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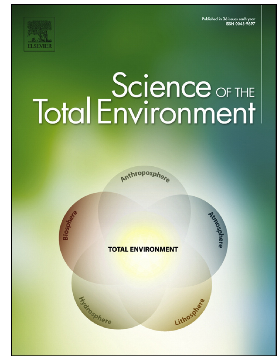

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To Burn or Retain Crop Residues on Croplands? An Integrated Analysis of Crop Residue Management in China

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To Burn or Retain Crop Residues on Croplands? An Integrated Analysis of Crop Residue Management in China

Abstract: Crop residue burning influences human health and global climate change. In China—the world's largest crop residue producer—farmers burn almost one quarter of their crop residues in the field after harvest, despite the government providing financial incentives such as subsidies to retain crop residues. This study combined economic analyses with simulations of soil carbon accumulation and carbon emission reduction associated with different residue management practices to determine the minimum level of incentives needed for Chinese farmers to shift from burning to retaining crop residues for generating carbon benefits. Simulation results showed that [1] the density of topsoil organic carbon in China's croplands would have increased from about 21.8 t ha^{-1} in 2000 to 23.9 t ha^{-1} in 2010, and soil organic carbon sequestration would have reached 24.4 Tg C yr^{-1} if farmers had shifted from burning to retaining crop residues on croplands during this period; and [2] retaining crop residues would have avoided about 149.9 Tg of CO_2 emission per year. Economic analyses showed that [1] existing subsidies in all regions of China, except Northeast China, only accounted for 18-82% of the incentives required for farmers to shift from burning to crop residue retention; [2] Northeast China required the lowest incentive (287 CNY ha^{-1}), while eastern China required the highest (837 CNY ha^{-1}); and [3] the prevailing market prices (1.4-60.2 CNY tCO_2e^{-1}) in China's seven pilot carbon markets seem to be below the required incentives (39.6-189.1 CNY $tCO_2e^{-1)}$. Our study suggests that the Chinese government should increase subsidies or seek innovative incentive schemes to encourage farmers to change their crop residue

management practices for global climate change mitigation and health benefits.

Key Words: Crop residue management, soil carbon, carbon emissions, economic analysis, China.

1. Introduction

Crop residue burning is practiced by farmers in many parts of the world, including India (Venkataraman et al., 2006) and Russia (McCarty et al., 2012), with serious health and environmental consequences. Globally, crop residue burning is a major contributor to total biomass burning. In Asia, crop residue burning in the field accounts for more than one-third of total biomass burning (Street et al., 2003). Since crop residue burning releases harmful pollutants such as PM_{10}, $PM_{2.5}$, and greenhouse gases (GHGs) including CO_2 and CH_4 (Cao et al., 2005; Li et al., 2013), it affects human health and contributes to global climate change.

Crop residue burning is particularly prominent in China, the world's largest crop residue producer, which produces about 600-800 Tg of crop residues per year (Cao et al., 2006, 2008; Liu et al., 2008; Cai et al., 2011; Jiang et al., 2012), accounting for around 20% of total global production (FAO, 2014). On average, about 20-30% of crop residues are burned by Chinese farmers in the field after harvest (Yan et al., 2006; Cao et al., 2008; Zhang et al., 2013; Li et al., 2016). About 2,200 Tg of crop residues, accounting for 22% of China's total crop residue production, were burned during 1996-2013. As a result, approximately 2,707 Tg of CO_2 was released, which is equivalent to 45% of the total CO_2 emission from China's residential coal consumption during the same period (Sun et al., 2016). On the other hand, farmers in

China retain around 30% of crop residues on their croplands (Cui et al., 2008; Jiang et al., 2012; Yu et al., 2013), which is less than that of other areas such as the US (55-90%) (Stewart and Moldenhauer, 1994; Lokupiyiya et al., 2012), Australia (70-80%) (Llewellyn and D'Emden, 2010), and Europe (40-70%) (Scarlat et al., 2010).

If Chinese farmers retained 20-30% of the crop residues that were burned on their croplands, the crop residue retention rate would reach around 50%, which would provide carbon benefits for society. When the crop residue retention rate increases from 25% to 50%, soil organic carbon (SOC) storage in China's cropland rises by 23.2 Tg C yr^{-1} (Yan et al., 2007). In a 12-year experimental study, Lou et al. (2011) demonstrated that a retention rate of 50% led to an increase in SOC, while a retention rate of 25% resulted in a loss of SOC. Hence, encouraging Chinese farmers to shift from burning to retaining crop residues can contribute to climate change mitigation efforts.

However, for farmers to change their management practices for carbon benefits, some appropriate level of incentive needs to be provided, such as subsidies or carbon prices. In fact, the government of China has been providing subsidies and other supporting policies to encourage farmers to retain crop residues in recent years (Table S1). For example, agricultural machinery operation subsidies and agricultural machinery purchase subsidies have been provided to farmers who retain their crop residues in 10 pilot provinces including Hebei, Shanxi, Inner Mongolia, Liaoning, Jilin, Heilongjiang, Jiangsu, Anhui, Shandong, and Henan. Subsidies for agricultural machinery operation alone range from 150-375 CNY ha^{-1} yr^{-1} in Southern China to 375-600 CNY ha^{-1} yr^{-1} in Northeast China. Meanwhile, seven pilot carbon markets were launched in 2013 and a nationwide carbon market was established in 2017 in China. However, Chinese farmers are still burning crop residues.

From economic point of view, for farmers to change their crop residue management practices, they need to at least reach a break-even point, i.e., they should have no net loss. Hence, it is important to conduct cost-benefit analyses. Nonetheless, there is still a dearth of quantitative studies conducted at national or regional scales to analyze the benefits and costs associated with shifting from burning to retaining crop residues. While Zhao et al. (2018) highlighted the importance of economic and policy incentives provided to farmers for SOC accumulation through changes in farming practices, most existing studies in China in this research area have largely focused on estimating greenhouse gas emissions from crop residue burning (e.g., Street et al., 2003; Zhang et al., 2008; Li et al., 2013; Sun et al., 2016; Chen et al., 2017), or assessing the effect of retaining crop residues on SOC (e.g., Tang et al., 2006; Lu et al., 2009, 2010, 2015; Lou et al., 2011). Some studies have explored the costs and benefits of retaining burned crop residues associated with crop residue management, but they were based on experiments conducted at field stations (e.g., Xia et al., 2014; Hu et al., 2016) or used qualitative methods (Mei, 2008; Keck and Hung, 2018).

This study attempts to fill gaps in the existing literature in China by combining biophysical simulations with economic analyses conducted at the national and regional scale. The simulations examined three crop residue management scenarios: [1] all crop residues are removed (scenario CR1); [2] business-as-usual crop residue retention (scenario CR2), and [3] crop residues that are burned are retained on croplands (scenario CR3). The net present value (NPV) of changes in crop residue management were calculated to estimate the minimum incentive required for farmers to shift from burning residues to retaining them for carbon benefits.

2. Methods

2.1 Crop residue management scenarios

Three scenarios with three different crop residue retention rates were developed: [1] CR1, the baseline scenario, where all crop residues are removed from cropland; [2] CR2, the business-as-usual residue management scenario, representing the residue management practices commonly adopted by farmers in China. For this scenario, a region-specific retention rate ranging from 15% to 33% was applied for seven different regions (Table 1); and [3] CR3, an improved crop residue management scenario, where crop residues that have been burned are retained on croplands. For this scenario, regional differences in burning rates were also considered (Table 1).

For each scenario, the region-specific residue retention rates and burning rates were used to account for differences in crop residue management practices and biophysical characteristics in seven regions of China, including Northeast China, North China, Northwest China, Southwest China, Central China, South China, and East China (Figure S1). The crop residue retention rate and the burning rate in these seven regions were calculated from provincial crop production data and provincial rates of crop residue retention and burning. Provincial crop production data were collected from the China Rural Statistical Yearbook released by the Ministry of Agriculture of China for the period 2000-2010 (NBSPRC, 2000-2010), and were combined with the residue-to-product ratios, i.e., 2 for corn, 1.366 for wheat, and 0.623 for rice (CAREI, 2000; Cao et al., 2006), to estimate provincial crop residue production. Due to a lack of official statistics, information on provincial residue retention and burning rates for three major crops were obtained from existing studies (Cao et al., 2005; Han et al., 2008). To obtain the rates at the regional scale, the provincial data were aggregated using provincial weights for crop residue production (Table 1).

2.2 SOC sequestration simulation

The Environmental Policy Integrated Climate (EPIC) model (Williams et al., 1985, 1989, 2006) was used to simulate spatial changes in SOC pools and SOC sequestration under alternative management scenarios during the period 2001-2010. The EPIC model is a comprehensive process-based cropping system model that is able to simultaneously simulate crop growth and soil processes. The model allows for interactions between climate, cropping systems, and soil and crop management involving SOC dynamics. Its major components encompass crop growth and yields, soil temperature and moisture, soil erosion, tillage, plant environment control, as well as hydrological, nutrient, and organic carbon cycling.

