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RESEARCH ARTICLE

A national assessment of the effect of intensive agro-land use practices on nonpoint source pollution using emission scenarios and geo-spatial data

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Abstract China's intensive agriculture has led to a broad range of adverse impacts upon ecosystems and thereby caused environmental quality degradation. One of the fundamental problems that face land managers when dealing with agricultural nonpoint source (NPS) pollution is to quantitatively assess the NPS pollution loads from different sources at a national scale. In this study, export scenarios and geo-spatial data were used to calculate the agricultural NPS pollution loads of nutrient, pesticide, plastic film residue, and crop straw burning in China. The results provided the comprehensive and baseline knowledge of agricultural NPS pollution from China's arable farming system in 2014. First, the nitrogen (N) and phosphorus (P) emission loads to water environment were estimated to be 1.44 Tg N and 0.06 Tg P, respectively. East and south China showed the highest load intensities of nutrient release to aquatic system. Second, the amount of pesticide loss to water of seven pesticides that are widely used in China was estimated to be 30.04 tons (active ingredient (ai)). Acetochlor was the major source of pesticide loss to water, contributing 77.65% to the total loss. The environmental impacts of pesticide usage in east and south China were higher than other parts. Third, 19.75% of the plastic film application resided in arable soils. It contributed a lot to soil phthalate ester (PAE) contamination. Fourth, 14.11% of straw produce were burnt in situ, most occurring in May to July (post-winter wheat harvest) in North China Plain and October to November (post-rice harvest days) in southeast China. All the above agricultural NPS pollution loadings were unevenly distributed across China. The spatial correlations between pollution loads at land unit scale were also estimated. Rising labor cost in rural China might be a possible explanation for the general positive correlations of the NPS pollution loads. It also indicated a co-occurred higher NPS pollution loads and a higher human exposure risk in eastern regions. Results from this research might provide full-scale information on the status and spatial variation of various agricultural NPS pollution loads for policy makers to control the NPS pollution in China.

Keywords Nutrient erosion loss · Scenario · Land use · Nonpoint source pollution · Pesticide · Agricultural plastic film · Straw burning

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Introduction

China is the world's most populous country with scarce land per capita and only 7% of the land area of the world. Thus, China's food security remains a world concern for generations and still a challenging task for governments. To keep up with the food demand for the growing population, the immediate response is a more excessive use of agrochemicals such as chemical fertilizers, which can result in a broad range of adverse impacts upon ecosystems and thereby cause environmental problems. The agricultural nonpoint source (NPS) pollution issues are becoming an increasing concern in China (Ouyang et al. 2008; Wang et al. 2006).

China faces greater NPS pollution challenges than other major countries (Liu and Diamond 2005). Water pollution, mainly deriving from intensive farming practices (FAO 2013; Basnyat et al. 1999), is one of the major threats to agricultural environmental sustainability (Leone et al. 2008). As the largest producer and consumer of fertilizers and pesticides in the world, China's average farm usage has widely exceeded the international standards for recommended application rates (ADB 2004). Most of the fertilizers and pesticides used are lost to water environment through runoff and leaching loss (Withers et al. 2000; Carvalho 2006). The intensive agro-land practices also can cause soil degradation (Lal 2002). As the FAO (2013) reported, over 180 million ha of cropland in China are polluted by pesticides. And mulching agricultural plastic film may also lead to soil contamination in China (Chen et al. 2015). In each year, more than 1 million tons of plastic films are used for plant covering in China (Chen et al. 2013). A large amount of plasticizers (e.g., phthalate esters (PAEs)) from the residual film therefore may be released into soils. In addition, China also faces severe air quality issues caused by crop residue burning. The open burning is a significant seasonal source of air pollution (Qu et al. 2012; Gadde et al. 2009), and it is also a source of greenhouse gases (Yoshinori and Kanno 1997). It is of great importance to address these "on- or off-site" environmental impacts on water, soil, and air due to the above agro-land management practices. Unfortunately, the total picture of agricultural pollutions is not yet clear at the state level (Chen et al. 2008). In this research, we mainly focused on agricultural NPS pollution that is caused by agro-land use practices including fertilization, pesticide use, plastic film mulching, and in situ straw residue burning, to give a full-scale baseline information at a national scale.

Understanding the status and spatial variation of NPS pollution potential is an essential foundation to enable preventative measures to control and reduce the NPS pollution (Yang et al. 2013). Information of the NPS spatial distribution highly depends on simulation modeling (Xu et al. 2013) because the NPS is dispersed across landscapes. These simulation models include the scenario method (Centofanti et al. 2008), the export coefficient model (Ding et al. 2010; Liu et al. 2009; Wang et al. 2014c; Chen et al. 2008; Chen et al. 2010; Shindo 2012), and complex hydrological model (Parajuli et al. 2009; Ouyang et al. 2008). Model selection relies mainly on the purposes of the model simulation, the scale of the research area, the data availability, the expected accuracy, and the affordable costs (Xu et al. 2013; Kovacs 2006). The hydrological mechanism models such as Soil and Water Assessment Tool (SWAT) (Parajuli et al. 2009; Ouyang et al. 2008; Tripathi et al. 2005; Akhavan et al. 2010; Huang et al. 2009b), Agricultural Nonpoint Source Pollution (AGNPS) (Suttles et al. 2003), Riverstrahler (Sferratore et al. 2005), and GWLF (Du et al. 2014) are the most rigorous and pinpoint methods for nutrient NPS evaluation. However, these basin or watershed scale models require mass data, complex simulation algorithms, and a large number of parameters which are difficult to acquire (Yang et al. 2013). It would be too expensive and time consuming if they are applied at larger scales, which hinders their universal application. In addition, only a few studies on the pesticide NPS pollution used hydrological models such as the SWAT (Boithias et al. 2011) and the PHYTOPIXAL method (Macary et al. 2014). Few mechanism models have been used to evaluate film and straw burning NPS.

In contrast, as the export coefficient model requires less data and parameters (or coefficients), it has been widely used to assess nutrient NPS pollution risk of agriculture at larger scales, such as the national level (Wang et al. 2014c; Chen et al. 2008; Chen et al. 2010; Shindo 2012) and the watershed scales (Yu et al. 2007; Wang et al. 2014b; Liu et al. 2008). The export coefficients used for nutrient calculation are mainly from various publications and associated derived coefficients (Sheldrick et al. 2002). A paper by Sheldrick et al. (2003) used an export coefficient-based nutrient audit model to illustrate the nutrient flows and balances of China's agricultural systems at the provincial and national level from 1961 to 1997. Chen et al. (2008, 2010) proposed a partial substance flow analysis (SFA) to describe the nutrient flow driven by China's agriculture and rural life in 2004. Wang et al. (2014c) mapped the uneven distribution of nutrient surplus across China in 2010 using remote sensing (RS) data and nutrient balance model. The main purpose of the export coefficient method is to build nutrient databases to calculate nutrient flows and balances (Sheldrick et al. 2003). The calculation of nutrient flow in this methodology mainly focuses on the different land use type contributions to NPS pollution (Yang et al. 2013); however, it does not take into account other hydrological factors and underlying surface conditions that affect the NPS pollution, such as soil, slope, crop, and land management. Until now, the NPS loads of pesticides, agricultural film, and straw burning using coefficient method on a large scale have been scarcely conducted.

Comparatively, the scenario method uses the key factors that affect the NPS pollution, such as soil, climate, slope, cropping, and land cover to characterize the diversity of agricultural and environmental conditions where NPS pollution occurs (Centofanti et al. 2008). Centofanti et al. (2008) established a large number of agro-environmental scenarios to characterize Pan-European agriculture to estimate the pesticide risk. The scenario method is an open and powerful tool, as it allows scenario developers to take into account the parameters which most influence the environmental fate of the NPS pollution. In essence, the scenario method is an adaptive methodology of the export coefficient model, for that the NPS loads are calculated based on export coefficients.

In this study, we combined land use gridded data and emission scenarios to calculate the NPS pollution loads. This effort had considered the spatial distribution of climate, soil, slope, cropping, and land use practices. It also filled the gap between the gridded data and emission scenarios derived from a large number of fixed point investigations across China. Using geographic information system (GIS) software, the NPS pollution loads were estimated. GIS provides an obvious and intuitively appealing means of storing and displaying spatially based information, such as land records and natural resource features. These attributes, such as spatial analysis, make it also a powerful tool for modeling purposes.

This paper aims to quantify the agricultural NPS pollution from various sources in China by combing the emission scenarios and the geo-spatial data with a focus on nutrient loss to water, pesticide risk, agricultural film residual, and straw in situ burning. The specific objectives are (1) to develop a large number of NPS emission scenarios of various sources based on fixed position experiments or surveys to characterize the diversity of agricultural and environmental conditions of China's arable systems, (2) to analyze the status and spatial variation of NPS pollution loads of nutrient, pesticide, plastic film residue, and crop straw in situ burning, and (3) to analyze the spatial correlation between different NPS pollution loads using data of land units.

Materials and methods

The methods applied in this paper are divided into three components: (1) developing of the NPS pollution emission scenarios and (2) matching the scenarios with geo-spatial data (3) to calculate the NPS pollution loads from agro-land use practices. The procedures of scenario development and attribute matching are summarized in Fig. 1.

Development of emission scenarios

All emission scenarios were spatially developed at provincial level in China, and 31 provinces (or cities) were considered excluding Hong Kong, Macao, and Taiwan for data unavailability. These scenarios identified the full range of agricultural and environmental conditions by unique combinations of agro-land practices, crops, soils, terrains, and agro-climates in each province. All the scenarios were summarized from the field experiments or surveys conducted by the first national pollution census institute (the China Pollution Source Census, http://cpsc.mep.gov.cn/). They are characterized with attribute information (or sample site information) that were recorded when sampling, such as crops planted, soil types, land slope, cropping pattern, and location. More detailed information on the first national pollution census project, the procedures of experiment data sampling, collecting and processing are presented in the handbook of the pollution source census (the China Pollution Source Census, http://cpsc.mep.gov.cn/).

Data validation is very important for the accuracy of NPS simulation. To improve NPS estimation accuracy, rules were applied to identify defective data. The first rule applied is to compare the field experiment data. The mean ± 3•SD (standard deviation (SD)) was ascribed as a reasonable range of the data that were used to generate a certain emission scenario (Wang et al. 2014c). The second rule is to compare the scenario parameters with public literature or open-access data if available. If the difference between the scenario parameters and the open data is more than 15% (Wang et al. 2014c), the scenario parameters and the corresponding experiment data need to be verified. In addition, to secure the validity of our calculation results, it also needs to compare the calculation results with the public literature (see "Comparison of results" section).

N/P runoff or leaching emission scenarios

The scenarios of nutrient runoff or leaching were constructed using the fixed position experiment data (232 and 140 positions throughout the whole country for runoff and leaching, respectively) conducted over 1 year (from 2007 to 2008), based on the examination of six plots (each over 20 m^2 in area) in each position. In total, 70 runoff scenarios and 28 leaching scenarios were summarized across China (see Appendix Table 2). As agro-climate conditions, land use management, slope, and soil type were the key factors that affect nutrient NPS, the details on administrative province, land use type, dominant crops, slope, soil type, N/P inputs, N/P runoff, or leaching were characterized in each N/P runoff or leaching emission scenario.

