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Carbon footprint of grain crop production in China – based on farm survey data

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ABSTRACT

Quantifying the carbon footprint of crop production can help identify key options to mitigate greenhouse gas emissions from agriculture. Using farm survey data from eastern China, the carbon footprints of three major grain crops (rice, wheat and maize) were assessed by quantifying the greenhouse gas emissions from individual inputs and farming operations with a full life cycle assessment methodology. The farm carbon footprint in terms of farm area was estimated to be 6.0 ± 0.1, 3.0 ± 0.2, and 2.3 ± 0.1 t CO_2-eq ha^{-1}, and the product carbon footprint in terms of grain produced was 0.80 ± 0.02, 0.66 ± 0.03, and 0.33 ± 0.02 t CO_2-eq t^{-1} grain for rice, wheat and maize, respectively. Use of synthetic nitrogen fertilizers contributed 44–79% and mechanical operations 8–15%, of the total carbon footprints. Irrigation and direct methane emission made a significant contribution by 19% and by 25%, on average respectively for rice production. However, irrigation was only responsible for 2–3% of the total carbon footprints in wheat and maize. The carbon footprints of wheat and maize production varied among climate regions, and this was explained largely by the differences in inputs of nitrogen fertilizers and mechanical operations to support crop management. Moreover, a significant decrease (22–28%) in the product carbon footprint both of wheat and maize was found in large sized farms, compared to smaller ones. This study demonstrated that carbon footprint of crop production could be affected by farm size and climate condition as well as crop management practices. Improving crop management practices by reducing nitrogen fertilizer use and developing large scaled farms with intensive farming could be strategic options to mitigate climate change in Chinese agriculture.

1. Introduction

Globally, the atmospheric concentrations of carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) have increased significantly, almost certainly as a consequence of anthropogenic activities since 1750 (IPCC, 2007, 2013). The increase in CO_2 emissions can be primarily attributed to fossil fuel combustion and land use change, while CH_4 and N_2O emissions have come mainly from agriculture (Smith et al., 2008). Thus the world agriculture sector has become increasingly important as a global solution to stabilize anthropogenic greenhouse gas (GHG) emissions. Quantifying carbon footprint (CF) has been widely accepted as an approach that can address the potential impact of production sectors or human activities on climate change, and can be assessed through characterizing the amount of greenhouse gas emissions "from cradle to grave" induced by a product or an activity based on the Life Cycle Assessment (LCA) principle (Wiedmann and Minx, 2008; WRI, 2010; BSI, 2008). Accordingly, CFs in agriculture have been used to explore mitigation measures in terms of GHG emissions associated with farming practices using the LCA method up to the farm gate (Lal, 2004a; Dubey and Lal, 2009).

Changes in land use and production systems in agriculture have increasingly been assessed for their potential impacts on climate

Abbreviations: CF, carbon footprint; FCF, farm carbon footprint; GHG, greenhouse gas; LCA, life cycle assessment; PCF, product carbon footprint.

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change, by quantifying CF of crop production in a life cycle up to harvest (Ponsioen and Blonk, 2012; Knudsen et al., 2014). Using the LCA methodology, St Clair et al. (2008) quantified the CFs for three bio-energy crops in the UK and was able to show the important role of land use before conversion to bioenergy cropping on net GHGs reduction. In a later study using a similar methodology by Hillier et al. (2009), the CFs of major staple crops under different farming systems in the UK were quantified, identifying N fertilizer as a main emission source in crop production. Dubey and Lal (2009) compared the CFs of crop production under different farm management practices in the US and India, showing a higher C-based sustainability in Ohio, USA with improved soil management involving straw return and conservation tillage. Similarly, the CFs of durum wheat production from Canada were compared in different cropping systems (Gan et al., 2011a) and production regions (Gan et al., 2011b), demonstrating the additional influence of climate on farming practices, and energy input. More recently, a similar approach was used by Schäfer and Blanke (2012) who compared the CFs of pumpkins from different farming and marketing systems, and showed good nutrient management but not farm size significantly influenced the CF. All these studies demonstrated that CFs using an LCA approach provide a powerful tool for understanding and developing cleaner food production systems.

China is one of the most important agricultural countries in the world. China's production of rice, wheat and maize contributed 28%, 18% and 21% to the world total (FAO, 2010), respectively. Meanwhile, agriculture contributed 11% to the total anthropogenic emissions, accounting for 57% and 74% of the total CH_4 and N_2O emissions respectively in China (NDRC, 2012a). According to the US–China Joint Announcement on Climate Change (The White House, 2014), China is committed to reach a peak in anthropogenic GHG emissions by 2030, and in the meantime to increase the share of non-fossil fuels in primary energy consumption to around 20% by 2030. For the last few decades, China's crop production has been developed under intensive agriculture with high inputs of fertilizers and pesticides but with limited use of conservation tillage (Jin et al., 2008). However, low carbon approaches have been encouraged with incentives under the national climate change policy of China (CAS, 2009; NDRC, 2012b). Rice, wheat and maize, as the three major staple crops, constituted 21%, 36% and 32% respectively of the national total grain production in 2010 (DRSES-SBS, 2011). In a previous study using national statistical data (Cheng et al., 2011), the CFs of crop production in China were shown to be generally higher than those in UK (Hillier et al., 2009) and in US (Dubey and Lal, 2009). So far, there have been few studies on quantification of CFs of individual grain crop production in China.

