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CLIMATE CHANGE AND AGRICULTURE RESEARCH PAPER

Carbon footprint of crop production in China: an analysis of National Statistics data

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SUMMARY

Assessing carbon footprint (CF) of crop production in a whole crop life-cycle could provide insights into the contribution of crop production to climate change and help to identify possible greenhouse gas (GHG) mitigation options. In the current study, data for the major crops of China were collected from the national statistical archive on cultivation area, yield, application rates of fertilizer, pesticide, diesel, plastic film, irrigated water, etc. The CF of direct and indirect carbon emissions associated with or caused by these agricultural inputs was quantified with published emission factors. In general, paddy rice, wheat, maize and soybean of China had mean CFs of 2472, 794, 781 and 222 kg carbon equivalent (CE)/ha, and 0·37, 0·14, 0·12 and 0·10 kg CE/kg product, respectively. For dry crops (i.e. those grown without flooding the fields: wheat, maize and soybean), 0·78 of the total CFs was contributed by nitrogen (N) fertilizer use, including both direct soil nitrous oxide (N_2O) emission and indirect emissions from N fertilizer manufacture. Meanwhile, direct methane (CH_4) emissions contributed 0·69 on average to the total CFs of flooded paddy rice. Moreover, the difference in N fertilizer application rates explained 0·86–0·93 of the provincial variations of dry crop CFs while that in CH_4 emissions could explain 0·85 of the provincial variation of paddy rice CFs. When a 30% reduction in N fertilization was considered, a potential reduction in GHGs of 60 megatonne (Mt) carbon dioxide equivalent from production of these crops was projected. The current work highlights opportunities to gain GHG emission reduction in production of crops associated with good management practices in China.

INTRODUCTION

Rapidly increasing anthropogenic greenhouse gas (GHG) emissions are making significant contribution to global climate change. As reported by IPCC (2007), the global mean temperature increased by 0·74 °C from 1906 to 2005; specifically, the linear warming trend over the 50 years was 0·13 °C per decade, which is nearly twice the rate for the last 100 years. Thus, concerns about reducing GHG emissions to mitigate climate change have recently inspired the assessment of carbon footprint (CF) for various activities and products. Carbon footprint, defined as a measure of the total amount of carbon dioxide (CO_2) emissions caused directly and indirectly by an activity or emitted over the life-cycle of a product, is analysed to measure the climate change impact of products and activities in terms of the amount of GHGs emitted in a whole life-cycle. To identify the climate change impact of a production sector and to develop climate-friendly policies in consumption, CF analysis and assessment has increasingly been concerned with, and widely applied in, agriculture (Ponsioen & Blonk 2012), industrial production (Virtanen *et al*. 2011), food consumption (Shirley *et al*. 2012) and international trading (Hertwich & Peters 2009; Peters 2010; Peters *et al*. 2011).

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In 2009, Hertwich & Peters found that global food production and consumption accounted for 0·20 of total anthropogenic GHG emissions (Hertwich & Peters 2009). As a primary food producer, GHG emissions from agriculture contribute 0·12–0·17 to global GHG emissions: this figure is increasing (Smith *et al.* 2007) and is likely to increase further in the coming decades (Smith & Gregory 2013). Growing interest in GHG mitigation in agriculture has provided a strong enticement to assess the CF for production of different crops. The CF of crop production could be quantified by taking into account the overall GHGs emissions from agricultural inputs used for crop production and protection and farm machinery in a single whole cycle of crop production (Adler *et al.* 2007). Carbon footprints have been evaluated for staple crops (Hillier *et al.* 2009), under different tillage practices (West & Marland 2002) and farming systems (Dubey & Lal 2009) as well as for a bulk sector of crop production in China (Cheng *et al.* 2011). The mean CF of major crops from the UK was reported as 351·7 kg carbon equivalent (CE)/ha/year, 0·75 of which was contributed by nitrogen (N) fertilizer use (Hillier *et al.* 2009). Dubey & Lal (2009) reported a higher CF for agricultural productions in Punjab, India (22·86 g CE/kg biomass) than that of Ohio in the USA (9·52 g CE/kg biomass). For agricultural practice, the study by West & Marland (2002) showed a lower CF in crop production under no-tillage than under conventional tillage in the USA.

China has committed to reduce the emissions per unit of gross domestic production by 25% and implemented a national climate change policy, which demands GHG emission reductions in all sectors. Greenhouse gas emissions from China's agriculture industry have been estimated at 819·97 megatonne (Mt) CO_2 equivalents (CO_2-eq), accounting for 0·11 of the nation's total emissions in 2005 (NDRC 2012). Characterizing the CF of crop production and the contribution by individual agricultural inputs is fundamental for identifying the key measures and their potentials to mitigate climate change in China's agriculture. In a previous study, a bulk CF of China's crop production sector was estimated at 0·78 t CE/ha/year, with N fertilizer-induced GHG emissions contributing an average proportion of 0·65 (Cheng *et al.* 2011). However, estimating crop and regional-specific CFs would be important for understanding the crop and regional variation and identifying crop and regional-specific mitigation options. The CF information will be critical for policy decisions about how to develop climate-friendly incentives for food consumption and domestic trading between regions as well as to improve management practices.

