Atmospheric Environment 50 (2012) 9–15

Contents lists available at SciVerse ScienceDirect

Atmospheric Environment

journal homepage: www.elsevier.com/locate/atmosenv

A high-resolution emission inventory of crop burning in fields in China based on MODIS Thermal Anomalies/Fire products

Xin Huang, Mengmeng Li, Jianfeng Li, Yu Song*

State Key Joint Laboratory of Environmental Simulation and Pollution Control, Department of Environmental Science, Peking University, Beijing 100871, China

ARTICLE INFO

Article history:
Received 24 August 2011
Received in revised form
23 November 2011
Accepted 5 January 2012

Keywords:
China
Crop residue
Fire emission
MODIS

ABSTRACT

Agricultural field burning plays an important role in atmospheric pollution and climate change. This work aims to develop a detailed emission inventory for agricultural burning in China with a high spatial and temporal resolution. Province-specific statistical data, distributed by the Chinese national government, and results from scientific literature were utilized to estimate the total emissions for the base year 2006. Emissions were allocated to a 1 km grid and a 10-day interval by using the Moderate Resolution Imaging Spectroradiometer (MODIS) Thermal Anomalies/Fire product (MOD/MYD14A1). The estimated annual emission ranges, with a 90% confidence interval, are 68 (51–85) Tg CO_2 yr^{-1}, 4 (2–7) Tg CO yr^{-1}, 0.25 (0.08–0.46) Tg CH_4 yr^{-1}, 2.2 (1.08–3.46) Tg NMOCs yr^{-1}, 0.23 (0.08–0.41) Tg NOx yr^{-1}, 0.09 (0.03–0.17) Tg NH_3 yr^{-1}, 0.02 (0.01–0.03) Tg SO_2 yr^{-1}, 0.03 (0.01–0.05) Tg BC yr^{-1}, 0.1 (0.04–0.17) Tg OC yr^{-1}, 0.27 (0.13–0.42) Tg $PM_{2.5}$ yr^{-1}, 0.31 (0.12–0.53) Tg PM_{10} yr^{-1}. Provinces with the highest emissions are Anhui, Guizhou and Hunan. Spatially, agricultural fires are mostly located in the North China Plain, where the occurrence of fires is concentrated in early and late June (over 75% of the whole year) with another smaller peak in early October. This pattern corresponds with sowing and harvesting times for the main crops: wheat and maize. The temporal fire variation of two other agricultural zones in northeast China and south China are also detailed in our study. Our inventory, with a relatively high spatiotemporal resolution (1 km grid and 10 days), could meet the need of global and regional air quality simulations.

1. Introduction

Agricultural field burning, which common both in rural agricultural regions and peri-urban areas, is one important kind of biomass burning (Yevich and Logan, 2003). Crop residue burning is a common management practice during the harvesting, post-harvesting or pre-planting periods. Crop residues are burned for a number of reasons, including to clear crop residue, provide short-lived ash fertilization, and manage pests (Korontzi et al., 2006). Agricultural burning was found to be the fourth largest type of biomass burning by Andreae and Merlet (2001). An estimated 250 Tg of crop residues were burned in Asia in 2000 (Streets et al., 2003a), and 400 Tg of crop residues were burned in all of the developing world in the 1990s (Yevich and Logan, 2003). Crop residue burning could be an important source of atmospheric trace gases and particulate matter (Dennis et al., 2002; Jenkins et al., 1992; Zhang et al., 2008), including carbon dioxide (CO_2), carbon monoxide (CO), methane (CH_4), Non-Methane Organic Compounds (NMOCs), black carbon and organic carbon (BC and OC), oxides of nitrogen (NOx), ammonia (NH_3), and sulfur dioxide (SO_2), and particulate matter (PM). Agricultural burning also contributes to atmospheric chemistry and global climate change (Crutzen and Andreae, 1990).

In China, a large agricultural country that ranks high in global crop production for several crops, agricultural field burning could play an important role on environmental pollution and climate change. The distribution of Moderate Resolution Imaging Spectroradiometer (MODIS) Thermal Anomalies/Fire product (MOD/MYD14) fire counts for various land cover types revealed that fires in croplands constituted approximately 30–40% of all fire detections in mainland China (Korontzi et al., 2006). An estimated 122 Tg of crop residues were burned in 2000, accounting for about 50% of all crop residue burning in Asia (Yan et al., 2006).

Nevertheless, fire occurrence and emissions caused by agricultural open fires in China has not been studied in great detail and is not well characterized. First, compared with forest and savanna fires, agricultural field burning are often missed by remote sensors because of their small size and temporal impermanency (Roy et al., 2008) or because the quantification of crop residue burning is completed through country-specific algorithms for satellite burned

* Corresponding author.
E-mail address: songyu@pku.edu.cn (Y. Song).

1352-2310/$ – see front matter © 2012 Elsevier Ltd. All rights reserved.
doi:10.1016/j.atmosenv.2012.01.017

Table 1
Dry weight ratios of straw to grain and combustion efficiency for different crops.

