Characterization of non-methane hydrocarbons emitted from open burning of wheat straw and corn stover in China

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Abstract

Field measurements were conducted to determine the characteristics of non-methane hydrocarbons (NMHCs) emitted from open burning of wheat straw and maize stover, two major agricultural residues in China. The headfire ignition technique was used with sampling downwind from the agricultural fire. Fifty-two NMHC species were quantified using gas chromatography/mass spectrometry. A carbon mass balance method was used to determine NMHC emission factors. The emission factors of the total speciated NMHCs from wheat straw and maize stover are 1690±580 mg kg⁻¹ and 1590±430 mg kg⁻¹, respectively. Propane, n-pentane, 2,3-dimethylbutane, 2-methylpentane, propene, benzene and toluene are the main species, together accounting for 55.3%–68.0% of the total NMHCs. On the basis of measured emission factors and the published maximum incremental reactivity values for NMHCs, we estimated the ozone forming potential (OFP) of speciated NMHCs. The results indicate that propene, 1-butene, isoprene, toluene and m, p-xylene have high OFP values and account for about 50% of the total OFP. Alkenes played the most important role in potential ozone formation, followed by aromatics and alkanes.

Keywords: agricultural residues, open burning, non-methane hydrocarbons, emissions, China

1. Introduction

The field burning of agricultural residues, a significant form of biomass burning in China, is a common way to clear land and dispose of waste after crop harvesting quickly. Recently, the government has promulgated regulations to prohibit field burning of agricultural residues. However, fire-count data from the FY-1 earth observation satellite indicate that the open burning of agricultural residues occurred frequently during harvest seasons in several provinces, including Hebei, Henan, Shandong, Jiangsu, and Anhui. Studies estimate that 17%–25.6% of the total agricultural residue production, or 110–157.5 Tg of agricultural residues, are burned in the field in China every year.

The open burning of agricultural residues releases a large amount of pollutants to the atmosphere, including particulate matter (PM), CO, volatile organic compounds (VOCs), and others, which cause serious local and regional environmental and health impacts.

This study was conducted using field measurements of particulate and gaseous pollutant emissions from the open burning of agricultural residues in China. Our previous work summarized PM emissions and trace gas emissions, including CO₂, CO, CH₄, total non-methane hydrocarbons (TNMHC), NOₓ, NH₃, N₂O and SO₂. In this letter, we speciate the NMHC emissions and assess the ozone forming potential (OFP) of the NMHC emissions. Currently there are few data on speciated NMHC emissions from...
the field burning of agricultural residues. Jenkins et al reported limited NMHC species emission, such as benzene, toluene, styrene, and xylene, based on wind tunnel simulation experiments [9]. Estimates of speciated non-methane volatile organic compounds (NMVOC) emissions from the field burning of agricultural residues in China [10–12] mainly adopted emission factors (EFs) from Andreae and Merlet [13]. Andreae and Merlet synthesized a large body of information on emissions of trace gases and aerosols from various types of biomass burning and presented a set of emission factors for individual species emitted from different types of biomass burning [13]. Due to the lack of available measurement data, the emission factors in their database for agricultural residues are primarily based on extrapolation, which may lead to high uncertainty in emission estimates. Our study provides speciated NMHCs EFs for the field burning of agricultural residues in China. These EFs can be used to improve the quality of China’s VOC emission inventory. In addition, source profiles of NMHCs may assist with source apportionment of the emissions. The data on individual NMHC emissions can also be used for modeling local or regional ozone and secondary organic aerosol (SOA) formation.

2. Methods

2.1. Experimental procedures

The experimental procedures have been described in our earlier paper [8] and are briefly summarized here.

Experiments were carried out in a rural area in Shandong province. Wheat straw and maize stover, which contribute about 59% of the total agricultural residues [14, 15], were chosen to represent agricultural residues. The wheat straw and maize stover was organized in windrows in an open area of the field. Several runs were conducted with about 40–60 kg of fuels burned in each. The density of the tested wheat straw and maize stover was about 1.5 kg m$^{-2}$ and 3.0 kg m$^{-2}$, respectively, similar to normal burning in the field. The headfire ignition technique was used to simulate common open burning practices. Measurements were conducted under sunlight, light wind (less than 2 m s$^{-1}$), with a relatively stable wind direction. The sampling instruments were on an agricultural vehicle creating a sampling height of about 2.5 m. The vehicle was moved to different locations to collect representative samples. The choice of locations was based on the wind direction, wind speed and fire location. Sampling sites were 5–10 m from the fire, far enough for smoke emissions to be diluted and cooled to ambient temperature before reaching the sampling equipment. The sampling time for each test ranged from 35 to 45 min, including the flaming and smoldering stages.

For each agricultural residue, three successful tests were conducted. Prior to the planned burning, ambient air samples were collected to measure background concentrations.

Before each test, the burned agricultural residues were weighed and a small bundle of the fuel was sealed and brought to the laboratory for proximate and ultimate analysis. The analysis results are shown in table 1 of our earlier paper [8]. After the tests, the ash was weighed and some was sealed and brought to the laboratory for carbon content analysis.

The experiment site was located far from any point emission sources and highways. To minimize impact from other area sources, the testing was staggered to avoid cooking hours in nearby villages.

