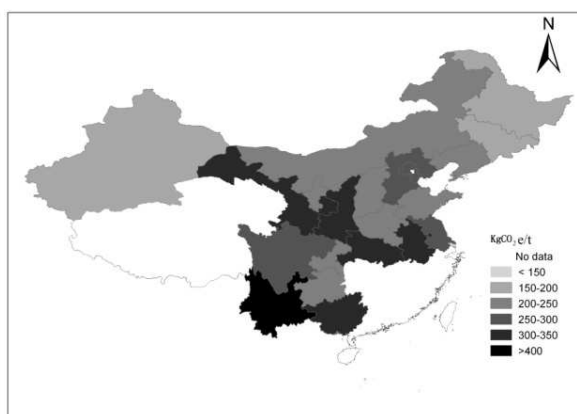
250 **Fig.1.** GHGI of rice (a), wheat (b) and maize (c) production in different provinces in 2006



(a)



(b)



(c)

251
252

253
254

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

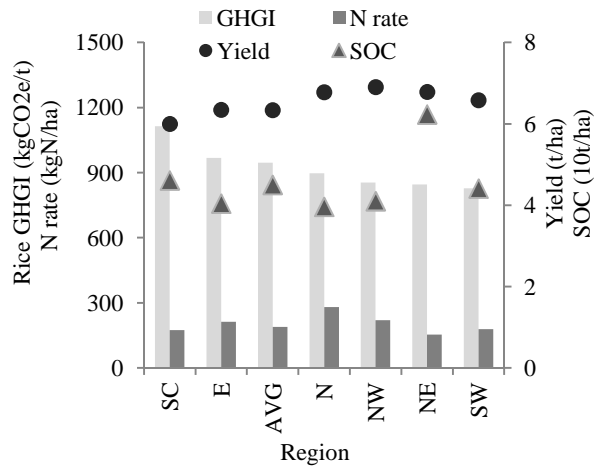
256 3.2. GHGI of wheat and maize production

257 The national average GHGI of wheat (Fig. 1b) and maize (Fig. 1c) for 2006 production were 265
 258 kgCO₂e/t and 230 kgCO₂e/t, respectively. Large spatial variability can be observed among provinces.
 259 For example, producing one ton of wheat in Ningxia emitted 3 times more N₂O than in Heilongjiang,
 260 attributable to significant differences in synthetic N input and wheat and maize yields between Chinese
 261 provinces. In general, synthetic N fertilizer made up at least 70% of total emissions and was therefore
 262 the primary emission contributor. Fig. 1 also shows that the trends of GHGI, which are affected by
 263 place-specific yield levels, were not necessarily consistent with those of per hectare N application rates.
 264 For instance, although the N application rate for maize in Ningxia (280 kgN/ha) was 30% higher than
 265 in Guangxi (215 kgN/ha), a much higher yield in Ningxia (6.97t/ha) than in Guangxi (3.88 t/ha) results
 266 in a lower maize GHGI in Ningxia. In contrast, a high N rate and low wheat productivity made
 267 Ningxia the most carbon intensive province for wheat cultivation.

268 The geographic variations of GHG emissions per ton of wheat (Fig. 2b) and maize (Fig. 2c) show
269 both similarities and differences. In general, similar levels of GHGI can be observed for wheat and
270 maize production (except for Ningxia); e.g. Yunnan was one of the most carbon intensive areas for
271 both wheat and maize production in 2006. The levels of maize GHGI converged to the range of
272 200-300 kgCO₂e/t, with obvious correlation with N rates and yields. Provincial discrepancies were
273 more evident for wheat GHGI. Among the five major wheat producing areas - Henan, Shandong,
274 Hebei, Anhui and Jiangsu, which contributed about 73% of the national production, GHGI levels in
275 Hebei and Jiangsu were superior to the national average. Among the major maize producing areas,
276 only Hebei had a higher GHGI than the national mean, while Jilin, Shandong, Henan and Heilongjiang
277 were lower.

