Greenhouse gas mitigation potential in crop production with biochar soil amendment—a carbon footprint assessment for cross-site field experiments from China

Xiangrui Xu1,* | Kun Cheng1,2,* | Hua Wu1 | Jianfei Sun1 | Qian Yue1 | Genxing Pan1,2

1Institute of Resource, Ecosystem and Environment of Agriculture, College of Resources and Environmental Sciences, Nanjing Agricultural University, Nanjing, China
2Jiangsu Collaborative Innovation Center for Solid Organic Waste Resource Utilization, Nanjing, China

Correspondence
Kun Cheng, Institute of Resource, Ecosystem and Environment of Agriculture, Nanjing Agriculture University, Nanjing, Jiangsu, China. Email: chengkun@njau.edu.cn

Funding information
Natural Science Foundation of Jiangsu, Grant/Award Number: BK20150684; National Key Research and Development Plan, Grant/Award Number: 2017YFD0200802; China Natural Science Foundation, Grant/Award Number: 41877097, 41501569; Fundamental Research Funds for the Central Universities, Grant/Award Number: KJQN201673

Abstract
Biochar soil amendment (BSA) had been advocated as a promising approach to mitigate greenhouse gas (GHG) emissions in agriculture. However, the net GHG mitigation potential of BSA remained unquantified with regard to the manufacturing process and field application. Carbon footprint (CF) was employed to assess the mitigating potential of BSA by estimating all the direct and indirect GHG emissions in the full life cycles of crop production including production and field application of biochar. Data were obtained from 7 sites (4 sites for paddy rice production and 3 sites for maize production) under a single BSA at 20 t ha⁻¹ across mainland China. Considering soil organic carbon (SOC) sequestration and GHG emission reduction from syngas recycling, BSA reduced the CFs by 20.37–41.29 t carbon dioxide equivalent ha⁻¹ (CO₂-eq ha⁻¹) and 28.58–39.49 t CO₂-eq ha⁻¹ for paddy rice and maize production, respectively, compared to no biochar application. Without considering SOC sequestration and syngas recycling, the net CF change by BSA was in a range of −25.06 to 9.82 t CO₂-eq ha⁻¹ and −20.07 to 5.95 t CO₂-eq ha⁻¹ for paddy rice and maize production, respectively, over no biochar application. As the largest contributors among the others, syngas recycling in the process of biochar manufacture contributed by 47% to total CF reductions under BSA for rice cultivation while SOC sequestration contributed by 57% for maize cultivation. There was a large variability of the CF reductions across the studied sites whether in paddy rice or maize production, due likely to the difference in GHG emission reductions and SOC increments under BSA across the sites. This study emphasized that SOC sequestration should be taken into account the CF calculation of BSA. Improved biochar manufacturing technique could achieve a remarkable carbon sink by recycling the biogas for traditional fossil-fuel replacement.

KEYWORDS
biochar, carbon footprint, climate change, crop production, greenhouse gas, soil organic carbon

These authors made equal contributions to this work.

This is an open access article under the terms of the Creative Commons Attribution License, which permits use, distribution and reproduction in any medium, provided the original work is properly cited.

© 2018 The Authors. GCB Bioenergy Published by John Wiley & Sons Ltd.
1 INTRODUCTION

Agriculture had been considered as the major emitter of non-CO$_2$ greenhouse gases (GHGs), contributing 52% and 84% to the global methane and nitrous oxide emissions, respectively, and 10%–12% to the global anthropogenic GHG emissions (IPCC, 2013; Smith et al., 2008). Hence, it had been urgent to develop effective management practices for GHG emissions reduction together with food production enhancement in global agriculture (Smith et al., 2013). Among the recommended management practices, soil organic carbon (SOC) sequestration had been considered a win–win strategy for mitigating climate change while sustaining food productivity through sustainable organic matter management (Paustian et al., 2016; Tubiello et al., 2015).

Biochar was produced upon pyrolysis of organic residues or bio-wastes produced in agriculture or food processing. Bio-waste or biomass was pyrolyzed at temperatures in a range of 350–750°C under limited oxygen conditions, which recycled nutrients associated with recalcitrant but nano-structured organic matter (Lehmann, 2007). During pyrolysis, volatile organic matter from the biomass feedstock was released as syngas and wood vinegar when condensed. The former could be recycled as the renewable and cleaner energy to replace fossil fuel (Karmee, 2016; Ning et al., 2013). The latter, as a liquid product, contained soluble nutrients and organic molecules beneficial to plants (Lou et al., 2015). Recycling of syngas to generate heat or power could bring biomass pyrolysis into GHG emission reduction though viable technologies were still under development (FAO 1987; Henrich, 2004).

With pyrolysis of mainly crop straw, biochar had been increasingly tested and advocated to apply in agriculture (Smith, 2016; Wang, Zhao, Xing, & Yang, 2013; Zhang, Yan, et al., 2016). Due to the significant recalcitrance of the organic matter with residence time up to thousands of years, biochar soil amendment (BSA) had been recommended for enhancing stable organic carbon storage with various ecosystem benefits (Lehmann et al., 2015; Sohi, 2012; Spokas et al., 2012). Some field studies also indicated that there was no simulative effect of biochar on soil respiration (Liu, Zheng, et al., 2016). Given these, biochar amendment has been regarded as a hopeful measure to mitigate climate change contributed by its favorable ability in SOC sequestration and N$_2$O emission reduction effects under soil amendment (Sohi, Lopez-Capel, Krull, & Bol, 2009; Woolf, Amonette, Street-perrott, Lehmann, & Joseph, 2010). For this, biochar application did not generally resulted in increased soil respiration (Liu, Zheng, et al., 2016; Zhou et al., 2017). Importantly, amendment of biochar had been found generally improving soil fertility and crop productivity, soil structure and moisture retention, increasing microbial abundance and metabolic efficiency of soil microbes, such as nitrobacteria and denitrrobacteria related to N cycles (Liu, Liu, et al., 2016; Omordi et al., 2016; Verhoeven et al., 2017; Wu et al., 2016; Zhang, Bian, et al., 2012; Zhang, Liu, Pan, Hussain, et al., 2012; Zhou et al., 2017). In particular, BSA had been shown effective for reducing N$_2$O emissions, thus improving N-use efficiency (Liu et al., 2012; Zhang, Liu, Pan, Hussain, et al., 2012). However, there were not consistent results concluding the effects of biochar addition on methane emissions in rice paddies (He, Cui, et al., 2017; Mukherjee & Lal, 2014). Nevertheless, BSA had been widely considered to have a great potential to mitigate climate change through enhanced SOC sequestration and N$_2$O emission reduction plus bioenergy recycling via biochar production (Sohi et al., 2009; Woolf et al., 2010).

A large number of field studies had been conducted in an attempt to quantify GHG emission reduction under BSA (Liu, Zheng, et al., 2016; Yang et al., 2017; Zhang, Bian, et al., 2012; Zhang, Yan, et al., 2016). Quantifying soil CO$_2$, N$_2$O, and CH$_4$ emissions monitored and measured over two consecutive rice seasons; Zhang, Bian, et al. (2012) reported a net increase in GHG emissions by 2,862 kg CO$_2$-eq ha$^{-1}$ in the first season but a reduction by 935 kg CO$_2$-eq ha$^{-1}$ in the second season compared to no biochar treatment, following a single BSA at 20 t/ha in a paddy field from East China. Using similar quantification approach, Bamminger, Poll, and Marhan (2018) reported no significant reduction in overall GWP under winter rapeseed and spring wheat rotation over 2 years following a BSA at 30 t/ha, in a field experiment from Germany. Such field experiments had been increasingly used in meta-analysis studies to quantify BSA performance on GHG emission reduction estimated with the calculation of response ratio of paired data. For example, a global meta-analysis declared a decrease by 54% in N$_2$O emissions under BSA in both laboratory and field experiments (Cayuela et al., 2014). Song, Pan, Zhang, Zhang, and Wang (2016) quantified a decrease by 20% in N$_2$O emission but an increase by 19% in CH$_4$ emissions in rice paddies under BSA, and Jeffery, Verheijen, Kamman, and Abalos (2016) argued that BSA resulted in a reduction in methane emissions from flooded fields, especially those in acid reaction. Nevertheless, few studies took into account SOC changes along with reductions of N$_2$O and CH$_4$ emissions though BSA was convinced for SOC sequestration. In addition, GHG emissions associated with biochar manufacture and application had not yet been considered when addressing BSA's roles in climate change mitigation.