Paddy rice, wheat, maize and soybean are the major staple crops in China, accounting for almost 0·86 of total arable land area (FAO 2012). In order to meet the food demand by the large and ever-increasing population, production of these crops has been achieved with maximum available input of resources to reach a maximum yield, although the total cropland area has been decreasing since the end of the 20th century (Huang 2013). Given that determining a CF of a certain crop is useful for identifying how a low-carbon economy could be implemented to abate climate change, the current study aimed to quantify the CFs of the four staple crops of China. Special attention has been given to the differences in both total CF and contribution by individual inputs between the crops and the regions. In addition, the current study also evaluates GHG mitigation potentials and key measures based on CF reductions.

MATERIALS AND METHODS

Carbon footprint quantification protocol

There have been various versions of the CF concept in the literature (Wiedema *et al.* 2008; Dubey & Lal 2009), although there is no one single definition widely accepted in the CF analysis (Peters 2010). However, CF of crop production was generally assessed by taking into account all the GHG emissions caused by or associated with material used, farm machinery operated, and irrigation and drainage power expended for crop production in a crop's life-cycle (Lal 2004; Hillier *et al.* 2009; Cheng *et al.* 2011). Greenhouse gas emissions from manufacturing of agricultural inputs such as fertilizer, pesticides, herbicides and irrigation, as well as farm machinery used for spraying and tillage, harvesting, packing and transportation were considered for CF calculation of a single crop season, i.e. up to harvest in the current study. In addition, direct nitrous oxide (N_2O) emissions from soil induced by chemical fertilizer-N application and methane (CH_4) emissions from rice paddy were also taken into account. While there has been increasing concern regarding CO_2 emissions due to land use change in CF assessment (Cederberg *et al.* 2011; Schmidinger & Stehfest 2012), potential soil carbon stock changes were not quantified in the current study due to a lack of available data relevant to a specific crop production.

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Carbon footprints calculated in the current study were expressed in CEs.

Sources of emissions and calculation

The first GHG emissions calculated were those induced by manufacturing (E_M, kg CE/ha) of individual inputs used for crop production, protection and management including fertilizer, pesticides, plastic film and diesel oil for machinery operation, using Eqn (1):

$$E_M = \sum AI \times EF \tag{1}$$

where *AI* denotes the amount of an agricultural input (kg/ha) and *EF* is emission factor of manufacturing a unit of the input material (kg CE/kg), which have been reported in previous studies (Cheng *et al.* 2011).

Secondly, the direct N_2O emissions (E_{N_2O}, kg CE/ha) induced by chemical fertilizer-N was estimated using Eqn (2):

$$E_{N_2O} = N \times EF_{N_2O} \times \frac{44}{28} \times 298 \times \frac{12}{44} \tag{2}$$

where *N* represents the chemical fertilizer-N application rate (kg N/ha); EF_{N_2O} is the emission factor of N_2O emission induced by N fertilizer application (kg N_2O–N/kg N fertilizer); 44/28 is the molecular weight of N_2 in relation to N_2O; 298 is net global warming potential (GWP) of N_2O in a 100-year horizon and 12/44 is the molecular weight of CO_2 in relation to CE. According to IPCC (2006), EF_{N_2O} is estimated as 0·01 for dry cropland and 0·003 for submerged rice paddy.

Thirdly, the annual CH_4 emissions from rice paddies were determined for the rice CF calculation. Here the region-specific emission factors (E_{CH_4}, kg CE/ha) reported by Yan *et al.* (2003) were adopted.

The fourth consideration was the energy used for irrigation, which would be one of the main GHG emission sources in farm operations. The main components of energy use associated with irrigation are related to pumping water to the fields, generally using either diesel or electricity (Wang *et al.* 2012). In general, GHG emissions induced by irrigation (E_{IRRI}, kg CE/ha) could be calculated using the approach developed by Wang *et al.* (2012):

$$E_{IRRI} = IR_{ij} \times EF_j \tag{3}$$

where IR_{ij} represents the amount of water irrigated for crop *i* in region *j* (m^3/ha) and EF_j is the emission factor of irrigation for region *j* (kg CO_2-eq/m^3). The GHG emission factor for irrigation in each province of China has been calculated: the details of this calculation were presented in the study of Wang *et al.* (2012).

Overall, total CF of the crop in a single cropping season in terms of land used (CF_A, kg CE/ha) was assessed by summarizing all of the individual GHG emissions mentioned above:

$$CF_A = E_M + E_{N_2O} + E_{CH_4} + E_{IRRI} \tag{4}$$

With the estimated CF_A, CF in terms of crop production (CF_Y, kg CE/kg product) (carbon intensity in other words) was evaluated using Eqn (4):

$$CF_Y = \frac{CF_A}{Y} \tag{5}$$

where *Y* denotes the yield of a given crop (kg/ha).

Data sources

Input data for crop production were collected from the national yearbook, which reported crop yields and cropland area, as well as the amounts of various agricultural inputs including fertilizer, pesticides, agricultural plastic film, irrigated water and diesel of the four crop production systems (rice, wheat, maize and soybean) in mainland China and each province of China in 2011. The original data are available from the archives (DP-NDRC 2011; SBS-DRSES 2011; WC-YCWR 2011).

RESULTS

Carbon footprint of the crop production

The CFs both in terms of land used and yield produced for the four major staple crops of China were calculated using Eqns (4) and (5), respectively (Table 1). The highest annual CF was observed for rice production (2472 kg CE/ha cropland and 0·37 kg CE/kg grain), and the lowest for soybean (222 kg CE/ha and 0·1 kg CE/kg product). Wheat has CFs of 794 kg CE/ha cropland and 0·14 kg CE/kg grain, which is higher than maize (781 kg CE/ha and 0·12 kg CE/kg grain).

