






## Estimating cropland carbon mitigation potentials in China affected by three improved cropland practices


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**Abstract:** Agriculture is a large source of carbon emissions. The cropland practices of fertilizer substitution, crop straw and conservation tillage are beneficial and help to rebuild local soil carbon stocks and reduce soil carbon emissions, in addition to reducing the consumption of fertilizers and fossil fuels. These improved cropland practices can directly and indirectly mitigate

carbon emissions, benefiting the sustainability of croplands. For these three improved practices, we estimated carbon mitigation potentials in rice, wheat and maize croplands in China. The combined contribution of these practices to carbon mitigation was 38.8 Tg C yr<sup>-1</sup>, with fertilizer substitution, crop straw return, and conservation tillage contributing 26.6, 3.6 and 8.6 Tg C yr<sup>-1</sup>, respectively. Rice, wheat and maize croplands had potentials to mitigate 13.4, 11.9 and 15.5 Tg C yr<sup>-1</sup>, respectively, with the combined direct and indirect

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potential of 33.8 and 5.0 Tg C yr<sup>-1</sup>. Because of differences in local climate and specific diets, the regional cropland carbon mitigation potentials differed greatly among provinces in China. In China, 18 provinces had a “target surplus” for which the carbon mitigation from these three practices was larger than the mitigation target set for 2020. At the national level, a net “target surplus” of 4.84 Tg C yr<sup>-1</sup> would be attained for Chinese croplands with full implementation of the three improved practices. Regional cooperation must be developed to achieve carbon mitigation targets using such measures as carbon trading, establishing regional associations, and strengthening research programs to improve practices.

**Keywords:** Cropland; Carbon mitigation potential; Cropland practices; Regional cooperation; China

## Introduction

Agriculture occupies 37% of global land, which is responsible for an estimated 1.4% to 10%-12% of the approximately 1.7 Pg CO<sub>2</sub>-eq yr<sup>-1</sup> of global greenhouse gas (GHG) emissions (Smith et al. 2007). GHG emissions come from a variety of practices in the agricultural sector, including soil drainage, soil tillage, biomass burning, flooding, and excess use of fertilizers (West and Marland 2002a; Smith et al. 2007; Li et al. 2011). Some of these practices can directly influence the soil properties and therefore are harmful to long-term cropping sustainability, particularly those that affect soil carbon storage. Current unsustainable practices result in carbon loss of 30-60 Mg C ha<sup>-1</sup> from agricultural soils, which accumulated to 78±12 Pg C (Lal 2007). Thus, to avoid further soil carbon loss, it is essential to improve cropland practices.

Many improved cropland practices are implemented, including reduced tillage or no-tillage, improved crop rotations, applications of organic amendments, return of crop residues, improved irrigation, organic farming, among other practices (Smith et al. 2007, 2008; Paustian et al. 2000; Lu et al. 2009; West et al. 2002a, 2002b, 2002c; Freibauer et al. 2004; Müller et al. 2015; Frank et al. 2015; Sanchez et al. 2016). With these improved cropland practices, the maintenance of

cropland soil carbon stocks improves, and therefore, the practices can directly or indirectly reduce cropland carbon emissions (Paustian et al. 2000; Lu et al. 2009). When implemented, these improved cropland practices could mitigate approximately 5500-6000 Mt CO<sub>2</sub>-eq yr<sup>-1</sup> globally (Smith et al. 2008).

As the world's largest carbon emitter (Peters et al. 2012), China has committed to a reduction of 40%-45% in its carbon emissions intensity (carbon emissions per GDP) by 2020, compared with that in 2005 (Fan and Xia 2012). Therefore, China faces a serious burden in mitigating carbon emissions in the near future, and the demand is high for further practices that mitigate carbon emissions for those sectors with high carbon intensities, like the agriculture sector. However, with the increase of population, food demand is also rapidly growing. Although cropland in China is limited, with 120 million ha, agriculture intensification has led to rapid increases in annual crop yields, supported by tremendous consumption of fertilizers and fossil fuels (Huang and Sun 2006; Qiu et al. 2009; Pan et al. 2009; Fan et al. 2012). With this intensification, environmental problems have occurred, such as declines in cropland soil carbon stocks, excessive N<sub>2</sub>O emissions, and increases in cropland CO<sub>2</sub> emissions. Additionally, other unsuitable practices also contribute to large increases in carbon emissions from the agriculture sector; for example, the burning of crop residues in open fields led to 2707.34 Tg of CO<sub>2</sub> emissions in China from 1996 to 2013 (Sun et al. 2016). Consequently, carbon emissions from cropland increased from 43.07 Tg C in 1993 to 78.43 Tg C in 2008, with an increasing rate of 4.08% (Li et al. 2011). Thus, the identification of practices that mitigate carbon emissions from cropland in China is essential.

Many studies have examined how improved cropland practices reduce carbon emissions in China (Xu et al. 2011; Yan et al. 2015; Chuai et al. 2015; Ponsioen and Blonk 2012; Wang et al. 2014, 2015; Nayak et al. 2015), including conservation tillage (Cheng et al. 2014; Xu et al. 2012; Han et al. 2010; Wu et al. 2007), crop straw management (Lu et al. 2009 2010; Li et al. 2011; Cheng et al. 2013; Huang et al. 2013), manure inputs (Pan et al. 2009; Shen et al. 2007; Zhu et al. 2012), organic farming (Ouyang et al. 2011), and crop rotation (Huang et al. 2012). From these studies, the potential of Chinese

agriculture for carbon sequestration was estimated to range from 0.4 to 0.9 Pg C yr<sup>-1</sup> (Pan et al. 2003; Han et al. 2005; Yan et al. 2007; Lu et al. 2009). However, some of these practices are neither environmentally-friendly nor sustainable (e.g., the N<sub>2</sub>O emissions associated with large-scale fertilizer use). To bridge the gaps in information for China, our study aimed to (1) estimate the cropland carbon mitigation potentials of three environmentally-friendly, improved cropland practices of fertilizer substitution, crop straw return and conservation tillage; (2) calculate the direct and indirect carbon mitigation potentials of rice, wheat and maize croplands at different levels; (3) compare cropland carbon mitigation potentials with their carbon reduction targets; and (4) discuss regional cooperation can better achieve the regional and national carbon reduction targets.

