Extension of residue retention increases net greenhouse gas mitigation in China's croplands

Guo Zhang ^{a, b}, Xiaoke Wang ^{a, c}, Hong Zhao ^a, Binfeng Sun ^a, Fei Lu ^{a, c, *}, Lirong Hu ^a

^a State Key Laboratory of Urban and Regional Ecology, Research Center for Eco-Environmental Sciences, Chinese Academy of Sciences, Beijing 100085, China

^b Institute of Karst Research, Guizhou Normal University, Guiyang 550001, China

^c Joint Center for Global Change Studies, Beijing 100875, China

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ABSTRACT

Crop residue retention plays an important role in greenhouse gas (GHG) mitigation for global agriculture. To clarify the current status of residue retention and burning as well as the contribution of these processes to GHG dynamics, we conducted a nationwide survey on residue management in China in 2011 and improved the Carbon Accounting and Net Mitigation of Straw return model with updated parameters. The results indicated that residue retention increased from 144 Mt yr⁻¹ in 2001 to 295 Mt yr⁻¹ in 2011, whereas burned residue reached 200 Mt yr⁻¹ in China. Because of these increases in residue retention, the net mitigation related to residue retention increased from 5.3 Tg Ce in 2001 to 13.3 Tg Ce in 2011. Further extension would sequester 15.8 Tg C, which is equivalent to a reduction in emissions of 3.9 Tg Ce from residue burning and a mitigate potential of 3.1 Tg Ce from replacing 1.5 Mt of nitrogen fertilizer. However, the national net mitigation from residue retention would achieve only 45–64% of the total soil C sequestration. The spatial heterogeneity of cropping systems and residue management was determined to have different effects on GHG dynamics. The net mitigation would have a negative value in eight provinces because of incremental increases in methane emissions from rice paddies. Increases in residue retention in the remaining 23 provinces would result in a maximum mitigation potential of 10 Tg Ce yr⁻¹ and offset the carbon dioxide emissions caused by fossil fuel burning by 0.5% from the national value in 2011. Therefore, residue retention has mitigated a substantial amount of GHGs, and extending this strategy has considerable mitigation potential for China's croplands. Our results indicate that the retention and burning of crop residues have an effect on the GHG dynamics in China and represent potential strategies for mitigating climate change via residue management.

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1. Introduction

Reducing greenhouse gas (GHG) emissions and enhancing GHG sinks for climate change mitigation represent global challenges (Lal, 2004a). Agricultural production directly contributes to soil organic carbon (SOC) losses and anthropogenic GHG emissions (Smith, 2012). The carbon (C) lost from China's croplands from 1850 to 2000 was estimated at 5.86 Pg C (Houghton, 2002). Furthermore, the uplands of China emitted 3.0 Pg Ce (100-yr time horizon) of nitrous oxide (N₂O), and rice paddies emitted 3.1 Pg Ce of methane (CH₄) between 1961 and 2005 (Tian et al., 2011). The considerable C

loss and current GHG emissions indicate that proper cropland management in China has a significant GHG mitigation potential (Lal, 2004b) because such management, including residue retention, recommended fertilization schedules and no-till farming, can mitigate atmospheric GHG levels by reducing emissions and sequestering C in biomass and soil (Jin et al., 2014).

The management of crop residues affects the C and nitrogen (N) cycles on croplands (Liu et al., 2014). In China, residue burning has been widely adopted during harvesting for faster removal with lower associated costs (Shi et al., 2014). In 2001, residue burning in China emitted 2.1 Tg Ce yr⁻¹ of CH₄ and 0.6 Tg Ce yr⁻¹ of N₂O (Lu et al., 2010a). Residue retention not only mitigates GHG emissions but also improves soil C sinks (Petersen et al., 2013; Zhao et al., 2014). Furthermore, residue retention can act as a substitute for synthetic N fertilizers (Gentile et al., 2009), thereby mitigating GHG emissions from N fertilizer production. However, residue retention

* Corresponding author. 18 Shuangqing Road, Haidian District, Beijing 100085, China.

E-mail address: feilu@rcees.ac.cn (F. Lu).

has also been found to increase CH₄ emissions from rice paddies (Cheng et al., 2014) as well as GHG emissions because of the additional fuel used for tilling and ploughing (Lu et al., 2010b). Therefore, all of these factors must be considered when estimating the net effects of residue retention on GHG emissions.

Several studies have focused on the GHG dynamics of residue retention in China. Using the Straw Return and Burning Model (SRBM) and provincial residue utilization information from 2001, the net mitigation potential (NMP) obtained by replacing residue burning with residue retention at the provincial scale was 6.3 Tg Ce yr⁻¹ (2010a), whereas when all residue from nine crops (e.g., rice, wheat, corn, sorghum, potato, rapeseed, sunflower, cotton and sugarcane) cultivated across China's croplands was returned, the NMP was 9.3 Tg Ce yr⁻¹ (2010b). Lun et al. (2016) estimated the combined effect of performing fertilizer substitution with manure, residue retention, and conservation tillage on C mitigation in rice, wheat and corn in 2008 and generated a mitigation target set for 2020 in China. However, in these studies, the scenario settings and parameters were based on the published literature rather than on first-hand information from systematic surveys, or they did not consider non-CO₂ GHG in the mitigation estimations.