Contributions of individual inputs

The proportions of different inputs into CF were calculated to assess the contribution of each input to total CF (Fig. 1). For rice, most (0·69) of the CF was derived from CH_4 emissions followed by N fertilizer use (0·16) and irrigation (0·07), whereas a minor contribution (0·08) was seen by all the other inputs and farming operations in rice production.

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Table 1. *Comparison of carbon footprint between different crops in China*

Crop	Area (Mha)	Yield (kg/ha)	CF per unit area (kg CE/ha)	CF per unit of production (kg CE/kg grain)	Overall GHG emissions (Mt CO_2-eq)
Rice	29·87	6716·25	2472·34	0·37	270·81
Wheat	24·26	5550·30	793·68	0·14	70·59
Maize	32·50	6791·10	781·17	0·12	93·09
Soybean	8·52	2220·45	221·91	0·10	6·93

CF: carbon footprint; CE: carbon equivalent; GHG: greenhouse gas.

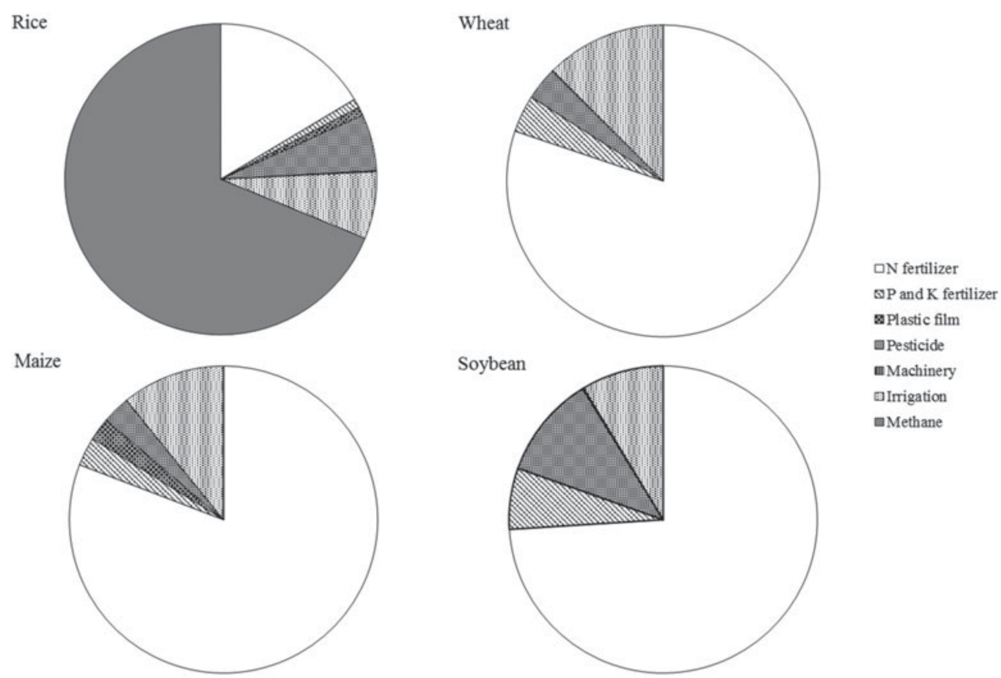

Fig. 1. Contributions of agricultural inputs to total carbon footprint of four crops.

For wheat, maize and soybean, however, N fertilizer contributed 0·80, 0·81 and 0·74, respectively, to the total CF, despite the N fertilizer application rate for soybean being about 0·25 of that for maize and wheat. Irrigation was the second biggest contributor (0·12 and 0·11 in wheat and maize, respectively). A minor contribution (0·03) was derived from pesticide use for wheat and maize production. The contribution of pesticide and irrigation was 0·11 and 0·09, respectively, to total CF.

As shown in Table 2, the regions surveyed were representative of 0·96–0·99 of total cropland, which showed them to be representative of China's crop production. If use of N fertilizers was reduced by 30%, the corresponding reduction in CF could be 6·53, 24·34 and 25·52% for rice, wheat and maize production, respectively.

Regional variations of carbon footprints of the four crops

Carbon footprints of four crops were calculated in different provinces of China (Fig. 2). The calculated province CF ranged from 0·14 to 0·48, 0·06 to 0·26, 0·07 to 0·18 and 0·02 to 0·18 kg CE/kg product for rice, wheat, maize and soybean, respectively. The highest CFs of rice, wheat and maize were observed in Yunnan province, with values of 0·48, 0·26 and 0·18 kg CE/kg grain; however, Heilongjiang province had the lowest CF for wheat and maize production (0·06 and 0·07 kg CE/kg grain), and the lowest CF for rice production (0·14 kg CE/kg grain) was found in Jilin province. The highest and lowest CFs in soybean production were observed in the provinces of Shaanxi and Chongqing (0·18 and 0·02 kg CE/kg product, respectively).

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Table 2. *Reduction in carbon footprint and greenhouse gas (GHG) emissions in surveyed regions when nitrogen application is reduced by 30%*

Crop	Number of surveyed provinces	Area (proportion of total)	CF_Y reduction (mean ± S.D.; %)	GHG emission Reduction (Mt CO_2-eq)
Rice	23	0·99	6·5 ± 2·06	18·23
Wheat	15	0·96	24·3 ± 2·02	16·85
Maize	20	0·97	25·5 ± 2·37	22·38

CF_Y, carbon footprint in production.