Crop	Production-to-residue ratios[a]	Combustion efficiency[b]
Rice	1.0	0.89
Wheat	1.0	0.86
Corn	2.0	0.92
Coarse cereals	1.0	0.80
Cotton	3.0	0.80
Legumes	1.5	0.68
Peanut	2.0	0.80
Rapeseed	2.0	0.82

[a] Values are from Lal (2005).
[b] Values are from de Zarate et al. (2005) and Turn et al. (1997).

area detection (McCarty et al., 2008). Thus, the emissions from crop residue field burning tend to be underestimated if they are based on global burned area products, which provide better estimations for nonagricultural fire sources (Song et al., 2010). Second, the burning of crop residues is strongly correlated with agricultural practices (harvesting cycles, types of crops, etc.). These factors are difficult to ascertain, especially in developing countries such as China. Emissions from agricultural open fires in China have been estimated in several publications mostly using annual statistic data (Cao et al., 2008; Streets et al., 2003b; Zhang et al., 2008). The limitation of these emission inventories is the low temporal resolution of one year. For most air quality simulations, a more detailed inventory with high temporal and spatial resolution is preferred.

In this study, CO_2, CO, CH_4, NMOCs, NOx, NH_3, SO_2, BC, OC, $PM_{2.5}$ and PM_{10} emissions from crop residue burning in fields in China (excluding small islands in the South China Sea) were estimated. We utilized the statistic data distributed by government offices to assess the provincial emissions initially. Next, the annual provincial results were allocated to 10-day intervals and 1 km grid emissions by combining daily MODIS Thermal Anomalies/Fire products and Global Land Cover 2000 for China (GLC-China) (Xu et al., 2005).

2. Methodology

2.1. Emission estimation

Crop residues, including residues from rice, wheat, corn, coarse cereals, cotton, legumes, peanut or rapeseed, are widely burned in fields in China (Cao et al., 2008; Gao et al., 2002). In this study, agricultural field burning emissions were initially estimated at a provincial level by multiplying the total mass of in-field burning crop residues and the corresponding emission factor (EF).

The provincial amounts of crop residues burned in the fields were calculated on the basis of the crop production using the following equation:

$$M_{i,k} = P_{i,k} \times R_k \times F_{i,k} \times CE_k \quad (1)$$

where i stands for each province; k for different crop species; $M_{i,k}$ is provincial mass of crop residue burned in the field in kg; $P_{i,k}$ is provincial crop production for various crops; R_k is crop-specific residue-to-production ratio (dry matter); $F_{i,k}$ is province and crop-specific percentage of crop residues burned in the field; CE_k is crop-specific percentage of combustion efficiency.

The total crop residue was the product of crop productions at the provincial level distributed by the government (NBSC, 2007) and the residue/crop ratio (de Zarate et al., 2005; Lal, 2005; Turn et al., 1997) (listed in Table 1). The values for the percentage of crop residues that were burned in fields were adopted from a large-scale investigation on the use of crop residues in different provinces (Gao et al., 2002); more details were presented in Table S1. Crop type-specific combustion efficiencies (Table 1) were compiled from Turn et al. (1997) and de Zarate et al. (2005).

Gaseous and particulate emissions from agricultural open fires were calculated by multiplying the burned mass and corresponding EFs. In recent years, considerable efforts have been made to quantify a wide range of emitted species from crop residue burning (McCarty, 2011; Zhang et al., 2008). The EFs for different pollutants have been summarized in two publications (Akagi et al., 2011; Andreae and Merlet, 2001). A summary of the EFs used in this study were listed in Table 2.

2.2. Spatial and temporal allocation

The goal of this study was to produce an extensive emission inventory with a higher resolution. Although crop residue burning in China may not be mapped by burned area products because of their small size, they could be located by fire count data (van der Werf et al., 2006). We selected MODIS Thermal Anomalies/Fire Daily L3 Global Product (MOD/MYD14A1) from 2003 to 2010, which provides data from both the Terra and Aqua satellites. The enhanced contextual fire detection algorithm was used MODIS Thermal Anomalies/Fire products through brightness temperatures derived from the MODIS 4-and 11-μm channels. The fire detection strategy was based on absolute detection of the fire, if the fire is strong enough, and on detection relative to the thermal emission of surrounding pixels to detect weaker fires (Giglio et al., 2003). The product is tile-based, with each product file spanning one of the 460 MODIS tiles, of which 326 contain land pixels, and in 1 km gridded cell over each daily (24 h) compositing period. Two observations per day are possible with the Terra overpass at 10:30 local time and the Aqua overpass at 14:30 local time (Giglio, 2010). Version 5 (V005) of the MODIS fire detection data was used in this analysis.

The 1 km land cover dataset, GLC-China, was used in this study to define croplands. This product covers the whole China region and has been found to be accurate for China (Song et al., 2009; Xu et al., 2005). Active fire detection that occurred on the land cover classes defined as “Farm” and “Mosaic of cropping” was identified as crop residue burning in fields.

The emissions in i-th grid (E_i) were calculated using the following equation:

$$E_i = \frac{FC_i}{FC_k} \times E_k \quad (2)$$

Where FC_i is the fire counts in i-th grid, FC_k is the total fire counts in province k, and E_k is the total emissions from crop residue burning in province k, estimated in section 2.1.

Table 2
Emission factor and its uncertainty for different pollutants (g Kg^{-1} dry crop residue).

	CO_2[a]	CO[a]	CH_4[a]	NMOCs[b]	NOx[a]	NH_3[a]	SO_2[b]	BC[a]	OC[a]	$PM_{2.5}$[a]	PM_{10}[b]
EF	1584.9	102.2	5.8	51.4	5.3	2.2	0.4	0.8	2.3	6.3	7.2
Uncertainty (CV)	6%	32%	61%	38%	58%	58%	50%	50%	50%	38%	50%

[a] Values are from Akagi et al. (2011).
[b] Values are from Andreae and Merlet (2001).

Table 3
Comparison of the fire emissions (Tg yr^{-1}) calculated in our study with other published estimates.