In the EPIC model, daily crop growth was calculated from intercepted photosynthetically active radiation using the energy-to-biomass conversion approach modified for a vapor pressure deficit and atmospheric CO_2 concentration effect (Monteith, 1977; Stockle et al., 1992). The model has been widely applied in European (van der Velde et al., 2009; Balkovič et al., 2018), Chinese (Zhao et al., 2013), and global agricultural research (Liu et al., 2007, 2013; Balkovič et al., 2014). The soil carbon and nitrogen (N) modules in EPIC were built on concepts from the Century model (Izaurralde et al., 2006). Crop residues were split into two litter compartments: metabolic and structural, depending on their N and lignin content. As a function of soil temperature and moisture, carbon in litter is allocated into three compartments: microbial biomass, slow humus, and passive humus, which differ in terms of their size, function, and turnover times (Izaurralde et al., 2006).

The EPIC model can simulate the amount of organic carbon in the soil to a plow layer depth of 20 cm (OCPD, t C ha^{-1}), thereby accounting for disturbance from tillage, irrigation, and fertilization, carbon respiration from soil, leaching of carbon, and carbon lost in runoff and eroded sediment (Elshout et al., 2015). This model has been successfully applied to estimations of SOC storage in China (Zhao, 2013) and elsewhere in the world (e.g., Causarano et al., 2007; Billen et al., 2009).

In this study, EPIC V0509 was used to simulate the impact of crop residue retention on topsoil organic carbon in croplands of China. Specifically, it was used to estimate OCPD in each pixel for each year of the study period (2000-2010). SOC sequestration in the crop residue retention scenarios (i.e., CR2-CR1, CR3-CR1, and CR3-CR2) were obtained by combining simulated yearly OCPD values for the different scenarios with crop cultivated areas in each region.

To simulate topsoil organic carbon dynamics, the EPIC model requires weather information, soil profile characteristics, topography, and crop management information as major inputs (Figure 1). Daily weather information, including minimum and maximum air temperature, precipitation, solar radiation, and relative humidity, was collected from China's Surface Climate Data from 660 meteorological stations for the period 2001-2010. Information on China's cropland was generated by aggregating 10 × 10 km^2 sub-areas (or pixels) classified as dryland or paddy field (Figure S2) from the National Land Cover Project Dataset (NLCD), based on Landsat ETM+ images acquired in 1999 and 2000 for China (Liu et al., 2003, 2005). Information on soil profiles was collected from the 1:1,000,000 soil dataset based on the Chinese Second Soil Survey Project (Shi et al., 2002; Liu et al., 2006). The weather and soil data were spatially interpolated and resampled to match the 10 km pixels. To characterize the major cropping systems in dryland and paddy fields, we used a wheat-corn crop rotation system and a double-cropping rice system, respectively. A set of management and equipment parameters were calibrated for the EPIC model by referring to relevant studies (Fan et al., 2012; Lin et al., 2013) and from field investigations in two experimental stations, i.e., Yucheng Comprehensive Agriculture Experiment Station (YCS) in Northern China and Qianyanzhou Ecological Experiment Station (QYZ) in Southern China; the wheat-corn crop rotation system and the double-cropping rice system are implemented in these two stations, respectively. The original crop growth and soil parameters were partly calibrated and then validated to account for characteristics of China's cropland (Table 2). Validation at the two local sites showed that simulation of SOC was consistent with measurements (YCS: R^2=0.85, P<0.01; QYZ: R^2=0.87, P<0.01), and could capture SOC dynamics with or without crop residue retention (Lin et al., 2013).

The different management scenarios were implemented through the crop residue model input (RSD, t ha^{-1}), which was defined as the yield of crop residues and the corresponding retention rate (variable among regions and scenarios) (Table 1). The yield of crop residues was obtained from the crop residue production (section 2.1) and the corresponding crop cultivated areas from the China Rural Statistical Yearbook (NBSPRC, 2000-2010).

2.3 Emissions calculations

A fraction of crop residues was burned in the CR2 scenario, leading to carbon and pollutant emissions into the atmosphere. These emissions were calculated using the amount of crop residues burned in the field and detailed emission factors obtained from the literature (Table 3). The amount of burned crop residues for each region was calculated based on the crop residue production and the fraction of crop residues burned in the field (Table 1). The emissions were products of the amount of each crop residue burned in the field and the corresponding emission factors for each pollutant. The regional and national emissions were the sum of each crop residue type. Emissions for each pollutant were calculated using the following equation:

$$E_{i,k} = \sum_{j=1}^{3} Y_{i,j} \times F_i \times EF_{j,k} \tag{1}$$

where Y is the amount of crop residue, F is the fraction of crop residue burned (Table 1), and EF represents the emission factor (Table 3); the subscript i, j, k represent the region, fuel type, and pollutant, respectively.

2.4 Economic analyses

The cost-benefit analysis focused on two scenarios (CR2 and CR3), i.e., shifting from the current crop residue management system to retaining that burned crop residue on the cropland. Costs mainly included labor and machinery costs for retaining the crop residue. The benefits focused on avoided CO_2 emissions and SOC sequestration changes from CR2 to CR3. However, other co-benefits, such as reduced emissions of CH_4 and other pollutants, were not considered for two reasons. First, CO_2 is the main pollutant associated with crop residue burning, where the emission factors of CO_2 for corn, wheat, and rice residues are 1261.5, 1557.9, and 791.3 g kg^{-1}, respectively (Table 2). Second, CO_2 contributes the most to the greenhouse effect, accounting for about 60%, and is the most important greenhouse gas (IPCC, 2000).

To calculate carbon benefits, it is important to define the baseline and "additionality." As the main purpose of the economic analysis in this study was to understand economic incentives for farmers who chose to retain crop residues during the period 2001-2010, the year 2000 was used as the baseline. The "additionality" is defined as the amount of SOC sequestration and avoided CO_2 emissions associated with a shift from burning to crop residue retention in each year. For SOC sequestration, non-permanence is considered to be a critical issue in the existing literature (Marland et al., 2001; Murray et al., 2007). In this study, non-permanence was also taken into account following the method used by Sohngen and Mendelsohn (2003), considering that carbon sequestered in soils may be released back to the atmosphere when farmers return to previous crop residue management methods or when there are some other incidences.

For the cost-benefit analysis, two cases were considered. In the first case, where non-permanence was not considered, the net present value (*NPV*) of a shift from CR2 to CR3 was computed as follows:

$$NPV_1 = \sum \frac{B_t - C_t}{(1+i)^t} = \sum \frac{P_1 \cdot (Q_{1,t} + Q_{2,t})}{(1+i)^t} - \sum \frac{C_t}{(1+i)^t} \tag{2}$$

where B_t is the benefit from sequestered carbon and carbon emissions avoided; C_t is the cost of retaining crop residue that would otherwise be burned, including labor and machinery costs; i is the discount rate, taking a value of 8%, which is the discount rate for public construction projects jointly announced by the National Development and Reform Commission (NDRC) and the Ministry of Construction; t is the time standing for years from 2001 to 2010; $Q_{1,t}$ is the amount of carbon not released to the atmosphere and $Q_{2,t}$ is the amount of carbon sequestered; P_1 is the required incentive for Chinese farmers to retain burned crop residues on croplands for each ton of additional carbon produced. In order for farmers to shift from burning crop residues to retaining them on croplands, incentives need to at least enable them to break-even. When NPV_1 was set at 0, the value of P_1, which was assumed to be constant over time, was obtained.

In the second case, non-permanence was considered. Following Sohngen and Mendelsohn (2003), a rental rate of carbon, which was the value of storing a ton of carbon for one year, was included in the *NPV* calculation:

$$NPV_2 = \sum \frac{B_t - C_t}{(1+i)^t} = P_2 \cdot \sum \frac{\sum_{i=2001}^{2010} Q_{1,t} + \sum_{i=2001}^{2010} Q_{1,t} \cdot rental_{rate} + Q_{2,2010}}{(1+i)^t} - \sum \frac{C_t}{(1+i)^t} \quad (3)$$

where $Q_{1,t}$ is the amount of carbon not released to the atmosphere and $Q_{2,t}$ is the amount of carbon sequestered; P_2 is the incentive required for Chinese farmers to retain burned crop residues on croplands for each ton of additional carbon produced; $rental_{rate}$ is the rental rate, which is equivalent to the market interest rate (5%). When NPV_2 was set at 0, the value of P_2, which was assumed to be constant over time, was obtained.