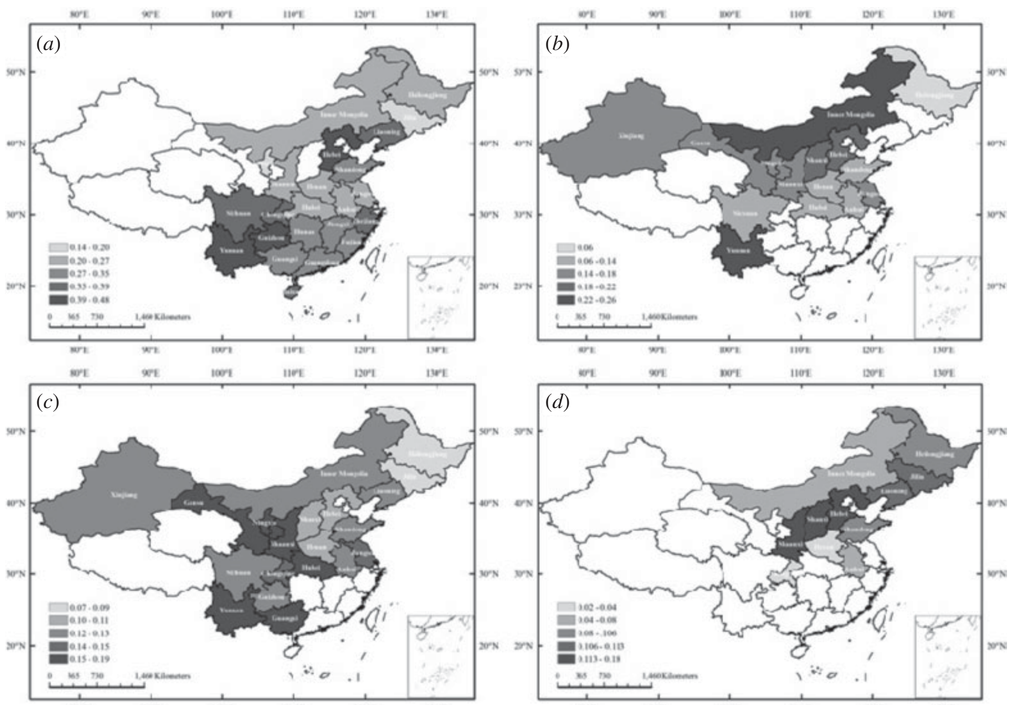

Fig. 2. Regional variation in carbon footprint of crops (*a*) rice; (*b*) wheat; (*c*) maize; (*d*) soybean (unit of carbon footprint is kg CE/kg grain).

Most of the CF in paddy rice production was derived from CH_4 emissions followed by N fertilizer use. For regional variation, the contribution of CH_4 emissions varied from 0·29 in Heilongjiang to 0·81 in Chongqing, and N fertilizer contributions varied from 0·10 in Guizhou to 0·37 in Ningxia. Furthermore, CH_4 emissions could explain 0·69 of the variation in the CF across provinces for paddy rice production (Fig. 3).

For the contributions of various agricultural inputs to total CF of wheat, the contribution of N fertilizer varied from 0·67 in Shanxi to 0·92 in Heilongjiang. Meanwhile, 0·86 to 0·93 of the variation in the CF across provinces for dry crop production could be explained by N fertilizer application (Fig. 3). In addition, the partial factor productivity from applied N (PFP_N) varied from 14·16 to 51·87 kg/kg N in different regions of China (Fig. 4). Irrigation also made significant contributions, varying from 0·00 in Heilongjiang to 0·27 in Hebei. For maize, N fertilizer made the largest contribution, from 0·67 in Gansu to 0·96 in Guangxi, and the contribution of irrigation varied from 0·00 in Hubei, Guangxi, Chongqing and Guizhou to 0·22 in Shanxi. For soybean, the largest contribution was found for N fertilizer in each province, which varied from 0·60 in Henan to 0·95 in Shaanxi. Irrigation also made a significant contribution, varying from 0·00 in Chongqing to 0·30 in Hebei.

DISCUSSION

Greenhouse gas emission cost of crop production in China

According to the Second National Climate Change Assessment Report, GHG emissions in China have