Pesticide runoff or leaching loss scenarios

The scenarios of pesticide runoff or leaching were developed based on 1-year (from 2007 to 2008) fixed position investigations (total 372 positions across China). We selected seven most commonly applied pesticides across China (Table 1). As agro-climate conditions, land use management, slope, and soil type were also the important factors that affect pesticide NPS; this research summarized 141 representative scenarios of pesticide runoff or leaching loss to aquatic environment based on these information. In each scenario, detailed pesticide applications were also listed (see Appendix Table 3).

Film residual scenarios

This research developed 36 representative scenarios of plastic film residual based on fixed position investigations (total 432 positions across China) conducted over 1 year

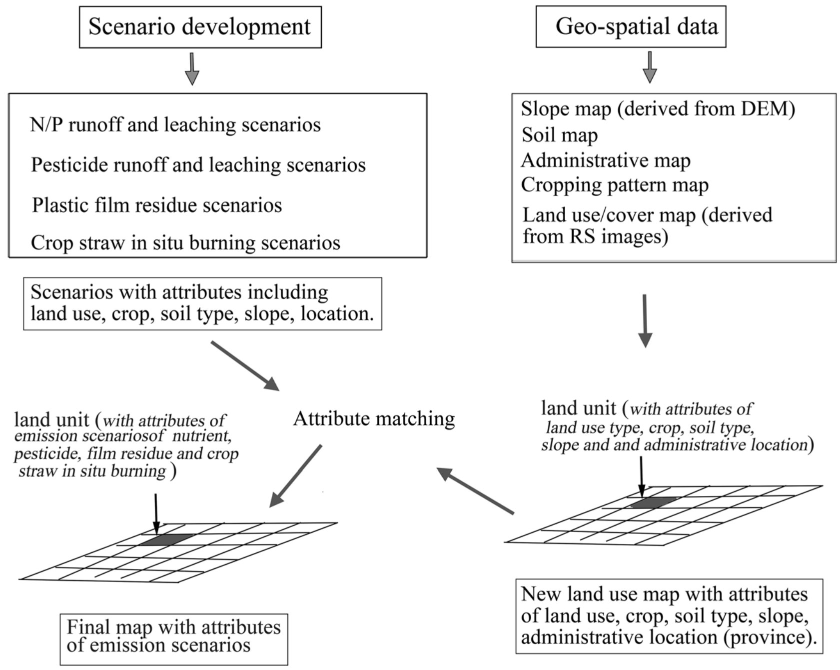

Fig. 1 Flowchart of scenario development and the attribute matching

(from 2007 to 2008), based on the examination of five pits (each 2 m^2 in area and 20 cm deep) in each mulching area with an area of about 667 m^2. As agro-climate conditions, land use management, film recycling pattern, soil type, and crops planted were the essential factors that affect film residual NPS, the details of film residual scenario included information on soil type, crops, film application, film recycling pattern, and residual (see Appendix Table 4). Two film disposal patterns (recycling and non-recycling) were considered in these scenarios.

Straw residue in situ burning scenarios

In total, 310 representative scenarios of straw residue in situ burning were constructed based on the field surveys across China conducted by the first national pollution census group from 2007 to 2008 (see Appendix Table 5). The straw produce and disposal of rice, wheat, maize, grain, soybeans, potato, oilseed rape, cotton, sugarcane, and sorghum were surveyed. The straw disposal in rural China included in site and in-door burning, returning to soils (incorporation), composting, livestock feeding, and raw materials for other uses. The in situ straw residue burning was a farmers' activity that was influenced by factors such as agro-climate conditions, crops grown, and other social economic factors.

Table 1 List of the pesticides widely used across China

Common name	CAS number	Application rate (kg/ha/year)	EIQ
Herbicide			
Acetochlor	2921-88-2	1.52	36.7
Atrazine	120068-37-3	1.79	33.2
Butachlor	13826-41-3	0.63	44.4
2,4-D	34256-82-1	0.80	22.6
Insecticide			
Chlorpyrifos	1912-24-9	1.83	52.8
Fipronil	23184-66-9	3.65	101.0
Imidacloprid	94-75-7	3.62	102.1

Geo-spatial data and GIS applications

The geo-spatial data required in this study contained land use maps, soil maps, digital elevation model (DEM), crop zoning maps, and administrative maps. The agricultural land use gridded data were generated by classifying Landsat ETM^+ satellite images with spatial resolution of 30 m acquired during 2014. ERDAS IMAGINE software (Leica Geosystems, Inc.) was used for the supervised image classification using maximum likelihood algorithm. Considering the vast data for land use, land use data were resampled at a resolution of 250 m and used in this study. The DEM, soil maps, and crop zoning maps were obtained from the Data Center for Resources and Environmental Sciences, Chinese Academy of Sciences. Land slope was derived from the DEM.

GIS offered major benefits to collecting, storing, retrieving, transforming, and displaying spatial data from the real world. One of the most useful applications of GIS in spatial data analysis was to overlay various thematic maps to derive useful results. In this study, we used the overlay analysis model of ArcGIS software (Esri, Inc.) to assign slope, soil, crop, and administrative map layer values to land use map layer, and thus we obtained a new land use map with attributes of land use, soil type, crops planted, and administrative province where the pixels were located. As the emission scenarios we created before also contained characterized information of land use, soil type, crops planted, and administrative province, a matching exercise between the new land use map and emission scenarios was easily worked out to generate a final map with certain emission scenarios in each resolution or pixel or land unit (see Fig. 1). This final map was then used to calculate the agricultural NPS pollution loads (see "NPS pollution loads" section).

Calculation of NPS pollution loads and pesticide risk

NPS pollution loads

To calculate the regional NPS loads, in this paper, the land unit was defined as a pixel in the final map (a resolution size, 250 m × 250 m), and it was supposed to be a minimal pollution discharge unit. For a particular emission source, NPS pollution loads of a land unit were given by multiplying the emission coefficient by the land unit area. The NPS pollution loads of a region were calculated as follows.

$$L = \sum SA_j \times SE_j \tag{1}$$

$$SA_j = LA \times N_j \tag{2}$$

Where L was the NPS loads from a particular source; SA_j was land area of the emission scenario j; SE_j was the NPS load of the emission scenario j (per ha); LA was land unit area (250 m × 250 m); N_j was the number of the land units with the emission scenario j in this region.

Pesticide risk assessment

Pesticide usage often spatially varied, and there were often more than one pesticides used in a land unit or a region (such as a province). The toxicities of different pesticides were usually different. Thus, the usage of pesticide loss was not enough to compare pesticide pollution at land unit or regional scale. In this research, the environmental impact quotient (EIQ) was applied as a pesticide risk indicator to evaluate the integrated potential environmental impact of pesticide usage (Kromann et al. 2011; Deihimfard et al. 2007). The EIQ could compare pesticide usage and assess its environmental impacts across space and different pesticides. It was a function that combines three general risk categories, pesticide hazard posed to farm workers (applicators and field workers), consumers (consumers and groundwater), and ecological component (aquatic organisms, bees, birds, and beneficial organisms):

$$EIQ = \left\{ \begin{array}{l} C[(DT \times 5) + (DT \times P)] \\ + \left[C \times \dfrac{S+P}{2} \times SY + L \right] \\ + \left[(F \times R) + \left(D \times \left(\dfrac{S+P}{2} \right) \times 3 \right) + (Z \times P \times 3) + (B \times P \times 5) \right] \end{array} \right\} / 3 \tag{3}$$

where C was chronic toxicity; DT was dermal toxicity; P was plant surface residue half-life; S was soil residue half-life; SY was systemicity; L was leaching potential; F was fish toxicity; R was surface loss potential; D was bird toxicity; Z was bee toxicity; B was beneficial arthropod toxicity. Values needed for calculating the EIQs of pesticides were obtained from the International Union of Pure, the Applied Chemistry (IUPAC) and the Extension Toxicology Network (EXTOXNET). Details on the weighting and scoring criteria for EIQ calculation could be referred to Deihimfard et al. (2007) and Gallivan et al. (2001). To assess and compare environmental risks at regional scale, the environmental impact (EI) of pesticides used per hectare (ha) in a given year was calculated as follows.

$$EI_{perHa} = \sum EIQ_i \times \%ai_i \times Dosage_i \tag{4}$$

where EI_{perHa} was the environmental impact of pesticides used per hectare (ha); EIQ_i was the EIQ of pesticide i; $\%ai_i$ was the percentage of ai (active ingredient) in pesticide i. $Dosage_i$ was pesticide i applied in kilograms per ha, which was calculated as follows.

$$Dosage_i = \frac{\sum SPA_{ik} \times SPE_{ik}}{LA \times M} \tag{5}$$

where SPA_{ik} was the land area of scenario k of pesticide i; SPE_{ik} was the application amount of pesticide i of the scenario k (per ha); M was the number of arable land units in a region.

Results

Current status of NPS pollution loads from intensive agro-land use practices in China

N/P erosion losses (runoff and leaching) from arable land

The average application rates (the amount of fertilizers used in per arable land area) of N and P fertilizers (including chemical fertilizers and manure) were 328.16 kg N and 69.47 kg P per ha in 2014. The intensive input of fertilizers contributed both the severe N and P erosion loss (Wang et al. 2014a). The arable emission loads of N and P into receiving water were estimated to be 1.44 Tg (1 Tg = 10^{12} g) and 0.06 Tg in China in 2014, respectively. The amount of N runoff and leaching loss were 0.38 and 1.06 Tg, respectively; that of P runoff and leaching loss were 0.05 and 0.01 Tg, respectively. For arable farming system, the N and P erosion export intensities were 13.13 and 0.89 kg/ha/year, respectively. In addition, the spatial distribution of N and P loss was unbalanced across the state (Fig. 2). Eastern provinces had much more intensive runoff and leaching loss of N and P per arable land area due to the more intensive cropping and fertilization practices. For example, Shandong province is one of the major vegetable-producing regions in China, with a vegetable planting area of 1.73 × 10^6 ha and fertilizer N application rate between 285 to 983 kg/ha in 2014. The leaching and runoff rate of vegetable fields there was between 43.82 and 100.57 kg/ha/year. Thus, measures and actions to reduce farmland fertilizer inputs have a great potential to reduce NPS pollution risk in these regions.