The objectives of the present study were, firstly, to quantify the CFs of the three major grain crops including rice, wheat and maize in China using farm survey data; secondly, to characterize the contribution of individual inputs involved in farming practices to the overall CFs; and lastly, to compare the CFs of crop production between climate regions and farm sizes. It aimed also to provide information for policy-makers to identify key options for climate change mitigation in China's agriculture.

2. Materials and methods

2.1. Carbon footprint, functional unit and system boundary

The CF was calculated for all the individual inputs used for grain production in rice, wheat and maize, based on the PAS 2050 protocol (BSI, 2008). Emissions of CO_2, CH_4 and N_2O were accounted and the results expressed in carbon dioxide equivalents (CO_2-eq) using their relative warming forcing values (IPCC, 2007). Two functional units for CF accounting were followed in the present study: the farm carbon footprint (FCF) expressed in terms of cropland area in t CO_2-eq ha^{-1}, and the product carbon footprint (PCF) expressed in terms of grain yield in t CO_2-eq t^{-1} grain.

Following the "farm gate" principle generally accepted for LCA in agriculture, the system boundary was set from seeding to harvesting of a cereal crop. As an attributional LCA study, land use change (LUC) did not apply. And carbon sequestration in soil was also excluded in accordance with the PAS 2050 guidelines.

2.2. Regions under study

Sites were selected representing the major crop production areas of China (Fig. 1). Double cropping of paddy rice in summer

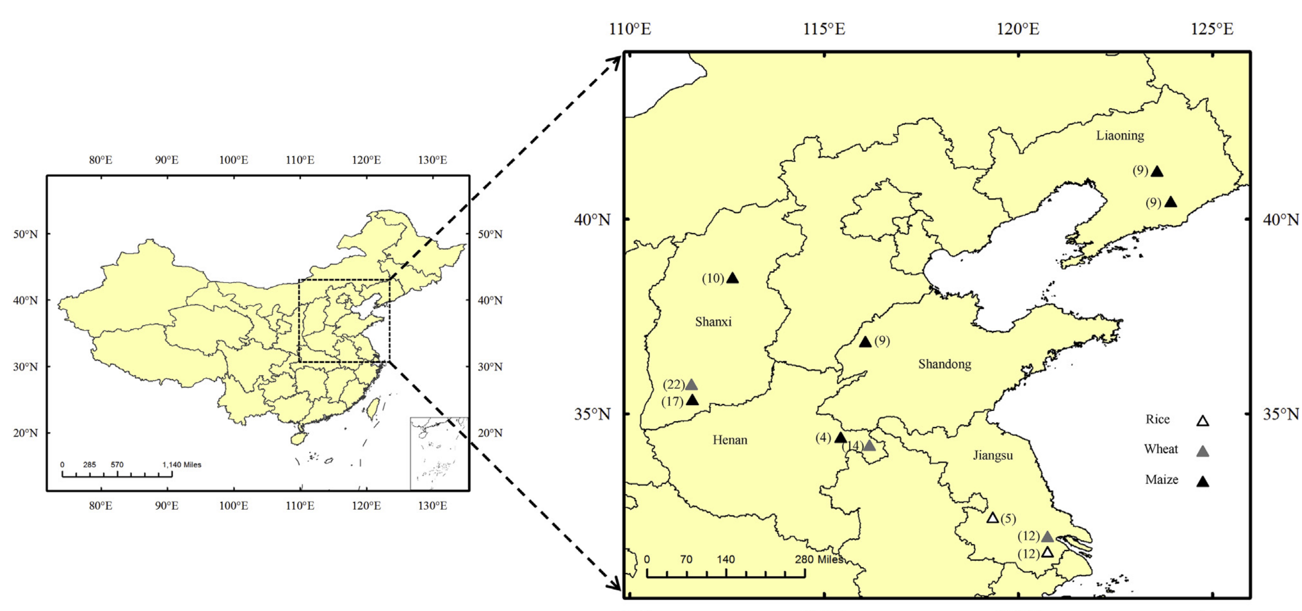

Fig. 1. Geographical distribution of sites surveyed in China (The value in parenthesis is the number of farms surveyed).

and wheat in winter was a typical pattern of crop production in Jiangsu with a warm and humid climate. Here rice paddy was irrigated normally under a water regime of intermittent flooding with midseason drainage in this region. While summer maize was a typical system of grain crop production in Liaoning and Shandong provinces with a humid climate, and summer maize and winter wheat in rotation were typical in Henan with a semi-humid climate. In addition, the province of Shanxi, with a semiarid temperate climate, was a typical region of China with single cropping of rainfed maize or wheat. Sites of the farm survey across these representative crop production areas are shown in Fig. 1.

2.3. Emission quantification protocol

Following the LCA methodology, the CF of crop production was estimated by quantifying the GHG emissions associated with agricultural inputs and farm management practices up to the farm gate (from sowing to harvest). In the present study, GHG emissions included both the direct and indirect emissions with production for a given crop. Here the indirect emissions were attributed to manufacture of agro-chemicals (e.g. fertilizers, pesticides and plastic films) and by electricity used for irrigation. In addition, direct emissions were N_2O from N fertilizer application and CH_4 emissions from rice paddy under submergence as well as energy consumption for farm mechanical operations such as seeding, tillage, transportation and harvesting. Soil carbon changes were not considered in accordance with the PAS 2050 protocol (BSI, 2008). Consequently, the GHG emissions from different sources were quantified using methods described below.