### 1 Data and Methodology

We estimate the technical carbon mitigation potentials in China for three improved cropland practices: (1) Fertilizer Substitution with Manure:

to replace N, P and K fertilizers with manure to fully use the livestock excreta that is currently disposed as waste; (2) Crop Straw Return: to return more of the crop residues to croplands, including those that are currently and directly burnt in the open field; and (3) Conservation Tillage: to replace conventional tillage. For these three improved cropland practices, we estimated the direct carbon mitigation potentials (including the increase in carbon sequestration and the reduction in carbon emissions) and the indirect carbon mitigation potentials (including less carbon emissions from less external consumption of fertilizers and fossil fuels). The reference year for all data was 2008, which included areas, yields and incomes of rice, wheat and maize croplands, and the consumption of N, P and K fertilizers. The information is obtained from China Rural Statistical Yearbook, China Statistical Yearbook, and China Statistical Yearbook on Environment.

Considering the local climates, cropping varieties and cultivation practices, Chinese cropland were divided into four primary regions (See Figure 1) as follows (Lu et al. 2009):

Region I: Northeast China, including



**Figure 1** The distribution of the four cropland regions in China.

Heilongjiang, Jilin, and Liaoning.

Region II: North China, including Inner Mongolia, Shanxi, Hebei, Henan, Beijing, Tianjin, and Shandong.

Region III: Northwest China, including Shaanxi, Gansu, Ningxia, Qinghai, and Xinjiang.

Region IV: South China, including Jiangsu, Anhui, Shanghai, Zhejiang, Fujian, Jiangxi, Hubei, Hunan, Guangdong, Guangxi, Chongqing, Sichuan, Guizhou, Yunnan, Hainan and Tibet, with Hong Kong, Macao and Taiwan not being included in this study because of the lack of data.

### 1.1 Fertilizer substitution by manure

Approximately 28.3% of livestock excreta in China is disposed as waste (Qiu et al. 2012). For the practice of fertilizer substitution by manure, we supposed that the manure replaced chemical fertilizers, leading to less consumption of fertilizers. In the year of 2008, 3000 million ton (Mt) of livestock excreta were produced in China, with 850 Mt being disposed as waste (Sun and Song 2008). Sun et al. (2005) estimated that 1 t of wasted livestock excreta can replace 5.65 t, 1.13 t and 4.26 t of N, P and K fertilizers, respectively. Therefore, the total 850 Mt of wasted excreta could substitute for 4.8 Mt, 0.96 Mt and 3.62 Mt of N, P and K fertilizers, respectively. In 2008, the total consumption of the three croplands was approximately 12.4 Mt of N, 4.2 Mt of P and 2.9 Mt of K fertilizers, respectively. Thus, if the total wasted excreta was fully used as manure, 39% of N, 23% of P and 100% of K fertilizers would be saved, equivalent to the substitution rates.

As noted by Triberti et al. (2008), application of livestock manure is beneficial for soil organic carbon sequestration and storage, with a carbon sequestration rate of about 0.26 t C ha<sup>-1</sup> yr<sup>-1</sup>. Therefore, the direct carbon mitigation by fertilizer substitution originates from the increase in soil carbon sequestration. Thus, the direct carbon mitigation potential by “fertilizer substitution” was estimated with the following equation:

$$MC_{direct} = a_{mc} \sum A_i \quad (1)$$

where  $MC_{direct}$  denotes the direct carbon mitigation by fertilizer substitution,  $a_{mc}$  is the increase in soil carbon sequestration from manure inputs, approximately 0.26 t C ha<sup>-1</sup> yr<sup>-1</sup> (Triberti et al.

2008), and  $A_i$  is the area of rice, wheat or maize cropland.

In addition to the increase in soil carbon sequestration, fertilizer application also leads to carbon emissions during the processes of production and consumption (West and Marland 2002a). Thus, the lower fertilizer consumption would reduce this portion of carbon emissions, which is an indirect form of carbon mitigation that results from fertilizer substitution. Additionally, because of the limited data in China, our calculations relied on two conservative assumptions: (1) the carbon emissions are identical for the transport and delivery of manure and fertilizers, and (2) the emissions of CH<sub>4</sub> and N<sub>2</sub>O are identical from the disposal of waste livestock excreta and the application as manure. Thus, the indirect carbon mitigation from fertilizer substitution was estimated as follows:

$$MC_{indirect} = \sum b_{mci} CF_{mci} \quad (2)$$

where  $MC_{indirect}$  is the indirect carbon mitigation from fertilizer substitution,  $b_{mci}$  are carbon emissions from consuming N, P and K fertilizers, with the emission rates of 857.5 KgC Mg<sup>-1</sup> for N, 165.1 KgC Mg<sup>-1</sup> for P, and 120.3 KgC Mg<sup>-1</sup> for K fertilizers (West and Marland 2002a);  $CF_{mci}$  are the substitution rates for N, P and K fertilizers, approximately 39%, 23% and 100%, respectively.

### 1.2 Crop straw return

Because crop straw is a primary precursor of soil organic carbon, the return of crop straw to soils increases soil carbon storage (Smith et al. 2008). In China, crop straw is used for biomass fuel (47%), livestock forage (20%), industrial raw materials (3%), and returned to the soil (15%); however, the remaining 15% of crop straw is burned in open fields as waste, which increases the environmental problem of air pollution (Wang et al. 2008). Therefore, with the return of the crop straw from open-field burning, soil organic carbon inputs increase and the local environment also benefits. With all the open-field burned crop straw being returned to the soil; we assumed that the proportion of crop straw return increased from 15% to 30%. As noted by Lu et al. (2009), soil organic carbon storage is linearly correlated with the amount of crop straw returned; the relationships

are summarized by region in China as follow:

Region I (Northeast China):  $SCS = 40.254S + 340.33$

Region II (North China):  $SCS = 40.607S + 181.9$

Region III (Northwest China):  $SCS = 17.116S + 30.553$

Region IV (South China):  $SCS = 43.548S + 375.1$

where  $S$  is the crop straw returned ( $t\ ha^{-1}\ yr^{-1}$ ), and  $SCS$  is the soil carbon sequestration ( $kg\ C\ ha^{-1}\ yr^{-1}$ ).

The annual yield of crop straw ( $S_i$ ) was estimated by crop yield ( $Y_i$ ) and the straw-grain ratio ( $SGR_i$ ), with  $SGR$ s of 0.623, 1.366 and 2.000 for rice, wheat and maize, respectively (Lu et al. 2009).

$$S_i = Y_i \times SGR_i \quad (3)$$

Thus, crop straw return leads to an increase in soil carbon storage, which is the direct carbon mitigation potential. Moreover, no indirect carbon mitigation results from crop straw return. Consequently, the direct carbon mitigation potential by crop straw return was calculated as follows:

$$SRC_{direct} = SCS_{30\%} - SCS_{15\%} \quad (4)$$

where  $SRC_{direct}$  denotes direct carbon mitigation from crop straw return and  $SCS_{30\%}$  and  $SCS_{15\%}$  are levels to soil carbon sequestration with crop straw return ratios of 30% and 15%, respectively.