In the above two cases, the required incentives were calculated based on the quantity of carbon sequestered or not released. This type of required incentive was denoted as a “quantity-based” incentive. Considering that subsidies for retaining crop residues on croplands in China are usually paid based on unit area, area-based incentives were then calculated. Denoting CO_2e produced per hectare of land as q, at the break-even point, the quantity-based incentive P was calculated as:

$$P = \sum \frac{C_t}{(1+\delta)^t} \Big/ \sum \frac{q \cdot area}{(1+\delta)^t} \quad (4)$$

which is a function of q, subject to non-permanence issues.

Denoting the area-based incentives as *P’*, at the break-even point, *P’* was calculated as:

$$P' = P \cdot q = \sum \frac{C_t}{(1+\delta)^t} \Big/ \sum \frac{area}{(1+\delta)^t} \quad (5)$$

which is not related to q and not subject to non-permanence issues.

The above cost-benefit analyses were conducted for each of the seven regions by considering regional differences in biophysical conditions, crop residue management, and the cost of retaining crop residues on croplands.

3. Results and discussion

3.1 Soil carbon sequestration through crop residue retention

Simulation results for the SOC dynamics for the whole of China are presented in Figure 2. In Figure 2a, the line representing CR3 is upward sloping, showing that the density of organic carbon in the soil to a plow layer depth of 20 cm (OCPD) in China's croplands continuously increased from 2000 to 2010. In contrast, the lines representing CR1 and CR2 are both downward sloping, showing that the density continuously decreased under both scenarios. Moreover, the line for CR1 has a steeper slope than that of CR2, indicating that the density in CR1 decreased at a faster rate than that in CR2 during the period 2000-2010. Specifically, under CR3, OCPD increased from about 21.78 t ha^{-1} in 2000 to 23.85 t ha^{-1} in 2010; under CR2, the density decreased from 21.78 t ha^{-1} to 20.64 t ha^{-1} between 2000 and 2010. The annual average loss of SOC during 2001-2010 was higher (28.59 Tg C yr^{-1}) in CR1 than in CR2 (10.30 Tg C yr^{-1}) (Table S2), which is consistent with previous studies (Liu et al., 2014; Lu et al., 2015). This result further confirms findings that a lack of crop residue retention is one cause of a net loss of SOC in China's croplands (Li et al., 2003; Tang et al., 2006).

In Figure 2b, the line representing CR3 is always above zero, while the two lines representing CR1 and CR2 remain below zero. The annual average change in OCPD in CR3 during 2000-2010 was around 0.21 t ha^{-1}, leading to an annual average increase of 14.08 Tg C; the annual average change in OCPD in CR2 during 2000-2010 was around -0.11 t ha^{-1}, resulting in a total loss of 10.30 Tg C per year in China. The above information shows that [1] there was a net loss of SOC due to crop residue burning (CR2) during 2000-2010, and [2] there would have been a net increase in SOC if the residues that were burned had been retained by farmers on the cropland (CR3).

The spatial heterogeneity in SOC dynamics under the three crop residue management scenarios is presented in Figure 3. In CR1, croplands in China had different degrees of net loss in SOC during 2000-2010. The Northeastern China Plain—where cropland has fertile black soil and high initial SOC levels (Xie et al., 2007)—had the highest loss. This result is consistent with previous findings that a significant decrease in SOC has occurred in Northeast China (Huang and Sun, 2006; Tang et al., 2006; Yu et al., 2012). In CR2, SOC loss was lower than in CR1, and the Northern China Plain showed a net increase in SOC (see Figure S3, Table S2). In CR3, there was a net increase in SOC in most regions, except for in croplands in Southwest China and paddy fields in several regions (Figure S3, Table S2).

The above spatial heterogeneity may be explained by differences in crop types, cropping systems, and soil types across different regions. For example, drylands with a rotation system of "wheat-corn" are concentrated in northern regions, while paddy fields with double-cropping rice are prevalent in southern regions (Figure S2). Regarding soil types, croplands in northern and western regions are mostly sandy while in southern regions, croplands are dominated by clay (Shangguan et al., 2012).

Regional differences in carbon sequestration in croplands, using removal of all crop residues as the baseline scenario (i.e., CR1), are presented in Figure 4. When farmers managed crop residues as today (CR2), croplands in Northeast China had the highest SOC sequestration among all regions. When farmers retained those crop residues that were burned on their croplands (CR3), croplands in the Northern China Plain showed the highest SOC sequestration, followed by those in Northeast China (Figure 4). Under CR3, croplands in Northwest China showed the lowest SOC sequestration. Northeastern China and the Northern China Plain may demonstrate higher SOC sequestration through crop residue retention as these two regions are major agricultural production regions and produce more crop residues. When the crop residue retention rate is increased, the carbon input to croplands is higher and the potential for SOC sequestration increases as well. It is estimated that crop residue retention on croplands (CR3 vs. CR1) in North and Northeast China together represented about 21.4 Tg of SOC sequestration per year (Figure 4), accounting for around half of the national annual total SOC sequestration for 2000-2010.

Differences in SOC sequestration also exist between drylands and paddy fields. The SOC sequestration in China's drylands was estimated to be 33.68 Tg yr^{-1} under CR3 during 2000-2010 (Figure 4), which is nearly 3.75 times more than that in paddy fields. Retaining crop residues that are currently burned on croplands sequestered about 20.01 Tg carbon per year in the drylands, while only 4.36 Tg carbon were sequestered per year in paddy fields during 2000-2010.

Using the business-as-usual scenario (CR2) as the baseline, results from simulations show that the annual SOC sequestration in CR3 during 2001-2010 was 24.4 Tg C yr^{-1} (Figure 4, Table S3), which is equivalent to 1.1% of China's total carbon emissions from fossil fuels in 2010. The SOC sequestration from changing CR2 to CR3 strongly suggests that retaining crop residues that were burned on croplands during 2001-2010 could have helped to mitigate climate change. Spatial differences also exist for SOC sequestration across different regions (Figure 4, S4). Among the seven regions studied, North China showed the highest potential for SOC sequestration, implying that encouraging farmers to change their crop residue management in this region may be most effective for SOC carbon sequestration.

3.2 Emissions from crop residue burning in China

Estimated emissions from crop residue burning in China during 2000-2010 in CR2 are presented in Figure 5. It is estimated that the annual average release of CO_2, CH_4, $PM_{2.5}$, and PM_{10} from crop residue burning was 149.89 Tg, 0.19 Tg, 1.23 Tg, and 0.77 Tg, respectively. In 2010 alone, about 179.12 Tg of CO_2 was released, equivalent to 2.1% of China's total carbon emissions (2.25 Pg) from fossil-fuel burning, cement manufacturing, and gas flaring in 2010 (Boden et al., 2011). During 2000-2010, North, Central, and Northeast China were the three largest emitters in terms of annual average emissions of CO_2 from crop residue burning. CO_2 emissions released from crop residue burning in these three regions, which are major agricultural production areas in China, accounted for 68% (30.8% in North China, 20.5% from Central China, and 16.8% from Northeast China) of total CO_2 emissions from crop residue burning in China during 2000-2010 (Table S4). The above results imply that CO_2 emissions would have been reduced if crop residues burned had been retained on croplands during 2000-2010.

3.3 Required incentives to retain burned crop residues

Results from the cost-benefit analyses for seven regions are presented in Table 4 (see details in the Supplementary Information, Table S5-S9). Columns 2 and 3 in Table 4 show the quantity-based incentives with and without considering non-permanence, respectively. Column 4 shows the area-based incentives.

The incentives required to encourage Chinese farmers to shift from burning to retaining crop residues varied across regions (Table 4). Without considering non-permanence issues, the required incentives ranged from 30 CNY per ton of CO_2 equivalent (tCO_2e) to 122 CNY tCO_2e^{-1}. Of all the regions, Northeast China had the lowest required incentive (30.44 CNY tCO_2e^{-1}), followed by North China. On the other hand, East China required the highest incentive (122.42 CNY tCO_2e^{-1}), followed by South China (112.41 CNY tCO_2e^{-1}). When non-permanence is considered (column 3 of Table 4), the required incentive across regions fell in the range of approximately 40-189 CNY tCO_2e^{-1}, higher than those when non-permanence was not considered. This shows that it is more expensive for farmers to provide carbon benefits by shifting from burning to retaining crop residues on their cropland when non-permanence issues are considered.