	CO_2	CO	CH_4	NMOC	NOx	NH_3	SO_2	BC	OC	$PM_{2.5}$	PM_{10}
This study	68	4	0.25	2.20	0.23	0.09	0.02	0.03	0.10	0.27	0.31
Streets et al. (2003b)	160	10	0.28	1.70	0.40	0.13	0.04	0.07	0.03	–	–

3. Results and discussion

The cropland fire emissions calculated in our study in 2006 and the comparison with other published estimates are listed in Table 3, which were somewhat lower than extensively used results for the base year 2000 in Streets et al. (2003b). The smaller agricultural burning emissions in our study were caused by the distinct EFs, as well as lower percentages of crop burning in fields. Streets et al. derived the emissions on the basis of the EFs summarized by Andreae and Merlet (2001). The updated and improved EFs, compiled by Akagi et al. (2011), were adopted in our calculation, which combined numerous recent measurements and corrected the undervalued EF for non-methane organic compounds. The differences included EF for CO (from 92 to 102 g Kg^{-1} in Andreae and Merlet, 2001 and Akagi et al., 2011 respectively), CH_4 (from 2.7 to 5.8 g Kg^{-1}), NH_3 (from 1.3 to 2.2 g Kg^{-1}), and $PM_{2.5}$ (from 3.9 to 6.3 g Kg^{-1}). In addition, Streets et al. (2003b) estimated crop burning in the fields with a unified ratio of 17% for all kinds of crops across the country, which are outdated data and were suitable for 1970s and 1980s in China (Hao and Liu, 1994). In our estimate, the provincial level and crop-specific percentages were based on surveys from 2000. On average, 6.6% of crop residues were burned in the field in this study. This percentage is reasonable and could reflect the status of crop residue burning in China in recent years. Crop residues have various uses in China's rural area, including use as fertilizer, fodder, fuel, and raw material, etc. In the 2000s, over 40% of wheat and rice straws were composted to increase soil fertility since crop production has increased rapidly in recent decades. Simultaneously, a larger portion of residues from coarse cereal and peanut are used as fodder because of the sustainable growth of the animal population. Additionally, agricultural fire activities have decreased gradually because the government has enacted a series of regulations that prohibit field burning since the 1990s. Farmers are encouraged to return crop residue to agricultural soils as fertilizer or to manufacture it into raw material. That's why 40 Tg of crop residues were burned outside in 2006 in our study but 110 Tg of crop residues were reported in Streets et al. (2003b). Consequently, the disparity of emissions between the results from Streets et al. and the estimates made in this study were to be expected.

3.1. Provincial and crop specific emissions

Provincial emissions of pollutants from crop residue burning in fields in China for 2006 were compiled in Table 4. CO was used as an illustrative example in this paper, as it was widely studied in the

Table 4
Provincial level agricultural open fire emissions (Gg) in China in 2006.

Province	CO_2	CO	CH_4	NMOC	NOx	NH_3	SO_2	BC	OC	$PM_{2.5}$	PM_{10}
Northeast											
Heilongjiang	1653	107	6.1	53.6	5.5	2.3	0.4	0.8	2.4	6.5	7.5
Inner Mongolia	2136	138	7.8	69.3	7.1	2.9	0.5	1.0	3.1	8.4	9.7
Jilin	3710	239	13.6	120.3	12.4	5.1	0.8	1.8	5.4	14.7	16.9
Liaoning	474	31	1.7	15.4	1.6	0.7	0.1	0.2	0.7	1.9	2.2
Northwest											
Gansu	203	13	0.7	6.6	0.7	0.3	0.0	0.1	0.3	0.8	0.9
Ningxia	420	27	1.5	13.6	1.4	0.6	0.1	0.2	0.6	1.7	1.9
Qinghai	132	9	0.5	4.3	0.4	0.2	0.0	0.1	0.2	0.5	0.6
Shaanxi	1343	87	4.9	43.6	4.5	1.8	0.3	0.6	2.0	5.3	6.1
Xinjiang	1447	93	5.3	46.9	4.8	2.0	0.3	0.7	2.1	5.7	6.6
Tibet	62	4	0.2	2.0	0.2	0.1	0.0	0.0	0.1	0.3	0.3
North											
Beijing	154	10	0.6	5.0	0.5	0.2	0.0	0.1	0.2	0.6	0.7
Hebei	4392	283	16.1	142.4	14.7	6.0	1.0	2.1	6.4	17.4	20.0
Henan	5224	337	19.2	169.4	17.4	7.2	1.2	2.5	7.6	20.6	23.7
Shandong	4700	303	17.3	152.4	15.7	6.4	1.0	2.2	6.8	18.6	21.4
Shanxi	730	47	2.7	23.7	2.4	1.0	0.2	0.4	1.1	2.9	3.3
Tianjin	203	13	0.8	6.6	0.7	0.3	0.0	0.1	0.3	0.8	0.9
Central											
Anhui	8038	518	29.5	260.7	26.8	11.0	1.8	3.8	11.7	31.8	36.5
Hubei	5367	346	19.7	174.1	17.9	7.4	1.2	2.5	7.8	21.2	24.4
Hunan	5943	383	21.8	192.7	19.8	8.2	1.3	2.8	8.6	23.5	27.0
Jiangxi	665	43	2.4	21.6	2.2	0.9	0.2	0.3	1.0	2.6	3.0
Southeast											
Fujian	991	64	3.6	32.1	3.3	1.4	0.2	0.5	1.4	3.9	4.5
Guangdong	1432	92	5.3	46.4	4.8	2.0	0.3	0.7	2.1	5.7	6.5
Hainan	597	38	2.2	19.4	2.0	0.8	0.1	0.3	0.9	2.4	2.7
Jiangsu	4103	265	15.1	133.1	13.7	5.6	0.9	1.9	6.0	16.2	18.6
Shanghai	153	10	0.6	5.0	0.5	0.2	0.0	0.1	0.2	0.6	0.7
Taiwan	273	18	1.0	8.9	0.9	0.4	0.1	0.1	0.4	1.1	1.2
Zhejiang	1702	110	6.3	55.2	5.7	2.3	0.4	0.8	2.5	6.7	7.7
Southwest											
Chongqing	883	57	3.2	28.6	2.9	1.2	0.2	0.4	1.3	3.5	4.0
Guangxi	1597	103	5.9	51.8	5.3	2.2	0.4	0.8	2.3	6.3	7.3
Guizhou	5951	384	21.9	193.0	19.9	8.2	1.3	2.8	8.6	23.5	27.0
Sichuan	1586	102	5.8	51.4	5.3	2.2	0.4	0.8	2.3	6.3	7.2
Yunnan	1618	104	5.9	52.5	5.4	2.2	0.4	0.8	2.4	6.4	7.4