China accounts for the third largest area of croplands in the world. The total residue product on China's cropland increased from 660 to 807 Mt yr⁻¹ between 2000 and 2009 (Jiang et al., 2012), whereas the use of residue as fuel decreased following economic development (Yao et al., 2012). Furthermore, Chinese governmental policies have been implemented to encourage the comprehensive utilization of crop residues (MEP, 1999; MOA, 2007). We hypothesized that these factors would stimulate an increase in the implementation of residue retention, which would affect the GHG budget of residue retention and burning. Our objectives were to (1) clarify the current status of residue retention and burning; (2) analyze the dynamics of realized net mitigation (RNM) of residue retention over the past decade; and (3) estimate the NMP under a full extension of residue retention in place of residue burning on China's croplands in 2011. To achieve these objectives, we conducted a nationwide survey of farmers between 2012 and 2013. By addressing these objectives, we can clarify the contributions of residue retention to GHG mitigation on China's croplands.

2. Materials and methods

We considered nine crops in China for this study: rice, wheat, corn, sorghum, potato, rapeseed, sunflower, cotton and sugarcane. The nine crops represent the main cereal or economic crops in China. The residue from other crops is typically used as forage or fuel because of the higher nutrient content or caloric content. The harvested area of these crops accounted for 67% of the total harvested area in China in 2011 (NBS, 2012).

2.1. Study region

In this study, Chinese croplands were divided into four cropping regions according to the heterogeneity of climate, cropping system and cultivation (Lu et al., 2009): northeast, north, northwest and south (Fig. 1). The cropping systems mainly involved one single crop per year in northeastern and northwestern China, double-cropping systems in northern China, and double or even triple cropping in southern China.

2.2. Field survey

From 2012 to 2013, we conducted a field survey of farmers on cropland management practices implemented in 2011 in 31

Chinese provinces (Fig. 1). The survey process involved a pilot survey, interviewer training, a formal survey and data entry and cleaning. This survey process is described in detail in Zhang et al. (2016). The survey included questions on the crop species and the percentage of croplands with residue retention and burning. In addition, triangulation questions were included on crop yields, residue retention types, and farmer attitudes toward residue retention, which were designed to ensure the questionnaire validity. We evaluated 752 valid farmer questionnaires, which included 1126 questionnaires from different crop seasons. The invalid questionnaires were those without any residue management or a sum of the percentage of all residue management practices that was more than 100. In addition, we conducted a survey on equipment use and the resulting fossil fuel consumption during cropland management from specialized agricultural machinery. Additionally, we obtained 40 valid questionnaires on machinery practices in China.

2.3. Data sources

In this study, the harvested area, cropland area and yield of the nine studied crops in 2001 and 2011 were obtained from the Chinese Statistical Yearbook from 2002 (NBS, 2012) and 2012 (NBS, 2012), respectively. The percentage of cropland with residue retention (PR) and burned residue (PB) in 2001 was obtained from studies by Lu et al. (2009) and Cao et al. (2008), and those for 2011 were obtained from the field survey.

Multiple cropping systems are widespread on China's croplands. In single-cropping systems, the harvested area equals the cropland area, whereas in multiple cropping systems, the harvested area is double or triple the size of the cropland area. The study area focused on cropland area that cultivated the nine studied crops (A_i), and it was calculated using formula 1 (Lu et al., 2009):

$$A_i = TCA_i \cdot \left(\sum HA_{ij} \right) / THA_i \quad (1)$$

where TCA_i represents the total cropland area of all crops in province i ; HA_{ij} represents the harvested area of crop j (rice, wheat, corn, sorghum, potato, rapeseed, sunflower, cotton and sugarcane) in province i ; and THA_i represents the total harvested area of all crops in 2011 province i (NBS, 2012). All areas have a unit of 10^3 ha.

2.4. Estimation of the GHG dynamics of residue retention and burning

We improved the SRBM model (Lu et al., 2010b) with a C Accounting and Net Mitigation of Straw return (CANM-Straw) model to inventory the GHG budget from residue retention and burning (Fig. 2). Both the CANM-Straw and SRBM include three parts: input, process and output. With the input of PR and PB, the models incorporated the effects of residue treatments when estimating the total effective GHG emissions (TEGEs) and subsequently determined the RNM and NMP generated via extension of residue retention (Fig. 2).

2.4.1. Inputs

The PR and PB from five scenarios were required to estimate the RNM for 2001 and determine the RNM and NMP for 2011. These treatments included the following: actual scenarios from 2001 to 2011, hypothetical scenarios for full burning in 2001 and 2011, and a hypothetical scenario for full retention in 2011. The sum of the PR and PB in the actual situation equaled either the PB under a full burning scenario or the PR under a full retention scenario.

Fig. 1. Map of the study area showing the distribution of surveyed counties.

Fig. 2. Structure of the carbon (C)- and nitrogen (N)-oriented Straw Return and Burning Model-II revised from Lu et al. (2010a).