The Chinese government launched a nationwide carbon market in late 2017. Comparing the required incentives to prevailing prices in seven pilot carbon markets launched in 2012, the required incentives are generally higher than the market prices. Even without considering non-permanence issues, the required incentives (30-122 CNY tCO_2e^{-1}) are much higher than prevailing carbon prices in seven pilot carbon markets from 2014 to 2018 (1.40-60.20 CNY tCO_2e^{-1}) (China Carbon Emissions Trading Network, 2018). To be more specific, even without considering the discount rate and inflation rate, the carbon prices in most parts of China, such as Central China and South China, are much lower than the required incentives to stop farmers from burning crop residues, except for in North China where the carbon price is occasionally higher than subsidies. The required incentive (111.83 CNY tCO_2e^{-1}) in Central China is more than four times the price of carbon (10.81-27.88 CNY tCO_2e^{-1}) (China Carbon Emissions Trading Network, 2018). Currently, it seems challenging to design an offset scheme that involves soil carbon sequestration to incentivize farmers to retain crop residues instead of burning them.

In terms of area-based required incentives, the last column in Table 4 shows that Northeast China (287.00 CNY ha^{-1}) requires the lowest incentives for farmers to shift from burning residues to retaining them on croplands for carbon benefits. In contrast, farmers in East China (836.49 CNY ha^{-1}) need the highest level of required incentive, followed by Central (828.53 CNY ha^{-1}) and Northwest China (781.45 CNY ha^{-1}). Farmers in these three regions need higher incentives than the current maximum subsidies for them to shift.

The existing subsidies are too low in most parts of China to incentivize farmers to shift from burning crop residues to retaining them on croplands. Particularly in East China, the subsidy level used in the pilot stage is only 150-375 CNY ha^{-1} (Table S1) while the required incentives are the highest (836.49 CNY ha^{-1}) of all seven regions. Indeed, this region has the highest share of crop residues burned (29%) among all regions. In North China and East China, the government subsidy is less than half of the required incentive. Only in Northeast China do subsidies (375-600 CNY ha^{-1}) (Table S1) exceed the required incentives (287 CNY ha^{-1}). In addition, the 10 pilot provinces for crop residue management do not cover Northwest China, Southwest China, Central China, or South China and the subsidy policies in these regions are lagging behind.

To summarize, the results in Table 4 show differences in the incentives required for farmers to shift from burning crop residues to retaining them on croplands across different regions. Farmers in Northeast China need the lowest incentive to shift while those in East China need the highest incentive. Among the seven regions, East China showed the highest rate (29%) of crop residues burned while Northeast China had the highest crop residue retention rate (33%) among all seven regions (Table 1). Given that Northeast China is a major food production area in China, this region also has high crop residue production. As 22% of crop residues are still burned in this region, and it had the lowest costs to incentivize farmers, this region should be prioritized for carbon benefits.

3.4 Limitations and uncertainties

Although the annual residues of rice, corn, and wheat account for more than 75% of the total crop residues (STEMOA, 2010), and the cultivated area is basically stable (Cheng et al., 2018), the reduction of crop systems to wheat-corn rotation systems in dryland and double-cropping rice systems in paddy fields is a simplification. Due to coarse spatial and temporal data on land use and cover, we did not consider any changes in cropland distribution during the simulated years. Besides which, the internal homogeneity of all input data for each simulated sub-area (a square area of 100 km^2) cannot be guaranteed. The above-mentioned facts could contribute to uncertainties in the SOC sequestration estimations in this study. In addition, SOC sequestration lasts decades (Lal, 2004a; West and Six, 2007) and the ten-year simulations used in this study cannot account for soil carbon saturation levels under long-term carbon inputs, which might cause an overestimation in the SOC sequestration potential of crop residue retention. More accurate spatial and temporal data and longer time periods would be helpful. It is worth noting that emissions from field burning of crop residues depend on complex factors, including the composition of crop residues, combustion temperatures, and ambient conditions. However, the measurements of emission factors for field burning are limited, especially in China (Cao et al., 2008). We used emission factors available in published studies, or even used average values (EF of PM_{10} for wheat residue). Experimental studies would validate the suitability of the EF values used in our study.

In this study we only focus on carbon benefits without considering all other possible co-benefits. Although retaining crop residues on croplands rather than burning them upon harvest may bring other benefits, such as improved soil fertility that is beneficial for crop yields (Witt et al., 2000; Lal, 2004b; Malhi and Kutcher, 2007; Zhao et al., 2015) and water availability (Bescansa et al., 2006; Sissoko et al., 2013), greater amounts of crop residues retained on croplands may also result in water percolation and promote nitrogen leaching, which may negatively affect crop growth and yields (Yang et al., 2016). In addition, when crop residues are retained on croplands, the survival of soil-borne plant pathogens in crop residues may also make diseases more problematic (Cook et al., 1978; Sturz et al., 1997; Bockus and Shroyer, 1998; Xia and Wu, 2013). Thus, our study acts as a starting point for future studies that could incorporate these additional costs and benefits.

The role of carbon markets in encouraging Chinese famers to retain crop residues on croplands instead of burning is largely uncertain. Based on the prevailing price of the seven pilot carbon markets during 2013-2016, it seems that carbon trading alone may not be sufficient to incentivize farmers to shift their crop residue practices. The carbon prices in the nascent national market are still unpredictable, given tremendous changes faced by seven pilot carbon markets in China, such as high transaction costs, lack of scientific data, and weak legal foundations (Auffhammer et al., 2015). Thus, given the current situation, it is difficult to judge whether designing an offset scheme involving soil carbon sequestration can be a viable solution for incentivizing farmers to provide carbon benefits through changes in crop residue management.

Last, it is important to note that owing to data limitations, this study only used aggregated data to analyze the required incentives for the period 2001-2010 to provide some benchmark information for future analyses. If more accurate estimations need to be conducted in the future, updated information must be collected through detailed surveys, preferably conducted at disaggregated levels, such as the household.

4. Conclusions

Through a combination of simulations with economic analyses, this study assessed the changes in C emissions and SOC dynamics associated with different crop residue retention rates. We found a net loss of SOC during 2000-2010 based on current retention rates. However, there would have been a net increase in SOC if the residues that were burned had been retained by farmers on croplands. If farmers could change their crop residue management practices from burning to retention on field, approximately 149.9 Tg yr^{-1} of CO_2 emission could be avoided and 24.4 Tg C yr^{-1} of SOC could be sequestered, albeit with significant heterogeneity among the seven regions studied. Northeast China and North China had the highest SOC sequestration. From an economic point of view, Northeast China is the lowest-cost region in China to incentivize farmers to change their residue management practices for carbon benefit, while East China is the most expensive. However, the subsidy standards set by the government in recent years are too low. Except for subsidies in Northeast China, which may be considered sufficient, the government subsidies in other parts of China still do not provide sufficient incentives for farmers to shift from burning to retaining crop residues. It can be speculated that current subsidies may not be sufficient to prevent Chinese farmers from burning crop residues. Besides, the incentives required to encourage farmers to retain crop residues are much higher than prevailing carbon prices in seven pilot carbon markets, making it impossible for carbon markets alone to provide sufficient incentives for farmers to change their crop residue management practices.

Taken together, neither the existing subsidies nor the carbon market provides sufficient incentives for farmers in China to change their crop residue management practices. As abolition of crop residue burning may be accompanied with other co-benefits, such as reduced health impacts because of reduced emissions of particulate matter, the government of China should consider providing higher subsidies or seeking other innovative solutions to encourage farmers to improve their crop residue management. Foremost, existing subsidies would need to be increased in North China and East China. The subsidies in North China need to increase by 100-150 CNY ha^{-1} and the subsidy in East China would have to be tripled (from 451 CNY ha^{-1} to 686 CNY ha^{-1}). Meanwhile, other non-pilot provinces should develop scientific subsidy policies as soon as possible. In addition, we suggest including soil carbon sink trade on the carbon market as a supplementary policy tool. Although the carbon market alone cannot solve the problem of crop residue burning, it can still be a supplementary policy tool, especially in North China.

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Reference

1. Ahmed, T., Ahmad, B., Ahmad W., 2015. Why do farmers burn rice residue? Examining farmers' choices in Punjab, Pakistan. Land Use Policy 47: 448-458.
2. Auffhammer, M., Gong, Y., 2015. China's carbon emissions from fossil fuels and market-based opportunities for control. Annu. Rev. Resour. Econ. 7(1): 11-34.
3. Balkovič, J., van der Velde, M., Skalský, R., et al., 2014. Global wheat production potentials and management flexibility under the representative concentration pathways. Global and Planetary Change 122: 107-121.
4. Balkovič, J., Skalský, R., Folberth, C., Khabarov, N., Schmid, E., Madaras, M., Obersteiner, M., & van der Velde, M., 2018. Impacts and Uncertainties of +2◦C of Climate Change and Soil Degradation on European Crop Calorie Supply, Earth's Future 6, 373–395.
5. Bescansa, P., Imaz, M. J., Virto, I., et al., 2006. Soil water retention as affected by tillage and residue management in semiarid Spain. Soil and Tillage Research 87(1): 19-27.
6. Billen, N., Röder, C., Gaiser, T., et al., 2009. Carbon sequestration in soils of SW-Germany as affected by agricultural management—calibration of the EPIC model for regional simulations. Ecological Modelling 220(1): 71-80.
7. Bockus, W. W., Shroyer, J. P., 1998. The impact of reduced tillage on soilborne plant pathogens. Annual review of phytopathology 36(1): 485-500.
8. Boden, T. A., et al., 2011. Global, Regional, and National Fossil-Fuel CO_2 Emissions. Carbon Dioxide Information Analysis Center, Oak Ridge National Laboratory, U.S. Department of Energy, Oak Ridge, Tenn., U.S.A. (http://cdiac.ornl.gov/)
9. Cai, Y., Qiu, H., Xu, Z., 2011. Evaluation on potentials of energy utilization of crop residual resources in different regions of China. Journal of Natural Resources 10(000).
10. Cao, G., Zhang, X., Wang, D., et al., 2005. Inventory of emissions of pollutants from open burning crop residue. Journal of Agro-Environment Science 24(4): 800-804.
11. Cao, G., Zhang, X., Zheng, F., 2006. Inventory of black carbon and organic carbon

emissions from China. Atmospheric Environment 40(34): 6516-6527.