### 1.3 Conservation tillage

The dominant tillage practice in China, conventional tillage with moldboard plowing, changes the structure of soil organic matter and also increases microbial activity. Thus, compared with conservation tillage (primarily no-tillage), conventional practice increases the decomposition of organic matter, leading to an increase in carbon emissions (Wright and Hons 2005; Fabrizzi et al. 2009; Xu et al. 2011). Additionally, tractors consume fossil fuel in plowing during conventional tillage, leading to carbon emissions of 0.3868  $kg\ C\ ha^{-1}$  (Lal 2004; Wu et al. 2007). Therefore, with conservation tillage (particularly no-tillage), the carbon mitigation potential is significant (West and Post 2002c; Lal 2004; Smith et al. 2008). For conservation tillage, we assumed that 50% of the

cropland in conventional tillage was converted to conservation tillage. For this practice, the direct carbon mitigation refers to the carbon emissions from conventional tillage, whereas the indirect mitigations from less fossil fuel consumption. Thus, the potentials for direct and indirect carbon mitigation under conservation tillage were determined as follow:

$$NTC_{direct} = NTSC_i A_i \quad (6)$$

$$NTC_{indirect} = 0.3868 \times NTA_i \quad (7)$$

where  $NTC_{direct}$  and  $NTC_{indirect}$  are the direct and indirect carbon mitigation by conservation tillage,  $NTSC_i$  is soil carbon emissions by conventional tillage, with 255, 157, 390 and 198  $kg\ C\ ha^{-1}\ yr^{-1}$  for Regions I, II, III, and IV, respectively (Lu et al. 2009), and  $NTA_i$  is the area in conservation tillage (50% of the cropland area).

## 2 Results

The potentials for carbon mitigation with the application of the three improved practices in the three types of cropland are shown in Table 1 for all provinces in China (except for Hong Kong, Macao and Taiwan).

### 2.1 Carbon mitigation potential by fertilizer substitution

Substituting chemical fertilizer with manure in cropland contributed a carbon mitigation potential of 26.6  $Tg\ C\ yr^{-1}$  in China, approximately 320.5  $kg\ C\ ha^{-1}\ yr^{-1}$ , which included the direct mitigation of 21.6  $Tg\ C\ yr^{-1}$  and indirect the potential of 5.0  $Tg\ C\ yr^{-1}$ . The difference in carbon mitigation potentials of rice, wheat and maize were not significant after fertilizer substitution, which were 9.6, 7.6, and 9.3  $Tg\ C\ yr^{-1}$ , respectively. Geographically, the largest overall contributor was Henan at 2.8  $Tg\ C\ yr^{-1}$ , which had the largest direct and indirect carbon mitigation potentials of 2.3 and 0.6  $Tg\ C\ yr^{-1}$ , respectively. Following Henan, fertilizer substitution in Shandong also contributed a large potential of 2.1  $Tg\ C\ yr^{-1}$ , with the direct and indirect contributions of 1.7 and 0.4  $Tg\ C\ yr^{-1}$ , respectively. The carbon mitigation from this practice was relatively small in Tibet (0.01  $Tg\ C\ yr^{-1}$ ) and Qinghai (0.03  $Tg\ C\ yr^{-1}$ ), because both

**Table 1** Cropland carbon mitigation potential from improved practices in China (Unit: Tg C yr<sup>-1</sup>)

Region	Province	Fertilizer substitution			Straw return	Conservation tillage			Total		
		Total	Direct	Indirect	Direct	Total	Direct	Indirect	Total	Direct	Indirect
Region I	Liaoning	0.83	0.66	0.16	0.16	0.33	0.33	4.9E-04	1.32	1.15	0.17
	Jilin	1.11	0.93	0.17	0.27	0.46	0.46	6.9E-04	1.84	1.66	0.17
	Heilongjiang	1.79	1.64	0.15	0.28	0.80	0.80	1.2E-03	2.88	2.73	0.15
Region II	Beijing	0.07	0.05	0.02	0.01	0.02	0.02	4.1E-05	0.10	0.08	0.02
	Tianjin	0.10	0.07	0.03	0.02	0.02	0.02	5.5E-05	0.14	0.11	0.03
	Hebei	1.73	1.39	0.35	0.28	0.42	0.42	1.0E-03	2.43	2.09	0.35
	Shanxi	0.62	0.54	0.08	0.10	0.16	0.16	4.0E-04	0.89	0.81	0.08
	Inner Mongolia	0.87	0.75	0.11	0.19	0.23	0.23	5.6E-04	1.28	1.17	0.11
	Shandong	2.09	1.70	0.39	0.40	0.51	0.51	1.3E-03	3.01	2.61	0.40
	Henan	2.83	2.26	0.57	0.47	0.68	0.68	1.7E-03	3.98	3.41	0.57
Region III	Shaanxi	0.80	0.63	0.17	0.04	0.47	0.47	4.7E-04	1.31	1.14	0.17
	Gansu	0.43	0.38	0.05	0.02	0.29	0.29	2.8E-04	0.74	0.69	0.05
	Qinghai	0.03	0.03	0.00	0.00	0.02	0.02	2.1E-05	0.05	0.05	0.00
	Ningxia	0.21	0.13	0.08	0.01	0.10	0.10	9.5E-05	0.31	0.24	0.08
	Xinjiang	0.49	0.40	0.09	0.04	0.30	0.30	3.0E-04	0.83	0.73	0.09
Region IV	Shanghai	0.06	0.04	0.02	0.01	0.02	0.02	3.0E-05	0.08	0.06	0.02
	Jiangsu	1.69	1.22	0.46	0.19	0.47	0.47	9.1E-04	2.34	1.88	0.46
	Zhejiang	0.36	0.26	0.10	0.03	0.10	0.10	2.0E-04	0.50	0.40	0.10
	Anhui	1.64	1.37	0.27	0.20	0.52	0.52	1.0E-03	2.37	2.09	0.28
	Fujian	0.32	0.23	0.09	0.02	0.09	0.09	1.7E-04	0.44	0.35	0.09
	Jiangxi	0.98	0.85	0.13	0.08	0.33	0.32	6.3E-04	1.38	1.25	0.13
	Hubei	1.20	0.90	0.30	0.12	0.34	0.34	6.7E-04	1.66	1.36	0.30
	Hunan	1.35	1.09	0.26	0.13	0.42	0.41	8.1E-04	1.89	1.63	0.26
	Guangdong	0.76	0.54	0.21	0.05	0.21	0.21	4.0E-04	1.01	0.80	0.21
	Guangxi	0.82	0.68	0.14	0.07	0.26	0.26	5.1E-04	1.16	1.01	0.15
	Hainan	0.11	0.09	0.03	0.01	0.03	0.03	6.3E-05	0.15	0.12	0.03
	Chongqing	0.43	0.34	0.08	0.06	0.13	0.13	2.5E-04	0.62	0.53	0.09
	Sichuan	1.46	1.21	0.25	0.18	0.46	0.46	9.0E-04	2.10	1.85	0.25
	Guizhou	0.51	0.44	0.07	0.07	0.17	0.17	3.3E-04	0.75	0.68	0.07
	Yunnan	0.89	0.72	0.17	0.10	0.27	0.27	5.4E-04	1.27	1.10	0.17
Tibet	0.01	0.01	0.00	0.00	0.00	0.00	8.2E-06	0.02	0.02	0.00	
Total		26.59	21.57	5.02	3.62	8.63	8.61	0.02	38.83	33.80	5.04