278 3.3. Implications for regional GHGI reduction strategies

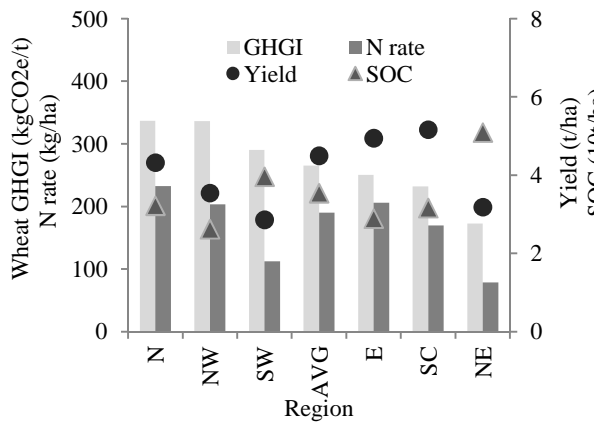
279 The GHGI, yield and synthetic N rate of rice, wheat and maize cultivation as well as the SOC content
280 at the regional scale in 2006 are illustrated in Fig. 3. In general, the southwest had lowest cereal yields,
281 albeit second highest SOC after the northeast. Conversely more N fertilizers were added to croplands
282 in northwest provinces to compensate poor soil fertility, resulting in elevated regional GHGI of crop
283 production. Fig. 3 reveals that yield levels do not necessarily correspond to local SOC status, since
284 productivity is also influenced by climate, precipitation and other factors. In this regard, regional
285 strategies to minimize GHGI and improve soil fertility should accommodate local climatic, soil and
286 water conditions and management practices. For example, in the northwest measures improving SOC
287 density (e.g. conservation tillage) should be favored to enhance soil fertility and land productivity. In
288 intensive cropping systems in east and north China where over-fertilization is prominent, more
289 efficient use of N fertilizer can allow N rates to be cut by 30 to 60% without sacrificing crop yields (Ju
290 et al. 2009). Although the northeast was the least carbon intensive region in cereal production, this
291 came at the expense of net carbon losses, especially in Heilongjiang Province (Pan et al. 2010; Yu et al.
292 2012), thus calling for better management practices to sustain soil fertility in this region.



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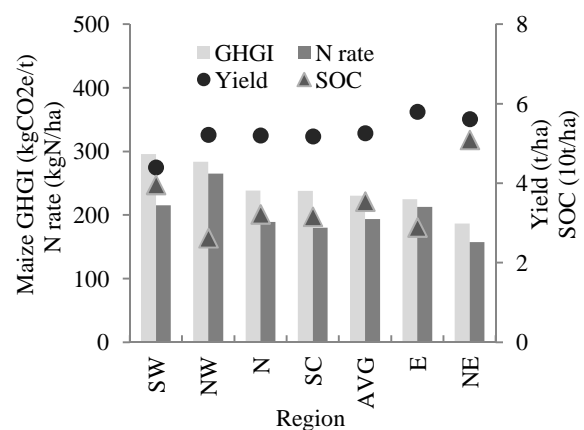
(a)



295

296

(b)



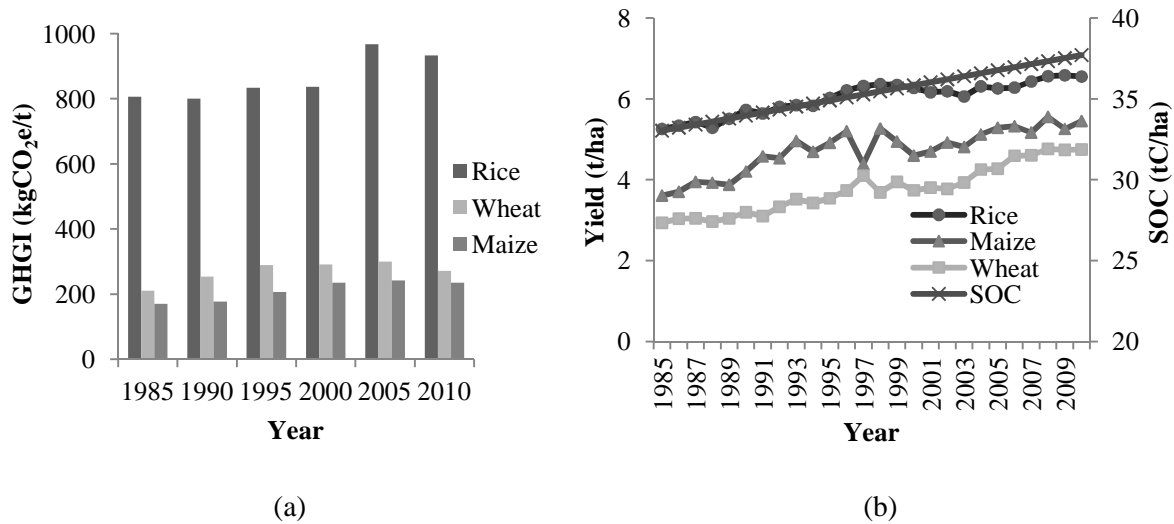
(c)

297 **Fig.3.** GHGI of rice (a), wheat (b) and maize (c) production in different regions in 2006 and its
 298 relationship with yield, N rates and SOC content. NE, N, NW, E, SC, SW and AVG refer to northeast,
 299 north, northwest, east, south and central, southwest China, and national average, respectively.