12. Cao, G., Zhang, X., Wang, Y. Q., et al., 2008. Estimation of emissions from field burning of crop straw in China. Chinese Science Bulletin 53(5): 784-790.
13. CAREI (Chinese Association for Rural Energy Industries), Strategic considerations for development and utilization of biological energy in China (in Chinese). July, 2000. Beijing, 61pp.
14. Causarano, H. J., Shaw, J. N., Franzluebbers, A. J., et al., 2007. Simulating field-scale soil organic carbon dynamics using EPIC. Soil Science Society of America Journal 71(4): 1174-1185.
15. Chen, J., Li, C., Ristovski, Z., et al., (2017). A review of biomass burning: Emissions and impacts on air quality, health and climate in China. Science of the Total Environment, 579, 1000-1034.
16. Cheng, W., Gao, X., Ma, T., et al., 2018. Spatial-temporal distribution of cropland in China based on geomorphologic regionalization during 1990-2015, Acta Geographica Sinica 73(9): 1613-1629.
17. China Carbon Emissions Trading Network, 2018. http://www.tanpaifang.com/
18. Cook, R. J., Boosalis, M. G., Doupnik, B., 1978. Influence of crop residues on plant diseases. Crop residue management systems (cropresiduemana): 147-163.
19. Cui, M., Zhao, L. X., Tian, Y. S., et al., 2008. Analysis and evaluation on energy utilization of main crop straw resources in China. Transactions of the CSAE 24(12): 291-296
20. Elshout, P. M. F., Van Zelm, R., Balkovic, J., et al., 2015. Greenhouse-gas payback times for crop-based biofuels. Nature Climate Change 5(6): 604.
21. Fan, L., Lu, C., Chen, Z., 2012. A Review of EPIC Model and Its Applications. Progress In Geography 31(5): 584-592.
22. FAO (Food and Agriculture Organization of the United Nations), 2014. The Food and Agriculture Organization Corporate Statistical Database. http://faostat3.fao.org
23. Haider M. Z., 2013. Determinants of rice residue burning in the field. Journal of

environmental management 128: 15-21.

24. Han, B., Wang, X. K., Lu, F., et al., 2008. Soil carbon sequestration and its potential by cropland ecosystems in China. Acta Ecologica Sinica 28(2): 612-619.
25. Hays, M. D., Fine, P. M., Geron, C. D., et al., 2005. Open burning of agricultural biomass: physical and chemical properties of particle-phase emissions. Atmospheric environment 39(36): 6747-6764.
26. Hu, N., Wang, B., Gu, Z., et al., 2016. Effects of different straw returning modes on greenhouse gas emissions and crop yields in a rice–wheat rotation system. Agriculture, Ecosystems & Environment, 223, 115-122.
27. Huang, Y., Sun, W., 2006. Changes in topsoil organic carbon of croplands in mainland China over the last two decades. Chinese Science Bulletin 51(15): 1785-1803.
28. IPCC, 2000. Special Report on Emissions Scenario, Working Group III, Intergovernmental Panel on Climate Change. Cambridge: Cambridge University Press.
29. Izaurralde, R. C., Williams, J. R., Mcgill, W. B., et al., 2006. Simulating soil C dynamics with EPIC: Model description and testing against long-term data. Ecological Modelling 192(3-4): 362-384.
30. Jiang, D., Zhuang, D., Fu, J., et al., 2012. Bioenergy potential from crop residues in China: Availability and distribution. Renewable and sustainable energy reviews 16(3): 1377-1382.
31. Kanabkaew, T., Oanh, N. T. K., 2011. Development of spatial and temporal emission inventory for crop residue field burning. Environmental Modeling & Assessment 16(5): 453-464.
32. Keck, M., Hung, D. T., 2018. Burn or bury? A comparative cost-benefit analysis of crop residue management practices among smallholder rice farmers in northern Vietnam. Sustainability Science 1-15.
33. Lal, R., 2004a. Soil carbon sequestration impacts on global climate change and food security. Science 304(5677): 1623-1627.
34. Lal, R., 2004b. Soil carbon sequestration to mitigate climate change. Geoderma 123(1-2):

1-22.

35. Li, C., Zhuang, Y., Frolking, S., et al., 2003. Modeling soil organic carbon change in croplands of China. Ecological Applications 13(2): 327-336.
36. Li, F., Wang, J., 2013. Estimation of carbon emission from burning and carbon sequestration from biochar producing using crop straw in China. Transactions of the Chinese Society of Agricultural Engineering 29(14): 1-7.
37. Li, J., Bo, Y., Xie, S., 2016. Estimating emissions from crop residue open burning in China based on statistics and MODIS fire products. Journal of Environmental Sciences 44: 158-170.
38. Li, X., Wang, S., Duan, L., et al., 2007. Particulate and trace gas emissions from open burning of wheat straw and corn stover in China. Environmental Science & Technology 41(17): 6052-6058.
39. Lin, F. Y., Wu, Y. J., Wang, S. Q., et al., 2013. Simulation and Prediction of Straw Return on Soil Carbon Sequestration Potential of Cropland in Jiangxi Province. Journal of Natural Resources 28(6): 981-993. doi: 10.11849/zrzyxb.2013.06.009.
40. Liu, C., Lu, M., Cui, J., et al., 2014. Effects of straw carbon input on carbon dynamics in agricultural soils: a meta- analysis. Global change biology 20(5): 1366-1381.
41. Liu, H., Jiang, G. M., Zhuang, H. Y., et al., 2008. Distribution, utilization structure and potential of biomass resources in rural China: with special references of crop residues. Renewable and Sustainable Energy Reviews 12(5): 1402-1418.
42. Liu, J., Williams, J. R., Zehnder, A. J. B., et al., 2007. GEPIC–modelling wheat yield and crop water productivity with high resolution on a global scale. Agricultural systems 94(2): 478-493.
43. Liu, J., Folberth, C., Yang, H., et al., 2013. A global and spatially explicit assessment of climate change impacts on crop production and consumptive water use. PLoS One 8(2): e57750.
44. Liu, J., Liu, M., Zhuang, D., et al., 2003. Study on spatial pattern of land-use change in China during 1995–2000. Science in China Series D: Earth Sciences 46(4): 373-384.

45. Liu, J., Tian, H., Liu, M., et al., 2005. China's changing landscape during the 1990s: Large- scale land transformations estimated with satellite data. Geophysical Research Letters 32(2).

46. Liu, Q. H., Shi, X. Z., Weindorf, D. C., et al., 2006. Soil organic carbon storage of paddy soils in China using the 1: 1,000,000 soil database and their implications for C sequestration. Global Biogeochemical Cycles 20(3).

47. Llewellyn, R. S., D'Emden, F., 2010. Adoption of no-till cropping practices in Australian grain growing regions. Australian Government, Grains Research and Development Corporation.

48. Lokupitiya, E., Paustian, K., Easter, M., et al., 2012. Carbon balances in US croplands during the last two decades of the twentieth century. Biogeochemistry 107(1-3): 207-225.

49. Lou, Y., Xu, M., Wang, W., et al., 2011. Return rate of straw residue affects soil organic C sequestration by chemical fertilization. Soil and Tillage Research 113(1): 70-73.

50. Lu, F., Wang, X., Han, B., et al., 2009. Soil carbon sequestrations by nitrogen fertilizer application, straw return and no-tillage in China's cropland. Global Change Biology 15(2): 281-305.

51. Lu, F., Wang, X., Han, B., et al., 2010. Net mitigation potential of straw return to Chinese cropland: estimation with a full greenhouse gas budget model. Ecological Applications 20(3): 634-647.