cropland and fertilizer consumption were limited there.

## 2.2 Carbon mitigation potential by crop straw return

In 2008,  $6.06 \times 10^8$  t of crop straw were produced in China, including  $1.20 \times 10^8$ ,  $1.54 \times 10^8$  and  $3.32 \times 10^8$  t from rice, wheat and maize. When crop straw return increased from 15% to 30%, soil carbon sequestration increased from 3.9 Tg C yr<sup>-1</sup> to 7.5 Tg C yr<sup>-1</sup>, an approximately 30% rate of increase. Thus, the direct carbon mitigation potential was 3.6 Tg C yr<sup>-1</sup> or 43.6 kg C ha<sup>-1</sup> yr<sup>-1</sup> because of the increase in crop straw return. The contribution of straw return to carbon mitigation potential showed large divergences among maize, rice and wheat croplands in China. The crop straw return in maize

contributed the largest carbon mitigation potential of 2.0 Tg C yr<sup>-1</sup>, accounting for 53.9% of the total potential mitigation by this practice. The potentials from straw return in rice and maize croplands were 0.8 and 0.9 Tg C yr<sup>-1</sup>, respectively. Henan and Shandong also had large mitigation potentials of 0.5 and 0.4 Tg C yr<sup>-1</sup> because of the large production of crop straw. Crop straw production in minimal in Tibet, and therefore, an increase in crop straws was only a small contribution, with carbon mitigation potential of 0.003 Tg C yr<sup>-1</sup>.

## 2.3 Carbon mitigation potential by conservation tillage

The total carbon mitigation potential with conservation tillage was 8.6 Tg C yr<sup>-1</sup> or 104.0 kg C ha<sup>-1</sup> yr<sup>-1</sup>, 99.8% of which was direct potential



mitigation. Under conservation tillage, the contributions to carbon mitigation from rice, wheat and maize croplands were 3.0, 2.4, and 3.2 Tg C yr<sup>-1</sup>, respectively. The largest mitigation potential of 0.8 Tg C yr<sup>-1</sup> was Heilongjiang in northeast China, followed by Henan in north China with 0.7 Tg C yr<sup>-1</sup>. Similar to the mitigation potential of the other practices, Tibet also had the smallest potential of 0.004 Tg C yr<sup>-1</sup> under conservation tillage, because of the limited area in cropland.

#### **2.4 The total carbon mitigation potential of the three practices combined**

Combined, the total carbon mitigation potential of the three practices was 38.8 Tg C yr<sup>-1</sup> in China, or 468.2 kg C ha<sup>-1</sup> yr<sup>-1</sup> (Table 1). For China, the carbon mitigation potential represented an equivalent of 49.4% of the total CO<sub>2</sub> emissions from cropland in 2008 and 1.6% of CO<sub>2</sub> emissions from fossil fuels in 2011 (Le Quéret et al. 2013; Li et al. 2011). Thus, compared with the rapid increase in carbon emissions in China, the total carbon mitigation by these practices was very limited; however, the contributions remain essential to achieve Chinese carbon mitigation targets. Additionally, the direct carbon mitigation summed to 33.8 Tg C yr<sup>-1</sup>, which was 87.0% of total potential of these three practices combined.

Furthermore, fertilizer substitution contributed the largest direct mitigation potential of 63.8%, followed by that of conservation tillage with 25.4%. The remaining mitigation was achieved by crop straw return. For the indirect carbon mitigation, fertilizer substitution with livestock manure was responsible for most of its mitigation, with approximately 5.02 Tg C yr<sup>-1</sup>, and the remaining mitigation potential of 0.02 Tg C yr<sup>-1</sup> occurred under conservation tillage. Overall, fertilizer substitution was the largest contribution to carbon mitigation potentials, followed by conservation tillage and crop straw return. When the mitigation potentials of the three improved practices were combined, maize cropland had the largest carbon mitigation potential of 14.5 Tg C yr<sup>-1</sup>, accounting for 37.3% of the total potential. Rice and wheat cropland also contributed large potentials of 13.4 and 11.0 Tg C yr<sup>-1</sup>, respectively. Geographically, south China had the largest carbon mitigation of 17.7 Tg C yr<sup>-1</sup>, followed by northern

China with 11.8 Tg C yr<sup>-1</sup>, northeast China with 6.0 Tg C yr<sup>-1</sup>, and western China with 3.3 Tg C yr<sup>-1</sup>. The two provinces with the largest mitigation potentials were Henan and Shandong because of the decrease in consumption of fertilizers and fossil fuels. However, the carbon mitigation potentials were only 0.05 Tg C yr<sup>-1</sup> and 0.02 Tg C yr<sup>-1</sup> in Qinghai and Tibet, respectively, because of the limited cropland and limited consumption of fertilizers and fossil fuels.

### **3 Discussion**

With the implementation of these improved practices, the total carbon mitigation potential was 38.8 Tg C yr<sup>-1</sup> for the three primary Chinese croplands, which was equivalent to 49.4% of CO<sub>2</sub> emission from agriculture and 1.6% of CO<sub>2</sub> emissions from fossil fuels. Although this mitigation potential is much smaller than the large increase in Chinese carbon emissions, this mitigation represents a substantial absolute potential to avoid further emissions. More importantly, these practices provide additional benefits for the long-term sustainability of crop cultivation in China, such as rebuilding soil carbon in depleted agricultural soils, reducing air pollution from open-field burning of crop straw, and improving soil organic carbon intensity.