300 3.4. Historical trends of regional GHGI of cereal production

301 Fig. 4a shows that national GHGI of rice production evolved at a different way to those of wheat and
 302 maize production, and the latter has always been the least carbon intensive of the three crops. Rice
 303 GHGI saw little variation between 1985 and 2000, which can be explained by nearly the same rate of
 304 growth in the CH₄ flux, yield (Fig. 4b) as well as the N application rate over this period. However,
 305 when rice yield reached a periodic peak in 1998 the CH₄ flux continued to climb, resulting in a sharp
 306 rise in GHGI in the first decade of the 21st century. Wheat and maize GHGIs had been steadily

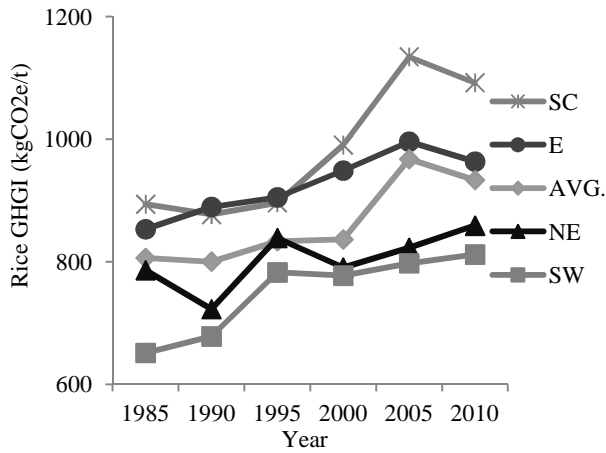
307 increasing from 1985 to 2000 since the growth rate of N application exceeded the rate of yield
 308 improvement. The GHGI began to stabilize or even decrease after 2000 as the combined effects of
 309 increasing yields, albeit at a lower rate, and a stabilized synthetic N rate promoted by the national “Soil
 310 testing and fertilizer recommendation program” (MOA 2005) initiated in 2005. At the national level,
 311 some studies (e.g. Pan et al. 2009) suggest a positive correlation between SOC improvement and
 312 cereal productivity increase (Fig. 4b).



313 **Fig.4.** Historical trends of national average GHGI (a) and yield (b) of rice, wheat and maize production

314 Fig. 5 illustrates that nearly all regional GHGI of rice(a), wheat(b) and maize(c) production
 315 reached a higher level in 2010 relative to 1985. For rice production (Fig. 5a), south and central and
 316 east regions have consistently been the most carbon intensive areas due to high temperature and
 317 greater level of organic matter application (Zhang et al. 2011a). In parallel, rice paddies in eastern,
 318 southern and central China are found to have experienced the greatest SOC increase (Zhang et al. 2007;
 319 Pan et al. 2010). In contrast, a lower level of crop residues, farm manure and green manure application
 320 enabled the southwest to emit least GHG in producing same amount of rice.
 321

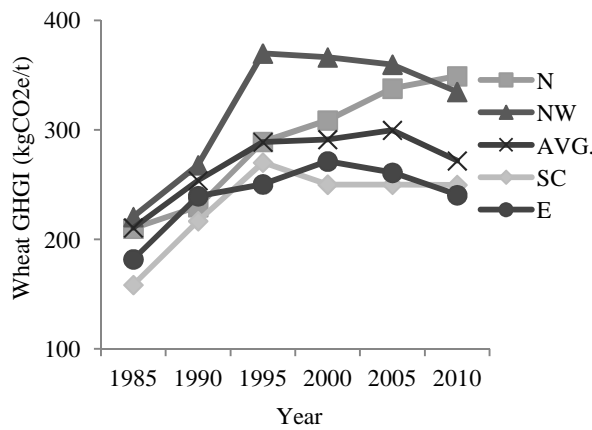
322 As to the GHGI of wheat production (Fig. 5b), all regions except north China exhibited the same
 323 trend as the national average. Consequently, reducing N rates should be advocated in northern
 324 provinces, confirming the findings of other experimental and theoretical studies (Ju et al. 2009, 2011).
 325 Maize GHGI evolution patterns (Fig. 5c) were more diverse between geographic regions, with
 326 northeast China having the lowest GHGI. The northwest has been characterized with the highest
 327 GHGI in both wheat and maize production.
 328