52. Lu, F., 2015. How can straw incorporation management impact on soil carbon storage? A meta-analysis. Mitigation and Adaptation Strategies for Global Change 20(8): 1545-1568.

53. Malhi, S. S., Kutcher, H. R., 2007. Small grains stubble burning and tillage effects on soil organic C and N, and aggregation in northeastern Saskatchewan. Soil and Tillage Research 94(2): 353-361.

54. Marland, G., Fruit, K., Sedjo, R., 2001. Accounting for sequestered carbon: the question of permanence. Environmental Science & Policy 4(6): 259-268.

55. McCarty, J. L., Ellicott, E. A., Romanenkov, V., et al., 2012. Multi-year black carbon

emissions from cropland burning in the Russian Federation. Atmospheric Environment 63: 223-238.

56. Mei, F. C., 2008. Cost-benefit analysis of crop burning pollution in Xinyang, Henan (in Chinese). Environmental Science and Management 33(1): 30-32.
57. Monteith, J. L., 1977. Climate and the Efficiency of Crop Production in Britain Philos. Trans. R. Soc. Lond. B. Biol. Sci. 281: 277-94.
58. Montgomery, D. R., 2007. Soil erosion and agricultural sustainability. Proceedings of the National Academy of Sciences 104(33): 13268-13272.
59. Murray, B. C., Sohngen, B., Ross, M. T., 2007. Economic consequences of consideration of permanence, leakage and additionality for soil carbon sequestration projects. Climatic Change 80(1-2): 127-143.
60. NBSPRC (National Bureau of Statistics of the People's Republic of China). China Rural Statistical Yearbook 2001-2010, Beijing: China Statistics Press.
61. Pimentel, D., Harvey, C., Resosudarmo, P., et al, 1995. Environmental and Economic Costs of Soil Erosion and Conservation Benefits. Science 267(5201): 1117-1123.
62. Scarlat, N., Martinov, M., Dallemand, J. F., 2010. Assessment of the availability of agricultural crop residues in the European Union: potential and limitations for bioenergy use. Waste management 30(10): 1889-1897.
63. Shangguan, W., Dai, Y., Liu, B., et al., 2012. A soil particle-size distribution dataset for regional land and climate modelling in China. Geoderma 171: 85-91.
64. Shi, X. Z., Yu, D. S., Pan, X. Z., Sun, W. X., Gong, Z. G., Warner, E. D., & Petersen, G. W., 2002. Framework for the 1: 1,000,000 soil database of China. In 17. World congress of soil science, Bangkok (Thailand) 14-21 Aug 2002.
65. Sissoko, F., Affholder, F., Autfray, P., et al., 2013. Wet years and farmers' practices may offset the benefits of residue retention on runoff and yield in cotton fields in the Sudan–Sahelian zone. Agricultural water management 119: 89-99.
66. Sohngen, B., Mendelsohn, R., 2003. An optimal control model of forest carbon sequestration. American Journal of Agricultural Economics 85(2): 448-457.

67. STEMOA (Department of Science and Technology and Education, Ministry of Agriculture of the People's Republic of China), 2010. Survey and access report of national crop straw source. http://d.wanfangdata.com.cn/Periodical_nygcjs201102002.aspx
68. Stewart, B. A., Moldenhauer, W. C., 1994. Crop residue management to reduce erosion and improve soil quality: Southern Great Plains. Conservation research report (USA).
69. Stockle, C. O., Williams, J. R., Rosenberg, N. J. and Jones, C. A., 1992. A method for estimating the direct and climatic effects of rising atmospheric carbon dioxide on growth and yield of crops: Part I—Modification of the EPIC model for climate change analysis Agric. Syst. 38 225–38.
70. Streets, D. G., Yarber, K. F., Woo, J. H., et al., 2003. Biomass burning in Asia: Annual and seasonal estimates and atmospheric emissions. Global Biogeochemical Cycles 17(4).
71. Sturz, A. V., Carter, M. R., Johnston, H. W., 1997. A review of plant disease, pathogen interactions and microbial antagonism under conservation tillage in temperate humid agriculture. Soil and Tillage Research 41(3-4): 169-189.
72. Sun, J., Peng, H., Chen, J., et al., 2016. An estimation of CO2 emission via agricultural crop residue open field burning in China from 1996 to 2013. Journal of Cleaner Production 112: 2625-2631.
73. Tang, H., Qiu, J., Van Ranst, E., et al., 2006. Estimations of soil organic carbon storage in cropland of China based on DNDC model. Geoderma 134(1-2): 200-206.
74. van der Velde, M., Bouraoui, F., Aloe, A., 2009. Pan- European regional- scale modelling of water and N efficiencies of rapeseed cultivation for biodiesel production. Global Change Biology 15(1): 24-37.
75. Venkataraman, C., Habib, G., Kadamba, D., et al., 2006. Emissions from open biomass burning in India: Integrating the inventory approach with high- resolution Moderate Resolution Imaging Spectroradiometer (MODIS) active- fire and land cover data. Global biogeochemical cycles 20(2).
76. West, T. O., Six, J., 2007. Considering the influence of sequestration duration and carbon

saturation on estimates of soil carbon capacity. Climatic Change 80(1-2): 25-41.

77. Williams, J. R., Renard, K. G., 1985. Assessments of soil erosion and crop productivity with process models (EPIC). Soil erosion and crop productivity (soilerosionandc): 67-103.

78. Williams, J. R., Jones, C. A., Kiniry, J. R., et al., 1989. The EPIC crop growth model. Transactions of the ASAE 32(2): 497-0511.

79. Williams, J.R., et al., 1995. The EPIC model. In: Singh, V.P. (Ed.), Computer Models of Watershed Hydrology. Water Resources Publications, Highlands Ranch, CO, pp. 909–1000.

80. Williams, J. R., et al., 2006. EPIC users guide v. 0509. Blackland Research and Extension Center, Temple, Texas.

81. Witt, C., Cassman, K. G., Olk, D. C., et al., 2000. Crop rotation and residue management effects on carbon sequestration, nitrogen cycling and productivity of irrigated rice systems. Plant and Soil 225(1-2): 263-278.

82. Xia, Y., Wu, Y., 2013. Research on Effect of Straws Back into Field on Pests and Diseases and Yield. North Rice 43(6): 37-39.

83. Xia, L., Wang, S., Yan, X. (2014). Effects of long-term straw incorporation on the net global warming potential and the net economic benefit in a rice–wheat cropping system in China. Agriculture, ecosystems & environment, 197, 118-127.

84. Xie, Z., Zhu, J., Liu, G., et al., 2007. Soil organic carbon stocks in China and changes from 1980s to 2000s. Global Change Biology 13(9): 1989-2007.

85. Yan, H., Cao, M., Liu, J., et al., 2007. Potential and sustainability for carbon sequestration with improved soil management in agricultural soils of China. Agriculture, ecosystems & environment 121(4): 325-335.

86. Yan, X., Ohara, T., Akimoto, H., 2006. Bottom-up estimate of biomass burning in mainland China. Atmospheric Environment 40(27): 5262-5273.

87. Yang, H., Xu, M., Koide, R. T., et al., 2016. Effects of ditch- buried straw return on water percolation, nitrogen leaching and crop yields in a rice–wheat rotation system.

Journal of the Science of Food and Agriculture 96(4): 1141-1149.

88. Yu, Y., Huang, Y., Zhang, W., 2012. Modeling soil organic carbon change in croplands of China, 1980–2009. Global and Planetary Change 82: 115-128.

89. Yu, Y., Huang, Y., Zhang, W., 2013. Projected changes in soil organic carbon stocks of China's croplands under different agricultural managements, 2011–2050. Agriculture, ecosystems & environment 178: 109-120.

90. Zhang, H., Ye, X., Cheng, T., et al., 2008. A laboratory study of agricultural crop residue combustion in China: emission factors and emission inventory. Atmospheric Environment 42(36): 8432-8441.

91. Zhang, Y., Shao, M., Lin, Y., et al., 2013. Emission inventory of carbonaceous pollutants from biomass burning in the Pearl River Delta Region, China. Atmospheric environment 76: 189-199.

92. Zhao, X., Hu, K., Stahr, K., 2013. Simulation of SOC content and storage under different irrigation, fertilization and tillage conditions using EPIC model in the North China Plain. Soil and Tillage Research 130: 128-135.

93. Zhao, Y., et al., 2015. Effects of tillage and straw returning on microorganism quantity, enzyme activities in soils and grain yield. Chinese Journal of Applied Ecology 26.6.

94. Zhao, Y., Wang, M., Hu, S., Zhang, X., Ouyang, Z., & Zhang, G., et al., 2018. Economics- and policy-driven organic carbon input enhancement dominates soil organic carbon accumulation in Chinese croplands. Proceedings of the National Academy of Sciences, 115(16), 4045-4050.