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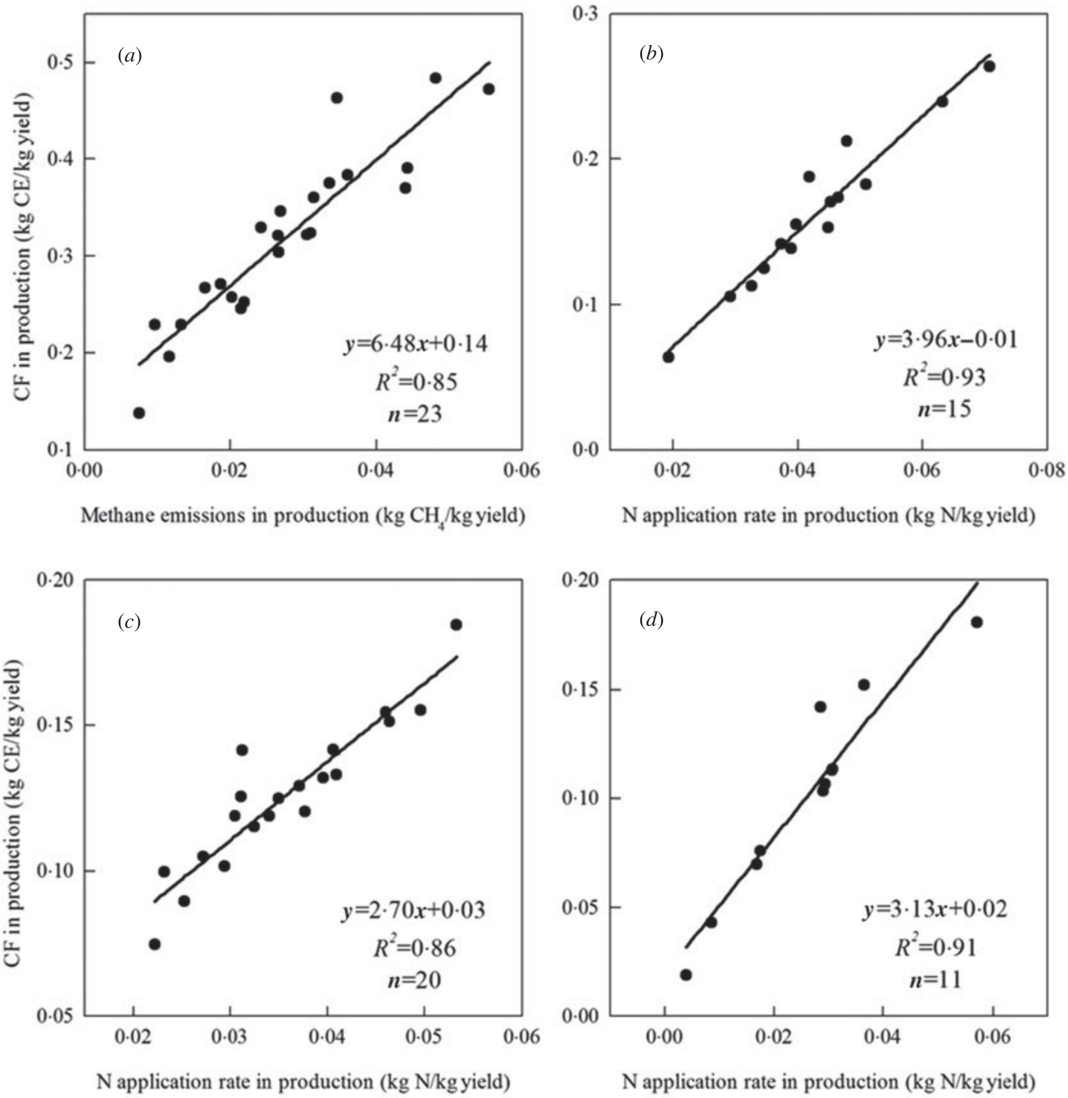

Fig. 3. Correlation of total carbon footprint (CF) with methane (CH_4) emissions and nitrogen (N) applications in (*a*) rice; (*b*) wheat; (*c*) maize; (*d*) soybean production.

been estimated at 7467 Mt CO_2-eq in 2005, which accounted for 0·11 of the total national GHG emissions (NDRC 2012). Total annual GHG emissions due to production of the four crops were estimated at 441·42 Mt CO_2-eq, which accounted for almost 0·06 of the nation's total emissions in 2005. This indicated that crop production played a significant role in China's GHG emissions. An area-weighted mean CF of dry crops (i.e. wheat, maize and soybean) was calculated as 713 kg CE/ha according to the current results, which showed a higher CF crop production in China than in the UK, without paddy rice (estimated at 351·7 kg CE/ha by Hillier *et al.* 2009). Specifically, CF of wheat in China was estimated in the current study at 794 kg CE/ha and 0·14 kg CE/kg grain, which seemed much higher than for the UK at 388 kg CE/ha (Hillier *et al.* 2009) and Canada at 0·09 kg CE/kg grain (Gan *et al.* 2012). Due to the N-fixing ability of legume crops, the N fertilizer application rate for soybean production was 54·42 kg N/ha: much lower than that for wheat and maize production (being 210·92 and 209·26 kg N/ha in China). The low N application rate resulted in a much lower CF for soybean production (222 kg CE/ha) compared to wheat (794 kg CE/ha) and maize (781 kg CE/ha). Hillier *et al.* (2009) also indicated a low CF in leguminous crop production compared to other crops; however, the CF of soybean in China was still much higher than the CF of legumes in the UK at 125 kg CE/ha (Hillier *et al.* 2009). Rice production in China had a CF of