Firstly, the GHG emissions from agricultural inputs including fertilizers, pesticides, and plastic films, electricity used for irrigation, and energy consumption for farm mechanical operation were estimated using the following equation:

$$\mathrm{CF_M} = \sum (I_i \times \mathrm{EF}_i) \tag{1}$$

where $\mathrm{CF_M}$ represents the sum of the GHG emissions induced by the ith agricultural input, in t CO_2-eq; i is the kind of agricultural input or source; I_i is the amount of the ith agricultural input or source (in t for fertilizer, pesticide and plastic film, or in L for diesel oil, or in kW h for electricity); EF_i is the GHG emission factor of the ith input or source when manufactured and/or applied, in t CO_2-eq per unit volume or mass.

Secondly, the direct N_2O emissions from N fertilizer application were estimated with the following equation:

$$\mathrm{CF_{N_2O}} = I_\mathrm{N} \times \mathrm{EF_{N_2O}} \times \frac{44}{28} \times 298 \tag{2}$$

where $\mathrm{CF_{N_2O}}$ represents the direct N_2O emissions from application of N fertilizer, in t CO_2-eq; I_N is quantity of N fertilizer applied in a single crop season, in t; $\mathrm{EF_{N_2O}}$ is the default emission factor of N_2O emission of applied N fertilizer, in t N_2O-N t^{-1} N fertilizer. Emission factors of synthetic N fertilizer use in dry crops and submerged rice paddies were adopted respectively from IPCC (2006) and Zou et al. (2007); 44/28 is the molecular conversion factor of N_2 to N_2O; 298, the relative molecular potential of warming forcing in a 100-year horizon (IPCC, 2007).

Thirdly, the direct CH_4 emissions from submerged paddy were estimated using the following equation:

$$\mathrm{CF_{CH_4}} = \mathrm{EF_d} \times t \times A \times 25 \tag{3}$$

where $\mathrm{CF_{CH_4}}$ represents the CH_4 emissions from rice cultivation in a single season, in t CO_2-eq; $\mathrm{EF_d}$ is a daily emission factor, in t CH_4 ha^{-1} day^{-1}; t is the rice growing period, in days; A is the farm area, in ha; and 25 is the relative molecular warming forcing of CH_4 in a 100-year horizon (IPCC, 2007).

Here $\mathrm{EF_d}$ was estimated with the following equation:

$$\mathrm{EF_d} = \mathrm{EF_c} \times \mathrm{SF_w} \times \mathrm{SF_p} \times \mathrm{SF_m} \times \mathrm{SF_{s,r}} \tag{4}$$

where $\mathrm{EF_c}$ is the baseline emission factor for continuously flooded fields without organic amendments; $\mathrm{SF_w}$ and $\mathrm{SF_p}$, is a scaling factor to account for the differences in water regime during the rice growing period and before rice transplantation, respectively; $\mathrm{SF_m}$, is a scaling factor to account for the differences in type and amount of organic amendment used for rice production; and $\mathrm{SF_{s,r}}$, is a scaling factor for soil type, rice cultivar, etc., if available. Following Yan et al. (2005), the rice growth period was set as 130 days and emission factors for methane in submerged rice paddies with intermittent flooding with midseason drainage were adopted. All the emission factors (EFs) used in this analysis for different inputs or sources are listed in Table 1.

Consequently, the total CF ($\mathrm{CF_t}$, t CO_2-eq) of a grain crop production was calculated by summarizing all the individual GHG emissions from different sources, using the following equation:

$$\mathrm{CF_t} = \mathrm{CF_M} + \mathrm{CF_{N_2O}} + \mathrm{CF_{CH_4}} \tag{5}$$

Finally, the farm carbon footprint (FCF) was expressed in terms of cropland area in t CO_2-eq ha^{-1}; and the product carbon footprint (PCF) was expressed in terms of grain yield in t CO_2-eq t^{-1} grain.

2.4. Data collection

Farm survey activities for this study were conducted during 2010–2011. The data collected in the field surveys with the questionnaire sheets obtained for individual interviewed farmers included: (1) amounts of nitrogen (N), phosphate (P), potassium (K) fertilizers, and pesticides used for each crop production; (2) farm mechanical operations (e.g. methods of soil tillage, harvesting); (3) water management practices such as tube or well irrigation; and (4) farm area and grain yield of each crop. Household farms were divided into two categories of small sized (<0.5 ha) and large sized household farms (>0.5 ha) according to the farm size data obtained in the survey. Overall, valid data from 123 questionnaires (17 for rice, 58 for wheat, and 48 for maize) were obtained to form a database shown in Table 2 (Table S1 in detail).

2.5. Data processing and statistical analysis

Data processing was performed using Microsoft Office Excel 2010 and all statistical analyses were conducted using JMP Ver. 7.0. One-way ANOVA and the least significant difference test (LSD) were used to check the differences between farm size classes and regions. The level of significance was defined at $p < 0.05$.

3. Results

3.1. Farm size, agricultural input and grain yield

The farm size, grain yield and agricultural inputs for crop production in the surveyed farms had a very wide variability (Table 2, Fig. 2). Most of the surveyed farms were 0.1–0.5 ha in size, showing the great fragmentation of China's croplands. 90% of total farmers visited owned small sized (<2 ha) farms while the other 10% owned relatively large (>2 ha) farms. Chemical fertilizers in the range of 250–350 kg ha^{-1} were used in over 50% of the total farms surveyed (Fig. 3). Moreover, nitrogen (N) fertilizer use ranged from

Table 1
Greenhouse gas emission factors for different inputs or sources used in the present study.