Table and figure legends

Table 1. Current rates of crop-residue burning and retention in seven regions of China

Table 2. Parameter calibration in the EPIC model

Table 3. Emission factors for carbon and pollutants (g kg^{-1})

Table 4. Incentives required to encourage farmers retain the burned crop residues on croplands

Figure 1. Flow diagram for the EPIC model simulation

Figure 2. (a) Average organic carbon in the soil to a plow layer depth of 20 cm (OCPD), and (b) its annual variation in croplands of China under different scenarios. CR1, CR2, and CR3 present the baseline, the business-as-usual and the improved crop residue management scenarios, respectively. All values are nationwide averages.

Figure 3. Spatial patterns of carbon changes in cropland of China under three management scenarios (a) CR1, (b) CR2, and (c) CR3. Variations in organic carbon in the soil to a plow layer depth of 20 cm (OCPD) were calculated from the OCPD of each year minus that of the year before. All values are annual averages calculated for the period 2001-2010.

Figure 4. SOC sequestration in seven regions. "CR2-CR1" and "CR3-CR1" represent the differences between SOC pools under crop residue retention (CR2 and CR3) and crop residue removal (CR1) scenarios; "CR3-CR2" represents the difference between SOC pools in the improved retention scenario (CR3) and the business-as-usual retention scenario (CR2), namely the SOC sequestration of retaining those burned crop residues. All values are annual averages for 10 years.

Figure 5. Time series of greenhouse gases and particulate matter emissions from crop residue burning. The annual emissions of (a) CO_2, (b) CH_4, (c) $PM_{2.5}$, and (d) PM_{10} from corn, wheat, and rice residue burning from 2000 to 2010. Each value represents the sum of each type of emissions across seven regions.

Table 1. Current rates of crop-residue burning and retention in seven regions of China

Region	Rate of crop-residue burning	Rate of crop-residue retention	Sum
Northeast China	22%	33%	55%
North China	24%	28%	52%
Northwest China	17%	17%	34%
Southwest China	17%	15%	32%
Central China	23%	22%	45%
South China	27%	23%	50%
East China	29%	23%	52%

Note: The Sum represents the largest percentage of crop-residue retained if the originally burned crop residues were retained on cropland (CR3).

Table 2. Parameter calibration in the EPIC model

Parameter	Description	Default value	Modified value
Crop related			
HI	harvest index	0.2 (rice) 0.5 (corn) 0.45 (wheat)	0.5 (rice) 0.55 (corn) 0.47 (wheat)
DMLA	maximum potential leaf area index	6 (rice) 6 (corn)	6.82 (rice) 7 (corn)
DLAI	fraction of the growing season when LAI begins to decline	0.8 (rice) 0.8 (corn) 0.6 (wheat)	0.9 (rice) 0.6 (corn) 0.5 (wheat)
HMX	maximum crop height (m)	2 (corn)	2.5 (corn)
RDMX	maximum root depth (m)	2 (corn)	2.5 (corn)
Soil related			
PARM20	microbial decay rate coefficient	1	1.5
PARM45	residue decay tillage coefficient	15	10
PARM47	slow humus transformation rate (d^{-1})	0.000548	0.00068
PARM48	passive humus transformation rate (d^{-1})	0.000012	0.000015
PARM61	soil erosion coefficient relates C factor to soil cover by flat and standing residue and growing biomass	1	2

Table 3. Emission factors for carbon and pollutants (g kg^{-1})

Fuel type	CO_2	CH_4	CO	NO	NO_2	NO_x	SO_2	$PM_{2.5}$	PM_{10}
Corn residue	1,261.5[a]	1.75[a]	114.7[a]	0.8 [a]	0.43[a]	1.28[a]	0.44[b]	11.7 [b]	4.3[d]
Wheat residue	1,557.9[a]	1.82[a]	141.2[a]	0.79[a]	0.32[a]	1.12[a]	0.85[b]	4.71[c]	8.05[e]
Rice residue	791.3[a]	0.72[a]	64.2[a]	1.02[a]	0.79[a]	1.81[a]	0.18[b]	12.95[c]	9.1[d]

[a]Zhang et al., 2008.

[b]Li et al., 2007.

[c]Hays et al., 2005.

[d]Kanabkaew et al., 2011.

[e]Owing to lack of data, the EF of PM_{10} for wheat residue was selected based on available EFs of PM_{10} for combined crop residues.

Table 4. Incentives required to encourage farmers retain the burned crop residues on croplands

Regions	Quantity-based		Area-based
	Not considering non-permanence (CNY tCO_2e^{-1})	Considering non-permanence (CNY tCO_2e^{-1})	(CNY ha^{-1})
Northeast China	30.44	39.60	287.00
North China	39.24	63.80	550.82
Northwest China	95.21	119.60	781.45
Southwest China	65.26	95.49	455.38
Central China	111.83	166.63	828.53
South China	112.41	186.15	465.08
East China	122.42	189.10	836.49

HIGHLIGHTS

- Farmers in China burn almost 1/4 of crop residues, causing environmental damages.
- Simulations and economic analyses combined to assess a shift in residue management.
- Residue retention avoided CO_2 emission by 150 Tg yr^{-1} and increased SOC sequestration by 24 Tg C yr^{-1}.
- Required incentives vary regionally; Northeast China required the lowest.
- Existing subsidies (150-600 CNY ha^{-1}) are lower than the incentives required (287-836 CNY ha^{-1}) to make a shift.

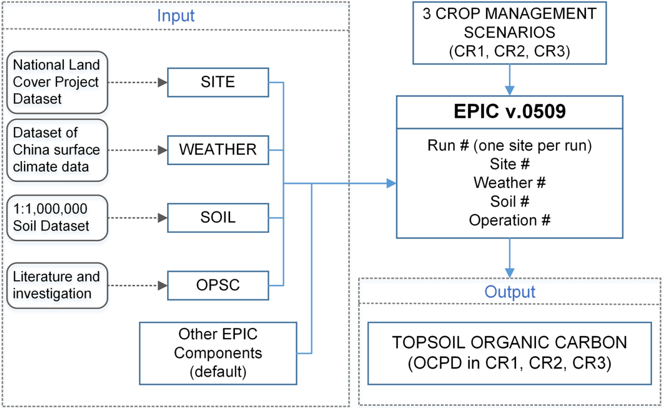

Input
National Land Cover Project Dataset
SITE
Dataset of China surface climate data
WEATHER
1:1,000,000 Soil Dataset
SOIL
Literature and investigation
OPSC
Other EPIC Components (default)
3 CROP MANAGEMENT SCENARIOS (CR1, CR2, CR3)
EPIC v.0509
Run # (one site per run)
Site #
Weather #
Soil #
Operation #
Output
TOPSOIL ORGANIC CARBON (OCPD in CR1, CR2, CR3)

Figure 1

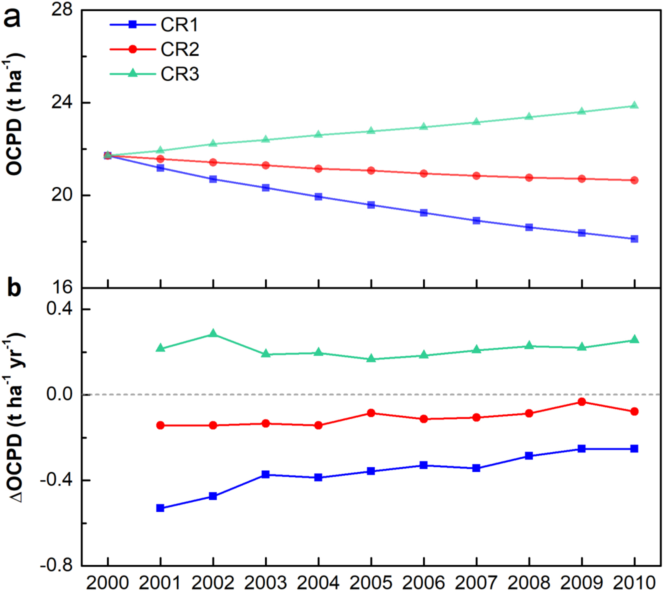

a
b
CR1
CR2
CR3
OCPD (t ha-1)
ΔOCPD (t ha-1 yr-1)
28
24
20
16
0.4
0.0
-0.4
-0.8
2000
2001
2002
2003
2004
2005
2006
2007
2008
2009
2010