Carbon footprint (CF), a tool to quantify the total GHG emissions of a product or activity, has been used widely to assess all the GHG emissions of crop production over the full life cycle (Roy & Dias, 2017). Carbon footprint of crop production was generally quantified by considering all the GHG emissions induced by or associated with agricultural material used, farm machinery operated, and irrigation...
power expended and direct field GHG emissions in a crop’s life cycle. There had been increasing studies developing methodologies to quantify the CF for various crop production. Among these, using national statistics and field survey data, respectively, Cheng et al. (2011) and Yan, Cheng, et al. (2015) reported CFs of China’s crop production, to which nitrogen fertilizer application contributed the most. With life cycle analysis (LCA)-based CF methodology, Liu, Liu, et al. (2016) was able to quantify the CF of rice production under BSA and demonstrate a net reduction of CF by 18%. Peters, Iribarren, and Dufour (2015) shown a significant carbon abatement potential with biochar from biomass pyrolysis to amend soil with benefits for crop production, compared to direct biomass combustion. So far, SOC storage changes had not yet been considered in CF assessment, even in the PAS 2050 protocol (Publicly Available Specification 2050), the most authoritative and widely used manual of LCA (BSI 2008, 2011).

Therefore, it would help better understand the biochar’s role by quantifying GHG reduction with LCA coupled CF methodology considering all emissions from biochar production to application, and taking into account the SOC changes. This study was aim to (a) develop an appropriate methodology for quantifying CF of crop production under BSA, (b) estimate a net GHG reduction potential under BSA by quantifying the CFs in different crop cultivations across multi-sites in China’s agro-regions, and (c) identify any spatial variability and key contributors of the CFs of crop production under BSA. We anticipated to provide sound information for a better understanding of GHG reduction with biochar and practicing biochar production and application for sustainable agriculture of China.

2 METHODS AND MATERIALS

2.1 Data for carbon footprint assessment

The systems of crop productions with BSA included dry crop production of maize and flooded paddy rice in this assessment, as the representative of cereal production of China. The sites used for this assessment included the maize cultivation fields located in Taian (TA), Xinzhou (XZ), and Shanqiu (SQ), and the paddy rice cultivation fields located in Changsha (CS), Jinxian (JX), Guanghan (GH), and Yixing (YX), across mainland China (Figure 1). The basic information of the field experiments obtained from the published literatures is summarized in Table 1. The biochar used in all the experiments was derived from the pyrolysis of wheat straw at temperatures in a range of 350–500°C in a vertical kiln by Sanli Energy Company, China. Before soil amendment, biochar was ground in powder in a particle diameter of 3mm and transported to the field. A single amendment at the rate of 20 t/ha was performed following the harvest of the preceding crop but before sowing of next crop, by spreading biochar to soil surface and evenly incorporating into topsoil by plowing and ranking. The detailed operation of soil amendment was firstly described by Zhang, Cui, et al. (2010). The organic carbon content, total nitrogen content, surface area, CEC, and pH of the biochar were 467.1 g/kg, 5.94 g/kg, 8.9 m²/g, 21.7 cmol/kg, and 10.4, respectively.

Data of soil properties and direct N₂O and CH₄ emissions were collected from the published papers of individual experiment. Additional information of biochar production was obtained accessing the shared database owned by the corresponding author’s Institute. Data for the agricultural inputs, activities, and outputs associated with crop production, such as fertilizer, pesticides, agricultural plastic film, diesel, irrigation, and grain yield, were obtained either from the previous studies or by the survey conducted for this study (Table 1; Supporting information Table S1).

2.2 Carbon footprint protocol

2.2.1 Carbon footprint definition and functional units

A CF of crop production was defined as the sum of each individual GHG release caused by various agricultural material and energy input and direct greenhouse gases emission over an entire life cycle of the crop production, expressed in carbon dioxide equivalent (CO₂-eq) using relative global warming potential (GWP) values of individual GHG proposed by IPCC (2013). In this study, a methodology of LCA was employed to assess the CF by taking into account all the GHG emissions caused by or associated with biochar and other inputs used, farm management, and power exhausted over the entire growing season of crop cultivation (ISO, 2006). The functional unit was defined as CF per unit cultivation area (t CO₂-eq ha⁻¹) and per kilogram of grain produced in the cropland (kg CO₂-eq kg⁻¹).

2.2.2 Scope, system boundary, and carbon footprint calculation

Carbon footprints of both paddy rice and maize production with and without biochar addition were estimated. The CF was calculated by summarizing all GHG emissions directly and indirectly caused by crop cultivation in the life cycle including the emissions by agricultural input manufacture (biochar, fertilizers, and pesticides), farm work (machinery operation and irrigation), soil N₂O emissions, and CH₄ emissions in flooded rice paddy.

Soil organic carbon changes were excepted from the system boundary unless land use changed as per PAS 2050 protocol (BSI, 2008). However, SOC sequestration in cropland with improved land management was also promised.
as a potential pathway to mitigate climate change in IPCC AR5, which ignored CO₂ emission in croplands (IPCC, 2013). SOC changes by land management could be quantified with a method provided in the 2006 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006). Noting an input of recalcitrant organic matter persisting hundreds of years from biochar (Wang, Xiong, & Kuzmakov, 2016) and a potential offset through recycling the syngas generated in biochar production, soil carbon changes, and biogas utilization were taken into account with four calculation scenarios in this study (Table 2).

Scenario A: According to the system boundaries defined above, a CF (CF, kg CO₂-eq ha⁻¹) of a crop production under BSA was quantified based on the PSA 2050 protocol using the following equation:

\[
CF_A = \frac{AI_i \times EF_i + BC \times EF_{BC1} + E_{N_2O} \times 265 + E_{CH_4} \times 28}{C_2}
\]

\[
E_{N_2O} = E_{N_2O, direct} + E_{N_2O, indirect}
\]

\[
E_{N_2O, indirect} = (AI_{N_{fert}} + BC \times C_{N_{BC}}) \times \left( L_{vola} \times EF_{vola} + L_{leac} \times EF_{leac} \right)
\]

where \( AI_i \) indicates the amount of agricultural input \( i \) in kg/ha, including chemical fertilizer, plastic film, pesticide,
diesel, and irrigated water; $E_F$ is the adopted emission factors of the $i$th agricultural material or activity $i$ as shown in Table 3. BC is the amount of biochar input in t/ha. The amount of $N_2O$ emissions ($E_{N_2O}$) includes the direct and indirect $N_2O$ emissions in this study. The direct $N_2O$ emissions ($E_{N_2O\_direct}$) and $CH_4$ emissions ($E_{CH_4}$) were both sourcing from the field monitoring, while the indirect $N_2O$ emissions ($E_{N_2O\_indirect}$) were calculated using equation 3. In equation 3, $L_{vola}$ and $L_{leac}$ are the volatilizing rate and leaching rate of input N, and $EF_{vola}$ and $EF_{leac}$ are the indirect $N_2O$ emission factors of volatilizing N and leaching N, respectively. The N input includes chemical fertilizer N ($A_{N\_fert}$) and the N from biochar with a N content of $CN_{BC}$ mentioned in Section 3. The numbers of 265 and 28 are the GWP values for $N_2O$ and $CH_4$, respectively (IPCC, 2013).

Scenario B: The CF under Scenario B with biogas recycled ($CF_B$, kg $CO_2$-eq ha$^{-1}$) was calculated using the following equation:

$$CF_B = A_i \times EF_i + BC \times EF_{BC1} + E_{N_2O} \times 265 + E_{CH_4} \times 28$$ (4)

where $EF_{BC2}$ indicates the emission factor of biochar manufacturing with biogas recycled.

Scenario C: Equations 5 and 6 were used for calculating the CF with SOC change considered.

$$CF_C = A_i \times EF_i + BC \times EF_{BC1} + E_{N_2O} \times 265 + E_{CH_4} \times 28 - \Delta SOCD$$ (5)

$$\Delta SOCD = (SOC_t - B_t - SOC_0 \times B_0) \times H \times H \times 44 \times 12 \times 100$$ (6)

where $\Delta SOCD$ represents the changes in SOC density after harvest compared to the beginning of the experiments in kg $CO_2$-eq ha$^{-1}$; $SOC_0$ and $SOC_t$ indicate the SOC concentrations before and after the experiments in kg C kg$^{-1}$; $B_0$ and $B_t$ are the bulk densities before and after the experiments in g/cm$^3$; $H$ means the sampling depth of soil profile, which was 0.15 m for paddy and 0.2 m for dry cropland; 44/12 is a coefficient that converts carbon into $CO_2$-eq.