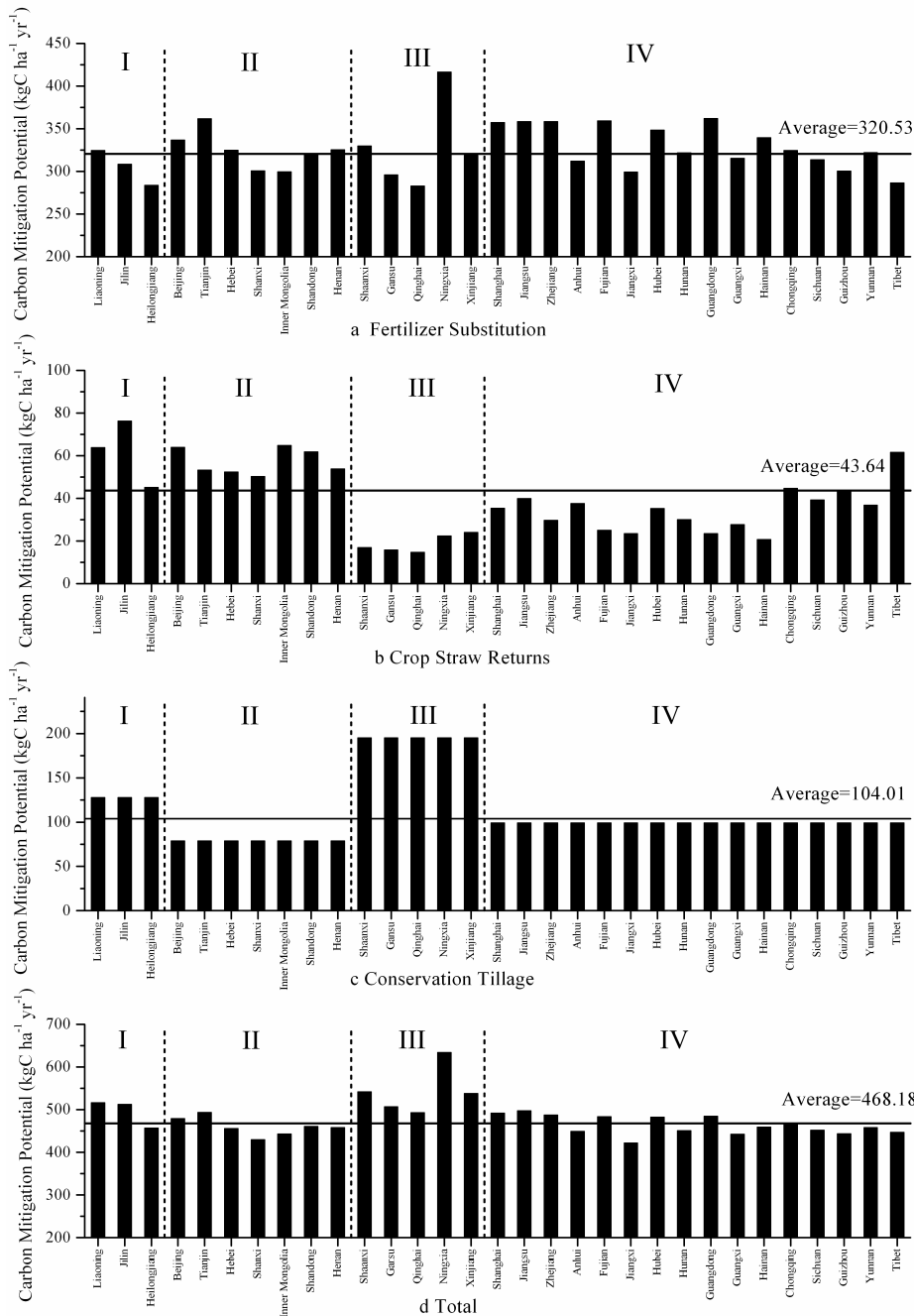
#### **3.1 Regional differences for carbon mitigation potential**

Geographically, south China contributed the largest potential of 17.7 Tg C yr<sup>-1</sup>, followed by northern China with 11.8 Tg C yr<sup>-1</sup>, northeast China with 6.0 Tg C yr<sup>-1</sup>, and western China with 3.3 Tg C yr<sup>-1</sup> (Table 2 and Figure 2). However, northwest China had the largest carbon mitigation potential per unit area, followed by northeast China, and both of their potentials were higher than the national average. The mitigation potentials were relatively small in north and south China.

Northwest China had the largest unit area potential of 538.9 kg C ha<sup>-1</sup> yr<sup>-1</sup>, with the contribution of fertilizer substitution being the largest mitigation potential. In the northwest China, all provinces had higher mitigation potentials per unit area than the national average, with the largest

**Table 2** The potentials of carbon mitigation per area in different regions by improved cropland practices in China (Unit: kg C ha<sup>-1</sup> yr<sup>-1</sup>)

Region	Fertilizer substitution		Crop straw return		Conservation tillage		All three practices together		
	Amount	Percent	Amount	Percent	Amount	Percent	Total Potential	Direct Potential	Indirect Potential
Region I	299.21	61.7%	57.94	11.9%	127.69	26.3%	484.84	445.44	39.40
Region II	319.63	70.3%	56.52	12.4%	78.69	17.3%	454.85	395.02	59.83
Region III	324.93	60.3%	18.83	3.5%	195.19	36.2%	538.95	473.83	65.12
Region IV	327.35	71.0%	34.19	7.4%	99.19	21.5%	460.73	393.19	67.54
China	320.53	68.5%	43.64	9.3%	104.01	22.2%	468.18	407.46	60.72



**Figure 2** Cropland carbon mitigation potential per area with improved cropland practices in China.

potential in Ningxia. The local climate is semi-arid in these areas and soil erosion caused by conventional tillage and deforestation can be extensive; thus, with the application of conservation tillage to this area, the carbon mitigation potential of 195.2 kg C ha<sup>-1</sup> yr<sup>-1</sup> was very high. Because of the long-term and historical soil erosion in this region, intensive fertilizer applications are required, and therefore, by substituting chemical fertilizers with manure, the carbon mitigation potential was large. However, because of limited crop straw production in northwest China, less crop straw would be returned back. Thus, the mitigation potential from the return of crop straw was much lower than that of the national level there.