Figure 2

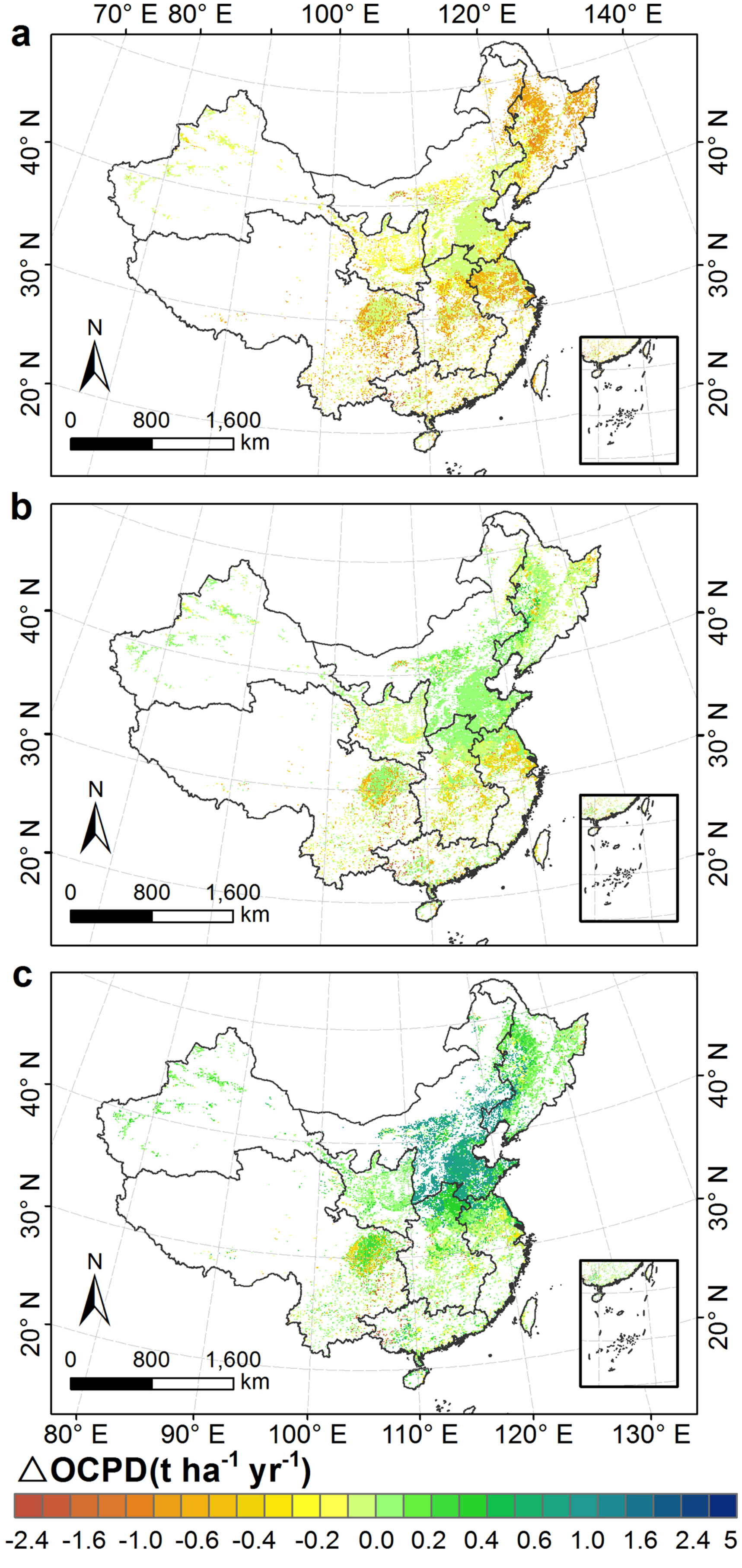
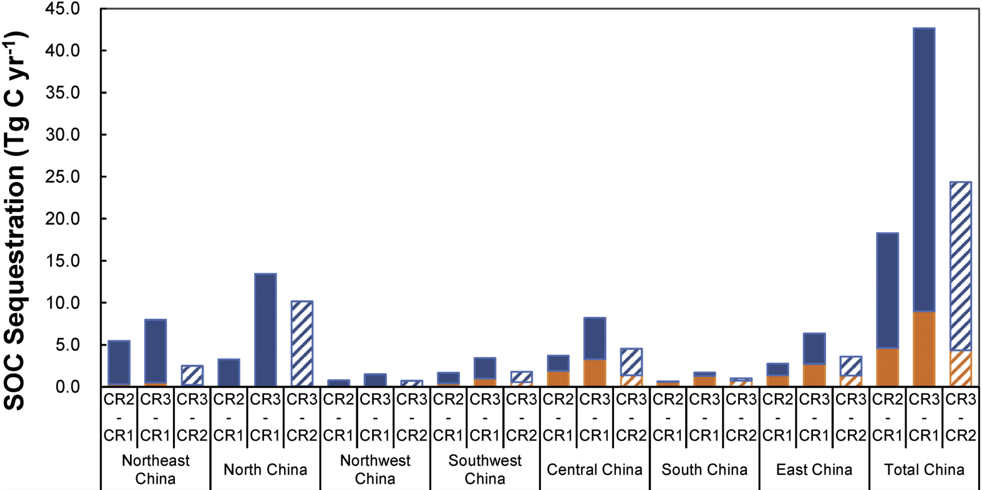

a
b
c
70° E 80° E
100° E
120° E
140° E
80° E
90° E
100° E
110° E
120° E
130° E
40° N
30° N
20° N
N
0
800
1,600
km
△OCPD(t ha^-1 yr^-1)
-2.4
-1.6
-1.0
-0.6
-0.4
-0.2
0.0
0.2
0.4
0.6
1.0
1.6
2.4
5

Figure 3



	Northeast China CR2-CR1	Northeast China CR3-CR1	Northeast China CR3-CR2	North China CR2-CR1	North China CR3-CR1	North China CR3-CR2	Northwest China CR2-CR1	Northwest China CR3-CR1	Northwest China CR3-CR2	Southwest China CR2-CR1	Southwest China CR3-CR1	Southwest China CR3-CR2	Central China CR2-CR1	Central China CR3-CR1	Central China CR3-CR2	South China CR2-CR1	South China CR3-CR1	South China CR3-CR2	East China CR2-CR1	East China CR3-CR1	East China CR3-CR2	Total China CR2-CR1	Total China CR3-CR1	Total China CR3-CR2
dryland	5.17	7.45	2.28	3.23	13.3	10.1	0.77	1.48	0.71	1.23	2.45	1.23	1.77	4.92	3.15	0.10	0.39	0.29	1.39	3.64	2.25	13.6	33.6	20.0
paddy field	0.31	0.53	0.23	0.02	0.09	0.06	0.02	0.04	0.02	0.44	1.00	0.56	1.92	3.31	1.39	0.58	1.31	0.74	1.36	2.72	1.36	4.63	8.99	4.36

Figure 4

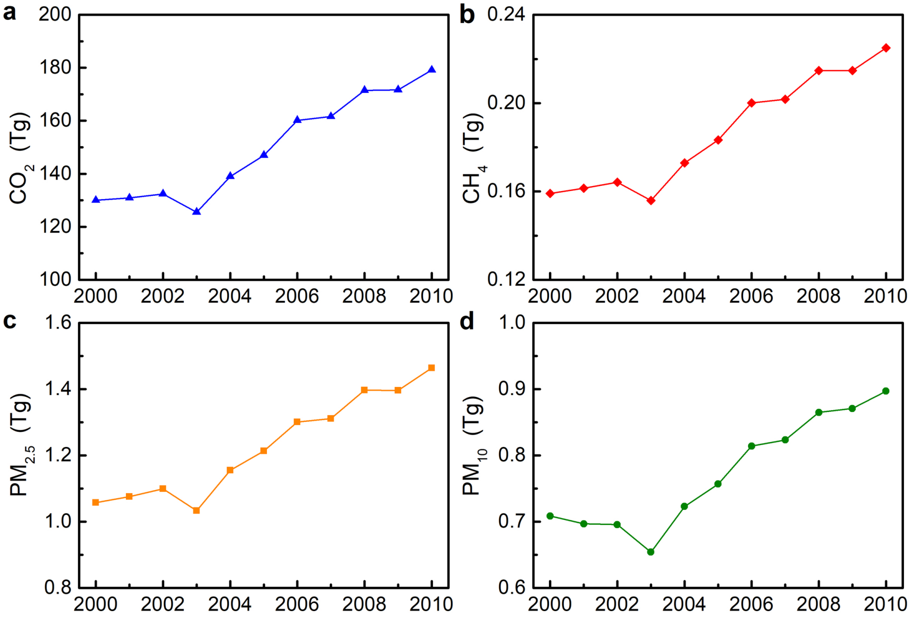

a
b
c
d
CO_2 (Tg)
CH_4 (Tg)
$PM_{2.5}$ (Tg)
PM_{10} (Tg)
200
180
160
140
120
100
0.24
0.20
0.16
0.12
1.6
1.4
1.2
1.0
0.8
1.0
0.9
0.8
0.7
0.6
2000
2002
2004
2006
2008
2010

Figure 5