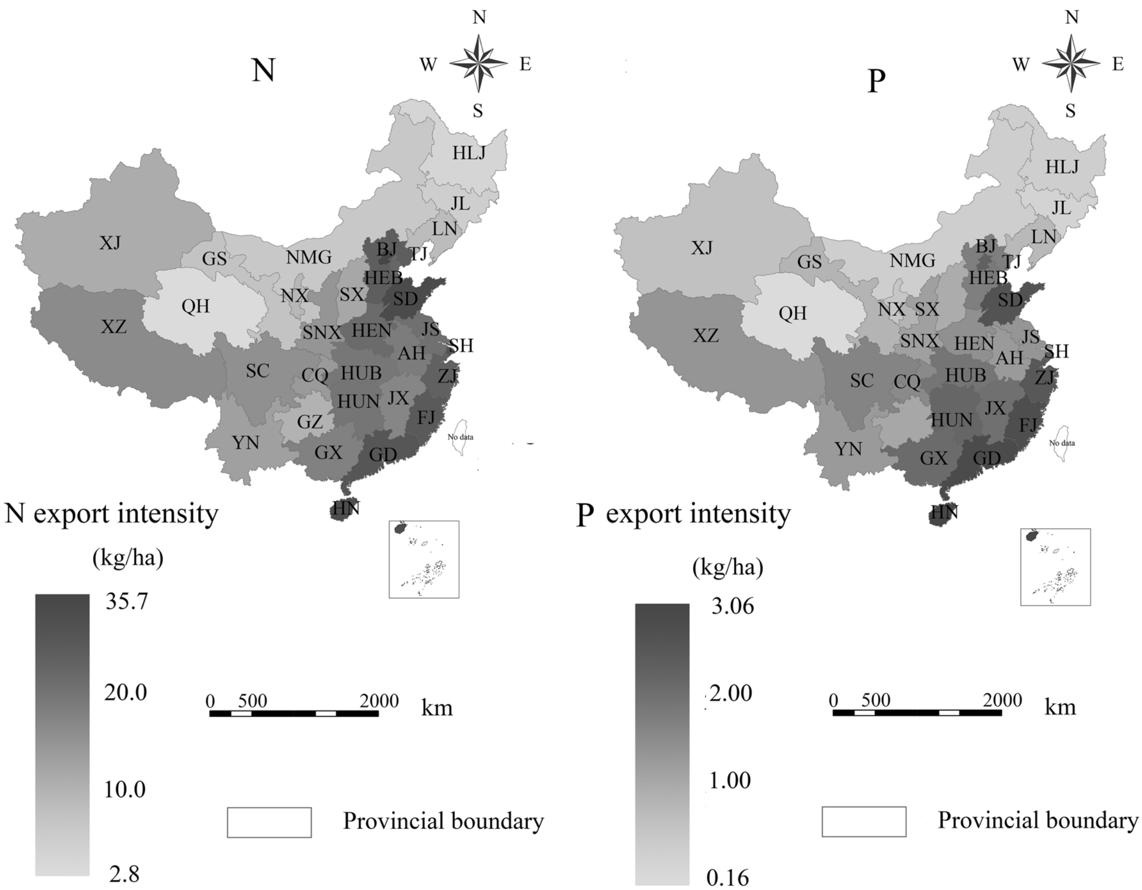

Fig. 2 N and P leaching and runoff export intensities of arable land in China in 2014 (BJ, Beijing; TJ, Tianjin; HEB, Hebei; SX, Shanxi; SNX, Shaanxi; NMG, Inner Mongolia; LN, Liaoning; JL, Jilin; HLJ, Heilongjiang; SH, Shanghai; JS, Jiangsu; ZJ, Zhejiang; FJ, Fujian; JX, Jiangxi; SD, Shandong; AH, Anhui; HEN, Henan; HUB, Hubei; HUN, Hunan; GD, Guangdong; GX, Guangxi; HN, Hainan; YN, Yunnan; GZ, Guizhou; SC, Sichuan; CQ, Chongqing; XZ, Tibet; GS, Gansu; QH, Qinghai; NX, Ningxia; XJ, Xinjiang)

Pesticide erosion loss and EIs

The amount of runoff and leaching loss of the seven pesticides (ai) in China was estimated to be 30.04 tons in 2014, and acetochlor, imidacloprid, and fipronil comprised 76.31, 9.40, and 9.95% of the total pesticide loss to water, respectively. Figure 3a shows that northeast China suffered higher pesticide loss per arable land (ai per ha) due to large application rates of acetochlor in Heilongjiang, Liaoning, Jilin, and Inner Mongolia. However, acetochlor had a relatively low EIQ (Table 1). The EIs per ha at province level combined EIQs and overall ai application rate of the seven pesticides used. The reason for the low EI per ha obtained in Tibet appeared related to lower ai application rates of atrazine, 2,4-D, and fipronil. The highest EIs per ha were observed in eastern and southern China (Fig. 3b) due to high ai application rate of chlorpyrifos, fipronil, and imidacloprid in these areas.

Regression analysis showed there was no significant relationship between EIs and pesticide loss (r^2 = 0.12, $p > 0.05$). For instance, provinces with high EIs (e.g., Shanghai, Hunan, Jiangsu) did not have higher pesticide loss values (in g/ha). In contrast, the highest pesticide loss per ha was obtained in Jilin, Liaoning, and Heilongjiang which did not have the largest EIs. Policies simultaneously banning some particular pesticide with high toxicity while encouraging production of pesticides with low toxicity and loss rate seemed to be providing a substantial potential of pesticide risk reduction.

Plastic film residual

The annual amount of plastic film used for cropping was 0.55 Tg (1 Tg = 10^{12} g), 17.97% of which resided in soil in 2014. The largest amount of plastic film application

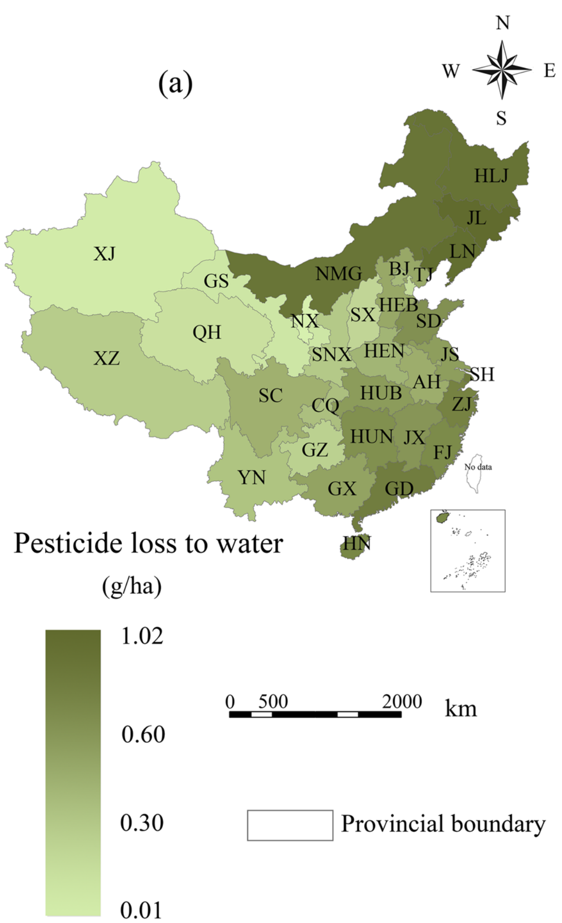

Fig. 3 **a**, **b** Pesticide loss to water through runoff and leaching, and EIs of pesticide use on provincial scale in China in 2014 (BJ, Beijing; TJ, Tianjin; HEB, Hebei; SX, Shanxi; SNX, Shaanxi; NMG, Inner Mongolia; LN, Liaoning; JL, Jilin; HLJ, Heilongjiang; SH, Shanghai; JS, Jiangsu; ZJ, Zhejiang; FJ, Fujian; JX, Jiangxi; SD, Shandong; AH, Anhui; HEN, Henan; HUB, Hubei; HUN, Hunan; GD, Guangdong; GX, Guangxi; HN, Hainan; YN, Yunnan; GZ, Guizhou; SC, Sichuan; CQ, Chongqing; XZ, Tibet; GS, Gansu; QH, Qinghai; NX, Ningxia; XJ, Xinjiang)

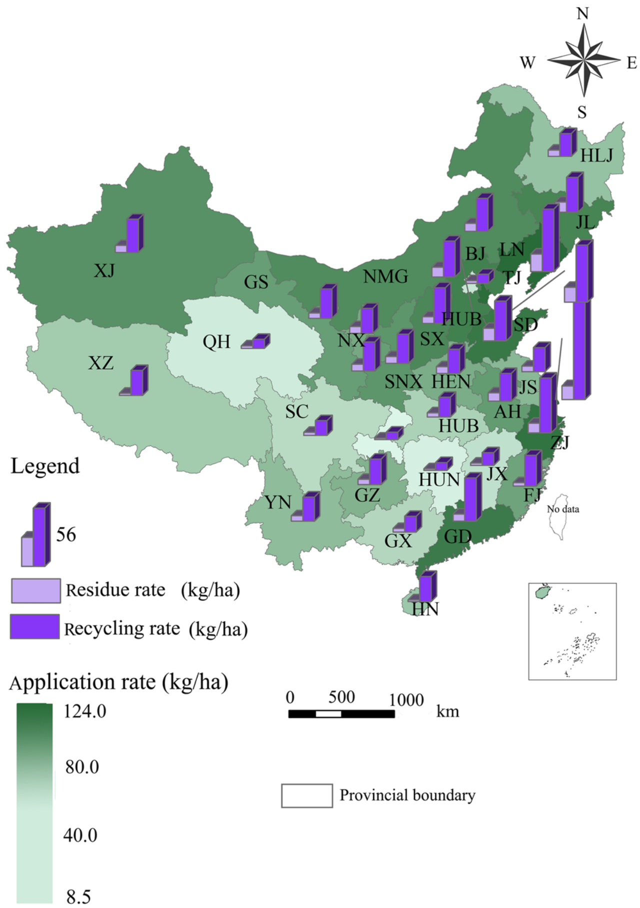

Fig. 4 Plastic film application, recycling, and residual across China in 2014 (BJ, Beijing; TJ, Tianjin; HEB, Hebei; SX, Shanxi; SNX, Shaanxi; NMG, Inner Mongolia; LN, Liaoning; JL, Jilin; HLJ, Heilongjiang; SH, Shanghai; JS, Jiangsu; ZJ, Zhejiang; FJ, Fujian; JX, Jiangxi; SD, Shandong; AH, Anhui; HEN, Henan; HUB, Hubei; HUN, Hunan; GD, Guangdong; GX, Guangxi; HN, Hainan; YN, Yunnan; GZ, Guizhou; SC, Sichuan; CQ, Chongqing; XZ, Tibet; GS, Gansu; QH, Qinghai; NX, Ningxia; XJ, Xinjiang)

occurred in Shandong province, accounting 19.22% of total plastic film used, and Xinjiang province, as the next largest contributor, accounting for 16.51% of which. Figure 4 indicates that northeast China suffered higher film residual pollution loads per arable land area, but southeast areas harbored less film residual. It was noted that largest amount of plastic film application (per ha) did not definitely result in largest plastic film residual (per ha) due to varying recycling practices among provinces. For example, Shanghai with highest film application of 124.00 kg/ha had a residual rate of 12.95 kg/ha less than that of Liaoning (16.79 kg/ha) and Tianjin (14.24 kg/ha), for that the recycling rate in Shanghai (91.01%) was relatively higher than that of Liaoning and Tianjin. Thus, residual plastic film was, to some extent, determined by film management (e.g., recycling). Encouraging film recycling could be an effective policy option for decision-makers to reduce NPS pollution caused by film residue in agricultural fields.