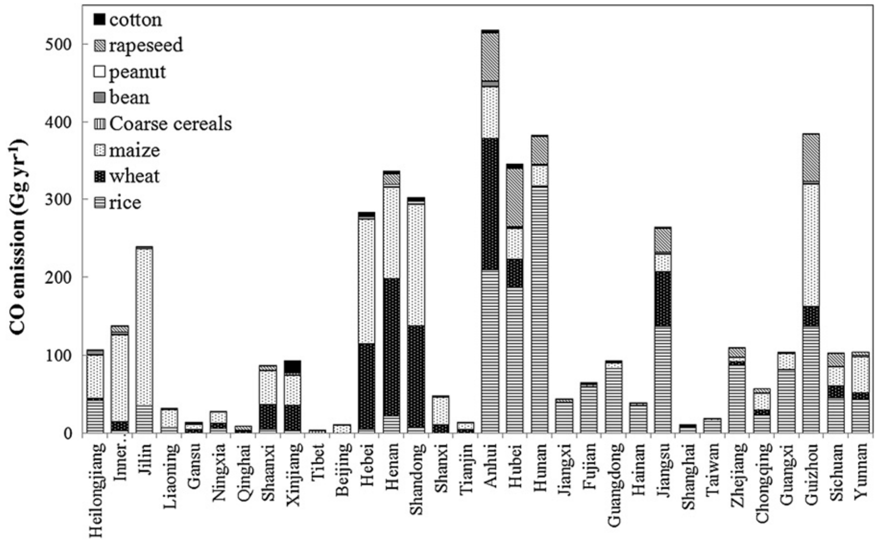

Fig. 1. Province level crop-specific CO emissions (Gg) in China in 2006.

open fire emission modeling (Streets et al., 2003b). CO emissions are primarily concentrated in Anhui (518 Gg yr^{-1}), Guizhou (384 Gg yr^{-1}) and Hunan (383 Gg yr^{-1}), which accounted for 12%, 9% and 9% of the total emissions, respectively. The smallest emitters of CO are Tibet and Qinghai, with emissions less than 10 Gg yr^{-1}. This result is slightly different with the results reported by Streets et al. (2003a,b) which gave more weight to the Sichuan and Jiangxi provinces. As mentioned above, we used a province-specific percentage of crop residues burning in fields in this study. According to a previous study (Gao et al., 2002), this percentage in Sichuan and Jiangxi is very small, and thus, Streets et al. (2003a,b) overemphasized these two provinces because the used an undifferentiated ratio for all the provinces.

Crop-specific emissions in each province were listed in Fig. 1. As shown, emissions due to burned rice, wheat and maize straws contributed about 90% of the emissions from all kinds of crop residues. From North to South, primary emission contributors generally transform from the burning of maize straws, to wheat straws and then rice straws. For instance, the greatest emitter in Jilin, located in northeast China, burns maize straws, accounting for 84% of the total emissions. While in Henan, wheat straws contribute 52% of the CO emissions. In contrast, rice straw contributes 94% of the CO emissions in Fujian. These results is consistent with China's cereal crop distribution: maize is extensively planted in Northeast China, a wheat-summer maize rotation system is used in the North China Plain and double or even triple cropping of rice is used in South China.

3.2. Spatiotemporal patterns of fire counts and fire emissions

A total of 273,418 fire counts (or fired pixels) were recorded in China from the year 2003 to 2010 on farmland. On the basis of eight-year data, the provincial fire counts are demonstrated in Fig. 2 and spatial and seasonal patterns of fire counts are shown in Fig. 3. Regionally, agricultural open fires are concentrated in three regions, the North China Plain (Henan, Shandong, Northern Anhui and Northern Jiangsu), Northeast China (Heilongjiang, Jilin and Liaoning), and South China (Guangdong, Guangxi and Yunnan province), accounting for approximately 75% of the whole country. Conversely, fire counts were sparsely scattered in the western part of China, such as Tibet, Qinghai and Gansu provinces.