Emission source	Abbreviation	Emission factor or scaling factor	Literature
N fertilizer	$EF_{fertilizer}$	6.38 t CO_2-eq t^{-1} N	Lu et al. (2008)
P fertilizer		0.61 t CO_2-eq t^{-1} P_2O_5	West and Marland (2002a)
K fertilizer		0.44 t CO_2-eq t^{-1} K_2O	West and Marland (2002a)
Pesticide	$EF_{pesticide}$	18.0 t CO_2-eq t^{-1} pesticide	West and Marland (2002a)
Plastic film	EF_{film}	2.5 t CO_2-eq t^{-1} film	Energy Source, China (2009)
Diesel oil for machinery	$EF_{machinery}$	2.63×10^{-3} t CO_2-eq L^{-1}	BP China (2007)
Electricity for irrigation	$EF_{irrigation}$	9.2×10^{-4} t CO_2-eq kW^{-1} h^{-1}	BP China (2007)
Direct N_2O emission from N fertilizer	EF_{N_2O}	Dry cropland, 0.01 t N_2O-N t^{-1} fertilizer-N	IPCC (2006)
		Rice paddy, 0.0073 t N_2O-N t^{-1} fertilizer-N	Zou et al. (2007)
CH_4 emission from rice field	EF_c	1.30×10^{-3} t CH_4 ha^{-1} day^{-1}	Yan et al. (2005)
	SF_w	0.52	Yan et al. (2005)
	SF_p	0.68	Yan et al. (2005)
	SF_m	1.00	Yan et al. (2005)

Table 2
Agricultural inputs and grain yield of rice, wheat and maize production in the surveyed regions (mean ± S.E.).

Crop	Region	Farm size (ha)	Grain yield (t ha^{-1})	N fertilizer (N kg ha^{-1})	P fertilizer (P_2O_5 kg ha^{-1})	K fertilizer (K_2O kg ha^{-1})	Pesticide (kg ha^{-1})	Diesel oil (L ha^{-1})	Electricity for irrigation (kW h ha^{-1})
Rice	Jiangsu (17)	1.7 ± 0.8	7.6 ± 0.1	269 ± 10	58 ± 11	55 ± 13	13.0 ± 2.9	178 ± 24	1216 ± 88
Wheat	Henan (14)	0.5 ± 0.1	6.5 ± 0.2	309 ± 23	127 ± 12	50 ± 8	0.8 ± 0.3	175 ± 8	0
	Jiangsu (12)	2.2 ± 1.2	4.8 ± 0.1	317 ± 33	83 ± 9	83 ± 9	2.6 ± 0.4	109 ± 20	0
	Shanxi (22)	0.6 ± 0.1	3.6 ± 0.3	107 ± 4	108 ± 4	107 ± 4	1.6 ± 0.2	206 ± 21	142 ± 79
Maize	Liaoning (18)	0.4 ± 0.1	7.5 ± 0.2	112 ± 18	52 ± 7	19 ± 3	2.3 ± 0.3	64 ± 27	0
	Shandong (9)	1.0 ± 0.2	9.1 ± 0.3	278 ± 21	53 ± 9	21 ± 3	7.0 ± 0.4	136 ± 18	0
	Henan (4)	0.6 ± 0.2	6.0 ± 0.3	134 ± 48	65 ± 22	64 ± 22	3.9 ± 0.3	104 ± 0	0
	Shanxi (27)	0.3 ± 0.0	5.8 ± 0.4	143 ± 10	102 ± 6	70 ± 12	2.2 ± 0.3	138 ± 14	94 ± 56

Note: Film was not used for crop production but in Shanxi, where 90–300 kg ha^{-1} films were used for maize production. The value in parenthesis is the number of farms surveyed.

28 kg N ha^{-1} to 460 kg N ha^{-1} across the farms surveyed. The mean N application rate was the highest for rice (269 kg N ha^{-1}) and the lowest for maize (154 kg N ha^{-1}). For wheat production, N was applied in a higher rate in Jiangsu than that in Liaoning and Shanxi. While for maize, the N application rate was higher in Shandong province than in the other areas (Table 2).

Overall, the mean grain yield was 7.6, 7.0 and 4.8 t $ha^{-1}$for rice, maize, and wheat respectively for all the surveyed farms. Grain yield of wheat was higher in Liaoning than in Jiangsu and Shanxi, and that of maize was higher in Shandong than in Henan and Shanxi (Table 2, Table S1).

3.2. Variation of carbon footprint with regions and farm size

Clearly, the estimated CFs of grain production varied with crops. Among the three crops, rice production possessed the highest CF at 6.0 t CO_2-eq ha^{-1} (FCF) and 0.8 t CO_2-eq t^{-1} grain (PCF) on average. This was followed by wheat and maize production, where the

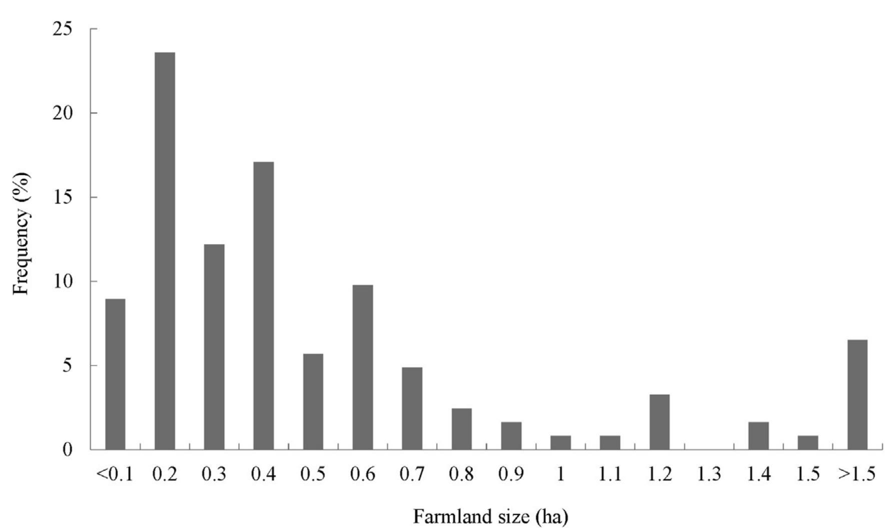

Fig. 2. Frequency distribution of farmland size in the survey.