Straw in situ burning

China was abound with straw resource, and the total output of crop straw was 868.13 Tg in 2014. North and east China were the main producing areas (Fig. 5a). Shandong, Henan, Heilongjiang, and Hebei had in total produced 38.05% of the straw produce in 2014. In situ burning of crop straw residue in China had reached 113.48 Tg in 2014, accounting for 13.07% of the straw produced. Overall, open straw burning distributed extensively across China with strong spatial variations (Fig.5b), and the average of in situ burnt straw was 1012 kg/ha. Intensive in situ burning was obtained in east China areas. The amount of straw in situ burning in Anhui province was the largest, accounting 15.66% of the national straw produce and 40.31% of the straw produced in this province. During post-harvest seasons, regions with large amount of in situ burning might suffer more seasonal air quality degradation. The share of field burning at provincial level ranged from 0.01 in Hebei to 0.44 in Hainan (see Appendix Table 6). The percentages of in situ burning were higher in south areas, while those of combustion for fuel energy were higher in northeast China.

Spatial correlation of NPS pollution loads

Figure 6 shows the spatial relationship between the NPS load intensities on the land unit scale. General trends of positive correlation were observed, except for the negative relationship between film residue and straw in situ burning, and between film residue and P erosion loss (Fig. 6). It indicated that intensive NPS pollution loads from multiple sources might co-occur in some regions (e.g., eastern China), which contributed to severe NPS pollution risk in these areas. The correlations between pesticide loss, straw burning, N and P release to water were all statistically significant ($p < 0.1$). In contrast, relationship between film plastic residue and other pollution loads was not well correlated. One possible reason behind the fact was discussed in "Behind the spatial correlation between NPS loads" section.

Discussion

Comparison of results

Intensive arable farming practices have contributed the most important factors in agricultural NPS pollution (Ouyang et al. 2013; Beaudoin et al. 2005). In China,

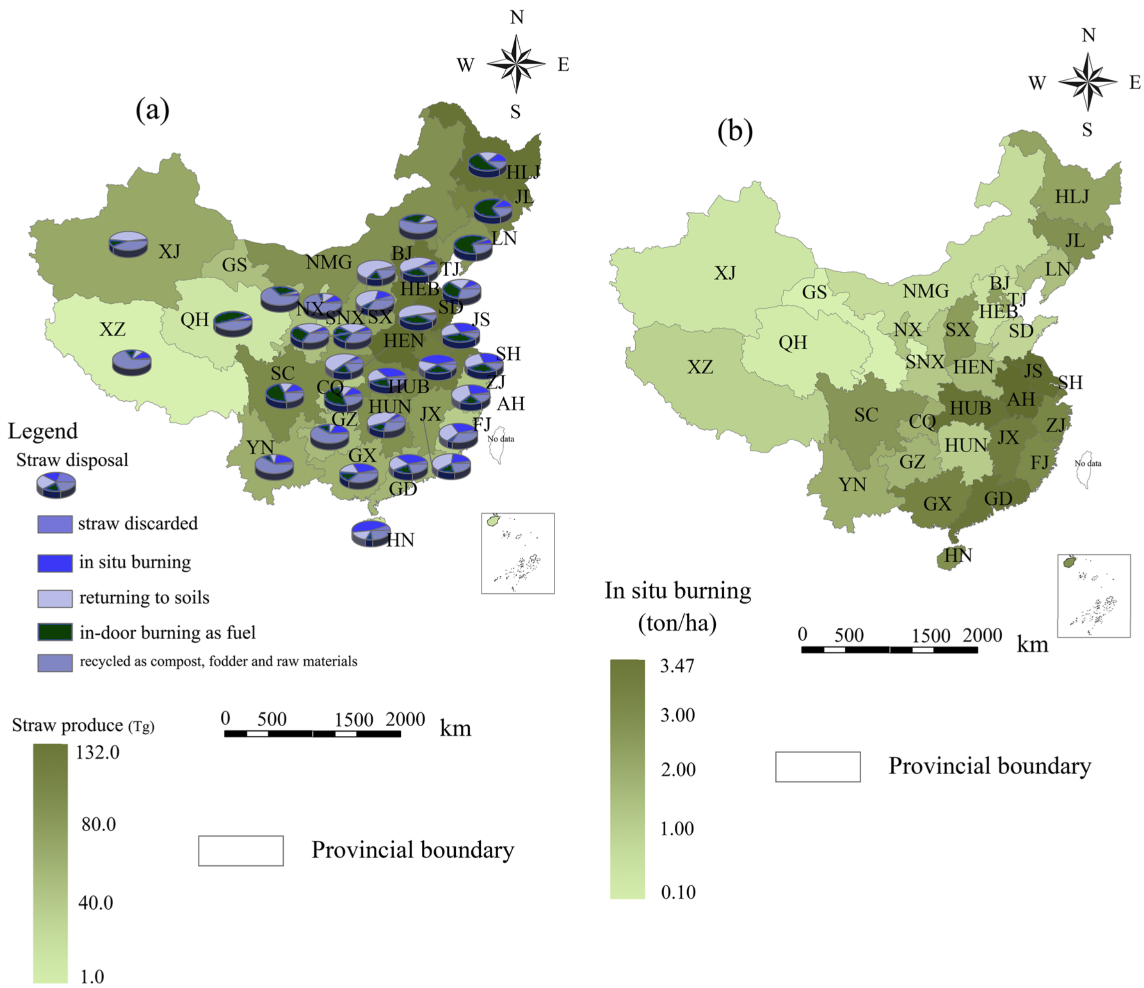

Fig. 5 Straw produce and disposal on provincial scale in China in 2014. **a** Straw produce and disposal. **b** Straw in situ burning per ha. (BJ, Beijing; TJ, Tianjin; HEB, Hebei; SX, Shanxi; SNX, Shaanxi; NMG, Inner Mongolia; LN, Liaoning; JL, Jilin; HLJ, Heilongjiang; SH, Shanghai; JS, Jiangsu; ZJ, Zhejiang; FJ, Fujian; JX, Jiangxi; SD, Shandong; AH, Anhui; HEN, Henan; HUB, Hubei; HUN, Hunan; GD, Guangdong; GX, Guangxi; HN, Hainan; YN, Yunnan; GZ, Guizhou; SC, Sichuan; CQ, Chongqing; XZ, Tibet; GS, Gansu; QH, Qinghai; NX, Ningxia; XJ, Xinjiang)

fertilizers have been applied in excess in the years since 1980s to increase crop yield (Pan 2014), exerting increasing nutrient export loads. The fertilizer use rates in China calculated in this study were 328.16 kg N and 69.47 kg P per ha in 2014, respectively, and these values were significantly higher than those in most other countries and the global average (FAOSTAT 2015). A partial substance flow analysis (SFA) of N and P flows in China's agricultural systems conducted by Chen et al. (2008, 2010) showed that the nutrient erosion losses from arable system were 1.36 Tg N and 0.04 Tg P in 2004, respectively, which were close to our estimates of 1.44 Tg N and 0.06 Tg P. Although a county-level nutrient balance model of Wang et al. (2014c) estimated much higher exporting loads of 4.18 Tg N and 0.09 Tg P in China in 2010, the spatial distribution of nutrient loss was in alignment with our estimates. The differences might be interpreted by the data sources and methods applied. Compared to the average N and P loss to waters from arable system in some western countries (Ulén et al. 2007; van Dijk et al. 2015; Zobrist and Reichert 2006), the values of 13.13 kg N per ha and 0.89 kg P per ha in China were much higher than that in these countries.

Due to differences in pesticide use intensity and categories, meteorological and surface conditions, and crop categories, the pesticide loss rates in China were distributed unevenly. The highest contributions to pesticide loss to aquatic environment agreed well with relative studies (Ouyang et al. 2016), which showed that herbicides including acetochlor contributed the most part loss to aquatic ecosystems, and the highest herbicide loss rate occurred in the northeastern provinces.

Application of agricultural plastic film was the major source of phthalate ester (PAE) pollutions in agricultural soils in China (Hu et al. 2003; Niu et al. 2014). According to Chen et al. (2013), the transform ratio (film residue to PAEs in soil) in China was 12.25, which indicated that

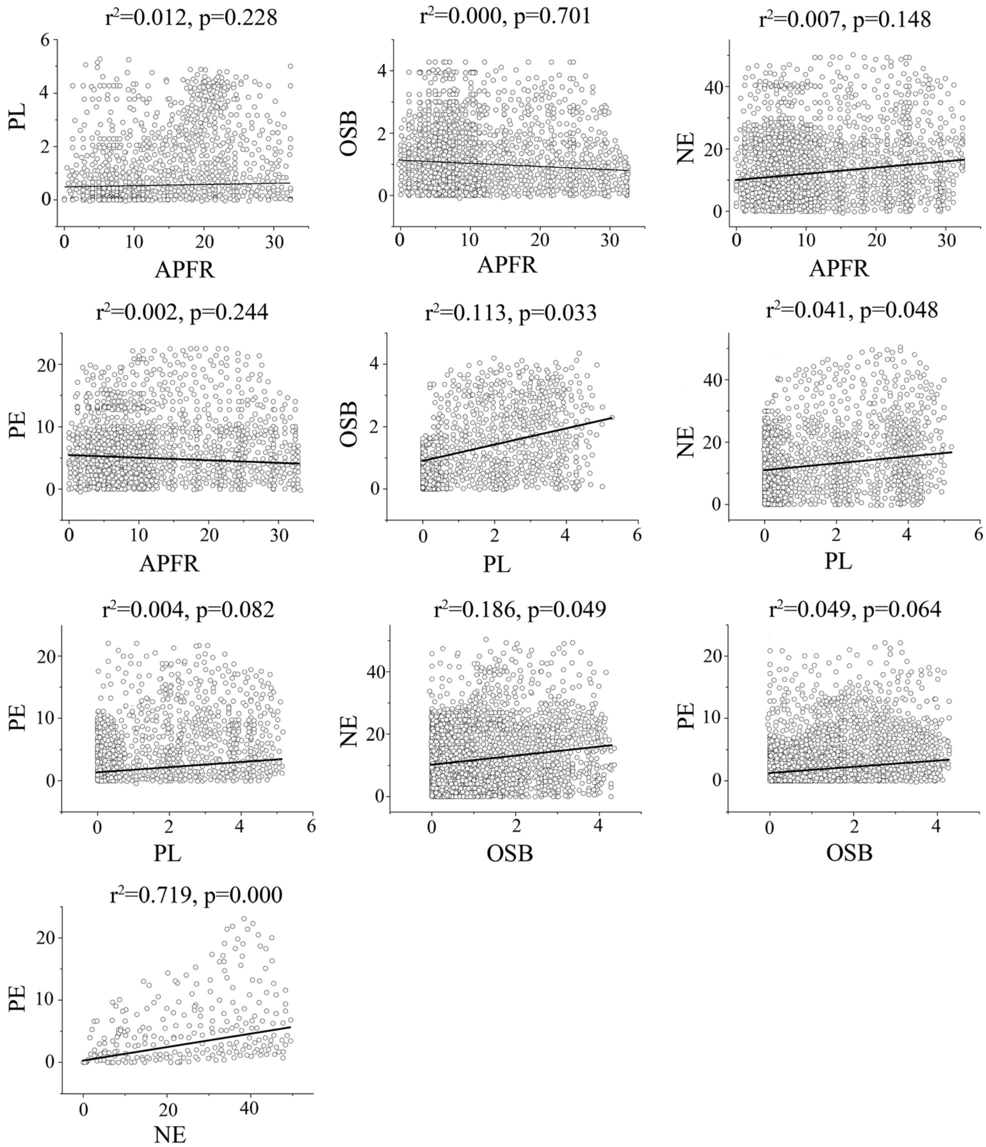

Fig. 6 Relationship between agricultural plastic film residue (APFR; kg/ha), pesticide (ai) loss (PL; g/ha), open straw burning (OSB; tons/ha), N emission (NE; kg/ha), and P emission (PE; kg/ha) to water at land unit level in China in 2014