Scenario D: The CF under Scenario D ($CF_D$, kg $CO_2$-eq ha$^{-1}$) with both SOC stock changes and biogas recycled concerned was calculated using the following equation:

$$CF_D = A_i \times EF_i + BC \times EF_{BC2} + E_{N_2O} \times 265 + E_{CH_4} \times 28 - \Delta SOCD$$ (7)

For these different scenarios, the emission factors of biochar without and with syngas recycled, $EF_{BC1}$ and $EF_{BC2}$, are included in Table 3.

Furthermore, the yield-scaled CF ($CF_y$, kg $CO_2$-eq kg$^{-1}$ yield) was also calculated to assess the amount of CF

| Emission source/sink | Agricultural inputs | CH$_4$ emission | $N_2O$ emission | SOC pool change | Syngas recycled |
|----------------------|---------------------|-----------------|----------------|-----------------|----------------|
| Scenario A | √ | √ | √ | × | × |
| Scenario B | √ | √ | √ | × | √ |
| Scenario C | √ | √ | √ | √ | × |
| Scenario D | √ | √ | √ | √ | √ |

**TABLE 1** Climate and soil condition of the experiment sites used in this study

| Crop Site | MAT (°C) | MAP (mm) | SOC (g/kg) | TN (g/kg) | BD (g/cm$^3$) | pH | Literature |
|-----------|----------|----------|------------|-----------|---------------|----|------------|
| Rice Changsha (CS) | 17.1 | 1500 | 18.76 | 1.79 | 0.9 | 6.21 | Qu et al. (2012) |
| Jinxian (JX) | 17.7 | 1400 | 17.7 | 1.59 | 1.1 | 4.89 | |
| Guanghan (GH) | 16.5 | 807 | 20.11 | 1.81 | 1.08 | 5.99 | Zhang, Liu, Pan, Zheng, et al. (2012) |
| Yixing (YX) | 15.7 | 1177 | 23.5 | 1.78 | 1.01 | 6.51 | Zhang, Cui, et al. (2010) |
| Maize Shangqiu (SQ) | 13.9 | 780 | 9.87 | 0.94 | 1.46 | 8.38 | Zhang, Liu, et al. (2012) |
| Taian (TA) | 13 | 697 | 8.44 | 0.83 | 1.56 | 5.93 | Li et al. (2014) |
| Xinzhou (XZ) | 9.3 | 445 | 4.30 | 0.38 | 1.30 | 8.39 | Zhang, Pan, et al. (2016) |

MAT, MAP, SOC, TN, and BD are annual mean temperature, mean annual precipitation, soil organic carbon content, soil total nitrogen content, and soil bulk density.
TABLE 3  Emission factors of each inputs used in the assessment

| Emission source      | Emission factor                          | Literature                        |
|----------------------|------------------------------------------|-----------------------------------|
| Urea                 | 7.48 kg CO₂-eq kg⁻¹N                    | Chen, Lu and Wang (2015)          |
| Calcium superphosphate | 0.72 kg CO₂-eq kg⁻¹ P₂O₅              |                                   |
| Potassium fertilizer | 0.62 kg CO₂-eq kg⁻¹ K₂O (Potassium chloride) 1.50 kg CO₂-eq kg⁻¹ K₂O (Potassium sulphate) |                                   |
| Biochar manufacture  | 293.4 kg CO₂-eq t⁻¹ (biogas not recycle) –678 kg CO₂-eq t⁻¹ (biogas recycle) | Ji et al. (2018)                  |
| Pesticide            | 55.0 kg CO₂-eq kg⁻¹ (paddy rice) 50.6 kg CO₂-eq kg⁻¹ (maize) | Zhang, Lu, Huang, Chen and Wang (2016) |
| Plastic film         | 19.0 kg CO₂-eq kg⁻¹                    | NDRCC (2010)                      |
| Diesel               | 3.18 kg CO₂-eq kg⁻¹                    | NDRCC (2011)                      |
| Irrigation           | 0.38 kg CO₂-eq m⁻³ (CS site) 0.30 kg CO₂-eq m⁻³ (JX site) 0.24 kg CO₂-eq m⁻³ (GH site) 0.29 kg CO₂-eq m⁻³ (YX site) 0.18 kg CO₂-eq m⁻³ (SO site) 0.26 kg CO₂-eq m⁻³ (CS site) | Wang et al. (2012)                |

| N volatilization rate | 0.1                                      | NDRCC (2011)                      |
| N leaching rate       | 0.2                                      |                                   |
| EFvola                | 0.01 kg CO₂-eq kg⁻¹N                    |                                   |
| EFleac                | 0.0075 kg CO₂-eq kg⁻¹N                  |                                   |

EFvola and EFleac are indirect N₂O emission factors of volatilizing and leaching N by N inputs.

induced by unit yield of crop production:

\[ \text{CF}_Y = \frac{\text{CF}}{Y} \]  \hspace{1cm} (8)

where \( Y \) is the grain yield in kg/ha.

Finally, to characterize the variability between different sites, the coefficient of variable (C.V.) was calculated using the following equation:

\[ \text{CV} = \frac{SD}{\overline{CF}} \times 100\% \]  \hspace{1cm} (9)

where CV is the coefficient of variable in %; \( \overline{CF} \) and SD indicate the mean and standard deviation of the CFs of 4 sites for paddy field or 3 sites for maize field.

3 | RESULTS

3.1 | Carbon footprint of crop production under biochar amendment

Rice production had the higher CFs being 14.36–16.79 t CO₂-eq ha⁻¹ than maize production being 4.29–5.46 t CO₂-eq ha⁻¹ under typical fertilization without biochar application (Table 4). Under BSA, however, the estimated CF ranged from −11.96 to 18.04 t CO₂-eq ha⁻¹ for paddy rice but from −28.98 to 10.66 t CO₂-eq ha⁻¹ for maize, varying with the scenario conditions. In detail, the area- and yield-scaled CFs estimated with Scenario A were 18.04 t CO₂-eq ha⁻¹ and 2.16 kg CO₂-eq kg⁻¹ under BSA compared to 14.36 t CO₂-eq ha⁻¹ and 1.80 kg CO₂-eq kg⁻¹ under typical fertilization for paddy rice production, respectively. With the calculation scenarios of B, C, and D considering SOC sequestration and/or biogas utilization, biochar amendment had the lower area- and yield-scaled CFs being −11.96–7.47 t CO₂-eq ha⁻¹ and −1.43–0.90 kg CO₂-eq kg⁻¹ than typical fertilization being 14.36–16.79 t CO₂-eq ha⁻¹ and 1.80–2.11 kg CO₂-eq kg⁻¹, respectively, showing a greater CF reduction than that with calculation Scenario A (Table 4). Biochar performance on GHG emissions was calculated with the CF differences between BSA and typical fertilization. BSA increased area and yield-scaled CF by 3.69 t CO₂ and 0.37 kg CO₂-eq kg⁻¹ with calculation Scenario A but decreased CF by 9.32–28.75 t CO₂-eq ha⁻¹ and 1.21–3.54 kg CO₂-eq kg⁻¹ with calculation scenarios of B, C, and D (Table 5).

For maize production with Scenario A, the area- and yield-scaled CFs were 5.46 t CO₂-eq ha⁻¹ and 0.61 kg CO₂-eq kg⁻¹ under BSA compared to 10.66 t CO₂-eq ha⁻¹ and 1.11 kg CO₂-eq kg⁻¹ under typical fertilization, respectively (Table 4). Using calculation scenarios of B, C, and D, both area- and yield-scaled CFs were greatly decreased with BSA, exerting the remarkable GHG mitigations of 13.84–33.27 t CO₂-eq ha⁻¹ and 1.38–3.42 kg CO₂-eq kg⁻¹ over typical fertilization, respectively (Table 5).

The coefficients of variation were calculated to characterize the variability between the 4 sites for rice paddy field or 3 sites for maize field. As shown in Table 5, the C.V.
of the changes in area- and yield-scaled CFs by BSA was as high as 33%–181% and 37%–203%, respectively, for paddy rice fields while that was 8%–41% and 2%–37% for maize fields. This demonstrated a very high variability of rice CF changes by BSA across the experiments sites.

3.2 | Carbon footprint profiles for crop production

The contributions of GHG emissions induced by various emission sources were presented in Figure 2. With Scenario A and Scenario B whereby SOC pool change not concerned, the largest contributors were methane emissions (8.68 t CO₂-eq ha⁻¹) followed by chemical fertilizer input (2.43 t CO₂-eq ha⁻¹) for paddy rice production, but chemical fertilizer input (2.72 t CO₂-eq ha⁻¹) followed by N₂O emission (2.35 t CO₂-eq ha⁻¹) for maize production under typical fertilization.