The actual PR and PB in 2001 were obtained from studies Lu et al. (2009), which was based on a literature survey, and Cao et al. (2008), respectively. Based on the field survey, the actual PR

(%) and PB (%) in 2011 were calculated using formulas 2 and 3, respectively:

$$PR_i = \sum_{j=1}^M \left(HA_{ij} \cdot \left(\sum_{k=1}^F PRF_{ijk} \right) / F \right) / \sum_{j=1}^M HA_{ij} \quad (2)$$

$$PB_i = \sum_{j=1}^M \left(HA_{ij} \cdot \left(\sum_{k=1}^F PRF_{ijk} \right) / F \right) / \sum_{j=1}^M HA_{ij} \quad (3)$$

where PRF_{ijk} and PBF_{ijk} (%) represent the PR and PB for crop j in province i for the k th farmer, F represents the number of surveyed farmers in province i who planted crop j , and M represents the number of crop species planted in 2011 in province i . The values of PR and PB were between 0 and 100.

2.4.2. Process

Five processes of residue retention and burning were involved in the CANM-Straw. (1) Soil C sequestration (SCS). Compared with the SRBM, the CANM-Straw was dependent on the amount of residue-C input rather than the amount of retained residue to linearly determine the soil C sequestration rate (SCSR). The linear relationships were based on 174 pairs of long-term published experimental data ($P < 0.05$, Fig. A1 in the Appendix). (2) Mitigation of N fertilizer substitution (MNS). The substitution effect from residue-N inputs could avoid GHG emissions from equivalent N fertilizer production. We adopted an emission factor for GHGs from a synthetic N fertilizer production rate of 2.116 kg Ce kg N⁻¹ as estimated by a life cycle analysis of N fertilizer production in China (Chen et al., 2015) rather than the value of 1.748 kg Ce kg N⁻¹ calculated from fossil fuel combustion or electric power consumption in production processes alone as used in the SRBM (Lu et al., 2010b). (3) Additional fuel consumption for residue retention (ED). After residue retention, additional tilling or ploughing is typically implemented to ensure the sowing quality of the next crop. A field survey of machinery practices indicated that the additional diesel consumption was 31 kg ha⁻¹ under residue retention compared with 14 kg ha⁻¹ as used in the SRBM, which was cited from Chen et al. (2008). (4) CH₄ emission from rice paddies (EP). The global warming potential (GWP) of CH₄ emission was estimated using the method recommended by the IPCC (2006). The emission factors for CH₄ were estimated using 272 published data points for CH₄ emissions from rice paddies. In addition, the scaling factors of CH₄ emissions were linearly determined based on the residue-C inputs from 41 published experimental data pairs ($P = 0.003$, Fig. A2 in Appendix). (5) CH₄ and N₂O emissions from residue burning (EB). The total GWP of CH₄ and N₂O emissions from residue burning was estimated using a method recommended by the IPCC (2006). Using the CANM-Straw, the GHG emissions and mitigation potential in five processes were calculated per unit area of cropland. The calculation steps in the CANM-Straw are listed in the Appendix section A.1.

2.4.3. Output

As one output of the CANM-Straw, the TEGEs were obtained for the five scenarios of residue retention and burning to estimate the RNM and NMP. In 2001, 2011, the RNM, NMP (Tg Ce yr⁻¹) and net mitigation rate (NMR, kg Ce ha⁻¹ yr⁻¹) were calculated using formulas 4, 5 and 6, respectively:

$$RNM_i = TEGE_{FBI} - TEGE_{CSI} \quad (4)$$

$$NMP_i = TEGE_{CSI} - TEGE_{FRI} \quad (5)$$

$$NMR_i = RNM_i \cdot 10^9 / (A_i \cdot PR_i / 100) \quad (6)$$

where $TEGE_{CSI}$, $TEGE_{FBI}$ and $TEGE_{FRI}$ represent the TEGEs for the current scenario, the full burning scenario and the full retention scenario for province i , respectively.

2.5. Statistical analysis

To test the validity of this survey, a paired-sample t -test was conducted to test the difference between our survey and the Chinese Statistical Yearbook data (NBS, 2012) for crop yields in 2011. The crops adopted in the t -test included only three cereals (rice, wheat and corn) because their harvested area accounted for 81% of the area of the nine crops. The analysis indicated no significant difference for crop yields between the two data sources ($p = 0.288$ for rice in 23 provinces, $p = 0.427$ for wheat in 21 provinces and $p = 0.091$ for corn in 27 provinces).

3. Results

3.1. Residue retention and burning

In 2011, the total cropland area for the nine studied crops in China was 81.6 M ha (Table 1). The total residue, residue C and residue N from the nine crops were 778 Mt, 273 Mt C and 5.7 Mt N, respectively. The residue-C on the croplands was 3.5, 4.2, 3.0 and 2.8 t C ha⁻¹ in the northeast, north, northwest and south, respectively. The residue-C exceeded 5 t C ha⁻¹ in Henan, Shandong and Xinjiang.

In 2001, the percentage of China's croplands implementing residue retention and burning were 28% and 22%, respectively (Table 2). In 2011, 38% of China's croplands implemented residue retention, whereas 27% implemented residue burning. Therefore, residue retention increased to 295 Mt and represented inputs of 103 Mt C and 2.1 Mt N in 2011. In addition, 200 Mt of residue was still burned on the fields. According to the percentages of residue retention, the cropping regions ranked as north (62%) > south (35%) > northwest (23%) > northeast (9%). The four provinces (i.e., Hebei, Shandong, Henan and Shanxi) where more than half of the croplands had implemented residue retention were located in northern China. The amount of burned residue was 98, 61, 40 and 12 Mt in the south, northeast, north and northwest, respectively.