soil film residue of 12.25 kg might transform to soil PAEs of 1 kg. Thus, the national level of PAEs in soils was estimated to be 1.09 mg/kg in this study, which was much higher than the Netherland's target level of PAEs for soils (0.1 mg/kg; Carlon 2007), indicating agricultural soil polluted by PAEs in China was ubiquitous. In north China, within non-recycling scenario, the total PAE concentration loads in greenhouse soils for planting vegetables ranged from 3.81 to 10.71 mg/kg, which were 3.34–7.29 times of those under recycling scenario. The crest value had exceeded the grade II limits for PAEs (10 mg/kg) in arable soils suggested by the Environmental Quality Standard for soil in China (China National Environmental Protection Agency 2008).

When compared with related studies conducted at national scale, the amount of in situ burning straw in this study fell within 16.81% (Cao et al. 2006) or 13.27% (Li and Wang 2013) of the estimate simulated. And the uneven distribution of in situ burning aligned well with the RS data (He et al. 2007; van der Werf et al. 2010). In north China, field burning occurred from May to July, most frequently in June when wheat was harvested (see Appendix Fig. 7). Wheat straw open burning combined with high industrial and vehicle emission under high humidity and prevailing south winds might produce regional heavy haze pollution with high particle concentrations and low visibility in the North China Plain (Wang et al. 2015). In particular, it could cause serious air pollution events in metropolis like Beijing (Li et al. 2008). Southern China was an important rice production area. Rice debris burning there most occurred in October and November (see Appendix Fig. 7). Phoothiwut and Junyapoon (2013) reported that the average concentration of particulate matter less than 1002 μm during haze period was 3.5 times higher than that in non-burning period. And fairly high values of the smoke tracer and levoglucosan concentrations were observed in rice straw burning episode (Lee et al. 2008).

Behind the spatial correlation between NPS loads

Labor costs have increased consistently with the farmers' income in rural China where labor markets have become relatively active in the past decades (Chen et al. 2008; Huang et al. 2009a). The income gap between off-farm and on-farm labor has been widening (Kung 2002). Increasingly, farmers have shifted their land practices, attempting to save labors with more material inputs, because these saved labors tend to gain higher economic returns from off-farm work. Intensive use of chemical fertilizers, pesticides, and in situ straw burning for next crops could save farm labors. The subsidy backed price policy for fertilizer and pesticides has often been blamed for that it has distorted the market price of these land inputs and has remained these prices relatively low to intensify resource use in the search for higher productivity from land. For the straw disposal, collection and recycling of straw residue is often time consuming, economic inefficient, and labor costly. In developed areas, such as Shanghai and Jiangsu, the share of straw in situ burnt is higher than some underdeveloped areas. As expected, the positive correlations between pesticide loss, open straw burning, N and P release were observed. In contrast, relationship between film plastic residue and other pollution loads was not well correlated. One possible reason for the difference is that film application is associated with higher intensity of labor input. It could not be generally considered as a labor-saving practice. In China, a large portion of plastic film was used to grow cash crops and vegetables with higher economic value, often indicating more labor inputs.

Implications for control of agricultural NPS pollution

The challenges facing China include how to produce enough food to feed the growing population and how to manage the agricultural NPS pollution in a sustainable way. An essential prerequisite to the latter is that a total picture of NPS pollution in China must be clear. This research provided the baseline information on the NPS pollution status and its spatial variation for policy makers to control NPS pollution risk. In particular, nutrient emission loads varied across China, and the values were high in south and east regions. Efforts targeted to control NPS pollution from arable nutrient loss should be given priority in these areas. As chemical fertilizer and manure contribute large part emission of N and P, increasing fertilizer N and P use efficiencies should be adopted to mitigate nutrient losses from arable farming. Several practices including reduction of tillage in erosion-prone areas, preservation of natural vegetation, and reducing fertilizer inputs in areas of excessive fertilization and adoption of eco-fertilizers should be considered in specific area with high NPS pollution risk. In addition, the role of animal manure disposal becomes important as the amount of livestock in China tend to increase (Li et al. 2011), which pose a great pressure on the recycling of animal waste as fertilizer in face of increasing labor cost. From a system perspective, to optimize the farmland-animal nutrient management in rural areas, planning (including spatial planning) of agricultural land use and livestock farms should be highlighted within regions.

This study has also highlighted the spatial variation of pesticide loss, film residue, and straw burning to help to identify potential reduction of NPS pollution. Management strategies like film recycling and application of plastic film easy to dissolve, temporary and seasonal bans on open straw burning, and straw disposal in more environmentally friendly ways such as returning to soils, banning some particular pesticide with high toxicity while encouraging production of less toxic pesticides, may be useful. However, the agricultural NPS pollution has proven to be difficult to control. Policy options to control NPS pollution need to be targeted to adapt to local environment due to the spatial variation of pollution sources, loads, and exposure. Except for the information available to the nature of the agricultural NPS pollution, policies to control NPS pollution also depend on socio-economic decisions about who should bear the costs and responsibilities of control (Grazhdani 2015). As economic incentives and initiatives are often put ahead of agricultural activities by farmers, governments should enhance NPS pollution reduction through programs of incentives for farmers to reduce NPS pollution particularly in the prior regions with higher pollution loads and human exposure risk. The instruments of control (standard, ban, or permission), planning (including spatial planning), taxes, or subsidies should be carefully used (Grazhdani 2015; Pena-Haro et al. 2010; Pan 1994).

In this paper, our focus was the NPS impact of agro-land use practices including fertilization, application of pesticide and plastic film, and crop straw burning. But the NPS pollution was multi-source. As a reviewer of this paper noted, besides the above agro-land use practices, other NPS sources like fine particulate matter (PM2.5) in China's urban areas could negatively impact the surrounding farmland areas, which was a significant NPS source of farmland surrounding urban cities in eastern and central China (Han et al. 2015). Although the NPS pollution caused by other NPS sources was not the main concerns of this research, strict environmental policies or actions should be adopted in heavy polluted areas.

Approach applied for agricultural NPS pollution assessment

This research combined the emission scenarios with geo-spatial data (e.g., RS images, soil maps, DEM) to estimate the NPS pollution from crop farming. Compared to some mechanism model (e.g., SWAT, AGNPS), this method simplified the hydrological and biochemical processes, making it easier for NPS pollution modeling with limited data. It could be an effective tool for modeling the NPS pollution, if mass data on soil, land use, climate, and topography were available. In this study, the use of export scenarios derived from field experiments and investigations, coupling with soil types, crops, and typography, provided a consistent and accurate NPS pollution load calculation. The model developed here can be applied in similar situations for estimating NPS pollution loads at multiple scales. However, several uncertainties might inherit to the modeling procedure.

The uncertainties mainly resulted from the spatial variation of NPS pollution loadings and the integration between geo-spatial data and NPS pollution scenarios developed. First, to secure the accuracy of the results, the export scenarios we developed must cover all the emission situations across China. But the fact was that farming practices were often considered to be "random" due to > 200 million farm smallholders in China. This predefinition contributed to some uncertainties in the spatial differences of fertilizer, pesticide, agricultural film uses, and straw disposal in a certain area. Fertilization and pesticide application might vary as households' livelihood desires changed in time and space. The data collation was often insufficient and costly and, sometimes, could not meet the research needs. Second, the integration of geo-spatial data improved the resolution of spatial modeling and facilitated the understanding of spatial distribution of NPS pollution at national scale; nevertheless, spatial heterogeneity might, to some extent, be neglected in the scaling-up processes. Future modeling method might improve estimate accuracy with higher resolution of spatio-temporal data and vast field experiment data inputs.

Another uncertainty was also needed to be discussed in this study. The NPS emission scenarios were derived from the field experiments or surveys conducted from 2007 to 2008. There was a temporal gap between the scenarios and the geo-data obtained in 2014. To our knowledge, until now, the field dataset used to generate the scenarios was the most updated national NPS census dataset in China. It might lead to more accurate results if more updated data were available.

Conclusions

Limited information on NPS pollution from agriculture has restricted agricultural NPS pollution management practices. While, to our knowledge, this research is the first integrated assessment of NPS pollution loads from arable system in China, we used the emission scenarios and geo-spatial data to calculate NPS loads of nutrient emission, pesticide, plastic film residue, and crop straw in situ burning. This research provided a total picture of agricultural NPS pollution in China. Our results showed that the agricultural NPS loads were unevenly distributed across China. And the spatial correlation of pollution loads indicated a higher co-occurrence and human exposure in populous areas, especially the eastern China. The results provided policy makers full-scale information on the agricultural NPS pollution status and its spatial variation to control NPS pollution risk in China. Policy options to control NPS pollution are needed to be targeted to adapt to local environment due to the spatial variation of the NPS sources, loads, and exposure.

Although the empirical export coefficient model and its derived MFA method were widely applied in NPS calculation at regional or larger scales, they did not take into account some key factors that affect NPS, such as the hydrological factors. This had limited their capability in understanding the spatial variation of NPS pollution. However, coupling the scenario development with spatial data not only helped better capture the total NPS loads but also helped gain better understanding of the spatial and temporal variation of the NPS loading. This method provided an avenue to accounting for these key factors, but it also had some limitations. Cautions should also be exercised when applying the scenario model for assessing the NPS. The accuracy of the scenario method was largely limited by the availability of the representative NPS emission scenarios that derived from vast field experiments. To generate the representative scenarios, the sampling or survey sites should consider the diversity and spatial variation of the factors that affect the NPS. Another prerequisite to use the scenario method was to obtain the high-quality and high-resolution data of the influence factors, such as land use type, soil type, and cropping patterns. If other social economic factors were took into account, these factors are also needed to be spatially digitalized. There remained a lot of work to be done before the method application.