With BSA, a great proportion of total CF was contributed by biochar manufacture with the GHG emissions of 5.87 t CO₂-eq ha⁻¹ (Scenario A). Nevertheless, these emissions could be offset when the biogas was recycled for energy substitution, and the net carbon sink of biochar manufacture being −13.56 t CO₂-eq ha⁻¹ could be achieved (Scenario B). Regard to SOC pool change, there was a loss of SOC pool at 2.44 t CO₂-eq ha⁻¹ for rice but a gain at 1.17 t CO₂-eq ha⁻¹ for maize under typical fertilization though a big spatial variation existed (Figure 2, Supporting information Table S2). Following a BSA, SOC pool was increased by 10.57 and 20.21 t CO₂-eq ha⁻¹ for rice and maize production, respectively, though much variable across sites.

The relative contributions of SOC sequestration, N₂O and CH₄ emission reductions, and energy offset by biochar manufacture to the total CF reductions under BSA were presented in Figure 3. With Scenario A, the significant reductions of N₂O or CH₄ emissions in maize or paddy field were offset by the GHG emissions resulted from biochar manufacture without biogas recycling. With Scenario D, the net carbon sink under rice production with BSA was almost shared by SOC sequestration and biogas recycling (45% and 47%, respectively). Comparatively, SOC sequestration and biogas recycling contributed 57% and 41% of the net carbon sink by BSA in maize production.

4 | DISCUSSION

4.1 | Roles of SOC sequestration and biogas recycling on carbon footprint quantification of biochar amendment

As per PAS 2050 and ISO protocol, SOC pool could not be considered in CF calculating process unless land use
was changed, due to the uncertain persistence against decomposition. Straw return could increase SOC pool in short term, with the added organic carbon to be decomposed mostly in several years (Liu, Wu, Lu, & Chen, 2014). The increase in SOC pool with no-till was also argued for the potential mineralization of the physically protected carbon (Powlson, Whitmore, & Goulding, 2011). Dominated by aromatic carbon via pyrolysis, organic carbon in biochar was hardly decomposable (Pignatello, Uchimiya, Abiven, & Schmidt, 2015), with a mean residence time of hundreds of years (Wang et al., 2016). Moreover, biomass pyrolysis, shifting the decomposable feedstock biomass into recalcitrant organic matter and retarding the turnover of the CO₂ photosynthesized, has been increasingly accepted as a negative emission technology with environmental co-benefits (Smith, 2016). As to our estimation, a significant difference in CF between Scenario A and Scenario C was reached to 13.01 and 19.03 t CO₂-eq ha⁻¹ for rice and maize production, respectively, which indicated a huge carbon sink of BSA sourced from SOC sequestration (Figure 3). This clearly showed a shift from a net source to net sink of GHG emissions of the systems, when the SOC pool change was taken into account CF quantification.

Organic matter enhancement through biochar amendment, despite of its long residence time up to millennium years, could have a wide range of ecosystem benefits (Smith, 2016; Sohi, 2012). This would discourage people to adopt BSA for climate change mitigation, along with the issue of economic cost by farmers (Clare, Barnes, McDonagh, & Shackley, 2014; Clare et al., 2015). A number of field experiments could evidence a set of ecosystem functions and cost-benefits improved following a BSA over multiple years (Liu et al., 2014; Wang, Li, et al., 2018; Wang, Liu, et al., 2018). As a decline in SOC pool had been a global trend (Smith et al., 2015), and the re-build SOC storage had been recommended by the 4 per mil initiative (Minasny et al., 2017), we strongly declare that SOC pool change induced by BSA should be included within the rational boundary of CF calculation in crop production. This is particularly important in China, where arable soil had been depleted of organic carbon (Song, Li, Pan, & Zhang, 2005) and carbon sequestration has a great potential to mitigate climate change (Pan, Smith, & Pan, 2009).

The existing pyrolytic techniques and biochar manufacture equipment differed in energy cost from 1.77 to 8.7 MJ/kg dry feedstock, varying with modes and

### Table 5 Net greenhouse gas mitigation effects (Mean ± S.D.) under biochar amendment compared to typical fertilization quantified with four calculation scenarios

| Crops | Scenario A | Scenario B | Scenario C | Scenario D |
|-------|------------|------------|------------|------------|
|       | Area-scaled (t CO₂-eq ha⁻¹) |            |            |            |
| Rice  | 3.69 ± 6.7  (181%) | −15.74 ± 6.7 (42%) | −9.32 ± 9.53 (102%) | −28.75 ± 9.53 (33%) |
|       | Yield-scaled (kg CO₂-eq kg⁻¹) |            |            |            |
|       | 0.37 ± 0.75 (203%) | −1.95 ± 0.85 (44%) | −1.21 ± 1.2 (99%) | −3.54 ± 1.32 (37%) |
| Maize | Area-scaled (t CO₂-eq ha⁻¹) |            |            |            |
|       | 5.19 ± 1.18 (23%) | −14.23 ± 1.17 (8%) | −13.84 ± 5.62 (41%) | −33.26 ± 5.62 (17%) |
|       | Yield-scaled (kg CO₂-eq kg⁻¹) |            |            |            |
|       | 0.51 ± 0.19 (37%) | −1.54 ± 0.23 (15%) | −1.38 ± 0.37 (27%) | −3.42 ± 0.07 (2%) |

The number in bracket was the value of variable coefficient. The positive values indicated a net greenhouse gas emission while the negative values indicated a net greenhouse gas mitigation by biochar amendment.

### Figure 2 Contributions of individual emission sources to total GHG emissions for crops production (TF, typical fertilization; BA, biochar amendment)
temperature of pyrolysis, and the handling way of by-product (Liu, Liu, et al., 2016; Shi et al., 2011). As a by-product of biomass pyrolysis, syngas contained CH₄, H₂, and CO with a heat value ranging from 2 to 12 MJ/kg (Crombie & Mašek, 2014), which indicated a big potential to alternate fossil fuel. In a previous study of LCA for biochar production based on a survey of business scale factory, biochar manufacturing process induced the GHG emissions of 293.4 kg CO₂–eq t⁻¹ biochar on average, while a net carbon sink reached 678 kg CO₂–eq t⁻¹ biochar in case biogas was recycled for electricity generation (Ji, Cheng, Nayak, & Pan, 2018).

With Scenario A not considering SOC change and syngas recycling, the CFs under BSA were even higher than that without biochar addition, which was because the direct GHG emission reduction was greatly overshadowed by the energy cost in biochar manufacture. While the calculation Scenario B brought a significant GHG abatement of 15.74 and 14.23 t CO₂–eq ha⁻¹ for rice and maize production, respectively, mainly due to the considerable carbon sink from syngas recycling.

Calculation Scenario D, considering both carbon pool change and syngas recycling, gave a great GHG reduction, which became largely shared by SOC sequestration and energy offset by syngas for power (92%–98%) rather than N₂O and/or CH₄ emission reduction (Figure 3). A global assessment also concluded that annual emission reductions could be as high as 1.86 PgC in biochar production and application compared to biomass combustion, of which soil carbon sequestration and fossil-fuel offsets contributing to the most being as high as 80% (Woolf et al., 2010). Given these, this study suggested that the CF methodology for BSA should take SOC change into account and biochar manufacturing technique should be highlighted. In addition, the SOC stock change factors and emission factors of biochar manufacture would be further developed to minimize the uncertainties of CF estimation due to the advancing and upgrading of biochar manufacturing technique and the effects of raw material and pyrolysis condition on biochar performance (Boateng, García-Pérez, Mašek, Brown, & del Campo, 2015; Jahirul, Rasul, Chowdhury, & Ashwath, 2012; McHenny, 2009).

### 4.2 Performance of biochar amendment in GHG mitigation of crop production

Although SOC sequestration and fossil-fuel offset by biogas recycling both contributed most to total GHG reduction by BSA, BSA still had the significant mitigation effects on non-CO₂ GHG emissions. In rice paddies, biochar amendment both lowered the yield-scaled CH₄ and N₂O emissions by 26% and 16%, respectively. Due to the inverse relationship between CH₄ and N₂O emissions during the rice-growing seasons, intermittent flooding and no-tillage reduced the yield-scaled CH₄ emissions by 62% and 26%, but increased the N₂O emissions by 275% and 12% in paddy rice production of China according to a meta-analysis using 24 published field studies (Feng et al., 2013). While the yield-scaled N₂O emissions were reduced by 31% or 17%, the CH₄ emissions were enhanced largely by 93% or 195% under manure or straw application which induced the increases in yield-scaled GWP by 54% and 154% (Feng et al., 2013). The application of N inhibitor or slow release N could also both mitigate the CH₄ and N₂O emissions by 0.35 and 0.51 t CO₂–eq ha⁻¹ in paddy rice production of China conclude by a meta-analysis using nearly 50 published field studies (Nayak et al., 2015), which could be compared to the CH₄ and N₂O emissions
mitigation being 2.05 and 0.13 t CO2-eq ha$^{-1}$ under BSA by this study. These indicated that BSA had the integrated benefit of CH4 and N2O reductions, which was superior to other agricultural practices, especially other kinds of organic matter addition.