Northeast China is famous for the local fertile soils with high



**Table 3** The carbon mitigation potentials in rice, wheat and maize croplands in four regions (Unit: million ha for area and Tg C yr<sup>-1</sup> for the carbon mitigation potential)

Region	Rice			Wheat			Maize		
	Area	Potential	Percent	Area	Potential	Percent	Area	Potential	Percent
Region I	3.71	1.69	28.0%	0.28	0.12	2.1%	8.45	4.22	70.0%
Region II	0.93	0.41	3.5%	12.52	5.57	47.1%	12.56	5.84	49.4%
Region III	0.28	0.16	5.0%	3.23	1.71	52.6%	2.51	1.38	42.4%
Region IV	24.32	11.14	62.8%	7.75	3.55	20.0%	6.39	3.03	17.1%
Total	29.24	13.40	34.5%	23.79	10.95	28.2%	29.92	14.48	37.3%

carbon storage; thus, compared with conventional tillage practices, conservation tillage was beneficial in the mitigation of carbon emissions from these soils. However, also because of the local fertile soils, fertilizers consumption in this region is low, and therefore, fertilizer substitution with manure proved only a relatively small carbon mitigation potential. Maize is the dominant crop in northeast China and there are a large area of maize cropland. Consequently, with high straw-grain ratio crop straw return contributed to a relatively high carbon mitigation potential, compared with other regions.

North and south China had relatively low carbon mitigation potential per unit area. The low soil fertility results in much larger consumption of fertilizers than that in other regions, and therefore, fertilizer substitution had relatively large carbon mitigation potential in these two regions. Moreover, conventional tillage had a much smaller effect on the local soil carbon stock, compared with that in northeast and northwest China. Rice is the dominant crop in south China, with a low straw-grain. Therefore, crop straw production in south China is relatively low, and crop straw return contributed only a small mitigation potential in south China.

Overall, with fertilizer substitution, northwest and south China had large carbon mitigation potential per unit area because of the local extreme consumption of fertilizers; for crop straw return, northeast and north China had relatively high carbon mitigation potentials per unit area; and with conservation tillage, northwest and northeast China had high carbon mitigation potentials per unit area.

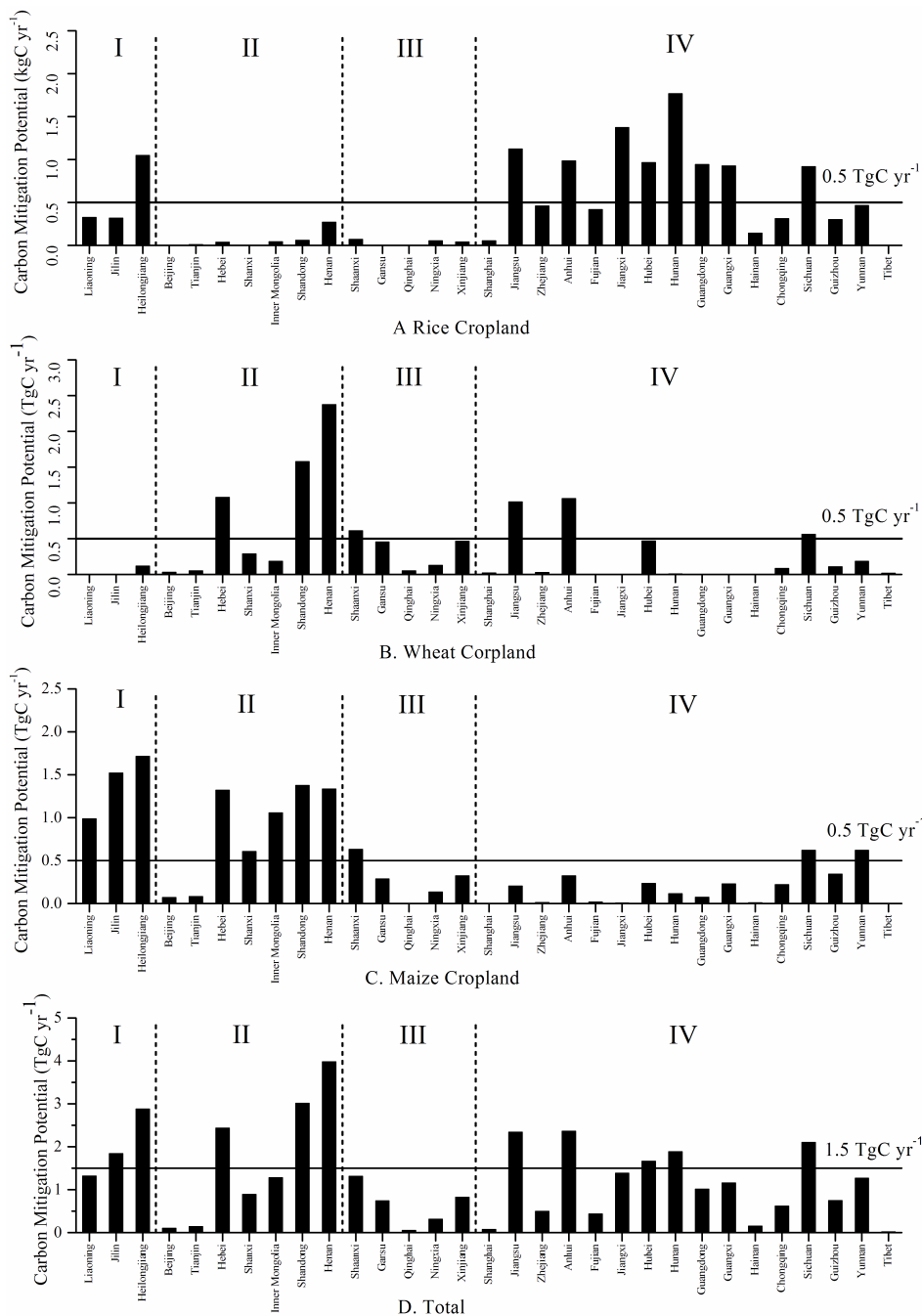
### 3.2 Comparison analysis among three croplands in China

For the combined effects of these three mitigation practices, maize cropland had the largest carbon mitigation potential of 14.5 Tg C yr<sup>-1</sup>,

followed by rice and wheat croplands (Table 3, Figure 3). In northeast China, rice and maize croplands contributed the most potential to the total carbon mitigation potentials, with the contribution of wheat cropland only being 2.1%. In northeast China, the scenario was similar with all provinces. In north China, wheat and maize contributed 47.1% and 49.4% to total mitigation potential, respectively. However significant differences were detected among provinces, because the dominant crops are different; wheat cropland are dominant in Shandong and Henan, while maize is the primary crop in the other 5 provinces. Maize contributed the largest proportion of mitigation potentials in all provinces in northwest China. In south China, the largest contribution was from rice in most provinces (with the only exceptions of Anhui and Tibet), which accounted for 62.8% of the total potential, but maize and wheat also contributed 3.55 Tg C yr<sup>-1</sup> and 3.03 Tg C yr<sup>-1</sup>, to carbon mitigation, respectively.