Funding information We thank financial support for this work from the National Science Foundation of China (No. 41130526).

Appendix

Table 2 Representative N/P runoff or leaching emission scenarios of arable land in China

Scenario code[a]	Provinces (or cities)[b]	Slope[c]	Land use	Dominant crops	Input (kg/ha/year)		Leaching/runoff (kg/ha/year)	
					N	P	TN	TP
1	I	S2	Upland	Wheat, soybean, potato, maize	207	33	8.12	4.08
2	I	S3	Upland	Millet, highland barley, oilseed rape, maize	168	38	2.63	6.60
3	I	S2	Upland	Maize	218	45	8.49	4.70
4	I	S2	Garden	Fruit tree	392	70	4.13	0.32
5	I	S2	Upland	Oilseed rape	59	39	1.80	1.32
6	I	S3	Upland	Potato	120	45	0.83	0.06
7	I	S2	Garden	Persimmon tree	182	36	4.89	1.56
8	I	S2	Upland	Wheat, maize	503	25	3.30	6.65
9	I	S3	Upland	–	180	38	0.66	0.05
10	I	S3	Upland	–	235	42	2.10	5.28
11	I	S3	Upland	–	180	38	3.30	0.24
12	I	S3	Upland	–	150	38	1.58	3.96
13	I	S2	Upland	–	225	30	9.83	4.89
14	I	S2	Upland	–	225	30	10.19	8.46
15	I	S2	Upland	–	180	38	4.13	0.32
16	II	S1	Paddy	Rice	172	28	5.96	1.50
17	II	S1	Upland	Maize	196	39	2.97	1.13
18	II	S1	Upland	–	150	30	2.70	1.98
19	II	S1	Upland	–	180	45	8.94	5.10
20	II	S1	Upland	–	150	75	4.95	0.38
21	III	S1	Upland	Welsh onion, cucumber	285	91	10.02	6.65
22	III	S1	Paddy	Rice	253	51	23.33	4.13
23	III	S1	Upland	Wheat	300	78	8.45	5.25
24	III	S1	Upland	–	375	120	8.97	5.40
25	III	S1	Upland	Vegetable crops	420	150	14.25	5.63
26	IV	S2	Upland	Wheat, sweet potato, barley	464	83	18.62	3.83
27	IV	S2	Upland	Wheat, maize, tobacco	126	32	7.01	9.66
28	IV	S3	Garden	Tea, fruit tree	278	76	5.19	1.83
29	IV	S2	Garden	Tea, fruit tree, longan	396	127	4.07	1.28
30	IV	S2	Garden	Tea, fruit tree	383	64	8.04	1.80
31	IV	S2	Paddy	Rice, oilseed rape	411	54	8.66	10.07
32	IV	S2	Paddy	Rice	281	44	27.72	23.21
33	IV	S3	Garden	Tea, fruit tree	240	76	10.14	4.34
34	IV	S2	Upland	Wheat, maize	165	59	7.53	1.67
35	IV	S3	Upland	Wheat, peanut	198	78	10.58	5.18
36	IV	S2	Upland	Wheat, maize	486	33	7.92	2.82
37	IV	S3	Upland	Chrysanthemum	330	106	6.90	6.74
38	IV	S3	Upland	Wheat, tobacco	104	24	9.42	8.22
39	IV	S2	Upland	Tea	612	0	2.61	1.08
40	IV	S2	Upland	Rice, tobacco	345	70	19.05	6.92
41	IV	S2	Upland	Soybean, peanut, tobacco	227	53	21.53	7.97
42	IV	S3	Upland	–	249	30	5.10	1.50

Table 2 (continued)

Scenario code[a]	Provinces (or cities)[b]	Slope[c]	Land use	Dominant crops	Input (kg/ha/year)		Leaching/runoff (kg/ha/year)	
					N	P	TN	TP
43	IV	S2	Paddy	Rice	863	20	16.88	9.62
44	IV	S3	Upland	–	225	90	3.86	2.97
45	IV	S3	Upland	–	225	75	7.05	3.45
46	IV	S3	Upland	–	300	105	5.00	3.86
47	IV	S3	Upland	–	450	135	22.50	12.75
48	IV	S3	Paddy	–	300	90	7.79	9.06
49	IV	S3	Paddy	–	315	60	15.20	8.66
50	IV	S3	Paddy	–	345	60	24.95	20.88
51	IV	S3	Paddy	–	375	75	21.62	4.05
52	IV	S3	Paddy	–	300	90	7.79	9.06
53	IV	S2	Upland	–	450	135	24.60	14.03
54	IV	S2	Upland	–	450	135	26.84	15.30
55	IV	S2	Upland	–	450	135	22.50	12.75
56	IV	S2	Paddy	–	360	75	21.62	4.05
57	IV	S2	Paddy	–	375	60	8.66	10.07
58	V	S1	Upland	Cabbage, potato, oilseed rape	567	143	21.96	13.10
59	V	S1	Paddy	Rice	361	48	26.85	9.24
60	V	S1	Paddy	Cabbage, radish	386	70	21.62	3.84
61	V	S1	Paddy	Wheat, rice	327	42	13.13	2.73
62	V	S1	Paddy	Rice, oilseed rape, peanut	317	68	16.85	4.20
63	V	S1	Paddy	Rice	171	32	15.71	5.45
64	V	S1	Upland	Sugarcane, sweet potato, peanut	315	46	14.39	13.01
65	V	S1	Upland	Wheat, potato, soybean, cotton,	333	52	15.78	0.62
66	V	S1	Garden	Tea, grape	418	84	12.83	7.71
67	VI	S1	Upland	Wheat	83	30	1.23	0.47
68	VI	S1	Paddy	Rice	192	83	0.33	0.00
69	VI	S1	Upland	–	750	120	8.94	5.10
70	VI	S1	Upland	–	405	225	5.13	3.09
71	II	S1	Upland	maize	207	33	7.50	1.01
72	II	S1	Upland	Vegetable crops	511	306	12.24	0.05
73	II	S1	Upland	Vegetable crops	171	44	6.14	1.53
74	II	S1	Upland	Soybean	44	25	6.63	0.00
75	II	S1	Garden	Fruit tree	173	75	5.82	0.00
76	II	S1	Upland	–	150	30	7.07	0.00
77	III	S1	Upland	Vegetable crops	953	292	40.10	0.96
78	III	S1	Upland	Wheat, maize	443	60	20.90	0.00
79	III	S1	Garden	Fruit trees	690	202	14.64	0.00
80	III	S1	Upland	Vegetable crops	542	204	23.03	0.15
81	III	S1	Upland	Wheat, sweet potato, maize	423	88	23.01	0.00
82	III	S1	Upland	Maize	239	45	11.51	0.00
83	IV,V	S1	Upland	Vegetable crops	568	109	34.94	2.97
84	IV,V	S1	Upland	Wheat, rice, vegetable crops	401	88	8.43	4.50
85	IV,V	S1	Upland	Wheat, soybean, vegetable crops, maize	590	100	49.35	0.21
86	IV,V	S1	Upland	Vegetable crops	230	65	40.41	13.25
87	IV,V	S1	Garden	Fruit trees	416	148	14.81	4.17
88	VI, I	S1	Upland	Wheat, oilseed rape, maize	225	43	10.13	1.44

Table 2 (continued)

Scenario code[a]	Provinces (or cities)[b]	Slope[c]	Land use	Dominant crops	Input (kg/ha/year)		Leaching/runoff (kg/ha/year)	
					N	P	TN	TP
89	VI, I	S1	Upland	Cotton	236	90	8.42	0.00
90	VI, I	S1	Upland	Vegetable crops	781	80	50.42	1.97
91	VI, I	S1	Upland	Vegetable crops	622	226	29.90	0.65
92	VI, I	S1	Garden	Fruit trees	446	280	19.83	1.34
93	VI	S1	Upland	Wheat, maize	558	95	7.08	0.00
94	VI	S1	Garden	Fruit trees	486	118	0.00	0.00
95	VI	S1	Upland	Wheat, maize	212	30	0.00	0.00
96	VI	S1	Upland	–	236	90	0.00	0.00
97	VI	S1	Upland	–	558	95	0.00	0.00
98	VI	S1	Upland	–	781	80	0.00	0.00

Data source: the first national pollution census of China. The details of soil types in each scenario are not presented in this table. Please find the details on the web sites of the first national pollution census, http://cpsc.mep.gov.cn/

– no data available

[a] Scenario codes 1–70: runoff loss; codes 71–98: leaching loss

[b] I, II, III, IV, V, and VI represent the six agro-climate zones across China (I: plateau and mountain areas of north China; II: semi-humid plains of northeast China; III: semi-humid plains of Yellow, Huai, and Hai Rivers; IV: mountain and hilly areas of south China; V: humid plains of south China; VI: arid and semi-arid plains of northwest China). Inner Mongolia Autonomous Region, Shaanxi province, Gansu province, Ningxia Hui Autonomous Region, and Shanxi province are within agro-climate zone I. Heilongjiang province, Liaoning province, and Jilin province are within II. Beijing city, Tianjing city, Hebei province, Shandong province, Henan province, and Jiangsu province are within III. Zhejiang province, Fujian province, Guangdong province, Guangxi Zhuang Autonomous Region, Hainan province, Guizhou province, Yunnan province, Shanghai city, and Jiangxi province are within IV. Chongqing city, Sichuan province, Hubei province, and Hunan province are within V. Tibet, Qinghai, and Xinjiang Uygur Autonomous Region are within VI

[c] S1: 0°–5°; S2: 5°–15°; S3: > 15°

Table 3 Representative pesticide runoff or leaching emission scenarios of arable land in China