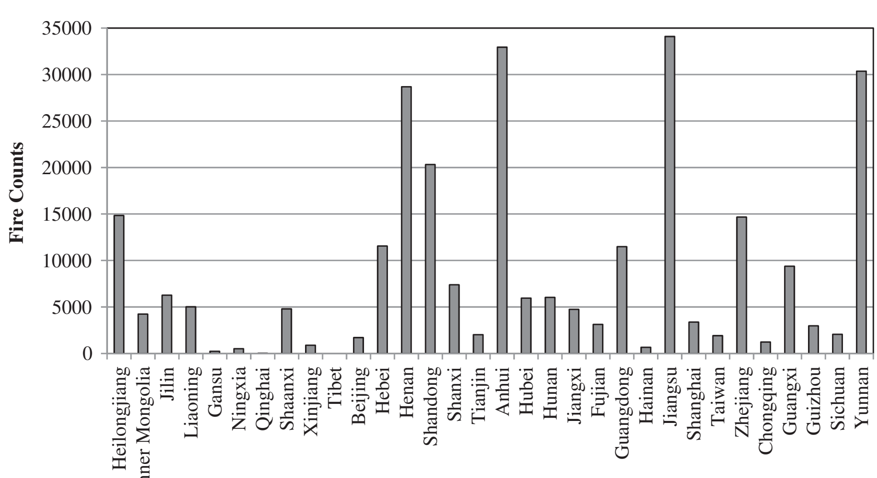

Fig. 2. Agricultural fire occurrences in each province of China.

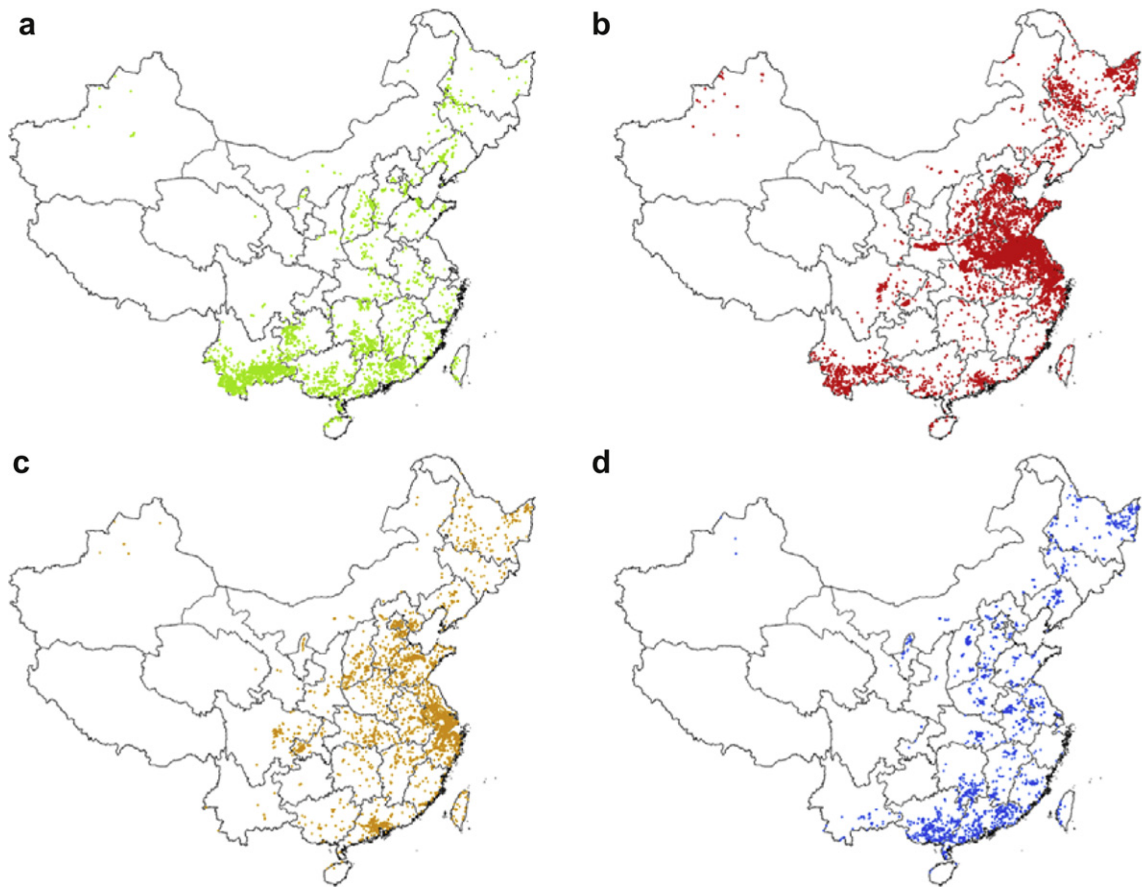

Fig. 3. Spatiotemporal distribution of agricultural fire occurrences in China during 2003–2010: (a) Spring (March–May); (b) Summer (June–August); (c) Autumn (September–November); (d) Winter (December–February).

The seasonal distribution of agricultural fires was presented in Fig. 3, since fire is affected by both natural and anthropogenic factors, which are variable spatially and temporally. The monthly variation of fire counts was extraordinarily concentrated in June (about 37% of total fires), followed by March to May (contributing averagely 9–10%), and was lowest during November to January (only occupying 2–3%). In the spring, fires are concentrated in the southern part of China, which is characterized by triple cropping cultivation where March to April is the time of the first-round harvest (CAAS, 1984). Summer fires mostly occur in central China. Simultaneously, a considerable number of fires also occur in South China in the summer, which may be attributed to the second-round harvest. In the autumn, crop residue burning is uniformly scattered over most of the agricultural zone in China since the autumn is the main harvest season for many kinds of crops. During the winter, fires are sparsely distributed but are a little more intense in South China because of the third-round harvest of late rice.