In the full retention scenario, the percentage of croplands implementing residue retention increased to 65% in China in 2011 (Table 2). According to the percentages, the cropping regions ranked as follows: north (76%) > south (71%) > northeast (41%) > northwest (37%). Residue retention in China resulted in 173 Mt C input on 50 M ha of croplands. The annual input was more than 10 Tg C in Shandong, Henan, Hebei, Heilongjiang, Jiangsu and Sichuan. The sum of these inputs represented 53% of the total input in China.

3.2. Soil carbon sequestration

The SCSR in China was 781, 590, 462 and 751 kg C ha⁻¹ yr⁻¹ in the northeast, north, northwest and south, respectively (Table 3). The total SCS was 11.7 Tg C yr⁻¹ in 2001 (Fig. 3) and increased to 19.0 Tg C yr⁻¹ in 2011 (Fig. 4). In 2011, the SCS was 1.0, 8.5, 1.0 and 8.5 Tg C yr⁻¹ in the northeast, north, northwest and south, respectively. The SCS was over 2 Tg C yr⁻¹ in Shandong, Henan and Hebei. Therefore, full residue retention would increase the total SCS to 34.8 Tg C yr⁻¹ in China.

Table 1
Cropland area and crop residue in 2011 in China.

Cropping region	Province	Cropland area (10 ³ ha)	Residue (t ha ⁻¹ yr ⁻¹)	Residue-C (t C ha ⁻¹ yr ⁻¹)
Northeast	Liaoning	2886	10.7	3.75
	Jilin	4375	11.9	4.14
	Heilongjiang	7879	8.7	3.04
North	Beijing	153	14.4	4.96
	Tianjin	337	9.8	3.43
	Hebei	4594	12.3	4.27
	Shanxi	2838	7.5	2.59
	Inner Mongolia	4816	8.2	2.86
	Shandong	5174	14.6	5.05
	Henan	5437	15.5	5.35
Northwest	Shaanxi	2914	6.5	2.25
	Gansu	3011	4.8	1.68
	Qinghai	363	4.5	1.61
	Ningxia	684	7.6	2.62
	Xinjiang	3144	13.8	5.04
South	Shanghai	110	9.2	3.23
	Jiangsu	3393	10.1	3.54
	Zhejiang	984	5.9	2.12
	Anhui	4090	9.3	3.26
	Fujian	576	6.6	2.34
	Jiangxi	2059	7.3	2.61
	Hubei	3183	9.2	3.31
	Hunan	2663	9.7	3.47
	Guangdong	1438	6.9	2.47
	Guangxi	2682	7.1	2.58
	Hainan	350	4.3	1.54
	Chongqing	1215	8.3	2.94
	Sichuan	3905	9.0	3.19
	Guizhou	2639	3.8	1.35
	Yunnan	3647	5.8	2.05
	Tibet	101	5.2	1.83
China		81639	9.5	3.35

Table 2
Percentages of croplands with residue retention (PR) and burning (PB) in 2001 and 2011 in China (%).

Cropping region	Province	2001		2011	
		PR	PB	PR	PB
Northeast	Liaoning	31.1	21.1	11.2	30.0
	Jilin	15.0	21.0	4.0	33.5
North	Heilongjiang	35.1	22.4	10.2	33.2
	Beijing	47.3	45.2	88.6	3.3
	Tianjin	47.3	40.6	29.1	18.1
	Hebei	47.3	24.2	80.8	5.8
	Shanxi	55.7	21.0	55.5	26.3
	Inner Mongolia	15.0	21.3	10.7	44.5
	Shandong	23.6	26.5	77.8	4.1
Northwest	Henan	34.6	19.7	60.8	11.7
	Shaanxi	32.0	14.8	33.2	5.2
	Gansu	26.8	16.6	13.5	16.6
	Qinghai	15.0	10.1	0.8	25.4
	Ningxia	7.0	20.1	30.0	41.8
	Xinjiang	15.0	19.2	19.3	14.1
	Shanghai	47.8	48.7	47.2	38.6
South	Jiangsu	31.9	34.2	26.8	60.6
	Zhejiang	23.9	42.6	47.2	38.6
	Anhui	30.3	18.7	23.6	47.3
	Fujian	35.5	32.5	26.8	47.9
	Jiangxi	65.1	21.7	40.0	18.9
	Hubei	38.0	23.4	28.7	41.5
	Hunan	20.0	22.3	43.1	38.5
	Guangdong	41.4	36.5	38.0	8.4
	Guangxi	15.0	16.7	45.3	35.0
	Hainan	38.0	30.2	29.1	53.6
	Chongqing	15.0	19.4	58.0	35.7
	Sichuan	14.3	19.9	49.7	37.8
	Guizhou	15.0	15.0	8.0	8.1
China	Yunnan	15.0	11.4	22.0	17.4
	Tibet	15.0	8.5	0.8	24.5
		28.0	21.6	37.9	27.4

3.3. Mitigation resulting from N fertilizer substitution

In 2001, 1.2 Mt of N fertilizer was substituted with residue retention, and this process mitigated 2.5 Tg Ce yr⁻¹ of GHG emissions (Fig. 3). In 2011, this substitution increased to 2.1 Mt N fertilizer and mitigated 149 kg Ce ha⁻¹ yr⁻¹ on China's croplands (Table 3). In addition, the mitigation was 0.2, 2.5, 0.3 and 1.5 Tg Ce yr⁻¹ in the northeast, north, northwest and south, respectively (Fig. 4). The total mitigation increased to 7.6 Tg Ce yr⁻¹ in the full retention scenario. The increase was highest in northeastern China (393%), whereas the lowest value was observed in northern China (24%).