Scenario code	Province[a]	Pesticide	Runoff/ leaching	Slope[b]	Land use	Dominant crops	Application (kg ai/ha/year)	Loss (g ai/ha/year)
1	I	2,4-D	R	S2	Upland	Wheat	0.65	0.00
2	IV	2,4-D	R	S2	Garden	Fruit tree	1.68	0.00
3	II	2,4-D	L	S1	Upland	Soybean	1.32	0.00
4	VI	2,4-D	L	S1	Upland	Wheat	0.43	0.00
5	I	Atrazine	R	S2	Upland	Maize	1.95	0.05
6	I	Atrazine	R	S2	Upland	Maize	1.13	0.00
7	II	Atrazine	R	S1	Paddy	Rice	0.23	0.00
8	V	Atrazine	R	S1	Upland	Sweet potato	0.72	0.00
9	V	Atrazine	R	S1	Upland	Sugarcane	4.50	0.00
10	V	Atrazine	R	S1	Upland	Vegetable	0.38	0.00
11	V	Atrazine	R	S1	Paddy	Rice	1.03	0.02
12	V	Atrazine	R	S1	Paddy	Rice	0.65	0.00
13	II	Atrazine	L	S1	Upland	Vegetable	0.18	0.00
14	II	Atrazine	L	S1	Upland	Maize	2.43	0.01
15	III	Atrazine	L	S1	Upland	Vegetable	0.46	0.00
16	V	Atrazine	L	S1	Upland	Vegetable	0.38	0.00
17	I	Imidacloprid	R	S2	Garden	Fruit tree	0.12	0.00
18	III	Imidacloprid	R	S1	Paddy	Rice	0.06	0.00
19	IV	Imidacloprid	R	S3	Garden	Tea, fruit tree	0.15	0.00
20	IV	Imidacloprid	R	S3	Upland	Maize	0.38	0.00
21	IV	Imidacloprid	R	S3	Upland	Fruit tree	0.19	0.00
22	IV	Imidacloprid	R	S3	Garden	Fruit tree	0.11	0.00
23	IV	Imidacloprid	R	S2	Garden	Tea and fruit tree	0.05	0.00
24	IV	Imidacloprid	R	S2	Upland	Wheat, maize	0.12	0.00
25	IV	Imidacloprid	R	S2	Upland	Tobacco	0.06	0.01
26	IV	Imidacloprid	R	S2	Garden	Strawberry	0.02	0.00
27	IV	Imidacloprid	R	S2	Upland	Peanut	0.04	0.00
28	IV	Imidacloprid	R	S2	Garden	Tea and fruit tree	0.25	0.07
29	IV	Imidacloprid	R	S2	Paddy	Rice	0.19	0.37
30	IV	Imidacloprid	R	S2	Paddy	Rice	0.02	0.03
31	V	Imidacloprid	R	S1	Upland	Oilseed rape, peanut, cotton, tobacco	0.06	0.02
32	V	Imidacloprid	R	S1	Upland	Oilseed rape	0.17	0.44
33	V	Imidacloprid	R	S1	Paddy	Rice	0.01	0.00
34	V	Imidacloprid	R	S1	Paddy	Rice	0.14	0.07
35	V	Imidacloprid	R	S1	Paddy	Rice	0.07	0.00
36	V	Imidacloprid	R	S1	Paddy	Rice	0.13	0.16
37	V	Imidacloprid	R	S1	Paddy	Rice	0.09	0.06
38	II	Imidacloprid	L	S1	Upland	Vegetable	0.02	0.00
39	III	Imidacloprid	L	S1	Upland	Vegetable	0.20	0.00
40	III	Imidacloprid	L	S1	Upland	Wheat, maize	0.06	0.00
41	III	Imidacloprid	L	S1	Upland	Vegetable	0.02	0.00
42	III	Imidacloprid	L	S1	Garden	Fruit tree	0.12	0.00
43	V	Imidacloprid	L	S1	Upland	Vegetable	0.14	0.08
44	II	Butachlor	R	S1	Paddy	Rice	1.52	0.00
45	III	Butachlor	R	S1	Paddy	Rice	0.63	0.00
46	IV	Butachlor	R	S3	Upland	Maize	0.38	0.02
47	IV	Butachlor	R	S3	Upland	Fruit tree	0.19	0.00
48	IV	Butachlor	R	S2	Paddy	Rice	0.19	0.00
49	V	Butachlor	R	S1	Paddy	Rice	0.60	0.00
50	V	Butachlor	R	S1	Paddy	Rice	0.95	4.27
51	V	Butachlor	R	S1	Paddy	Rice	0.32	0.00
52	II	Butachlor	L	S1	Upland	Vegetable	1.38	0.00
53	II	Butachlor	L	S1	Upland	Maize	2.03	0.00
54	III	Butachlor	L	S1	Upland	Vegetable	0.35	0.00
55	V	Butachlor	L	S1	Upland	Vegetable	0.53	0.05
56	VI	Butachlor	L	S1	Garden	Grape	0.17	0.00
57	I	Chlorpyrifos	R	S3	Upland	Sweet potato	0.75	0.06
58	I	Chlorpyrifos	R	S2	Upland	Soybean, maize	1.28	0.37
59	I	Chlorpyrifos	R	S2	Upland	Wheat, maize	0.48	0.00
60	I	Chlorpyrifos	R	S2	Upland	Peanut	1.88	0.00
61	II	Chlorpyrifos	R	S1	Paddy	Rice	2.25	0.00
62	III	Chlorpyrifos	R	S1	Paddy	Rice	3.90	0.00
63	IV	Chlorpyrifos	R	S3	Upland	Maize	1.44	0.00

Table 3 (continued)

Scenario code	Province[a]	Pesticide	Runoff/ leaching	Slope[b]	Land use	Dominant crops	Application (kg ai/ha/year)	Loss (g ai/ha/year)
64	IV	Chlorpyrifos	R	S3	Garden	Tea	1.50	0.00
65	IV	Chlorpyrifos	R	S3	Upland	Maize	3.75	0.00
66	IV	Chlorpyrifos	R	S3	Upland	Wheat	1.28	0.00
67	IV	Chlorpyrifos	R	S3	Garden	Fruit tree	1.13	0.00
68	IV	Chlorpyrifos	R	S3	Upland	Fruit tree	1.08	0.00
69	IV	Chlorpyrifos	R	S2	Upland	Wheat, maize	2.00	0.00
70	IV	Chlorpyrifos	R	S2	Upland	Sugarcane	2.25	0.00
71	IV	Chlorpyrifos	R	S2	Upland	Wheat, sweet potato, maize, barley	1.82	0.00
72	IV	Chlorpyrifos	R	S2	Garden	Fruit tree	3.00	0.00
73	IV	Chlorpyrifos	R	S2	Upland	Rice	0.69	0.00
74	IV	Chlorpyrifos	R	S2	Garden	Fruit tree	0.43	0.00
75	IV	Chlorpyrifos	R	S2	Paddy	Rice	0.48	0.00
76	IV	Chlorpyrifos	R	S2	Paddy	Rice	1.88	0.00
77	V	Chlorpyrifos	R	S1	Upland	Oilseed rape, maize	4.95	0.00
78	V	Chlorpyrifos	R	S1	Upland	Cotton	5.02	0.00
79	V	Chlorpyrifos	R	S1	Upland	Vegetable	2.90	0.00
80	V	Chlorpyrifos	R	S1	Garden	Fruit tree	1.81	0.00
81	V	Chlorpyrifos	R	S1	Paddy	Rice	3.21	0.02
82	V	Chlorpyrifos	R	S1	Paddy	Rice	1.79	0.01
83	V	Chlorpyrifos	R	S1	Paddy	Rice	1.43	0.01
84	V	Chlorpyrifos	R	S1	Paddy	Rice	2.71	0.00
85	V	Chlorpyrifos	R	S1	Paddy	Rice	2.63	0.00
86	II	Chlorpyrifos	L	S1	Upland	Vegetable	7.79	0.00
87	II	Chlorpyrifos	L	S1	Upland	Maize	1.43	0.00
88	III	Chlorpyrifos	L	S1	Upland	Vegetable	1.66	0.27
89	III	Chlorpyrifos	L	S1	Upland	Soybean, wheat, maize	0.86	0.00
90	III	Chlorpyrifos	L	S1	Upland	Wheat, maize	2.93	0.00
91	III	Chlorpyrifos	L	S1	Upland	Vegetable	5.49	0.00
92	III	Chlorpyrifos	L	S1	Garden	Fruit tree	2.66	0.00
93	V	Chlorpyrifos	L	S1	Upland	Vegetable	1.50	0.00
94	V	Chlorpyrifos	L	S1	Upland	Vegetable	2.25	0.00
95	V	Chlorpyrifos	L	S1	Garden	Vegetable	0.53	0.02
96	VI	Chlorpyrifos	L	S1	Upland	Wheat, maize	1.50	0.00
97	VI	Chlorpyrifos	L	S2	Upland	Vegetable	2.63	0.00
98	VI	Chlorpyrifos	L	S3	Upland	Cotton	1.69	0.00
99	VI	Chlorpyrifos	L	S4	Garden	Fruit tree	1.39	0.00
100	IV	Fipronil	R	S3	Garden	Fruit tree	0.07	0.00
101	IV	Fipronil	R	S2	Paddy	Rice	0.05	0.00
102	IV	Fipronil	R	S2	Paddy	Rice	0.13	0.51
103	V	Fipronil	R	S1	Upland	Peanut	0.02	0.00
104	V	Fipronil	R	S1	Garden	Fruit tree	0.18	0.00
105	V	Fipronil	R	S1	Paddy	Rice	0.11	0.30
106	V	Fipronil	R	S1	Paddy	Rice	0.05	0.10
107	V	Fipronil	R	S1	Paddy	Rice	0.09	0.05
108	V	Fipronil	R	S1	Paddy	Rice	0.05	0.00
109	V	Fipronil	R	S1	Paddy	Rice	0.07	0.43
110	V	Fipronil	L	S1	Upland	Vegetable	0.23	0.00
111	V	Fipronil	L	S1	Garden	Vegetable	0.05	0.00
112	I	Acetochlor	R	S3	Upland	Sweet potato	0.75	0.05
113	I	Acetochlor	R	S3	Upland	Millet	0.75	0.00
114	I	Acetochlor	R	S2	Upland	Soybean	0.60	3.78
115	I	Acetochlor	R	S2	Upland	Maize	1.13	0.00
116	I	Acetochlor	R	S2	Upland	Peanut	1.88	2.29
117	IV	Acetochlor	R	S3	Garden	Tea and fruit tree	1.28	0.00
118	IV	Acetochlor	R	S3	Upland	Peanut	1.50	0.00
119	IV	Acetochlor	R	S2	Garden	Fruit tree	3.08	0.00
120	IV	Acetochlor	R	S2	Upland	Peanut	1.50	0.00
121	IV	Acetochlor	R	S2	Upland	Wheat	0.38	0.00
122	IV	Acetochlor	R	S2	Upland	Peanut	3.60	0.00
123	IV	Acetochlor	R	S2	Paddy	Rice	0.48	0.00
124	IV	Acetochlor	R	S2	Paddy	Rice	0.73	0.00
125	V	Acetochlor	R	S1	Upland	Soybean, maize	2.70	1.96
126	V	Acetochlor	R	S1	Upland	Cotton	2.79	0.43

Table 3 (continued)

Scenario code	Province[a]	Pesticide	Runoff/ leaching	Slope[b]	Land use	Dominant crops	Application (kg ai/ha/year)	Loss (g ai/ha/year)
127	V	Acetochlor	R	S1	Upland	Vegetable	0.55	0.00
128	V	Acetochlor	R	S1	Paddy	Rice	1.50	0.00
129	V	Acetochlor	R	S1	Paddy	Rice	1.20	0.00
130	V	Acetochlor	R	S1	Paddy	Rice	1.01	0.00
131	II	Acetochlor	L	S1	Upland	Maize	2.93	0.00
132	II	Acetochlor	L	S1	Upland	Soybean	1.32	0.20
133	III	Acetochlor	L	S1	Upland	Vegetable	0.90	2.05
134	III	Acetochlor	L	S1	Upland	Wheat, maize, soybean	0.99	0.00
135	III	Acetochlor	L	S1	Upland	Wheat, maize	1.37	0.66
136	III	Acetochlor	L	S1	Upland	Vegetable	0.77	0.00
137	III	Acetochlor	L	S1	Garden	Fruit tree	1.27	0.00
138	V	Acetochlor	L	S1	Upland	Wheat, soybean	0.68	0.00
139	VI	Acetochlor	L	S1	Upland	Wheat	0.60	0.00
140	VI	Acetochlor	L	S1	Upland	Cotton	1.35	0.00
141	VI	Acetochlor	L	S1	Garden	Grape	1.71	0.00