**Rice cropland:** The largest contribution of rice to the mitigation potential was 1.77 Tg C yr<sup>-1</sup> in Hunan, followed by Jiangxi with 1.37 Tg C yr<sup>-1</sup>. Nine other provinces had potentials from rice larger than 0.5 Tg C yr<sup>-1</sup>, including Heilongjiang, Jiangsu, Anhui, Jiangxi, Hubei, Guangdong, Hunan, Guangxi, and Sichuan. Because of the local weather, only a few cropland areas of rice are found in northwest China and north China; thus no province in these two regions contributed more than 0.5 Tg C yr<sup>-1</sup> of carbon mitigation potential from rice cropland.

**Wheat cropland:** With the climate suitable, wheat is the primary important food crop in north and northwest China, and large areas of wheat cropland are found in these two areas; thus, wheat cropland had relatively large carbon mitigation potentials in the provinces of these regions. In northeast and south China, because of the limited wheat cropland, the contribution of most provinces to carbon mitigation potentials was limited.



**Figure 3** Carbon mitigation potentials in rice, wheat and maize croplands in China.

However, wheat cropland contributed more than 0.5 Tg C yr<sup>-1</sup> of carbon mitigation in Jiangsu, Anhui, and Sichuan.

**Maize cropland:** Maize has an important role in local diet in north and northeast China, which is supported by large areas of maize cropland in these two regions. With the exceptions of Beijing and Tianjin, the potential from maize cropland was more than 0.5 Tg C yr<sup>-1</sup> in all other provinces, particularly in Heilongjiang, Jilin,

Shandong, Henan and Hebei. In northwest and south China, carbon mitigation potentials were relatively small from maize cropland and only three provinces (Shaanxi, Sichuan, and Yunnan) had mitigation potential of more than 0.5 Tg C yr<sup>-1</sup>.

### 3.3 Cropland carbon mitigation and regional cooperation

At the Conference of the Parties 15 of the United Nations Framework Convention on Climate Change in Copenhagen, the Chinese government committed to the implementation of the 2020 carbon reduction target of 40%-45% below the 2005 level of carbon emission intensity (Fan and Xia 2012). As calculated above, the three improved practices provided a total carbon mitigation potential of 38.8 Tg C yr<sup>-1</sup> in China, equivalent to 49.4% of CO<sub>2</sub> emissions from agriculture and 1.6% of CO<sub>2</sub> emissions from fossil

fuels. However, significant differences were found for carbon mitigation potentials among provinces in China. For example, based on a comparison of carbon emissions in one previous study (Li et al. 2011), the combined mitigation potential of the three practices in Heilongjiang was 95% of the cropland carbon emissions; whereas in Zhejiang, the combined contribution was only 19% of total cropland carbon emissions.

Soil carbon sequestration and carbon emission

mitigation were included in the Kyoto Protocol and can be used for carbon trading for a “Clean Development Mechanism” (CDM) or in an “Emission Trading Scheme” (ETS) (Jawson et al. 2005). Thus, we compared the cropland carbon mitigation targets with soil carbon sequestration potential (i.e., the direct cropland carbon mitigation). When the mitigation potential was larger than the target, we assumed it was a “target surplus”; whereas, we assumed it was a “target deficit” when the mitigation potential was less than the reduction target.

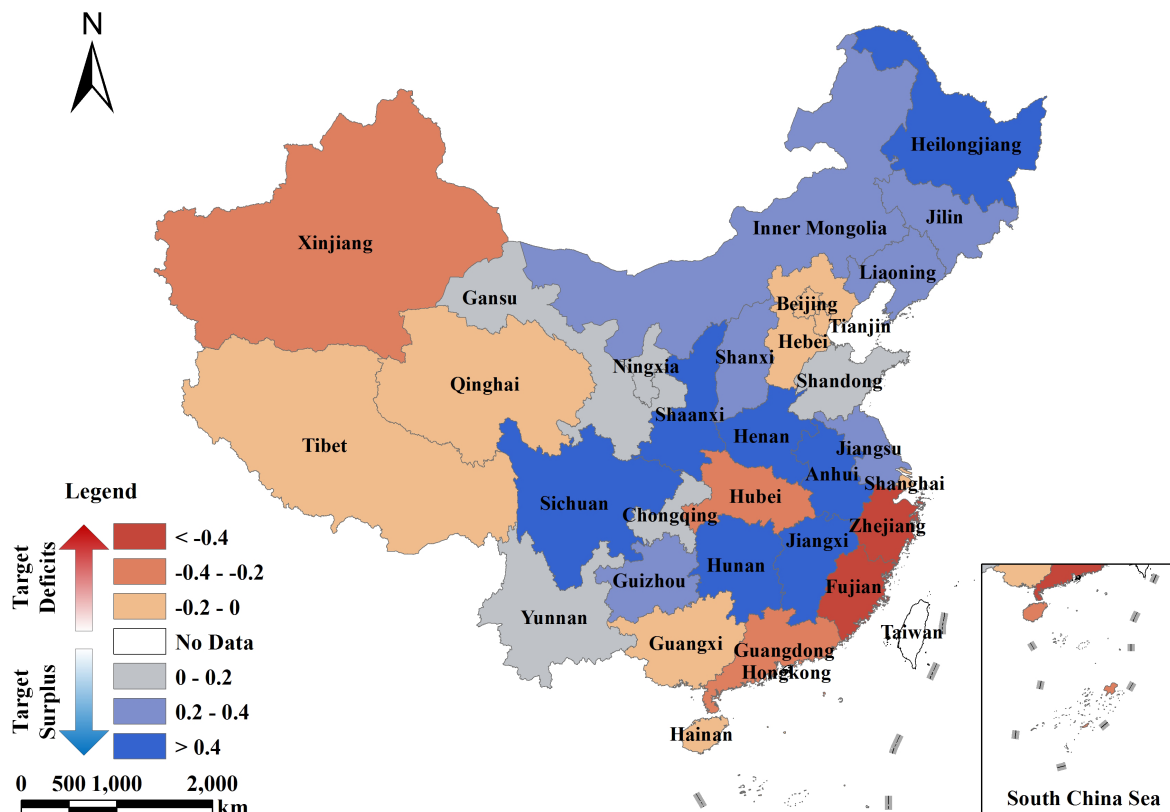
Using the results from Li et al. (2011), we calculated the carbon reduction target (40% of carbon emissions in 2005) for Chinese cropland by 2020. The relationship between cropland carbon mitigation target and soil carbon sequestration potential is illustrated in Figure 4. We concluded that 18 provinces had the “target surplus”, with the implementation of the improved cropland practices.