3.4. Additional fossil fuel consumption caused by residue retention

The additional fuel use associated with residue retention increased from 0.8 to 1.3 Mt yr⁻¹ during the decadal study of China's croplands. The GHG emissions from these fossil fuels were between 17 and 55 kg C ha⁻¹ yr⁻¹ (Table 3). The total emissions were 0.7 Tg C yr⁻¹ in 2001 (Fig. 3) and increased to 1.1 Tg C yr⁻¹ in 2011 (Fig. 4). In the full retention scenario, the total emissions from the additional fuel use increased to 1.9 Tg C yr⁻¹, including 0.12, 0.71, 0.10 and 0.96 Tg C yr⁻¹ from the northeast, north, northwest and south, respectively.

3.5. CH₄ emissions from rice paddies

Compared with rice paddies without residue retention, the CH₄ emissions doubled to 2646 kg Ce ha⁻¹ yr⁻¹ from paddies with residue retention (Table 3). Rice paddies emitted 49 Tg Ce of CH₄ in 2001 (Fig. 3), and these emissions increased to 53 Tg Ce of CH₄ in 2011 in China (Fig. 4). Furthermore, the emissions increased to 68 Tg Ce yr⁻¹ under the full retention scenario. The CH₄ emissions

Table 3
Greenhouse gas (GHG) emissions and mitigations in 2011 in China (kg Ce ha⁻¹ yr⁻¹).

Cropping region	Province	SCSR	N fertilizer substitution with RR	Avoidance of non-CO ₂ GHG emission from residue burning		GHG emission from additional fossil fuel use	Seasonal CH ₄ emission from rice paddy	
				Without RR	With RR		CH ₄	N ₂ O
Northeast	Liaoning	823	168	156	48	21	445	979
	Jilin	886	184	172	53	20	445	1062
North	Heilongjiang	707	137	128	39	19	445	940
	Beijing	689	210	210	65	55	1937	3969
	Tianjin	493	153	145	45	35	1937	4234
	Hebei	600	183	183	56	45	1937	4160
	Shanxi	386	112	109	34	38	1937	3545
	Inner Mongolia	421	131	120	37	45	1937	4530
	Shandong	700	211	218	67	39	1937	4448
Northwest	Henan	737	213	234	72	39	1937	4210
	Shaanxi	362	97	97	30	30	445	939
	Gansu	281	83	72	22	25		
	Qinghai	271	81	69	21	22		
South	Ningxia	414	122	112	35	33	445	1026
	Xinjiang	760	255	212	66	32	445	1036
	Shanghai	856	138	143	44	36	1483	3412
	Jiangsu	930	147	156	48	40	1483	3394
	Zhejiang	586	100	92	28	29	1445	3105
	Anhui	863	135	143	44	43	1448	2908
	Fujian	639	115	102	32	39	1409	2803
	Jiangxi	705	122	113	35	42	1356	2662
	Hubei	876	152	142	44	47	1431	3209
	Hunan	914	161	149	46	54	1378	2790
	Guangdong	670	117	106	33	36	1338	2584
	Guangxi	697	123	108	33	30	1345	2519
	Hainan	446	73	66	20	23	1351	2408
	Chongqing	785	144	125	39	51	1483	3173
	Sichuan	845	145	137	42	51	1482	3256
	Guizhou	399	70	57	18	40	1483	2625
	Yunnan	570	98	87	27	39	1471	2956
	Tibet	517	69	81	25	17	1483	2934
China		675	149	143	44	36	1281	2646

SCSR, soil carbon sequestration rate; RR, residue retention. Note: no rice was planted in Gansu and Qinghai in 2011.

in southern China accounted for 90% of the total emissions in China under all scenarios (Figs. 3 and 4). The CH₄ increase in south China also played a dominant role in the overall increase throughout China.

3.6. GHG emissions from residue burning

Residue retention in 2001 mitigated 3.1 Tg Ce yr⁻¹ of non-CO₂ GHG emissions (including CH₄ and NO₂) from residue burning in the full burning simulation (Fig. 3). This mitigation was 5.8 Tg Ce yr⁻¹ in 2011 (Fig. 4). However, 1.0, 0.8, 0.2 and 2.0 Tg Ce yr⁻¹ of GHG emissions in 2011 occurred in the northeastern, northern, north-western and southern China, respectively. Therefore, mitigation reached 3.9 Tg Ce yr⁻¹ under the full retention scenario.