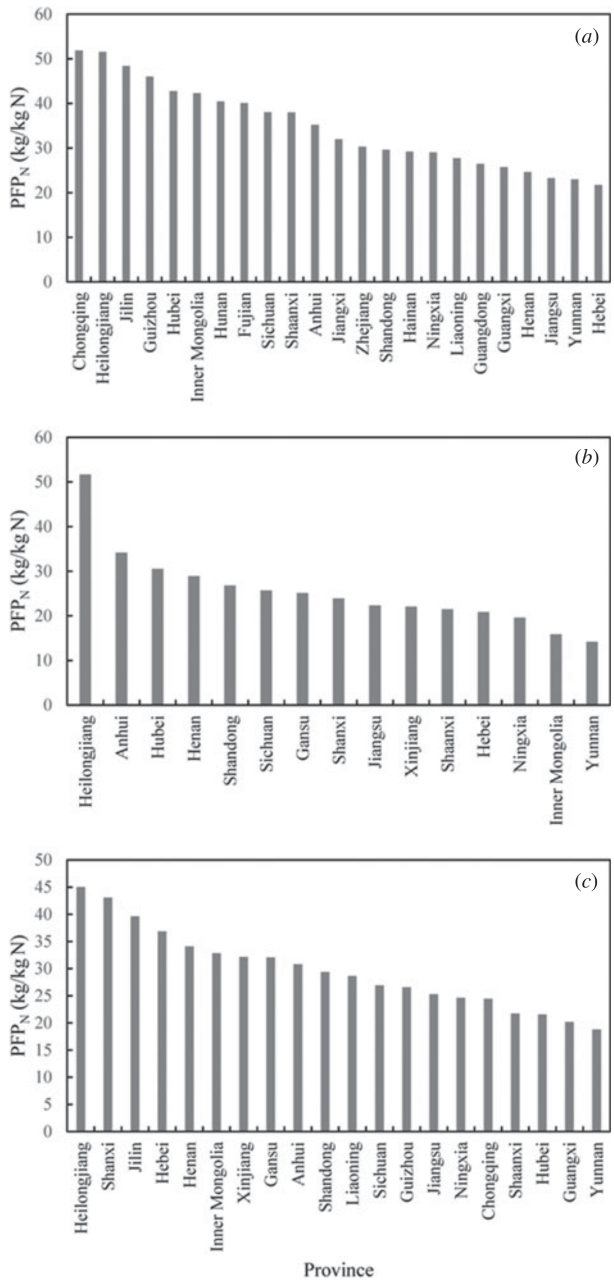

Fig. 4. Partial factor productivity from applied nitrogen (PFP_N) of (*a*) rice; (*b*) wheat; (*c*) maize production in different provinces of China.

0·37 kg CE/kg grain, close to the value estimated by Pathak *et al.* (2010) who reported a CF of 0·333 (rice) and 0·413 kg CE/kg grain (basmati rice) in India. The estimated crop-specific CF in the current study shows that China has a higher CF for dry crop production than the developed countries, which implies an opportunity to understand and formulate GHG mitigation strategies for farming management practices in China.

According to the results, 0·69 of rice CF was derived from CH_4 emissions, and a correlation of CH_4 emissions *v.* CF in different provinces had an R^2 of 0·85, demonstrating the sensitivity of CF to CH_4 emissions in rice production. Methane has been ranked as the second highest radiative force among the long-lived GHGs after CO_2, and rice paddies account for *c.* 0·32 of agricultural CH_4 emissions in China (Schimel 2000; Smith *et al.* 2007; NDRC 2012). The results of simulations by Zhang *et al.* (2012) showed that CH_4 emissions from irrigated rice cultivation in China will increase by 14% by 2040 compared with that in 2009, due to global warming and high crop residue retention, which is indicative of the important role of CH_4 emissions in the GHG mitigation of rice agriculture. As indicated in the current results, a reduction of 10% in CH_4 emissions could reduce the total CF by nearly 9·8%. To address this issue, proper field management, e.g. promoting water use efficiency and selecting rice varieties, should be developed for rice paddies in the future.

In 2012 it was reported by Xinhua net (2013) that the cultivation areas of rice, wheat and maize were increased but that of soybean decreased, especially in Heilongjiang. However, the total cropland area would expand in Northeast China due to warming. Maize cultivation has been forecast to extend in this area, which could result in soil organic carbon loss due to wetter soil becoming dry cropland. This must be accounted for in future assessments (Cederberg *et al.* 2011; Schmidinger & Stehfest 2012), particularly for Heilongjiang province. Hence, the changes of crop production structure will affect the GHG emissions induced by crop production in China in the future.

Nitrogen fertilizer induced emissions and the reduction potential

Averaging the contributions of N fertilizer to the total CF in the three dry crops, the relative mean proportion of 0·78 was close to the mean proportion of 0·75 in UK cropland (Hillier *et al.* 2009). China is currently the largest consumer of N fertilizers in the world. In the last 20 years (1992–2011), annual N fertilizer application increased from 17·56 to 23·81 Mt (a 36% increase) (SBS-DRSES 2011). Kahrl *et al.* (2010) calculated a specific emission factor range of 15–31 t CO_2-eq/t N for China's N fertilizer manufacturing and application, and implied that synthesized N fertilizer application could lead to total emissions of 400–840 Mt CO_2-e, equivalent to 0·08–0·16 of China's energy-related CO_2

emissions in 2005. In the current study, synthesized N fertilizer application in four crop production systems showed the total emissions of 181 Mt CO_2-e, equivalent to 0·20–0·50 of GHG emissions from China's N fertilizer manufacturing and application. Meanwhile, environmental problems related to overuse of N fertilizers have become more serious. For example, excess N entering water bodies can negatively affect the quality of drinking water and may cause eutrophication (Ju *et al.* 2009). Large inputs of synthetic N fertilizer could also lead to acid rain and soil acidification (Guo *et al.* 2010). Therefore, improving N use efficiency and avoiding overuse of synthetic N fertilizer would be a key issue for China's agriculture, not only for the sake of mitigation but also for the environmental quality of China.