N2O emissions were mitigated by 0.67 t CO2-eq ha$^{-1}$ under BSA in maize field (Figure 3), compared to N2O emission reduction rate at 0.8 t CO2-eq ha$^{-1}$ by reduced N fertilization according to a meta-analysis using 76 papers from North China (Xu et al., 2017). Nayak et al. (2015) further found that application of N inhibitor or slow release N could mitigate N2O emission by 0.01–0.66 t CO2-eq ha$^{-1}$ but had no effect on SOC pool. While manure application had no effect or even increased the N2O emissions by 0.13 t CO2-eq ha$^{-1}$ though the SOC pool was enhanced by 0.7 or 1.43 t CO2-eq ha$^{-1}$ according to two meta-analyses by Xia, Lam, Yan, and Chen (2017) and Nayak et al. (2015) for China's dry cropping systems. Given these, BSA had the higher N2O mitigation than manure management but was similar to or lower than some practices such as improved N fertilization or N inhibitor application.

The estimated CF reduction with BSA by Scenario D on average was 33.27 t CO2-eq ha$^{-1}$ for maize production and 28.75 t CO2-eq ha$^{-1}$ for rice production (in Figure 3). The greater GHG reduction in maize field could be explained by its higher topsoil organic C pool improvement (20.21 t CO2-eq ha$^{-1}$ on average vs. 10.57 t CO2-eq ha$^{-1}$ on average in paddy field; Supporting information Table S2). The major pathways of biochar C loss included biochemical mineralization and physical movement (Lehmann et al., 2015; Zimmermann et al., 2012); however, some incubation and field experiments indicated that biochar amendment did not largely enhance soil respiration (Bruun et al., 2011; Liu, Zheng, et al., 2016). As biochar could be subject to physical movement and lost via leaching in rice paddy field, the lower SOC gain could be due to horizontal and vertical movement of biochar with water stream from rainfall and/or irrigation (Rumpel, Ba, Darboux, Chaplot, & Planchon, 2009; Wang, Zhang, Hao, & Zhou, 2013; Zhang, Niu, et al., 2010). As the amendment was generally performed before paddy rice transplanting under flooding, such water stream could be significant (Fang et al., 2010; Huang, Rozelle, Lohmar, Huang, & Wang, 2006).

A large spatial variability of CF reductions was observed for paddy rice production in this study, with a CV up to 203% compared to 41% for maize production. However, the variability was mainly contributed by variation of non-CO2 GHG reductions (CVs up to 291%) for rice compared to 148% for maize production (Supporting information Figure S1). Being sensitive to moisture and soil texture as well as N level (Cayuela et al., 2014; Shcherbak, Millar, & Robertson, 2014), N2O emission and the reduction showed a large site variation with a CV of 198%. On the other hand, changes in CH4 emissions by BSA in rice paddy fields varied from site to site due to inconsistent response between CH4 emissions and CO2 emissions, between biochar type/rate as well as environmental variables (He, Zhou, et al., 2017; Jeffery et al., 2016; Mukherjee & Lal, 2014; Song et al., 2016). For sound understanding and convincing quantification, identifying the key variables and characterizing the site variation of GHG emissions with BSA should be among the priority needs.

Nevertheless, there had been often economic concerns of BSA due to uncertain biochar price and benefit feedback (Galgani, van der Voet, & Korevaar, 2014; Li et al., 2015; Wang, Li, et al., 2018; Wang, Liu, et al., 2018). The rate of biochar application was 20 t/ha for all the experiments in this study which was set up corresponding to direct return of nearly 5 years’ crop straw (with biomass yield about 10 t per hectare per year for both rice and maize; Supporting information Table S1). Although this application may not be very economically viable in the first year, the eco-environmental and economic benefits of biochar application could sustain for several years (Major, Rondon, Molina, Riha, & Lehmann, 2010; Hernandez‐Soriano et al., 2016; Wang, Li, et al., 2018; Wang, Liu, et al., 2018). A biochar amendment at 20 t ha$^{-1}$ increased maize grain yield over 4 years following the amendment in a dryland soil from Colombia (Major et al., 2010). Despite of a significant cost needed for biochar amendment operation, a BSA at 24 t ha$^{-1}$ increased the rice yield and thus economic income while decreased GHG emissions over 4 years following the single amendment in the central subtropics of China (Wang, Liu, et al., 2018). As a particular case, the rice production in the site of YX in this study was assessed for its sustainability and cost-effectiveness, which was shown more or less consistent throughout 6 years following a single BSA at 20 t ha$^{-1}$ in 2010 (Wang, Li, et al., 2018). Indeed, long-term observation could promise “One single amendment for multiple year benefits,” making the BSA cost-effective based on the continuing economic gain through increased yield and/or improved quality as well as reduced use of farm operation and pesticides, and so on (Clare et al., 2014, 2015; Wang, Li, et al., 2018). In spite of these, the long-term comprehensive assessments of eco-environmental and economic effects of BSA are still needed in future.

Household responsibility system is still the major crop-land management pattern in China until now, which may lead to small scale household farms with intense land fragmentation (Tan, Heerink, & Qu, 2006). A field survey-based CF assessment had been concluded that the CF of rice production under household farm was higher than that...
under aggregated farm (Yan, Luo, et al., 2015). For small household farmers, it could be increasingly infeasible to adopt simple BSA in their own farms due to poor access to improved biochar production system and limited carbon sink to attend carbon trading. This could be a shame for biochar use as it could not ease the farmer’s challenge of cost for their production (Clare et al., 2014, 2015). Fortunately, aggregating small farmlands into big farm company is prevailing across China, which could help to improve the cost-benefit through BSA in China’s agriculture (Ju, Gu, Wu, & Galloway, 2016; Wang, Liu, Li, & Chen, 2015).

ACKNOWLEDGEMENTS

This work was financially supported by the Natural Science Foundation of Jiangsu Province under a grant number BK20150684 and National Key Research and Development Plan under a grant number 2017YFD0200802. This work was also supported by China Natural Science Foundation under a grant number 41877097 and 41501569 and “the Fundamental Research Funds for the Central Universities” under a grant number KJQN201673.

ORCID

Kun Cheng http://orcid.org/0000-0002-6101-0558
Genxing Pan http://orcid.org/0000-0001-9755-0532

REFERENCES

Bamming, C., Poll, C., & Marhan, S. (2018). Offsetting global warming-induced elevated greenhouse gas emissions from an arable soil by biochar application. Global Change Biology, 24(1), e318–e334. https://doi.org/10.1111/gcb.13871

Boateng, A. A., Garcia-Bamminger, C., Poll, C., & Marhan, S. (2018). Offsetting global warming—nately, aggregating small farmlands into big farm company is prevailing across China, which could help to improve the cost-benefit through BSA in China’s agriculture (Ju, Gu, Wu, & Galloway, 2016; Wang, Liu, Li, & Chen, 2015).

under aggregated farm (Yan, Luo, et al., 2015). For small household farmers, it could be increasingly infeasible to adopt simple BSA in their own farms due to poor access to improved biochar production system and limited carbon sink to attend carbon trading. This could be a shame for biochar use as it could not ease the farmer’s challenge of cost for their production (Clare et al., 2014, 2015). Fortunately, aggregating small farmlands into big farm company is prevailing across China, which could help to improve the cost-benefit through BSA in China’s agriculture (Ju, Gu, Wu, & Galloway, 2016; Wang, Liu, Li, & Chen, 2015).

ACKNOWLEDGEMENTS

This work was financially supported by the Natural Science Foundation of Jiangsu Province under a grant number BK20150684 and National Key Research and Development Plan under a grant number 2017YFD0200802. This work was also supported by China Natural Science Foundation under a grant number 41877097 and 41501569 and “the Fundamental Research Funds for the Central Universities” under a grant number KJQN201673.