3.7. Net GHG mitigation from residue retention

Residue retention in place of burning induced different effects on the NMR across the 31 provinces (Fig. 5a and b). The average NMR in China was 250 kg Ce ha⁻¹ in 2001 (Fig. 5a), which increased to 480 kg Ce ha⁻¹ in 2011 (Fig. 5b). In addition, the NMR values changed from negative to positive in Hubei, Guangxi and Chongqing. However, the NMR changed from positive to negative in Jiangsu. In 2011, five provinces (Liaoning, Jilin, Beijing, Shandong and Xinjiang) had NMRs > 1000 kg Ce ha⁻¹ yr⁻¹ among the 23 provinces with positive NMRs. In addition, the NMRs from eight provinces in south China (Shanghai, Jiangsu, Zhejiang, Fujian, Jiangxi, Hunan, Guangdong and Hainan) were negative.

The RNM was 5.3 Tg Ce yr⁻¹ in 2001 (Fig. 3) and 13.3 Tg Ce yr⁻¹ in 2011 for Chinese croplands (Fig. 4). The RNM in northern China increased by 1.5 fold, which explained 95% of the increase in the total RNM in China from 2001 to 2011. However, the RNM decreased by 43% and 28% in the northeast and south, respectively. Compared with 2011, a full extension of residue retention could mitigate an additional 8 Tg Ce yr⁻¹ on China's croplands (Fig. 4). The highest NMPs were 2.1 and 1.7 Tg Ce yr⁻¹ in Heilongjiang and Jilin, respectively. Moreover, the maximum total NMP would be 10 Tg Ce yr⁻¹ if residue retention was not implemented in the eight provinces with negative NMRs.

4. Discussion

4.1. Residue retention and burning

Residues from the nine studied crops increased from 553 Mt in 2001 to 778 Mt in 2011 on China's croplands. A residue increase was also observed from 2000 to 2009 (Jiang et al., 2012) as a result of a gradual increase in crop productivity (Huang et al., 2007). However, the percentage of biomass fuel consumption in the total rural energy consumption decreased by 4% from 2001 to 2008 (Yao et al., 2012). This drop was attributed to the economic advancement of farmers (Cao et al., 2008) and the popularization of fossil fuel use (such as coal and natural gas). Hence, the use of residue required management before the next crop could be sown.

In this study, the PR increased from 28% to 38% from 2001 to 2011 (Table 2). In addition, the amount of residue retention

Greenhouse gas emission and mitigation (Tg Ce yr^{-1}) Greenhouse gas emission and mitigation (Tg Ce yr^{-1})

Fig. 3. Impacts of residue retention and burning on greenhouse gas emissions in 2001 under scenarios of full burning (left column) and the current situation (right column) in China. SCS = soil carbon sequestration of residue-C inputs; MNS = mitigation of synthetic nitrogen fertilizer substitution; ED = global warming potential (GWP) of greenhouse gas emissions from additional diesel oil use; EP = GWP of CH_4 emissions from rice paddies; and EB = GWP of CH_4 and N_2O emissions from residue burning. ED, EP and EB are expressed as positive values, whereas SCS and MNS are expressed as negative values.

increased from 144 (Lu et al., 2010b) to 295 Mt yr^{-1} . The increases in Shandong, Henan and Hebei accounted for 73% of the total increases in China. However, the increase in residue productivity and decrease in residue used as fuel also enhanced the amount of residue burning on croplands. The total burned residue increased from 140 Mt in 2000 (Cao et al., 2008) to 200 Mt in 2011 (this study). Bao et al. (2014) reported that the percentage of burned residue was 12% in northwestern China, which was similar to the level in our survey (14%). The status and trend of residue burning implied the potential for extending residue retention on Chinese croplands.

4.2. Soil carbon sequestration

SOC dynamics are typically driven by anthropogenic forces rather than by climate change (Lugato et al., 2014). China's agricultural soils lost 30%–50% of the antecedent SOC pool, which was partly a result of soil mismanagement (Lal, 2004b). Under less residue retention, the results of the DNDC model indicated a soil C loss of 79–95 Tg C yr^{-1} in the 1990s from China's croplands (Li et al.,

2003; Tang et al., 2006). Qin et al. (2013) estimated that the total SCS potential was 2.7 Pg C before a saturated SOC level was reached. If residue retention in the provinces with positive NMRs was maintained for approximately 20 years, as assumed in the present study, then the total SCS (0.58 Pg C) would make an important contribution to increasing the SOC content of China's croplands and could achieve approximately 20%–26% of the SCS potential of 2.21–2.95 Pg C before reaching the C-saturation level (Cheng et al., 2014). Under residue retention increases of 15%–30%, the SCS was 3.6 Tg C yr^{-1} in 2008 (Lun et al., 2016), whereas under residue retention increases of 28% in 2001 to 38% in 2011 (this study), it was estimated at 7.3 Tg C yr^{-1} . This difference was mainly attributed to a residue retention increase of 91 Mt in the former study and 151 Mt in this study. Wu et al. (2015) reported that the factor with the most significant impact on SOC change was residue management (49.4%), followed by CO_2 enrichment (37%), tillage (6.2%) and fertilizer (1.6%) in a study from Iowa in the United States of America. Therefore, residue retention could be regarded as one of the most direct and promising contributions to soil C stocks on croplands (Lu

Fig. 4. Impacts of residue retention and burning on greenhouse gas emissions in 2011 under scenarios of full burning (left column), the current situation (center column) and full residue retention (right column) in China. SCS = soil carbon sequestration of residue-C inputs; MNS = mitigation of synthetic nitrogen fertilizer substitution; ED = global warming potential (GWP) of greenhouse gas emissions from additional diesel oil use; EP = GWP of CH₄ emission from rice paddies; and EB = GWP of CH₄ and N₂O emissions from residue burning. ED, EP and EB are expressed as positive values, whereas SCS and MNS are expressed as negative values.

et al., 2009).