The spatial distribution of fire counts were used to allocate the emission. A 1 km grid CO emission was chosen as illustration in Fig. 4. Temporal distribution of emissions exactly agrees with the agricultural timing. Three important zones, listed in Fig. 4, are discussed in detail on temporal variation of fire emissions.

Most fire emissions originated in the North China Plain, which is the largest agricultural zone in China, with about 34% national rural population, 27% cultivated land and 35% crop yield (NBSC, 2007). Staple crops here include maize and wheat, because of the widespread use of the wheat-maize rotation system. Zone 1, which is shown in Fig. 4 including eastern Henan, southern Shandong, northern Anhui and northern Jiangsu, is the region with the most extensive fires. The temporal variation of the fire occurrences in this area is described in Fig. 5(a). The fire occurrences are predominantly in early and middle June, in which over 75% of the fires for the whole year occur, with another small peak in early October. This fluctuation is in general agreement with agricultural timing. Winter wheat (sown in mid-October and reaped in the end of May) and summer maize (sown in mid-June and reaped in the end of September) are the two most important crops (MOA, 2000–2011), both with a tremendous production of 76 Tg yr^{-1} and 50 Tg yr^{-1}, respectively (NBSC, 2007). The peak in fire occurrences in June is caused by the harvest of winter wheat. Just after the harvest, large-scale burning of wheat straw occurs to increase the soil fertility for subsequent maize cultivation. Another small peak in early October could be attributed to the maturity of maize and burning of maize straw afterward.

Another notable area with extensive burning is Northeast China (Zone 2 in Fig. 4), which has just one harvest for the whole year, primarily for japonica rice, spring maize and bean. In the three northeast provinces of Heilongjiang, Jilin and Liaoning, an average of 44 Tg of maize is produced annually. In this area, japonica rice and spring maize are sown in late April, and the sowing time for beans is mid-May. Early October is regarded as the harvest season for almost all crops (MOA, 2000–2011). The two peak emissions periods, in April and late October, are presented in Fig. 5(b). Fires in October result from the redundant crop straws after reaping, while the higher fire frequency in April could be attributed to the local practice of clearing the farmland to prepare for sowing. Crop residues are not completely burned immediately after the harvest, since a majority of farmland is vacant during the winter, except some under cultivation of winter rapeseed.

South China, which has two or even three harvest times each year, also contributes considerably to the national emissions as

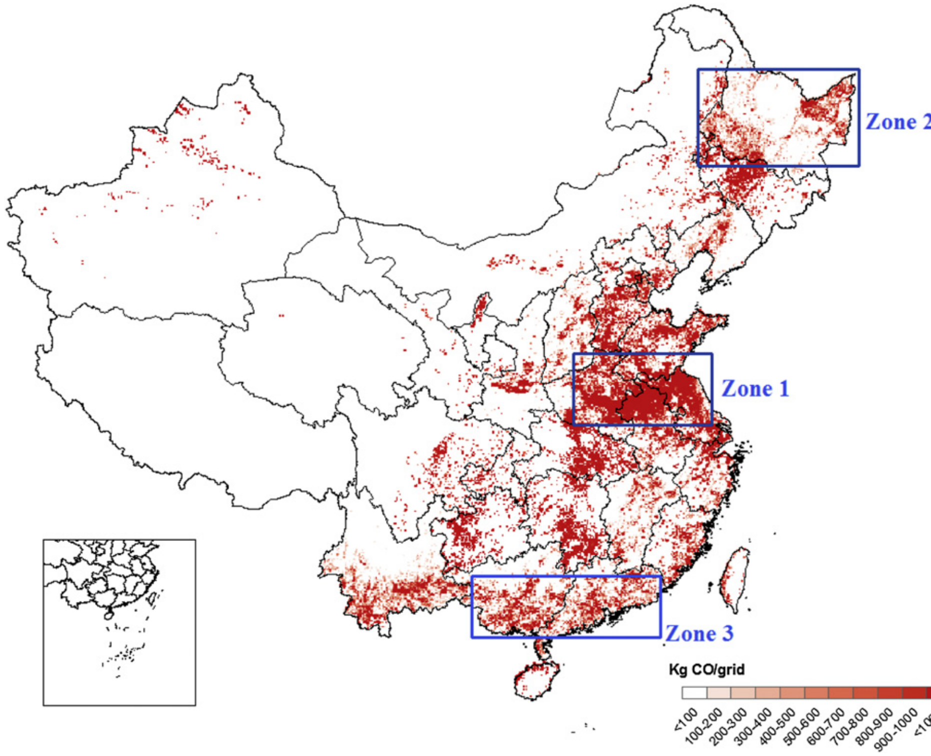

Fig. 4. Spatial distribution of CO emission in 2006 in 1 km grid cell (Kg grid^{-1}).

a result of crop residue burning in fields. Zone 3, which is shown in Fig. 4, comprises the South part of Guangdong and Guangxi provinces. This zone is a good example for presenting the temporal pattern of agricultural open fires, where multiple cropping rice, winter wheat and winter rapeseed possess larger yield. As shown in Fig. 5(c), agricultural fires range over almost every month during the year with several peaks. This region is located in the tropics with abundant water and sunshine, where various crops are planted in multiple crop rotations. February to March, late August and early December are slightly concentrated with agricultural fires. The distribution of these three peak periods is consistent with local sowing and harvest times. Crops in South China are usually sowed in February to March, which is earlier than the northern part of China, when plenty of crop residues are burned to fertilize the soil. In early August and November, early season rice and late rice mature in succession (MOA, 2000–2011), which may lead to more frequent burning during these periods.