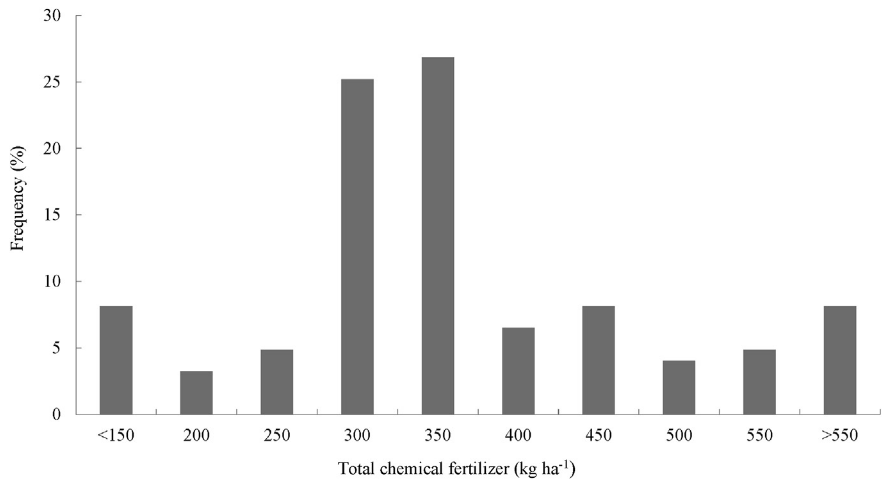

Fig. 3. Frequency distribution of total fertilizer application for all the crops surveyed.

estimated CF was on average 3.0 and 2.3 t CO_2-eq ha^{-1} (FCF), and 0.66 and 0.33 t CO_2-eq t^{-1} (PCF), respectively (Table 3). As seen also in Table 3, the grain yield and CF varied with climatic regions for both wheat and maize, though rice was surveyed only in the single region of Jiangsu. Wheat was produced with a significantly higher yield (by 60%), PCF (by 20%) and FCF (by 90%) in humid Jiangsu and semi-humid Henan than in semiarid Shanxi. In contrast, maize in semiarid Shanxi was produced with lower yield (by 20%) but higher PCF (by 40%) in comparison to humid Shandong and Liaoning.

Data in Table 4 illustrated the variation of PCF (CF in term of grain production) with farm size classes. While no difference was observed for rice production, the mean PCF of wheat and of maize in large sized (>0.5 ha) household farms was lower by about 22% and 28% than small sized (<0.5 ha) ones, respectively.

Table 4
Variation of product carbon footprint with farm size classes (Mean ± S.E.).

Crop	Region	Carbon footprint (t CO_2-eq t^{-1} grain)	
		Small sized farm	Large sized farm
Rice	Jiangsu	0.84 ± 0.03a	0.80 ± 0.05a
Wheat	Jiangsu	0.86 ± 0.09a	0.65 ± 0.14a
	Shanxi	0.68 ± 0.07a	0.53 ± 0.04b
Maize	Liaoning	0.27 ± 0.04a	0.16 ± 0.01b
	Shandong	0.43 ± 0.02a	0.36 ± 0.01b
	Shanxi	0.39 ± 0.05a	0.38 ± 0.06a

Different letters indicate significant differences between farm size classes at $p < 0.05$.

3.3. Contribution of individual inputs

The average proportions of individual inputs or sources to the total CFs for rice, wheat and maize are presented in Fig. 4. Nitrogen fertilizer use was the biggest single contributor to the total CF, accounting for on average 44%, 79% and 75% for rice, wheat, and maize, respectively. This was followed by mechanical operation, which was on average responsible for 8%, 15% and 14% for rice, wheat, and maize production respectively. A minor contribution (overall 4–6%) came from pesticides, P and K fertilizers. There were relatively small changes in the proportion of individual inputs with the different crops except for irrigation. The use of plastic films, for example, accounted for about 3% of the total CF for maize production. However, irrigation (19% on average) and direct CH_4 emissions (25% on average) made a significant contribution for rice production, compared to a small contribution of irrigation (2–3%) for both wheat and maize.

4. Discussion

4.1. GHG emissions from crop production and the mitigation significance

For all the grain crops studied, the FCF and PCF was estimated on average to be 3.1 t CO_2-eq ha^{-1} and of 0.5 t CO_2-eq t^{-1} grain respectively. In comparison with the reported studies from different countries, the CFs for wheat and maize in this study were apparently higher than those from US, Canada and even from India (Table 5). However, the estimated CF for rice in our study was lower than that from India, where rice production was comparatively low yielding but there was the high energy cost for irrigation (Pathak et al., 2010).

Table 3
Grain yield and carbon footprint of wheat, maize and rice from different climate regions (Mean ± S.E.).