In this study, the temporal dynamics of SCS were primarily attributed to changes in the crop yield and the cropland area under residue retention. For instance, over the studied decade, the residue retention area increased from 86 to 262%, and the crop yields increased by 17–42% in Shandong, Henan, Hebei and Sichuan (NBS, 2012). These changes increased the SCS by more than 1 Tg Ce in each province. However, although the crop yield increased in Jilin (70%) and Liaoning (53%) (NBS, 2012), the SCS decreased by approximately one-half because of decreases in residue retention of more than 60% from 2001 (Lu et al., 2010b) to 2011 in the two provinces studied. The amounts of SCS were determined by both the crop yield and cropland area under residue retention.

4.3. Mitigation of N fertilizer substitution

The substitution effect from residue retention depends on the implementation of precision fertilization on croplands (Lu et al., 2010a). In the present study, the mitigation resulting from the substitution of N fertilizer could contribute to 30% of the total NMP in the full retention scenario for 2011. However, under excessive N fertilization, residue N would increase the additional soil N₂O emissions and result in soil acidification (Ju et al., 2009). Therefore, the extension of residue retention on China's croplands should be implemented together with improvements in fertilizer management (Li et al., 2003).

Fig. 5. Net mitigation rate (NMR) from residue retention in place of burning on China's croplands in 2001 (a) and 2011 (b).

4.4. Additional fossil fuels used for residue retention

The results of the residue retention scenarios indicate that diesel consumption increased from 0.8 to 1.3 Mt yr⁻¹ on Chinese croplands during the study decade. This consumption would increase by 2.2 Mt diesel yr⁻¹ and emit 2 Tg Ce yr⁻¹ of GHG under an expansion of residue retention. However, these emissions could be reduced by reducing tillage or with no-till practices. Dikgwatlhe et al. (2014) implied that under no-till, more C would be sequestered compared with plow tillage on croplands with residue retention on the North China Plain. The additional diesel consumption had a relatively low contribution to GHG emissions but increased the economic cost for farmers. Suitable tillage practices could promote the extension of residue retention in China's croplands.

4.5. CH₄ emissions from rice paddies

Residue retention increases the CH₄ emission from rice paddies in China. These additional CH₄ emissions offset 96%, 78% and 84% of the total SCS from residue retention in the actual scenarios during 2001, 2011 and the full retention scenario, respectively, and even reached 156% of the SCS in southern China in 2011. Hence, these emissions were the greatest GHG leakage factor under all scenarios. The spatio-temporal variation of CH₄ emissions increases were attributed to two factors: the amount of residue retained in rice paddies and the area of paddy rice using residue retention. The increases in residue retention enhanced the CH₄ emissions from paddies in Hunan and Sichuan by 2.12 and 1.16 Tg Ce, respectively. However, the CH₄ emission increased (0.51 Tg Ce) in Heilongjiang was mainly attributed to the expansion of rice paddies (88%) (NBS, 2012). The total rice paddy area accounted for 48% of the harvested area of the nine crops in southern China and reached 67% in the eight provinces that exhibited negative NMRs in this study. Hence, the negative NMRs were mainly caused by the increase in CH₄ emissions in these provinces.

The implementation of suitable practices can promote SCS or limit CH₄ emissions from rice paddies. For example, residue retention together with mineral fertilization in paddies could extend the duration of C sequestration and double the SCS potential compared with mineral fertilization alone (Tian et al., 2015). Zhang et al. (2014) also indicated that a reduction in C inputs during the rice season along with improvements in N-use efficiency to 40% would mitigate GHG emissions by 30 Tg Ce yr⁻¹ and sequester 22 Tg C yr⁻¹ in China's rice paddies. Moreover, optimized irrigation patterns, such as changing from continuous flooding to intermittent flooding, would greatly reduce the CH₄ emissions from rice paddies (Wang et al., 2012). Therefore, retaining residue under proper circumstances and with suitable water management favors CH₄ emission mitigation in China's rice paddies under the expansion of residue retention.

4.6. GHG emissions from residue burning

Agricultural biomass burning led to substantial GHG emission and accounted for 12% of the GWP from global agricultural non-CO₂ GHG emissions (Smith et al., 2012). Compared with the conditions one decade ago (Lu et al., 2010a), the GHG emissions from residue burning increased by 78% in this study as a result of increases in residue production and residue burning. However, these emissions decreased in Shandong, Hebei and Henan in 2011 compared with the levels in 2000 (Cao et al., 2008). Our results also indicated that the emissions decreased as a result of residue retention in place of burning in these provinces from 2001 to 2011. The sum of GHG emissions from northeastern and southern China accounted for 74%

of the total emissions from residue burning. Residue burning not only increases GHG emissions but also reduces crop yields and increases soil erosion (Smith et al., 2012). Furthermore, residue burning is one of the main contributing factors to air pollution during harvest periods in China (Hong et al., 2016). The expansion of residue retention could overcome these disadvantages resulting from of residue burning.