ORCID

Kun Cheng http://orcid.org/0000-0002-6101-0558
Genxing Pan http://orcid.org/0000-0001-9755-0532

REFERENCES

Bamming, C., Poll, C., & Marhan, S. (2018). Offsetting global warming-induced elevated greenhouse gas emissions from an arable soil by biochar application. Global Change Biology, 24(1), e318–e334. https://doi.org/10.1111/gcb.13871

Boateng, A. A., Garcia-Perez, M., Mašek, O., Brown, R., & del Campo, B. (2015). Biochar production technology. In J. Lehmann & S. Joseph (Ed.), Biochar for environmental management (pp. 95–120). Abingdon, UK: Routledge.

Bruun, E. W., Hauggaard-Nielsen, H., Ibrahim, N., Eggsgaard, H., Ambus, P., Jensen, P. A., & Dam-Johansen, K. (2011). Influence of fast pyrolysis temperature on biochar labile fraction and short-term carbon loss in a loamy soil. Biomass and Bioenergy, 35(3), 1182–1189. https://doi.org/10.1016/j.biombioe.2010.12.008

BSI (2008) PAS 2050: 2008 Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. London, UK: British Standards Institution. BSI British Standards.

BSI (2011). PAS 2050: Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. London, UK: BSI British Standards.

Cayuela, M. L., Van Zwieten, L., Singh, B. P., Jeffery, S., Roig, A., & Sánchez-Monedero, M. A. (2014). Biochar's role in mitigating soil nitrous oxide emissions: A review and meta-analysis. Agriculture, Ecosystems & Environment, 191, 5–16. https://doi.org/10.1016/j.agee.2013.10.009

Chen, S., Lu, F., & Wang, X. K. (2015). Estimation of greenhouse gases emission factors for China's nitrogen, phosphate, and potash fertilizers. Acta Ecologica Sinica, 35(19), 6371–6383.

Cheng, K., Pan, G., Smith, P., Luo, T., Li, L., Zheng, J., ... Yan, M. (2011). Carbon footprint of china's crop production—an estimation using agro-statistics data over 1993–2007. Agriculture Ecosystems & Environment, 142(3–4), 231–237. https://doi.org/10.1016/j.agee.2011.05.012

Clare, A., Barnes, A., McDonagh, J., & Shackley, S. (2014). From rhetoric to reality: Farmer perspectives on the economic potential of biochar in China. International Journal of Agricultural Sustainability, 12(4), 440–458. https://doi.org/10.1080/14735903.2014.927711

Clare, A., Shackley, S., Joseph, S., Hammond, J., Pan, G., & Bloom, A. (2015). Competing uses for China's straw: The economic and carbon abatement potential of biochar. GCB Bioenergy, 7(6), 1272–1282. https://doi.org/10.1111/gcbb.12220

Climate Change Department of National Development and Reform Commission. Guidelines for the provincial inventory of greenhouse gas (Trial) [R]. Beijing, China: National Development and Reform Commission of China (NDRCC), 2011.

Crombie, K., & Mašek, O. (2014). Investigating the potential for a self-sustaining slow pyrolysis system under varying operating conditions. Bioresource Technology, 162, 148–156. https://doi.org/10.1016/j.biortech.2014.03.134

Fang, Q., Ma, L., Yu, Q., Ahuja, L. R., Malone, R. W., & Hoogenboom, G. (2010). Irrigation strategies to improve the water use efficiency of wheat–maize double cropping systems in north china plain. Agricultural Water Management, 97(8), 1165–1174. https://doi.org/10.1016/j.agwat.2009.02.012

FAO (1987) Simple technologies for charcoal making, second printing. FAO Forestry Paper 41, Rome, Italy: Food and Agriculture Organization of the United Nations. www.fao.org/docrep/S5328e/x5328e00.htm

Feng, J., Chen, C., Zhang, Y., Song, Z., Deng, A., Zheng, C., & Zhang, W. (2013). Impacts of cropping practices on yield–scaled greenhouse gas emissions from rice fields in China: A meta-analysis. Agriculture, Ecosystems & Environment, 164, 220–228. https://doi.org/10.1016/j.agee.2012.10.009

Galgani, P., van der Voet, E., & Korevaar, G. (2014). Composting, anaerobic digestion and biochar production in Ghana. Environmental–economic assessment in the context of voluntary carbon markets. Waste Management, 34(12), 2454–2465. https://doi.org/10.1016/j.wasman.2014.07.027

He, G., Cui, Z., Ying, H., Zheng, H., Wang, Z., & Zhang, F. (2017). Managing the trade-offs among yield increase, water resources inputs and greenhouse gas emissions in irrigated wheat production systems. Journal of Cleaner Production, 164, 567–574. https://doi.org/10.1016/j.jclepro.2017.06.085

He, Y., Zhou, X., Jiang, L., Li, M., Du, Z., Zhou, G., ... Wallace, H. (2017). Effects of biochar application on soil greenhouse gas fluxes: A meta-analysis. GCB Bioenergy, 9(4), 743–755. https://doi.org/10.1111/gcbb.12376

Henrich, E. (2004). Fast pyrolysis of biomass with a twin screw reactor: A first BTL step. PyNe Newsletter, 17, 6–7.

Hernandez-Soriano, M. C., Kerré, B., Goos, P., Hardy, B., Dufey, J., & Smolders, E. (2016). Long-term effect of biochar on the stabilization of recent carbon: soils with historical inputs of charcoal. GCB Bioenergy, 8(2), 371–381.
Huang, Q., Rozelle, S., Lohmar, B., Huang, J., & Wang, J. (2006). Irrigation, agricultural performance and poverty reduction in China. Food Policy, 31(1), 30–52. https://doi.org/10.1016/j.foodpol.2005.06.004

IPCC (2006). Cropland. In H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara & K. Tanabe (Eds). 2006 IPCC guidelines for national greenhouse gas inventories, prepared by the national greenhouse gas inventories programme (pp. 5.15–5.23). Tokyo, Japan: IGES.

IPCC (2013). Climate Change 2013: Working Group Contribution to the IPCC Fifth Assessment Report: The Physical Science Basis. Cambridge, UK: Cambridge University Press.

ISO (2006). Technical Committee ISO/TC 207, Environmental management. Subcommittee SC 5, Life cycle assessment. Environmental Management: Life Cycle Assessment: Principles and Framework.

Jahirul, M. I., Rasul, M. G., Chowdhury, A. A., & Ashwath, N. (2012). Biofuels production through biomass pyrolysis—A technological review. Energies, 5(12), 4952–5001. https://doi.org/10.3390/en5124952

Jeffery, S., Verheijen, F. G., Kammann, C., & Abalos, D. (2016). Biochar effects on methane emissions from soils: A meta-analysis. Soil Biology and Biochemistry, 101, 251–258. https://doi.org/10.1016/j.soilbio.2016.07.021

Ji, C., Cheng, K., Nayak, D., & Pan, G. (2018). Environmental and economic assessment of crop residue competitive utilization for biochar, briquette fuel and combined heat and power generation. Journal of Cleaner Production, 192, 916–923. https://doi.org/10.1016/j.jclepro.2018.05.026

Ju, X., Gu, B., Wu, Y., & Galloway, J. N. (2016). Reducing China’s fertilizer use by increasing farm size. Global Environmental Change, 41, 26–32. https://doi.org/10.1016/j.gloenvcha.2016.08.005

Karmee, S. K. (2016). Liquid biofuels from food waste: Current trends, prospect and limitation. Renewable and Sustainable Energy Reviews, 53, 945–953. https://doi.org/10.1016/j.rser.2015.09.041

Lehmann, J. (2007). Bio-energy in the black. Frontiers in Ecology and the Environment, 5(7), 381–387. https://doi.org/10.1890/1540-9295(2007)5[381:BITBB]2.0.CO;2

Lehmann, J., Abiven, S., Kleber, M., Pan, G., Singh, B. P., Sohi, S. P., … Joseph, S. (2015). Persistence of biochar in soil. Biochar for Environmental Management: Science, Technology and Implementation, 2, 233–280.

Li, B., Fan, C.-H., Zhang, H., Chen, Z. Z., Sun, L. Y., & Xiong, Z. Q. (2015). Combined effects of nitrogen fertilization and biochar on the net global warming potential, greenhouse gas intensity and net ecosystem economic benefit in intensive vegetable agriculture in southeastern China. Atmospheric Environment, 100, 10–19. https://doi.org/10.1016/j.atmosenv.2014.10.034

Li, X., Zhang, J.-w., Li, L.-q., Pan, G.-x., Zhang, X.-h., Zheng, J.-f., … Wang, J.-f. (2014). Effect of biochar amendment on maize growth and soil properties in Huang-Huai-Hai Plain. Soils, 46(2), 269–274.