Crop	Region	Grain yield (t ha^{-1})	PCF (t CO_2-eq t^{-1} grain)	FCF (t CO_2-eq ha^{-1})
Wheat	Semiarid (22): Shanxi	3.6 ± 0.3b	0.59 ± 0.04b	2.0 ± 0.1b
	Humid (26): Henan, Jiangsu	5.8 ± 0.2a	0.71 ± 0.05a	3.9 ± 0.2a
	Mean (48)	4.8 ± 0.2	0.66 ± 0.03	3.0 ± 0.2
Maize	Semiarid (27): Shanxi	6.2 ± 0.3b	0.39 ± 0.02a	2.3 ± 0.2a
	Humid (31): Henan, Shandong, Liaoning	7.7 ± 0.2a	0.28 ± 0.02b	2.2 ± 0.2a
	Mean (58)	7.0 ± 0.2	0.33 ± 0.02	2.3 ± 0.1
Rice	Mean (17)	7.6 ± 0.1	0.80 ± 0.02	6.0 ± 0.1

Note: FCF, farm carbon footprint; PCF, product carbon footprint; Different lower case letters indicate significant differences between climate regions for the same crop at $p < 0.05$. The value in parenthesis is the number of farms surveyed.

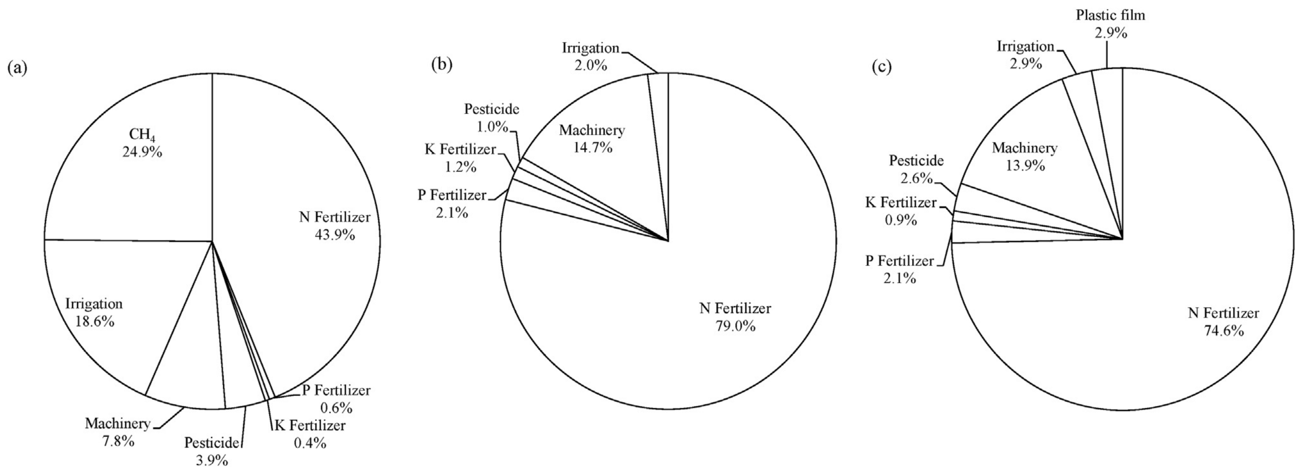

Fig. 4. Contribution of different inputs or sources to the total carbon footprint for rice (a), wheat (b), and maize (c) production.

The above comparison revealed a generally high CF for the major staple crops in China. As reported by Kitzes et al. (2008), China's major food crop production has been associated with high carbon emissions. This therefore provides a challenge to the sustainability of China's food production as China has prioritized increasing productivity as a pathway to improve food security in line with the ambition to reduce the carbon intensity of the nation's economy.

Agricultural management practices have a strong impact on CFs for crop production. The high CF for rice could be attributed to the direct CH_4 emissions under submergence and the energy use for irrigation, which was necessarily performed for rice cultivation in the farms surveyed. Whereas, the high CF for wheat was largely due to the high emissions from N fertilizer applications, and associated with high EF of the N applied. Conservation tillage, which reduces the energy use for machinery, has been increasingly extended to maize croplands in the North of China under a national project since 2003 (Zhao et al., 2012). This could explain, in part, the relatively low CFs for maize production from Henan, Shandong, Liaoning and Shanxi of northern China. West and Marland (2002a,b) also indicated that crop production had a smaller carbon intensity (PCF in this study) under no-tillage in comparison to conventional tillage in US. Besides the crop type, the CFs could vary with farm types and cropping systems. For example, Hillier et al. (2009) showed that organic farms had significantly lower CFs than conventional and integrated farm types in the UK, mainly due to the avoidance of synthetic N fertilizer use. As reported by Gan et al. (2011a), durum wheat production had a significantly higher CF under cereal-based monoculture systems in comparison to more diverse cropping systems in Canada, where oilseed, pulse, and cereal crops were grown in well-defined cropping sequences. In a similar study by Yang et al. (2014), reported that the high CF associated with conventional intensive crop production systems could be significantly reduced with appropriate diversification of crop rotation systems in the North China Plain.

China has experienced significant climate change impacts on agriculture with increasing frequencies of drought and pest invasion across the major grain production regions (Pan et al., 2011). Consequently, crop production has required higher inputs of chemicals (fertilizers and pesticides) and greater irrigation costs for rice production. Thus, crop production of China could potentially become increasingly carbon intensive under future climate change. Using data of PCF in Table 3, substitution of rice with maize could potentially save about 0.5 t CO_2-eq per ton of grain production. Roughly, about 100 Mt CO_2-eq could be avoided if all rice would be replaced by maize. Therefore, carbon sustainability would be greatly enhanced if food consumption changed alongside climate change trends, which could allow maize to extend across China's croplands. With a fast increasing trend of both yield and cultivation area (Meng et al., 2013), enhanced maize production would be beneficial to reducing the carbon cost of crop production in China and would contribute to future adaptation.