The rapid increase of synthetic N fertilizer application resulted in a decrease of the PFP_N from 55 to 20 kg yield/kg N fertilizer from 1977 to 2005 (Ju *et al.* 2009). Dobermann (2005) stated that PFP_N with 40–70 kg/kg N is appropriate for N use in crop production; however, PFP_N of many provinces were much lower than the appropriate range suggested by Dobermann (2005). In addition, it has been suggested that China's agriculture has considerable potential to reduce existing synthetic N fertilizer applications by 30–50% without reducing crop production (Ju *et al.* 2009; Cheng *et al.* 2010). Given this option, CF reductions for four crops in different provinces were calculated on the basis of reducing N application rate by 30%. The reduction of CF under a 30% reduction in N fertilizers could total 57·46 Mt CO_2-eq in the surveyed regions of China. As indicated by Cheng *et al.* (2014) who simulated the GHG mitigation potentials for Chinese cropland by the DAYCENT ecosystem model, N saving could be a key option for cutting down crop-production-induced emissions. Thus, improving N use efficiency in crop production would be a key option for both saving farmers' cost for unit crop production and mitigating climate change. Some techniques and strategies have been developed recently to improve N use efficiency and reduce N fertilizer application. For example, recommended fertilization has been initiated with a national project to greatly reduce N use and to balance nutrients, mainly with combined organic/inorganic fertilization in China (MOA 2011). Biochar amendments in agricultural soils have been shown to reduce N loss through reducing N_2O emissions and nitrate leaching (Sohi *et al.* 2010; Zhang *et al.* 2012), which is beneficial for reducing N losses from agriculture.

Overall, the production of rice, wheat, maize and soybean in China shows carbon footprints of 0·37, 0·14, 0·12 and 0·10 kg CE/kg product, respectively, mainly due to N fertilizer inputs, and in particular CH_4 emissions from flooded paddy rice. There was a large regional variation for crop CF, mainly in N input rates in dry crop production and CH_4 emissions in rice production. Quantification of soil carbon change due to land used would be required to characterize the impact of crop cultivation structure change on overall CF of a crop in the future. Nevertheless, the present study highlights opportunities for GHG mitigation by improving N fertilizer use and reducing paddy rice CH_4 emissions of crop production in China. Finally, the current work may be relevant for policy with governmental incentives to achieve low CF in crop production and food consumption as well as regional trading in China's future economy.

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REFERENCES

Adler, R. A., Del Grosso, S. J. & Parton, W. J. (2007). Life-cycle assessment of net greenhouse-gas flux for bioenergy cropping systems. *Ecological Applications* **17**, 675–691.

Cederberg, C., Persson, U. M., Neovius, K., Molander, S. & Clift, R. (2011). Including carbon emissions from deforestation in the carbon footprint of Brazilian beef. *Environmental Science and Technology* **45**, 1773–1779.

Cheng, C., Shi, Y. & Wen, T. (2010). *The Real Cost of Nitrogen Fertilizer*. Beijing: Greenpeace. Available from: http://www.greenpeace.org/china/zh/publications/reports/food-agriculture/2010/cf-n-rpt/ (verified 5 June 2012).

Cheng, K., Pan, G., Smith, P., Luo, T., Li, L., Zheng, J., Zhang, X., Han, X. & Yan, M. (2011). Carbon footprint of China's crop production: an estimation using agro-statistics data over 1993–2007. *Agriculture, Ecosystems and Environment* **142**, 231–237.

Cheng, K., Ogle, S. M., Parton, W. J. & Pan, G. (2014). Simulating greenhouse gas mitigation potentials for Chinese croplands using the DAYCENT ecosystem model. *Global Change Biology* **20**, 948–962.

https://doi.org/10.1017/S0021859614000665

DP-NDRC (Department Of Price In National Development and Reform Commission Of China) (2011). *Compilation of the National Agricultural Costs and Returns*. Beijing: China Statistics Press.

Dobermann, A. (2005). Nitrogen use efficiency-state of the art. In *Proceedings of the IFA International Workshop on Enhanced-Efficiency Fertilizers, June 28–30 2005, Frankfurt, Germany*. pp. 1–18. Paris, France: International Fertilizer Industry Association.

Dubey, A. & Lal, R. (2009). Carbon footprint and sustainability of agricultural production systems in Punjab, India, and Ohio, USA. *Journal of Crop Improvement* **23**, 332–350.

FAO (2012). *FAOSTAT Database*. Rome: FAO. Available from: http://faostat.fao.org (verified June 2013).

Gan, Y., Liang, C., Campbell, C. A., Zentner, R. P., Lemke, R. L., Wang, H. & Yang, C. (2012). Carbon footprint of spring wheat in response to fallow frequency and soil carbon changes over 25 years on the semiarid Canadian prairie. *European Journal of Agronomy* **43**, 175–184.

Guo, J. H., Liu, X. J., Zhang, Y., Shen, J. L., Han, W. X., Zhang, W. F., Christie, P., Goulding, K. W. T., Vitousek, P. M. & Zhang, F. S. (2010). Significant acidification in major Chinese croplands. *Science* **327**, 1008–1010.

Hertwich, E. G. & Peters, G. P. (2009). Carbon footprint of nations: a global, trade-linked analysis. *Environmental Science and Technology* **43**, 6414–6420.

Hillier, J., Hawes, C., Squire, G., Hilton, A., Wale, S. & Smith, P. (2009). The carbon footprints of food crop production. *International Journal of Agricultural Sustainability* **7**, 107–118.

Huang, J. (2013). China's agricultural development in the new era: opportunities, challenges, and strategies. *Bulletin of Chinese Academy of Sciences* **28**, 295–300. (in Chinese with English summary)

IPCC (2006). *2006 IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme* (Eds H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara & K. Tanabe). Tokyo, Japan: IGES.

IPCC (2007). *Climate Change 2007: Synthesis Report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change* (Eds Core Writing Team, R. K. Pachauri & A. Reisinger). Geneva, Switzerland: IPCC.

Ju, X. T., Xing, G. X., Chen, X. P., Zhang, S. L., Zhang, L. J., Liu, X. J., Cui, Z. L., Yin, B., Christie, P., Zhu, Z. L. & Zhang, F. S. (2009). Reducing environmental risk by improving N management in intensive Chinese agricultural systems. *Proceedings of the National Academy of Sciences USA* **106**, 3041–3046.