3.3. Uncertainty

Emission uncertainty is associated with the amount of burned crop residues, the percentage of crop residues burned in fields, combustion efficiency and emission factors. Given the larger presumed uncertainties of statistics for such informal energy use, the

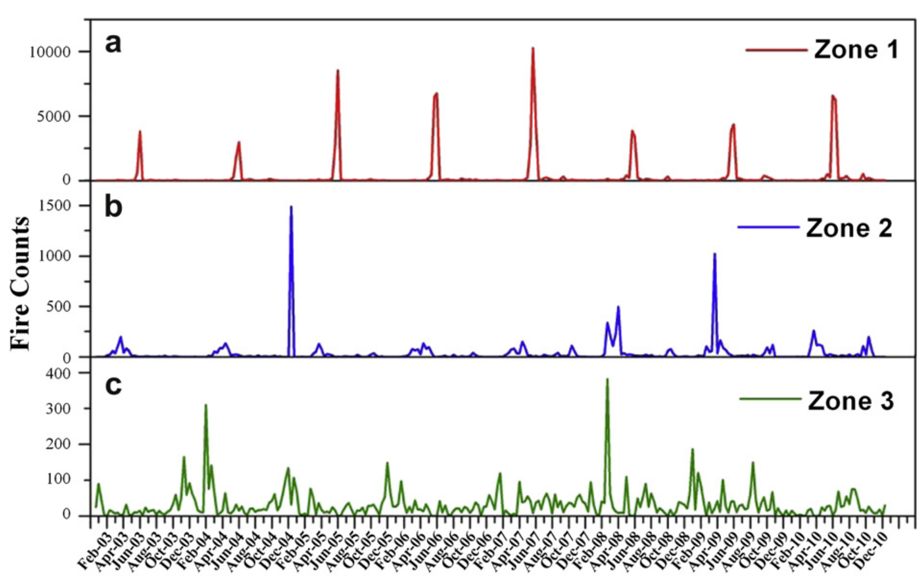

Fig. 5. Agricultural fire counts in 10-day interval during 2003–2010 for three typical zones (zone 1: North China Plain; zone 2: Northeast China; zone 3: South China).

probability of burned crop residue was assumed to have a normal distribution, with a coefficient of variation (CV) of 30% (Zhao et al., 2011). We assumed that the combustion efficiency was within an uncertainty range of about ±30% around the mean value. The uncertainty of the EF is species dependent (Akagi et al., 2011); more details can be found in Table 2. We ran 20,000 Monte Carlo simulations to estimate the range of fire emissions with a 90% confidence interval. The estimated average emission ranges were 51–85 Tg CO_2 yr^{-1}, 2-7 Tg CO yr^{-1}, 0.08–0.46 Tg CH_4 yr^{-1}, 1.08–3.46 Tg NMOCs yr^{-1}, 0.08–0.41 Tg NOx yr^{-1}, 0.03–0.17 Tg NH_3 yr^{-1}, 0.01–0.03 Tg SO_2 yr^{-1}, 0.01–0.05 Tg BC yr^{-1}, 0.04–0.17 Tg OC yr^{-1}, 0.13–0.42 Tg $PM_{2.5}$ yr^{-1}, 0.12–0.53 Tg PM_{10} yr^{-1}.

4. Conclusion

We developed a comprehensive emission inventory for crop residue field burning in China by using daily 1 km MODIS Thermal Anomalies/Fire products and the land cover dataset, GLC-China. Spatially, the most important contributors are the North China Plain, Northeast China and South China. Temporally, the emissions are significantly associated with local sowing and harvest timing. This work provides a detailed agricultural open fires emission inventory with a spatial resolution of 1 km and temporal resolution of 10 days. This inventory could be used for a global and regional air quality simulation.

Acknowledgments

The MODIS Thermal Anomalies/Fire products were provided by Land Process Distributed Active Archive Center (LPDAAC), USA. This study was funded by National Natural Science Foundation of China (40975088) and the Public Welfare Projects for Environmental Protection (200809018).

Appendix. Supplementary material

Supplementary data related to this article can be found online at doi:10.1016/j.atmosenv.2012.01.017.