4.2. N fertilizer and yield

Overall, N fertilizer induced GHG emissions contributed to 44–79% of the total CFs for all the farms surveyed. This contribution

Table 5
Carbon footprint and N fertilizer contribution for crop production from different countries.

Country	Crop	Carbon footprint (t CO_2-eq t^{-1} grain)	Contribution by N fertilizer use	Literature
UK	Staple-wheat mostly	–	75%	Hillier et al. (2009)
USA	Wheat	0.25–0.35	67–75% (NT)	Snyder et al. (2009)
	Maize	0.12–0.22 (NT)	67–75% (NT)	Snyder et al. (2009)
		0.14–0.25 (CT)	45–60% (CT)	Snyder et al. (2009)
Canada	Wheat	0.27–0.50	–	Gan et al. (2011a,b)
India	Rice	1.2–1.5	<10%	Pathak et al. (2010)
	Wheat	0.12	75%	Pathak et al. (2010)
China	Bulk	0.4–0.5	55%	Cheng et al. (2011)
	Rice	0.80	44%	This study
	Wheat	0.65	79%	This study
	Maize	0.33	75%	This study

Note: NT, no-till; CT, conventional till.

seemed considerably higher than the proportion of 57–53% to the overall CF of crop production in China estimated with statistical data in our previous study (Cheng et al., 2011). Hillier et al. (2009) reported a similar high proportion (75% on average) by N fertilizer use for the staple crops in the UK, though the total CF was much lower. In contrast, N fertilizer use accounted for a much lower proportion (30–40%) of the total CF for durum wheat production under various cropping systems across Canada (Gan et al., 2011a). In this study, the CFs of crop production were significantly correlated with N fertilizer application rates (Fig. 5), showing a strong N-dependence of GHGs emission in China's crop production. Therefore, the CF of wheat and maize rather than rice was more strongly dependent on N fertilizer use. It could be worthy to note that high grain yields (7.5 t ha^{-1} on average of all the crops) could be achieved with N fertilizer application rates of 200–300 kg N ha^{-1}. In other words, use of N fertilizer over 300 N ha^{-1} did not lead to a higher yield but did result in a much higher carbon cost (Fig. 6). Our results show that there has been surplus GHG emissions of approximately of 1000–2000 kg CO_2-eq ha^{-1}, which were associated with N use that was beyond that needed for maximum production.

This high GHG emission induced by N fertilizer use raised again the serious concern of China's agricultural sustainability. And reducing N application and improving N use efficiency should be urgently considered as a route to improving the sustainability of crop production (Janzen et al., 2003; Lal, 2004b). In western countries, N uptake by plants is typically reported as being 40–60% of total N applied (Ladha et al., 2005). In China, however, typically only 30–35% of the total N applied is recovered by crops (Ju and Zhang, 2003; Zhu, 1998). Such problems of low N use efficiency are attributed mainly to widespread over use and unbalanced fertilization of synthetic N by individual farmers in their fragmented farmlands (Ju et al., 2009). Thus high yields could be again sustained with reduced N application and improved N use efficiency in China. Of course, some gain in yield with further increase in N application could still be possible in under-fertilized regions. This issue has been evaluated in a national project involving soil testing and recommended fertilization in China since the late 1990s (Sun and Huang, 2012). Nevertheless, the present study suggests that an urgent need remains to cut N fertilizer use, particularly in the small sized household farms of China. Recently, biochar soil amendments were advocated as a potential option to enhance N use efficiency (Liu et al., 2012; Zhang et al., 2012). Fortunately, N-saving and slow-release biochar-based fertilizer have been developed and increasingly used in China, which could offer an option to avoid excess N use and reduce direct N_2O emission from N fertilized croplands (Joseph et al., 2013; Qian et al., 2014). With the accessibility of biochar production technology granted by the state low carbon development framework (Pan et al., 2011; NDRC, 2014), use of biochar in agriculture could offer a great opportunity to achieve sustainable development of low carbon crop production in China (Bell et al., 2014).

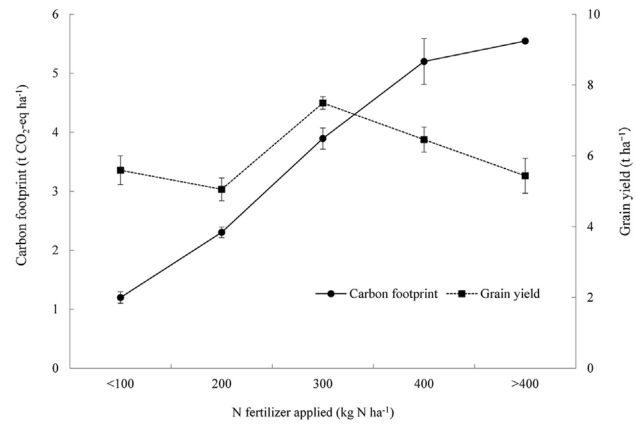

Fig. 6. Variation of carbon footprint and yield with different N fertilizer application rate.