Kahrl, F., Li, Y. J., Su, Y. F., Tennigkeit, T., Wilkes, A. & Xu, J. C. (2010). Greenhouse gas emissions from nitrogen fertilizer use in China. *Environmental Science and Policy* **13**, 688–694.

Lal, R. (2004). Carbon emission from farm operations. *Environment International* **30**, 981–990.

MOA (Ministry Of Agriculture Of China) (2011). *Work Program of Popularization Operation of Recommended Fertilization in 2011*. Beijing: MOA. Available from: http://www.moa.gov.cn/zwllm/tzgg/tz/201103/t20110310_1869341.htm. (verified 5 June 2012).

NDRC (National Development and Reform Commission Of China) (2012). *Second National Communication on Climate Change. The People's Republic of China*. Beijing: National Development and Reform Commission of China. Available from: http://nc.ccchina.gov.cn/WebSite/NationalCCC/UpFile/File116.pdf (verified 5 June 2012)

Pathak, H., Jain, N., Bhatia, A., Patel, J. & Aggarwal, P. K. (2010). Carbon footprints of Indian food items. *Agriculture, Ecosystems and Environment* **139**, 66–73.

Peters, G. P. (2010). Carbon footprints and embodied carbon at multiple scales. *Current Opinion in Environmental Sustainability* **2**, 245–250.

Peters, G. P., Minx, J. C., Weber, C. L. & Edenhofer, O. (2011). Growth in emission transfers via international trade from 1990 to 2008. *Proceedings of the National Academy of Sciences USA* **108**, 8903–8908.

Ponsioen, T. C. & Blonk, T. J. (2012). Calculating land use change in carbon footprints of agricultural products as an impact of current land use. *Journal of Cleaner Production* **28**, 120–126.

SBS-DRSES (State Bureau Of Statistics In Department Of Rural Social Economical Survey) (2011). *China Rural Statistical Yearbook*. Beijing, China: China Statistics Press.

Schimel, J. (2000). Rice, microbes and methane. *Nature* **403**, 375–377.

Schmidinger, K. & Stehfest, E. (2012). Including CO_2 implications of land occupation in LCAs-method and example for livestock products. *International Journal of Life Cycle Assessment* **17**, 962–972.

Shirley, R., Jones, C. & Kammen, D. (2012). A household carbon footprint calculator for islands: case study of the United States Virgin Islands. *Ecological Economics* **80**, 8–14.

Smith, P. & Gregory, P. J. (2013). Climate change and sustainable food production. *Proceedings of the Nutrition Society* **72**, 21–28.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, B. & Sirotenko, O. (2007). Agriculture. In *Climate Change 2007: Mitigation of Climate Change. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change* (Eds B. Metz, O. R. Davidson, P. R. Bosch, R. Dave & L. A. Meyer), pp. 497–540. Cambridge, UK: Cambridge University Press.

Sohi, S. P., Krull, E., Lopez-Capel, E. & Bol, R. (2010). A review of biochar and its use and function in soil. *Advances in Agronomy* **105**, 47–82.

Virtanen, Y., Kurppa, S., Saarinen, M., Katajajuuri, J. M., Usva, K., Mäenpää, I., Mäkelä, J., Grönroos, J. & Nissinen, A. (2011). Carbon footprint of food-approaches from national input–output statistics and a LCA of a food portion. *Journal of Cleaner Production* **19**, 1849–1856.

WANG, J., ROTHAUSEN, S. G. S. A., CONWAY, D., ZHANG, L., XIONG, W., HOLMAN, I. P. & LI, Y. (2012). China's water–energy nexus: greenhouse-gas emissions from groundwater use for agriculture. *Environmental Research Letters* **7**, 014035. doi: 10.1088/1748–9326/7/1/014035.

WEST, T. O. & MARLAND, G. (2002). Net carbon flux from agricultural ecosystems: methodology for full carbon cycle analyses. *Environmental Pollution* **116**, 439–444.

WIEDEMA, B. P., THRANE, M., CHRISTENSEN, P., SCHMIDT, J. & LØKKE, S. (2008). Carbon footprint: a catalyst for life cycle assessment? *Journal of Industrial Ecology* **12**, 3–6.

WC-YCWR (Writing Committee For Yearbook Of China Water Resources) (2011). *Yearbook of China Water Resources*. Beijing: China Water Conservancy & Hydropower Press.

Xinhua net (2013). *The Planting Area of Soybean in Heilongjiang was Decreased by 50% for the Eighth Consecutive Year*. Beijing, China: Xinhua net. Available from: http://www.hlj.xinhuanet.com/info/2013-11/13/c_132883640.htm (verified 13 November 2013). In Chinese.

YAN, X., CAI, Z., OHARA, T. & AKIMOTO, H. (2003). Methane emission from rice fields in mainland China: amount and seasonal and spatial distribution. *Journal of Geophysical Research: Atmospheres* **108**, 4505. doi: 10.1029/2002JD003182.

ZHANG, A., BIAN, R., PAN, G., CUI, L., HUSSAIN, Q., LI, L., ZHENG, J., ZHENG, J., ZHANG, X., HAN, X. & YU, X. (2012). Effects of biochar amendment on soil quality, crop yield and greenhouse gas emission in a Chinese rice paddy: a field study of 2 consecutive rice growing cycles. *Field Crops Research* **127**, 153–160.