References

Akagi, S.K., Yokelson, R.J., Wiedinmyer, C., Alvarado, M.J., Reid, J.S., Karl, T., Crounse, J.D., Wennberg, P.O., 2011. Emission factors for open and domestic biomass burning for use in atmospheric models. Atmos. Chem. Phys. 11, 4039–4072.
Andreae, M.O., Merlet, P., 2001. Emission of trace gases and aerosols from biomass burning. Global Biogeochem. Cy. 15, 955–966. doi:10.1029/2000GB001382.
Chinese Academy of Agricultural Sciences (CAAS), 1984. Agricultural Regionalization of China (In Chinese). China Agriculture Press, Beijing, 275 pp.
Cao, G.L., Zhang, X.Y., Wang, Y.Q., Zheng, F.C., 2008. Estimation of emissions from field burning of crop straw in China (in Chinese). Chinese Sci. Bull. 53, 784–790.
Crutzen, P., Andreae, M., 1990. Biomass burning in the tropics: impact on atmospheric chemistry and biogeochemical cycles. Science 250, 1669–1678.
de Zarate, I.O., Ezcurra, A., Lacaux, J.P., Van Dinh, P., de Argandona, J.D., 2005. Pollution by cereal waste burning in Spain. Atmos. Res. 73, 161–170.
Dennis, A., Fraser, M., Anderson, S., Allen, D., 2002. Air pollutant emissions associated with forest, grassland, and agricultural burning in Texas. Atmos. Environ. 36, 3779–3792.
Gao, X., Ma, W., Ma, C., Zhang, F., YunhuaWang, 2002. Analysis on the current status of utilization of crop straw in China (in Chinese). Journal of Huazhong Agricultural University 21, 242–247.
Giglio, L., 2010. MODIS Collection 5 Active Fire Product User's Guide Version 2.4. Science Systems and Applications, Inc.
Giglio, L., Descloitres, J., Justice, C.O., Kaufman, Y.J., 2003. An enhanced contextual fire detection algorithm for MODIS. Remote Sens. Environ. 87, 273–282.
Hao, W.M., Liu, M.H., 1994. Spatial and temporal distribution of Tropical biomass burning. Global Biogeochem. Cy. 8, 495–503. doi:10.1029/94GB02086.
Jenkins, B.M., Turn, S.Q., Williams, R.B., 1992. Atmospheric emissions from agricultural burning in California – determination of burn fractions, distribution factors, and crop-specific contributions. Agr. Ecosyst. Environ. 38, 313–330.
Korontzi, S., McCarty, J., Loboda, T., Kumar, S., Justice, C., 2006. Global distribution of agricultural fires in croplands from 3 years of Moderate Resolution Imaging Spectroradiometer (MODIS) data. Global Biogeochem. Cy. 20. doi:10.1029/2005GB002529.
Lal, R., 2005. World crop residues production and implications of its use as a biofuel. Environ. Int. 31, 575–584.
McCarty, J.L., 2011. Remote sensing-based estimates of annual and seasonal emissions from crop residue burning in the contiguous United States. J. Air Waste Manag. Assoc. 61, 22–34.
McCarty, J.L., Loboda, T., Trigg, S., 2008. A hybrid remote sensing approach to quantifying crop residue burning in the United States. Appl. Eng. Agric. 24, 515–527.
Ministry of Agriculture of the People's Republic of China (MOA), 2000. Database of Farming Season (In Chinese). Ministry of Agriculture of the People's Republic of China. http://www.zzys.gov.cn/.
National Bureau of Statistics of China (NBSC), 2007. Chapt. 13. Agriculture in:China Statistical Yearbook 2007 (in Chinese). China Statistics Press, Beijing. Available at: http://www.stats.gov.cn/tjsj/ndsj/2007/indexeh.htm.
Roy, D.P., Boschetti, L., Justice, C.O., Ju, J., 2008. The collection 5 MODIS burned area product – global evaluation by comparison with the MODIS active fire product. Remote Sens. Environ. 112, 3690–3707.
Song, Y., Chang, D., Liu, B., Miao, W.J., Zhu, L., Zhang, Y.H., 2010. A new emission inventory for nonagricultural open fires in Asia from 2000 to 2009. Environ. Res. Lett. 5. doi:10.1088/1748-9326/5/1/014014 Artn 014014.
Song, Y., Liu, B., Miao, W.J., Chang, D., Zhang, Y.H., 2009. Spatiotemporal variation in nonagricultural open fire emissions in China from 2000 to 2007. Global Biogeochem. Cy. 23. doi:10.1029/2008GB003344.
Streets, D.G., Bond, T.C., Carmichael, G.R., Fernandes, S.D., Fu, Q., He, D., Klimont, Z., Nelson, S.M., Tsai, N.Y., Wang, M.Q., Woo, J.H., Yarber, K.F., 2003a. An inventory of gaseous and primary aerosol emissions in Asia in the year 2000. J. Geophys. Res. Atmos. 108. doi:10.1029/2002jd003093.
Streets, D.G., Yarber, K.F., Woo, J.H., Carmichael, G.R., 2003b. Biomass burning in Asia: annual and seasonal estimates and atmospheric emissions. Global Biogeochem. Cy. 17, 1099,. doi:10.1029/2003gb002040.
Turn, S.Q., Jenkins, B.M., Chow, J.C., Pritchett, L.C., Campbell, D., Cahill, T., Whalen, S.A., 1997. Elemental characterization of particulate matter emitted from biomass burning: wind tunnel derived source profiles for herbaceous and wood fuels. J. Geophys. Res. Atmos. 102, 3683–3699. doi:10.1029/96JD02979.
van der Werf, G.R., Randerson, J.T., Giglio, L., Collatz, G.J., Kasibhatla, P.S., Arellano, A.F., 2006. Interannual variability in global biomass burning emissions from 1997 to 2004. Atmos. Chem. Phys. 6, 3423–3441.
Xu, W., Wu, B., Yan, C., Huang, H., 2005. China land cover 2000 using SPOT VGT S10 data (in Chinese). J. Remote Sens. 9, 204–213.
Yan, X.Y., Ohara, T., Akimoto, H., 2006. Bottom-up estimate of biomass burning in mainland China. Atmos. Environ. 40, 5262–5273.
Yevich, R., Logan, J.A., 2003. An assessment of biofuel use and burning of agricultural waste in the developing world. Global Biogeochem. Cy. 17, 1095,. doi:10.1029/2002gb001952.
Zhang, H.F., Ye, X.N., Cheng, T.T., Chen, J.M., Yang, X., Wang, L., Zhang, R.Y., 2008. A laboratory study of agricultural crop residue combustion in China: emission factors and emission inventory. Atmos. Environ. 42, 8432–8441.
Zhao, Y., Nielsen, C.P., Lei, Y., McElroy, M.B., Hao, J., 2011. Quantifying the uncertainties of a bottom-up emission inventory of anthropogenic atmospheric pollutants in China. Atmos. Chem. Phys. 11, 2295–2308.