4.3. Regional and farm size impacts on carbon footprint of crop production

In the present study, grain yield and CF varied with climate regions for wheat and maize. Wheat was produced with a significantly higher yield and CF in the eastern humid region of Henan and Jiangsu than in the semiarid region of Shanxi, probably due to high inputs of N fertilizer and machinery for the high productivity in Jiangsu and Henan (Table 2). Such regional differences were also reported by Gan et al. (2011b) who found a significantly lower CF (by 25%) for spring wheat in the semiarid Brown soil zone than that in the more humid Black soil zone from western Canada. In contrast, maize was produced with lower yield but a higher PCF in semiarid Shanxi, compared to humid Shandong and Liaoning. Generally, maize was produced with high phosphorus input and irrigation cost in small sized farmlands in Shanxi, where the soils were mostly poor in soil nutrients, and particularly poor in phosphorus under rainfed conditions (Meng et al., 2013). Among the major staple crops, maize production exerted a higher sustainability than wheat in terms of yield and CF (Dubey and Lal, 2009). Rather than wheat production, improvement of maize production could be seen as a strategic option to meet food demand and to

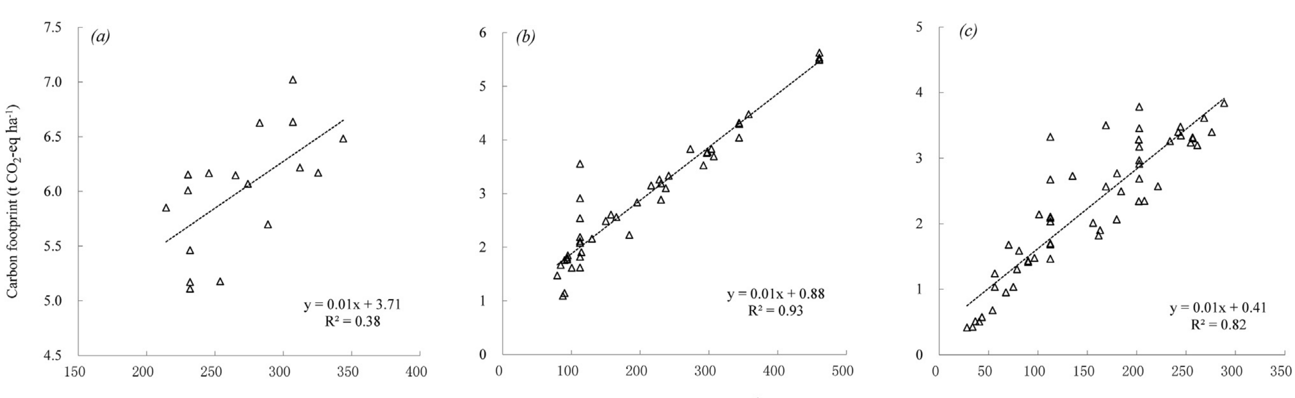

Fig. 5. Correlation of carbon footprint with N fertilizer application rate for rice (a), wheat (b), and maize (c) production.

reduce GHG emissions intensity at the same time (Cui et al., 2013). While agriculture in humid temperate region of China has undergone much intensification with double cropping of rice-wheat or wheat-maize rotations, the overall carbon-based sustainability would need further study.

The study has also shown the difference in PCF with farm size classes. The PCF in large sized farms was significantly lower than that in small sized household farms both for wheat and maize. This was in agreement with the findings of Feng et al. (2011) who reported that topsoil SOC storage could be 30% higher in larger sized farms (>0.7 ha, 10 Chinese *mu*) than in smaller ones (<0.7 ha). In a similar study using a questionnaire survey data, Sefeedpari et al. (2013) found that farms less than 1 ha in size had a higher total energy input of 17%, 21% and 34% than those of 1–4 ha, 4–10 ha and >10 ha respectively for rain-fed wheat production from central Iran. The wheat PCF could be more sensitive to management practices than maize. The role of the fragmentation of farmlands needs to be further addressed as crop production nowadays had been operated primarily under a household management system in China (Huang and Wang, 2008). Increasing farm size with cooperative small householders or aggregating small farms into large scaled units could offer an opportunity to help mitigate GHG emissions from crop production in China, without additional technical inputs. Therefore, improving farming system management, and farm intensification in particular, are critical issues to consider in the planning for the development of China's agriculture.

5. Conclusions

The present study, using questionnaire survey data from individual household farms, quantified the CFs for rice, wheat and maize crop production from China. The results showed that the CFs for the three major grain crops in China were higher than those from the developed countries. Moreover, N fertilizer use was seen as the most important contributor (44–79%) to the total CF of crop production, which was significantly correlated with N fertilizer application rate. Rice had a higher PCF (0.80 ± 0.02 t CO_2-eq t^{-1} grain) than wheat (0.66 ± 0.03 t CO_2-eq t^{-1}) and maize (0.33 ± 0.02 t CO_2-eq t^{-1}), mainly due to the high CH_4 emission from rice fields. It demonstrated the carbon intensive production of major staple crops in China, which could be cut down primarily through reducing excessive N fertilizer use and potentially through substantial substitution of rice and wheat for maize. In addition, the CF for wheat production was lower in the semiarid region than that in the humid region. In contrast, maize had a higher PCF in the semiarid region compared to the humid region, probably as a result of the poor soil nutrient status and irrigation costs associated with semiarid regions. Our study also indicated that large sized farms had a lower PCF (by 22–28%) than small fragmented (household) farms, for wheat and maize production. The CFs of crop production could be affected by farm size and climate condition as well as crop management practices. Therefore, improving farming management efficiency, upscaling farm size and developing farming intensification could be important measures to realize low carbon agriculture and climate change mitigation in China's crop production.

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Appendix A. Supplementary material

Supplementary data related to this article can be found online at http://dx.doi.org/10.1016/j.jclepro.2015.05.058.

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