However, 13 provinces in China (with the exceptions of Hong Kong, Macao and Taiwan) remained with “target deficits”, including Beijing, Tianjin, and Hebei in northeast China, Qinghai and Xinjiang in northwest China, in addition to

Shanghai, Zhejiang, Fujian, Hunan, Guangxi, Hainan and Tibet in south China. In these provinces, the cropland carbon mitigation potentials of these improved practices did not meet the mitigation targets. The largest “target deficit” was 0.54 Tg C yr<sup>-1</sup> in Zhejiang, indicating that further improvements in cropland practices are required to mitigate carbon emissions or carbon credits must be bought to meet their targets.

The use of these improved practices in Heilongjiang contributed to the largest “target surplus” of 1.62 Tg C yr<sup>-1</sup> in China, followed by Henan (0.70 Tg C yr<sup>-1</sup>) and Sichuan (0.63 TgC yr<sup>-1</sup>). This “target surplus” can be credited for domestic or international carbon trading. Taking all provinces as a whole, the total “target surplus” was 7.10 Tg C yr<sup>-1</sup>, whereas the total “target deficits” was only 2.26 Tg C yr<sup>-1</sup>. Thus, “target deficits” in the 13 provinces could be fully compensated for domestic carbon trading. Additionally, a further 4.84 Tg C yr<sup>-1</sup> of “carbon mitigation potential” from Chinese cropland could be traded with the implementation of the three improved practices combined.

Apart from “CDM” or “ETS”, regional



**Figure 4** The map of cropland “target surplus” and “target deficits” in China.

cooperation for carbon mitigation can also be achieved with regional associations (such as the “Association of Southeast Asian Nations”) and research programs (such as GRACEnet (greenhouse gas reduction through agricultural carbon enhancement network) in the U.S) (Jawson et al. 2005; Lee et al. 2013). ASEAN aims to cooperate in the adaptation to and the alleviation of global warming with the principles of equity, flexibility, and effectiveness (Lee et al. 2013). For GRACEnet, the aims are to identify and develop agricultural strategies that will increase soil C sequestration and reduce greenhouse gas emissions, in addition to providing a scientific basis for possible carbon credit and trading programs (Jawson et al. 2005). Thus, regional associations and research programs can also be developed to enhance soil carbon sequestration, to reduce carbon mitigation requirements, and to provide credits for carbon trading.

### 3.4 Discussion of other greenhouse gases, mitigation practices and interactive effects

After CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O are the largest contributors to human-induced climate change (Denman et al. 2007). Manure input and straw return increase soil carbon sequestration and avoid some of the carbon emissions; however, the emissions of cropland CH<sub>4</sub> and N<sub>2</sub>O also increase with these practices. With the improved cropland practices, however, the currently disposed of useless livestock excreta and crop straw is used, which also emit CH<sub>4</sub> and N<sub>2</sub>O during decomposition and open-field burning. Thus, based on the net balance, the improved practices in this study did not reduce the further increase in CH<sub>4</sub> and N<sub>2</sub>O emissions. In this study, we adopted a conservative approach because of the lack of sufficient regional data and considered that there was no change in total emissions between the two possible fates of manure and crop residues.

Moreover, we did not consider other mitigation practices that reduce emissions of CH<sub>4</sub> and N<sub>2</sub>O (Smith et al. 2007; Smith et al. 2013). For example, improved water management for rice cultivation leads to significant reductions in CH<sub>4</sub> emissions by shifting from continuous flooding to mid-season drainage (Zhou et al. 2013; Li et al.

2005). For N<sub>2</sub>O emissions, many options are available to reduce excess nitrogen inputs and to improve nitrogen use efficiency in cropland systems (Tian et al. 2013).

Additionally, apart from the improved practices discussed above, many other improved practices increase soil carbon sequestration. In recent studies, the persistent carbon sink is C occluded within plant phytoliths, and rice, wheat and maize are the predominant dominant phytolith producing crop species (Parr and Sullivan 2011; Song et al. 2014; 2015; Guo et al. 2015). The size of this soil carbon sink has doubled since 1978, which should attract more attention to this persistent pool, and many improved agricultural management practices can further increase the sink of cropland phytolith C (Song et al. 2014).

Furthermore, in the current study, we focused on the three practices separately to estimate their carbon mitigation potentials and then summed the potentials of the three practices. However, when the three practices are applied together, the interactive effects among these three practices may influence the total carbon mitigation potential. Additionally, different cropland with different land-use type could have different carbon emissions, which is also should be considered in future research. Therefore, in the future, the influences of combined practices on carbon mitigation potential should be the focus of research, in addition to examining the mitigation potentials of other practices.

## 4 Conclusion

We estimated the carbon mitigation potential for rice, wheat and maize croplands in China with the implementation of three improve cropland practices, i.e., fertilizer substitution, crop straw return and conservation tillage. Combined, the three practices had a total carbon mitigation potential of 38.8 Tg C yr<sup>-1</sup> (468.2 kg C ha<sup>-1</sup> yr<sup>-1</sup>) in China. However, these three practices affected mitigation potentials differently, with the largest mitigation potential of 26.6 Tg C yr<sup>-1</sup> with fertilizer substitution. For the different croplands, the carbon mitigation potentials were not significantly different.

However, because of local diets and weather,

carbon mitigation potentials and the effects of the improved practices diverged significantly among provinces in China. When cropland carbon mitigation potentials were compared with their carbon reduction targets, 18 provinces met their cropland carbon mitigation targets with these three practices, whereas the other 13 provinces remained with a “target deficit”. However, with the three practices combined, the national cropland carbon reduction target could be achieved through regional cooperation, with an additional carbon reduction of 4.84 Tg C yr<sup>-1</sup> available for international or national carbon trading. Thus, the three improved agriculture practices could be very

useful in mitigating carbon emissions in China. Additionally, to better achieve carbon mitigation targets, regional cooperation is extremely important in implementing carbon trading, establishing regional associations, and conducting research programs, in addition to improving other management practices.

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