Liu, Q., Liu, B., Ambus, P., Zhang, Y., Hansen, V., Lin, Z., … Wang, X. (2016). Carbon footprint of rice production under biochar amendment—a case study in a Chinese rice cropping system. GCB Bioenergy, 8(1), 148–159. https://doi.org/10.1111/gcbb.12248

Liu, X. Y., Qu, J. J., Li, L. Q., Zhang, A. F., Jufeng, Z., Zheng, J. W., & Pan, G. X. (2012). Can biochar amendment be an ecological engineering technology to depress N2O emission in rice paddies?—A cross site field experiment from South China. Ecological Engineering, 42, 168–173. https://doi.org/10.1016/j.ecole ng.2012.01.016

Liu, W., Wu, W., Lu, H. H., & Chen, Y. X. (2014). Carbon flux from decomposing 13 C-labeled transgenic and nontransgenic parental rice straw in paddy soil. Journal of Soils and Sediments, 14(10), 1659–1668. https://doi.org/10.1007/s11368-014-0899-z

Liu, X., Zheng, J., Zhang, D., Cheng, K., Zhou, H., Zhang, A., … Kazuyakov, Y. (2016). Biochar has no effect on soil respiration across Chinese agricultural soils. Science of the Total Environment, 554, 259–265. https://doi.org/10.1016/j.scitotenv.2016.02.179

Lou, Y., Joseph, S., Li, L., Graber, E. R., Liu, X., & Pan, G. (2015). Water extract from straw biochar used for plant growth promotion: An initial test. BioResources, 11(1), 249–266.

Major, J., Rondon, M., Molina, D., Riba, S. J., & Lehmann, J. (2010). Maize yield and nutrition during 4 years after biochar application to a Colombian savanna oxisol. Plant and Soil, 333(1–2), 117–128. https://doi.org/10.1007/s11104-010-0327-0

Mchenry, M. P. (2009). Agricultural bio-char production, renewable energy generation and farm carbon sequestration in western Australia: Certainty, uncertainty and risk. Agriculture Ecosystems & Environment, 129(1), 1–7. https://doi.org/10.1016/j.agee.2008.08.006

Minasny, B., Malone, B. P., McBratney, A. B., Angers, D. A., Arrouays, D., Chambers, A., … Field, D. J. (2017). Soil carbon 4 per mille. Geoderma, 292, 59–86. https://doi.org/10.1016/j.geoderma.2017.01.002

Mukherjee, A., & Lal, R. (2014). The biochar dilemma. Soil Research, 52(3), 217–230. https://doi.org/10.1071/SR13359

National Development Reform Commission of China (NDRCC) (2010). Deputy Director of the Committee Answered the Questions About Energy Conservation and Climate Change http://www.xzndrc.gov.cn/wszb/t20100310334122.htm. 2010-3-10.

Nayak, D., Saetnan, E., Cheng, K., Wang, W., Koslowski, F., Cheng, Y. F., … Yan, X. (2015). Management opportunities to mitigate greenhouse gas emissions from Chinese agriculture. Agriculture, Ecosystems & Environment, 209, 108–124. https://doi.org/10.1016/j.agee.2015.04.035

Ning, S. K., Hung, M. C., Chang, Y. H., Wan, H. P., Lee, H. T., & Shih, R. F. (2013). Benefit assessment of cost, energy, and environment for biomass pyrolysis oil. Journal of Cleaner Production, 59, 141–149. https://doi.org/10.1016/j.jclepro.2013.06.042

Omondi, M. O., Xia, X., Nahayo, A., Liu, X., Korai, P. K., & Pan, G. (2016). Quantification of biochar effects on soil hydrological properties using meta-analysis of literature data. Geoderma, 274, 28–34. https://doi.org/10.1016/j.geoderma.2016.03.029

Pan, G., Smith, P., & Pan, W. (2009). The role of soil organic matter in maintaining the productivity and yield stability of cereals in China. Agriculture, Ecosystems & Environment, 129(1–3), 344–348. https://doi.org/10.1016/j.agee.2008.10.008

Paustian, K., Lehmann, J., Ogle, S., Reay, D., Robertson, G. P., & Smith, P. (2016). Climate-smart soils. Nature, 532(7597), 49–57. https://doi.org/10.1038/nature17174

Peters, J. F., Iribarren, D., & Dufour, J. (2015). Biomass pyrolysis for biochar or energy applications? A life cycle assessment. Environmental Science & Technology, 49(8), 5195–5202. https://doi.org/10.1021/acs.est.0c01397
Pignatello, J. J., Uchimiya, M., Abiven, S., & Schmidt, M. W. (2015). Evolution of biochar properties in soil. *Biochar for Environmental Management—Science, Technology and Implementation, 1*, 195–233.

Powis, D. S., Whitmore, A. P., & Goulding, K. W. T. (2011). Soil carbon sequestration to mitigate climate change: A critical re-examination to identify the true and the false. *European Journal of Soil Science, 62*(1), 42–55. https://doi.org/10.1111/j.1365-2389.2010.01342.x

Qu, J. J., Zheng, J. W., Zheng, J. F., Zhang, X. H., Li, L. Q., & Pan, G. X. (2012). Effects of wheat-straw based biochar on yield of rice and nitrogen use efficiency of late rice. *Journal of Ecology and Rural Environment, 28*(3), 288–293. (in Chinese).

Roy, P., & Dias, G. (2017). Prospects for pyrolysis technologies in the bioenergy sector: A review. *Renewable and Sustainable Energy Reviews, 77*, 59–69. https://doi.org/10.1016/j.rser.2017.03.136

Rumpel, C., Ba, A., Darboux, F., Chaplot, V., & Plancho, O. (2009). Erosion budget and process selectivity of black carbon at meter scale. *Geoderma, 154*(1), 131–137. https://doi.org/10.1016/j.geoderma.2009.10.006

Shcherbak, I., Millar, N., & Robertson, G. P. (2014). Global meta-analysis of the nonlinear response of soil nitrous oxide (N2O) emissions to fertilizer nitrogen. *Proceedings of the National Academy of Sciences, 111*(25), 9199–9204. https://doi.org/10.1073/pnas.1322434111

Shi, J. L., Chang, C. Y., Chen, C. S., Shaw, D. G., Chen, Y. H., Kuan, W. H., Ma, H. K. (2011). Energy life cycle assessment of rice straw bio-energy derived from potential gasification technologies. *Bioresource Technology, 102*, 6735–6741. https://doi.org/10.1016/j.biortech.2011.02.116

Smith, P. (2016). Soil carbon sequestration and biochar as negative emission technologies. *Global Change Biology, 22*(3), 1315–1324. https://doi.org/10.1111/gcb.13178

Smith, P., Cotrufo, M. F., Rumpel, C., Paustian, K., Kuikman, P. J., Elliott, J. A., … House, J. I. (2015). Biogeochemical cycles and biodiversity as key drivers of ecosystem services provided by soils. *Soil Discussions, 2*(1), 537–586. https://doi.org/10.5194/soild-2-537-2015

Smith, P., Haberl, H., Popp, A., Erb, K. H., Lauk, C., Harper, R., … Masera, O. (2013). How much land-based greenhouse gas mitigation can be achieved without compromising food security and environmental goals? *Global Change Biology, 19*(8), 2285–2302. https://doi.org/10.1111/gcb.12160

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., … Scholes, B. (2008). Greenhouse gas mitigation in agriculture. *Philosophical Transactions of the Royal Society B: Biological Sciences, 363*(1492), 789–813. https://doi.org/10.1098/rstb.2007.2184

Sohi, S. P. (2012). Carbon storage with benefits. *Science, 338*(6110), 1034–1035. https://doi.org/10.1126/science.1225987

Sohi, S., Lopez-Capel, E., Krull, E., & Bol, R. (2009). Biochar, climate change and soil: A review to guide future research. *CSIRO Land and Water Science Report, 5*(9), 17–31.