329

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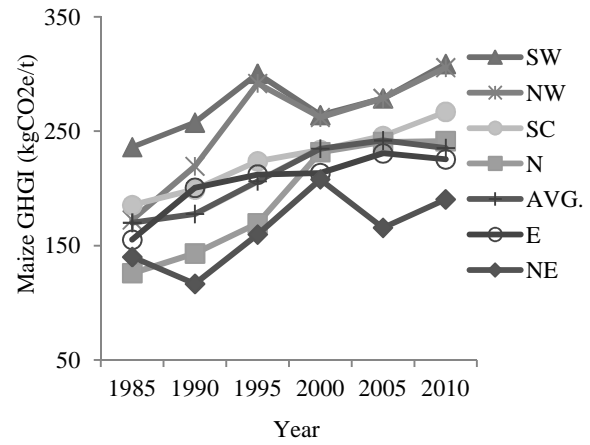
(a)



331

332

(b)



331

332

(c)

333 **Fig.5.** Historic evolution of regional GHGI of rice (a), wheat (b) and maize(c) production

334 3.5. Ways to improve GHGI of cereal production while safeguarding food security

335 Over the past 50 years, food production growth in China has been primarily driven by increasing yield
 336 per unit area rather than the expansion of cropping area. For example, from 1961 to 2010 there was an
 337 8.5-fold increase in wheat productivity, with only a 30% increase in total cereal cropping area (FAO
 338 2013). Ensuring food security in China in the future will still rely on yield improvement since rapid
 339 industrialization and urbanization will continue to encroach on China's arable land (UNDP 2013). Fan
 340 et al. (2012) argue that extension of existing technologies or better agricultural practices comprise the
 341 most effective near-term strategy for achieving higher resource (fertilizers and water) use efficiency,
 342 improving crop productivity and alleviating environmental impacts. In the longer term, continued

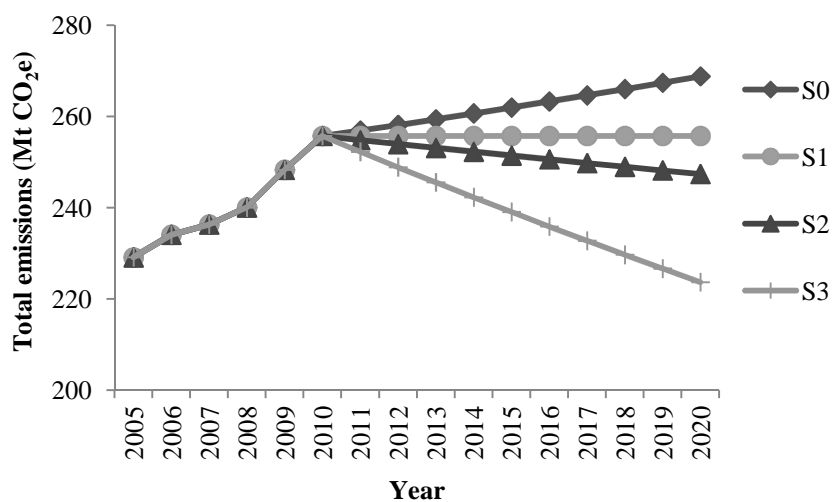
343 genetic improvement through plant breeding will be crucial to ensure future food security.

344 Integrated soil-crop management systems and better nutrient management techniques are
345 advocated to address the key constraints to yield improvement (Fan et al. 2012; Zhang et al. 2012a).
346 Extensive overuse of synthetic N fertilizers is well documented in China (Chen et al. 2011; Cui et al.
347 2010), resulting in significant losses and serious environmental externalities (Guo et al. 2010). Zhang
348 et al. (2013) suggest a possible 42% nationwide cut of N fertilizer use applying the balance concept to
349 equalize N input and above ground N removal. In parallel to optimum quantity, application time, right
350 placement and appropriate product are also essential to better nutrient management. Postponing N
351 application to a later stage of crop growth and popularizing fertilizer deep placement by using
352 appropriate machines for top-dressing could improve crop N uptake and minimize losses compared
353 with conventional practices of applying large amount of N fertilizer on the surface before planting or
354 at the early stages (Cui et al. 2008; Zhang et al. 2011b). Replacing a proportion of ammonium-based
355 fertilizers with nitrate-based fertilizers in places where denitrification dominates N₂O generation can
356 help minimize N₂O emissions and ammonia losses (Zhang et al. 2013). NUE can also be improved by
357 applying fertilizers added with nitrification inhibitors (NI) and/or urease inhibitors (UI) and slow- and
358 controlled-released fertilizers. Global meta-analysis results (Akiyama et al. 2010) suggest that NIs
359 addition can lower N₂O emission by 34% in upland fields and 30% in rice paddies on average,
360 compared with those of conventional fertilizers.