The details of soil types in each scenario are not presented in this table

R runoff, *L* leaching

[a] I, II, III, IV, V, and VI represent the six agro-climate zones across China (I: plateau and mountain areas of north China; II: semi-humid plains of northeast China; III: semi-humid plains of Yellow,, Huai and Hai Rivers; IV: mountain and hilly areas of south China; V: humid plains of south China; VI: arid and semi-arid plains of northwest China). Inner Mongolia Autonomous Region, Shaanxi province, Gansu province, Ningxia Hui Autonomous Region, and Shanxi province are within agro-climate zone I. Heilongjiang province, Liaoning province, and Jilin province are within II. Beijing city, Tianjing city, Hebei province, Shandong province, Henan province, and Jiangsu Province are within III. Zhejiang province, Fujian province, Guangdong province, Guangxi Zhuang Autonomous Region, Hainan province, Guizhou province, Yunnan province, Shanghai city, and Jiangxi province are within IV. Chongqing city, Sichuan province, Hubei province, and Hunan province are within V. Tibet, Qinghai, and Xinjiang Uygur Autonomous Region are within VI

[b] S1: 0°–5°; S2: 5°–15°; S3: > 15°

Table 4 Representative agricultural film mulching and residual scenarios of arable land in China

Scenario code	Province[a]	Crops planted in greenhouse	Crops	Film management scenario	Application (kg/ha/year)	Residual rate (%)
1	I	Yes	Vegetable	Non-recycling	70.43	11.5
2	I	Yes	Vegetable	Recycling	85.71	1.4
3	I	No	Grain crops	Non-recycling	48.43	12.7
4	I	No	Grain crops	Recycling	57.04	7.1
5	I	No	Vegetable	Non-recycling	45.00	65.0
6	I	No	Vegetable	Recycling	66.39	6.1
7	II	Yes	Vegetable	Non-recycling	–	75.1
8	II	Yes	Vegetable	Recycling	82.81	9.6
9	II	No	Grain crops	Non-recycling	86.84	17.1
10	II	No	Grain crops	Recycling	84.09	6.6
11	II	No	Vegetable	Non-recycling	76.42	42.4
12	II	No	Vegetable	Recycling	77.03	3.7
13	III	Yes	Vegetable	Non-recycling	60.00	36.0
14	III	Yes	Vegetable	Recycling	56.04	9.1
15	III	No	Grain crops	Non-recycling	41.21	27.3
16	III	No	Grain crops	Recycling	36.72	19.2
17	III	No	Vegetable	Non-recycling	–	43.4
18	III	No	Vegetable	Recycling	49.21	18.9
19	IV	Yes	Vegetable	Non-recycling	–	15.3
20	IV	Yes	Vegetable	Recycling	103.85	2.6
21	IV	No	Grain crops	Non-recycling	54.63	33.5
22	IV	No	Grain crops	Recycling	62.73	16.5
23	IV	No	Vegetable	Non-recycling	125.89	5.6
24	IV	No	Vegetable	Recycling	78.57	6.3
25	V	Yes	Vegetable	Non-recycling	–	27.8
26	V	Yes	Vegetable	Recycling	98.28	5.8
27	V	No	Grain crops	Non-recycling	51.78	47.8
28	V	No	Grain crops	Recycling	122.41	8.7
29	V	No	Vegetable	Non-recycling	39.09	30.7
30	V	No	Vegetable	Recycling	55.97	13.4
31	VI	Yes	Vegetable	Non-recycling	–	29.8
32	VI	Yes	Vegetable	Recycling	–	5.7
33	VI	No	Grain crops	Non-recycling	47.93	50.7
34	VI	No	Grain crops	Recycling	50.39	12.8
35	VI	No	Vegetable	Non-recycling	–	77.5
36	VI	No	Vegetable	Recycling	33.50	30.0

The details of soil types in each scenario are not presented in this table

– no data available

[a] I, II, III, IV, V, and VI represent the six agro-climate zones across China (I: plateau and mountain areas of north China; II: semi-humid plains of northeast China; III: semi-humid plains of Yellow, Huai, and Hai Rivers; IV: mountain and hilly areas of south China; V: humid plains of south China; VI: arid and semi-arid plains of northwest China). Inner Mongolia Autonomous Region, Shaanxi province, Gansu province, Ningxia Hui Autonomous Region, and Shanxi province are within agro-climate zone I. Heilongjiang province, Liaoning province, and Jilin province are within II. Beijing city, Tianjing city, Hebei province, Shandong province, Henan province, and Jiangsu province are within III. Zhejiang province, Fujian province, Guangdong province, Guangxi Zhuang Autonomous Region, Hainan province, Guizhou province, Yunnan province, Shanghai city, and Jiangxi province are within IV. Chongqing city, Sichuan province, Hubei province, and Hunan province are within V. Tibet, Qinghai, and Xinjiang Uygur Autonomous Region are within VI

Table 5 In situ burned straw residue of in situ burning scenarios across China

Province	Rice	Wheat	Maize	Grain	Soybean	Potato	Oilseed rape	Cotton	Sugarcane	Sorghum
Beijing	745[a]	257	223	156	83	735	27	24	0	211
Tianjin	1111	346	243	119	75	841	0	38	0	187
Hebei	147	58	38	43	17	116	11	4	0	45
Shanxi	1645	378	434	265	97	573	108	66	0	368
Inner Mongolia	396	78	110	78	41	222	20	16	0	183
Liaoning	690	198	194	178	87	587	53	35	0	348
Jilin	1466	0	473	718	138	1451	0	73	0	885
Heilongjiang	1420	274	378	452	136	1032	396	0	0	719
Shanghai	3747	806	969	0	467	4112	297	181	1339	0
Jiangsu	3934	1097	742	418	447	3777	400	131	1775	400
Zhejiang	2708	657	514	0	401	2258	242	116	1510	0
Anhui	3956	1623	1021	1620	328	2024	464	119	1625	908
Fujian	2603	575	536	818	453	2596	194	67	1633	1002
Jiangxi	1863	302	396	546	249	1791	126	101	873	264
Shandong	196	65	47	44	24	226	19	5	0	47
Henan	515	189	120	60	45	318	52	15	307	43
Hubei	4278	941	797	1689	445	2131	345	125	1293	1394
Hunan	328	82	89	0	47	292	25	14	171	112
Guangdong	2893	838	769	813	576	3301	198	0	2989	1479
Guangxi	3251	386	820	895	364	1997	184	129	2611	1040
Hainan	3360	0	961	0	849	3164	0	0	3018	1371
Chongqing	1409	279	336	0	158	996	113	27	451	462
Sichuan	1547	318	346	0	163	969	141	39	509	696
Guizhou	1346	241	308	290	88	902	130	31	917	354
Yunnan	1705	250	452	927	287	1167	159	120	1162	222
Tibet	955	484	299	0	259	1292	135	0	0	0
Shaanxi	636	142	137	92	56	282	53	29	173	176
Gansu	355	66	90	52	43	216	30	18	0	148
Qinghai	0	108	139	0	55	301	41	0	0	0
Ningxia	1232	211	364	0	42	378	116	0	0	0
Xinjiang	340	95	87	0	46	254	31	16	0	116

Unit: kg/ha/year

[a] In Beijing, the average amount of crop waste in situ burned in arable land for rice is 745 kg/ha/year

Table 6 Share of straw produce burned in situ in each province or city across China

Province	Share of straw produce burned in situ	Province	Share of straw produce burned in situ
Liaoning	0.06	Zhejiang	0.24
Jilin	0.12	Guangdong	0.33
Heilongjiang	0.13	Guangxi	0.36
Shandong	0.01	Henan	0.04
Chongqing	0.12	Jiangxi	0.20
Sichuan	0.13	Hebei	0.01
Qinghai	0.04	Guizhou	0.16
Jiangsu	0.29	Tibet	0.10
Beijing	0.07	Hunan	0.03
Shanghai	0.28	Xinjiang	0.02
Tianjin	0.09	Yunnan	0.19
Inner Mongolia	0.03	Hainan	0.44
Anhui	0.41	Shanxi	0.15
Gansu	0.03	Ningxia	0.09
Shaanxi	0.05	Fujian	0.27
Hubei	0.34		

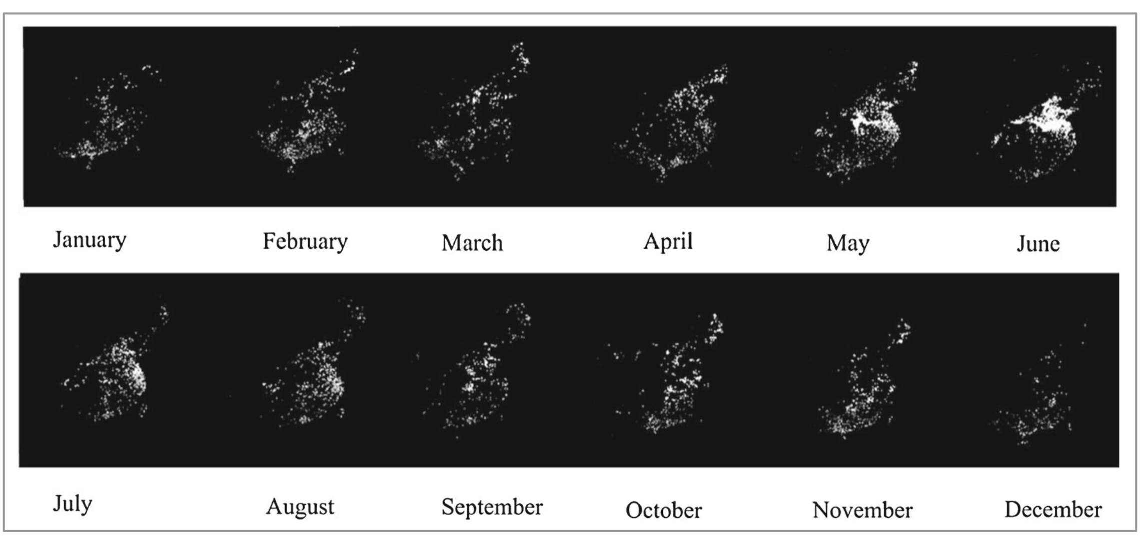

Fig. 7 Spatial variation of monthly straw waste open burning areas across China in 2012. Data source: Global Fire Emission Database (GFED4s, van der Werf et al. 2010)

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