Song, G., Li, L., Pan, G., & Zhang, Q. (2005). Topsoil organic carbon storage of China and its loss by cultivation. *Biogeochemistry, 74*(1), 47–62. https://doi.org/10.1007/s10533-004-2222-3

Song, X., Pan, G., Zhang, C., Zhang, L., & Wang, H. (2016). Effects of biochar application on fluxes of three biogenic greenhouse gases: A meta-analysis. *Ecosystem Health and Sustainability, 2*(2), e01202. https://doi.org/10.1002/ehs2.1202

Spokas, K. A., Cantrell, K. B., Novak, J. M., Archer, D. W., Ippolito, J. A., Collins, H. P., … Nichols, K. A. (2012). Biochar: A synthesis of its agronomic impact beyond carbon sequestration. *Journal of Environmental Quality, 41*(4), 973–989. https://doi.org/10.2134/jeq2011.0069

Tan, S., Heerink, N., & Qu, F. (2006). Land fragmentation and its driving forces in China. *Land Use Policy, 23*, 272–285. https://doi.org/10.1016/j.landusepol.2004.12.001

Tubilio, F. N., Salvatore, M., Ferrara, A. F., House, J., Federici, S., Rossi, S., … Prosperi, P. (2015). The contribution of agriculture, forestry and other land use activities to global warming, 1990–2012. *Global Change Biology, 21*(7), 2655–2660. https://doi.org/10.1111/gcb.12865

Verhoeven, E., Pereira, E., Decock, C., Suddick, E., Angst, T., & Six, J. (2017). Toward a better assessment of biochar-nitrous oxide mitigation potential at the field scale. *Journal of Environmental Quality, 46*(2), 237–246. https://doi.org/10.2134/jeq2016.10.0396

Wang, L., Li, L., Cheng, K., Ji, C., Yue, Q., Bian, R., & Pan, G. (2018). An assessment of energy, energy, and cost-benefits of grain production over 6 years following a biochar amendment in a rice paddy from China. *Environmental Science and Pollution Research, 25*, 9683–9696. https://doi.org/10.1007/s11356-018-1245-6

Wang, G., Liu, Y., Li, Y., & Chen, Y. (2015). Dynamic trends and driving forces of land use intensification of cultivated land in China. *Journal of Geographical Sciences, 25*(1), 45–57. https://doi.org/10.1007/s11442-015-1152-4

Wang, C., Liu, J., Shen, J., Chen, D., Li, Y., Jiang, B., & Wu, J. (2018). Effects of biochar amendment on net greenhouse gas emissions and soil fertility in a double rice cropping system: A 4-year field experiment. *Agriculture, Ecosystems & Environment, 262*, 83–96. https://doi.org/10.1016/j.agee.2018.04.017

Wang, J., Rothausen, S. G., 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*(1), 014035. https://doi.org/10.1088/1748-9326/7/1/014035

Wang, J., Xiong, Z., & Kuzyakov, Y. (2016). Biochar stability in soil: Meta-analysis of decomposition and priming effects. *GCB Bioenergy, 8*(3), 512–523. https://doi.org/10.1111/gcbb.12266

Wang, D., Zhang, W., Hao, X., & Zhou, D. (2013). Transport of biochar particles in saturated granular media: Effects of pyrolysis temperature and particle size. *Environmental Science & Technology, 47*(2), 821–828. https://doi.org/10.1021/es303794d

Wang, S., Zhao, X., Xing, G., & Yang, L. (2013). Large-scale biochar production from crop residue: A new idea and the biogas-energy pyrolysis system. *BioResources, 8*(1), 8–11.

Woolf, D., Amonette, J. E., Streeterpott, F. A., Lehmann, J., & Joseph, S. (2010). Sustainable biochar to mitigate global climate change. *Nature Communications, 1*(5), 56.

Wu, H., Zeng, G., Liang, J., Chen, J., Xu, J., Dai, J., … Li, F. (2016). Responses of bacterial community and functional marker genes of nitrogen cycling to biochar, compost and combined amendments in soil. *Applied Microbiology and Biotechnology, 100*(19), 8583–8591. https://doi.org/10.1007/s00253-016-7614-5

Xia, L., Lam, S. K., Yan, X., & Chen, D. (2017). How does recycling of livestock manure in agroecosystems affect crop productivity,
reactive nitrogen losses, and soil carbon balance? Environmental Science & Technology, 51(13), 7450–7457. https://doi.org/10.1021/acs.est.6b06470

Xu, C., Han, X., Bol, R., Smith, P., Wu, W., & Meng, F. (2017). Impacts of natural factors and farming practices on greenhouse gas emissions in the North China Plain: A meta-analysis. Ecology and Evolution, 7(17), 6702–6715. https://doi.org/10.1002/ece3.3211

Yan, M., Cheng, K., Luo, T., Yu, Y., Pan, G., & Rees, R. M. (2015). Carbon footprint of grain crop production in China – based on farm survey data. Journal of Cleaner Production, 104, 130–138. https://doi.org/10.1016/j.jclepro.2015.05.058

Yan, M., Luo, T., Bian, R., Cheng, K., Pan, G., & Rees, R. (2015). A comparative study on carbon footprint of rice production between household and aggregated farms from Jiangxi, China. Environmental Monitoring and Assessment, 187(6), 332. https://doi.org/10.1007/s10661-015-4572-9

Yang, X., Lan, Y., Meng, J., Chen, W., Huang, Y., Cheng, X., … Gao, J. (2017). Effects of maize stover and its derived biochar on greenhouse gas emissions and C-budget of brown earth in Northeast China. Environmental Science and Pollution Research, 24(9), 8200–8209. https://doi.org/10.1007/s11356-017-8500-0

Zhang, A., Bian, R., Pan, G., Cui, L., Hussain, Q., Li, L., … Xinyan, Y. (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(127), 153–160. https://doi.org/10.1016/j.fcr.2011.11.020

Zhang, A., Cui, L., Pan, G., Li, L., Hussain, Q., Zhang, X., … Crowley, D. (2010). Effect of biochar amendment on yield and methane and nitrous oxide emissions from a rice paddy from Tai Lake plain, China. Agriculture, Ecosystems & Environment, 139(4), 469–475. https://doi.org/10.1016/j.agee.2010.09.003

Zhang, A., Liu, Y., Pan, G., Hussain, Q., Li, L., Zheng, J., & Zhang, X. (2012). Effect of biochar amendment on maize yield and greenhouse gas emissions from a soil organic carbon poor calcareous loamy soil from Central China Plain. Plant and Soil, 351(1–2), 263–275. https://doi.org/10.1007/s11104-011-0957-x

Zhang, B., Liu, X.-Y., Pan, G.-X., Zheng, J.-F., Chi, Z.-Z., Li, L.-Q., … Zheng, J.-W. (2012). Changes in soil properties, yield and trace gas emission from a paddy after biochar amendment in two consecutive rice growing cycles. Scientia Agricultura Sinica, 45(23), 4844–4853.

Zhang, W., Niu, J., Morales, V. L., Chen, X., Hay, A. G., Lehmann, J., & Steenhuis, T. S. (2010). Transport and retention of biochar particles in porous media: Effect of pH, ionic strength, and particle size. Hydrology and Earth System Sciences, 14, 397–408. https://doi.org/10.1002/eco.160

Zhang, G., Lu, F., Huang, Z., Chen, S., & Wang, X. (2016). Estimations of application dosage and greenhouse gas emission of chemical pesticides in staple crops in China. Chinese Journal of Applied Ecology, 27(9), 2875–2883. (in Chinese with English summary)

Zhang, D., Pan, G., Wu, G., Kibue, G. W., Li, L., Zhang, X., … Liu, X. (2016). Biochar helps enhance maize productivity and reduce greenhouse gas emissions under balanced fertilization in a rainfed low fertility inceptisol. Chemosphere, 142, 106–113. https://doi.org/10.1016/j.chemosphere.2015.04.088

Zhang, D., Yan, M., Niu, Y., Liu, X., van Zwieten, L., Chen, D., … Zheng, J. (2016). Is current biochar research addressing global soil constraints for sustainable agriculture? Agriculture, Ecosystems & Environment, 226, 25–32. https://doi.org/10.1016/j.agee.2016.04.010

Zhou, H., Zhang, D., Wang, P., Liu, X., Cheng, K., Li, L., … van Zwieten, L. (2017). Changes in microbial biomass and the metabolic quotient with biochar addition to agricultural soils: A Meta-analysis. Agriculture, Ecosystems & Environment, 239, 80–89. https://doi.org/10.1016/j.agee.2017.01.006

Zimmermann, M., Bird, M. I., Wurster, C., Saiz, G., Goodrick, I., Barta, J., … Smernik, R. (2012). Rapid degradation of pyrogenic carbon. Global Change Biology, 18(11), 3306–3316. https://doi.org/10.1111/j.1365-2486.2012.02796.x

SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

How to cite this article: Xu X, Cheng K, Wu H, Sun J, Yue Q, Pan G. Greenhouse gas mitigation potential in crop production with biochar soil amendment—a carbon footprint assessment for cross-site field experiments from China. GCB Bioenergy. 2019;11:592–605. https://doi.org/10.1111/gcbb.12561