361 Better recycling of organic manures including animal excreta, crop residues and green manure
362 enables further improvement in NUE, SOC content and land productivity. Adopting conservation
363 tillage is found to be conducive to accumulate SOC density, improve water availability and reduce
364 water and wind erosion, especially on land of poor productivity (Xu et al. 2007; He et al. 2010). Such
365 practices shall be extended to wider areas supported by the MOA (2009). Finally, biochar addition can
366 be beneficial to soil quality and yield increase (Zhang et al. 2012b), therefore offering substantial
367 mitigation potential when it becomes economically available. As to CH₄ emissions from rice paddies,
368 upgrading irrigation regimes from mid-season drainage (F-D-F), currently being practiced in most rice
369 cultivation regions, to intermittent irrigation (F-D-F-M) or controlled irrigation, could avoid as much
370 as 1.256 CO₂e per hectare according to nationwide meta-analysis results (Wang et al. 2014).

371 3.6. Implication for mitigation potential from cereal production

372 Fig. 6 illustrates that total GHG emissions from rice, wheat and maize production have grown by 12%
 373 from 2005 to 2010 caused by an 11% increase in cropping area and a 5% increase in average yield (Fig.
 374 3b). In the S0 baseline scenario, although yields improve at the same rate of increase in N inputs,
 375 resulting in constant GHGI, total GHG emissions will still go up because of higher production levels.
 376 However, if no more N input is needed to enhance yields, emissions will stop increasing (scenario S1)
 377 and GHGIs will decrease. In contrast, if better fertilization practices are promoted to suppress the
 378 overuse of N fertilizers, total emissions will decline (scenario S2) by 8% compared to S0. Scenario S3
 379 assumes substantial efforts are dedicated to minimizing the GHGI of cereal production by eradicating
 380 N over-application, adopting better water management in rice paddies and improving yield levels. In
 381 this case, I_{N_2O} of rice, wheat and maize shall decline by 2.5%, 3% and 3.4% respectively, and I_{CH_4} by
 382 1.5% annually. Under this scenario, total GHG emissions are estimated to be 224MtCO₂e, a 17%
 383 decrease relative to S0 enabled by an 18% decrease in N input, 0.5-1.5% improvement in yields and 1%
 384 cut in average CH₄ flux. Such a mitigation scenario is feasible since the 18% cut in N use falls under
 385 the lower range of suggested 30-60% reduction (Ju et al. 2012; Zhang et al. 2013) and the 546 Mt
 386 cereal production meets the target for ensuring national food security.



387

388 **Fig.6.** GHG emission scenarios from rice, wheat and maize production to 2020 in China

389 Apart from the emission reduction potential, SOC density is projected to continue to increase at a
 390 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.
 391 2013). This implies that even the C inputs (including manure and crop residue) to Chinese croplands
 392 remain unchanged with no improvement in tillage practices, aggregate national SOC stocks will still
 393 increase over the period 2010-2020. If improved agricultural management practices are widely adopted,

394 as much as 70MtCO₂ could be sequestered in the cropland soils. Carbon sequestration is therefore
395 able to compensate 31% of GHG emissions under scenarios S3.

396 **4. Conclusions**

397 A low carbon development pathway implies minimization of emissions while increasing food
398 production and GHGI is an indicator combining both objectives. As such it is a central element of any
399 definition of sustainable intensification (Godfray and Garnett 2104). Our results on the GHGI of rice,
400 wheat and maize production show substantial heterogeneities among provinces/regions and indicate
401 considerable scope for improving carbon performance of cereal production. Under the BAU scenarios
402 where food production must grow to meet the demand of about 1.45 billion population, total GHG
403 emissions will continue to increase albeit with constant GHGIs. Controlling GHG emissions from
404 arable land thus requires additional mitigation efforts. Many abatement practices that improve crop
405 yields will not only enable emission reductions but also improve soil fertility via carbon sequestration,
406 therefore providing a triple win. Such findings can inform a broad range of policy, practitioner and
407 investment discussions on GHG mitigation strategies, and can also serve as benchmark values for
408 allocating quotas or as the baseline for generating carbon credits for any market-based mechanism.

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

414

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