4.7. Net GHG mitigation from residue retention

In this study, the variability in the RNM indicated spatial heterogeneity caused by changes in crop types and residue management. In northeastern China, the decrease in RNM was mainly attributed to the increased rice paddy area (NBS, 2012) and the decreased residue retention. The increase in RNM in northern China represented 95% of China's overall increase because the residue retention increased by 1.6 fold over the 10-year period. In northwestern China, the increase in SCS represented 64% of the total increase in RNM. The paddy area decreased by 0.4 M ha (NBS, 2012), which reduced the additional net GHG emission from residue retention in southern China. This decrease in paddy area also changed the values of RNM from negative to positive in Guangxi and Chongqing. In addition, the RNM changed from negative to positive in Hubei, which was mainly due to the decrease in paddy area with residue retention. However, the paddy area increased by 12% in Jiangsu (NBS, 2012); hence, residue retention, which was performed to mitigate GHGs in 2001, led to more GHG emissions in 2011.

The C and N processes other than C sequestration affected the contribution of residue retention to GHG mitigation. Compared with the upland areas, the paddy area had a higher SCS potential in China (Qin et al., 2013) and a faster recovery rate of SOC level due to residue retention in northeastern China (Ouyang et al., 2014). However, our results indicate that the CH₄ emissions from paddies would exceed the SCS and lead to negative NMRs under the implementation of residue retention in the eight provinces. The net GHG mitigation of other SCS practices may also be affected by changes in fuel consumption and non-CO₂ GHG emissions (Lal, 2004a). For instance, the SCS of N fertilization could be fully offset by the considerable GHG emissions from fertilizer production and enhanced soil N₂O emissions (Lu et al., 2008; Schlesinger, 2010). The SCS from no-till practices would also be partially counteracted by increased N₂O emissions (Powlson et al., 2011). Thus, the full GHG budget of C and N cycles in these practices must be considered rather than the dynamics of soil C stock alone.

The increased mitigation (i.e., 8 Tg Ce yr⁻¹) from 2001 to 2011 may persist for 10–20 years, with 20 years representing the time at which soil C becomes saturated. The NMP under full retention would also function for approximately two decades. These values would generate a total net GHG mitigation of 320–400 Tg Ce in the future. After soil C saturation, the relatively persistent mitigation from synthetic N substitution and the avoidance of non-CO₂ GHG emissions from residue burning would still contribute to climate change mitigation, although the net mitigation might be much smaller. Therefore, residue retention has substantial GHG mitigation potential for China's croplands and could represent a “buying time” measure to allow for the development of more permanent mitigation approaches (Lu et al., 2010b).

4.8. Limitations and suggestions

In this study, the insignificant difference between the survey data and the statistical data supported the validity of our survey to an extent; however, certain limitations were observed. First,

although the survey samples covered the main cropping regions of China, the sample size remained a considerable challenge because huge spatial heterogeneity occurs in residue management in China. To address this challenge, a larger-scale or national-scale survey is needed in the future. Second, the questionnaires related to rice, wheat and corn represented 89% of the total questionnaires. Residue retention from these three crops was more popular compared with the other crops due to highly developed retention technology (NBS, 2012). Hence, our results might slightly overestimate the percentage of croplands using residue retention. However, an area-weighted averaging method was adopted at the provincial level to reduce the uncertainties in the estimations. Lastly, due to spatial heterogeneity and data availability, we expect that more long-term data of residue retention will be available to validate the model in the future.

The extension of residue retention requires governmental promotion and farmer support. Therefore, certain details on expanding residue retention require further attention: (1) residue retention should differentially expand due to the spatial heterogeneity in climate, soil and cropping systems in China; (2) residue retention should be preferably implemented with other suitable practices, including precision fertilization, no-till practices, suitable water management and rice varieties that have low CH₄ emission; and (3) the government should compensate farmers for the economic costs of residue retention. In summary, crop residue retention should be prioritized among residue management strategies to meet global food demands (Lal, 2009).

5. Conclusions

In this work, a field survey for farmers throughout China was conducted from 2012 to 2013, and a CANM-Staw was developed to estimate the GHG emission changes and mitigation potential of residue retention and burning from 2001 to 2011. Over the studied decade, the residue retention increased from 144 to 295 Mt, which accounted for C inputs of 103 Mt; however, 200 Mt of residue was still burned on China's croplands in 2011. Although the RNM increased from 5.3 to 13.3 Tg C yr⁻¹ over the studied decade, it was 55% and 30% lower than the SCS in 2001 and 2011, respectively. The NMP would further increase by 7.8 Tg C yr⁻¹ under full residue retention expansion in place of residue burning in China. This expansion would lead to negative NMRs as a result of the increase in CH₄ emissions from rice paddies in the eight provinces in southern China. Hence, expansion in the remaining 23 provinces would result in a maximum net mitigation of 10 Tg C yr⁻¹, which would offset 0.5% of the CO₂ emissions from fossil fuel combustion that occurred in China in 2011 (IEA, 2013). Substantial mitigation potential for China's croplands exists in the targeted expansion of residue retention. Our study outlined the GHG budgets related to crop residue retention and burning in 2001 and 2011 and clarified the effect of spatio-temporal patterns on GHG emissions and mitigation, thereby providing a potential strategy for improving agriculture residue management and mitigating climate change.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at <http://dx.doi.org/10.1016/j.jclepro.2017.07.077>.

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