2.2. Sampling and analysis

Carbon-containing gases were collected using a sampling configuration that included, from upstream to downstream of the sampling train, a filter holder, a subminiature sampling pump, and a 20 l Tedlar bag. Teflon tubes were used to connect each section. The flow rate of the pump was adjusted to fill the Tedlar bag throughout the entire sampling period. After sampling, aliquots of gas samples were taken out of the 20 l Tedlar bags and placed into 1 l Tedlar bags for various subsequent chemical analyses. These included the determination of gas concentration of CO$_2$, CO, CH$_4$, and TNMHC. A gas chromatograph flame ionization detector (GC-FID) equipped with different packed columns was used to measure CO$_2$, CO, and CH$_4$. TNMHC were determined by subtracting CH$_4$ from total hydrocarbons (THC) which were quantified as CH$_4$ concentration using FID with a blank column.

The measurement of concentrations of speciated NMHCs involved filling 2L stainless-steel vacuumed SUMMA canisters (Entech, USA) with the samples collected in the 20 l Tedlar bags. The canisters were pre-cleaned using ultrapure N$_2$ (>99.999%) before sampling. The sample storage time before analysis was less than 30 days. The analytical procedure for speciated NMHCs was the same as reported by Duan et al [16] and is summarized below. Samples were first concentrated in a Model 7100 pre-concentrator (Entech Instrument Inc., USA). The concentrated samples were then thermally desorbed and carried by ultra-pure helium into an HP6890 gas chromatography coupled to an HP 5973 mass-selective detector (GC-MSD). A RESTEK RTX-1 capillary column, 60 m × 0.2 mm id × 1.0 μm, was used in the system. The initial temperatures were held at 40°C for 2 min, and then increased at a rate of 6°C min$^{-1}$ to 230°C and held for 5 min. Compounds were identified by their retention times and their mass spectra. Standard gas mixtures (1.0 ppm, Supelco TO-14 calibration mix) were first dynamically diluted with zero air, then sampled and analyzed using identical conditions to those for the field samples. A calibration curve was constructed using seven concentrations for each compound. Correlation coefficients, ranging from 0.996 to 1.000, showed good linearity between the integral area of peaks and the concentrations of target compounds. For QA/QC, the pre-concentrator was baked after each analysis, and the GC column was also baked after every 5 min. In total, 52 C3-C11 NHMC species were quantified.

PM$_{2.5}$ was collected by two parallel middle-volume samplers at 78 l min$^{-1}$ (Dizhi Instrument Factory, China). The sampler with quartz-fiber filter collected samples for organic and elemental carbon (OC and EC). OC and EC were determined by a thermal/optical carbon analyzer (DRI Model 2001, USA) using the IMPROVE protocol [17].

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2.3. Determination of emission factors

The carbon mass balance method was used to determine EFs [18]. This method has been widely used to evaluate emission factors during field experiments [19–21]. Dhammapala et al. found that the emission factors determined by the carbon mass balance method were in good agreement with those by determined by a direct method [22].

The carbon mass balance method hypothesizes all the burned carbon is emitted into the atmosphere as carbonaceous particles and carbonaceous gases such as CO₂, CO, CH₄, and TNMHC. To utilize this method, a complete carbon analysis is required. Thus, we measured carbon mass in the biomass fuel and the ash, and carbon in all emission products (CO₂, CO, CH₄, TNMHC, and particles). The sum of OC and EC in PM₂.₅ are categorized as particle carbon. Using this method and the data from the complete carbon analysis, we firstly calculated CO₂ EFs. The EFs of i (e.g. individual speciated NMHC) can be calculated using the following equation:

\[
EF_i = ER_i \times EF_{CO_2} \times \frac{MW_i}{MW_{CO_2}}
\]

where EF₁ and EF₂ are the emission factors of i and CO₂, respectively (unit: g kg⁻¹ biomass); MWᵢ and MW₂ are the molecular weight of i and CO₂, respectively; and ERᵢ is the molar ratio of i to CO₂, a dimensionless unit, and obtained from the following equation:

\[
ER_i = \frac{C_{i, smoke} - C_{i, background}}{C_{CO_2, smoke} - C_{CO_2, background}}
\]

where \(C_{i, smoke}\) and \(C_{i, background}\) are the concentration of i measured in the smoke and background air, in ppm or any other molar concentration equivalent unit; \(C_{CO_2, smoke}\) and \(C_{CO_2, background}\) are the CO₂ concentration measured in the corresponding smoke and background air.

3. Results and discussions

3.1. Emission factors of speciated NMHCs

The EFs of 52 NMHCs species were quantified from the field burning of wheat straw and maize. The EFs are presented in Table 1. The average EFs for total NMHCs are 1690 ± 580 mg kg⁻¹ and 1590 ± 430 mg kg⁻¹ for wheat straw and maize stover, respectively. The 52 NMHC species were then classified into three categories: alkanes (25 species), alkenes (14 species), and aromatics (13 species). The EFs of alkanes, alkenes and aromatics from wheat straw are 690 ± 170 mg kg⁻¹, 140 ± 190 mg kg⁻¹, and 590 ± 230 mg kg⁻¹, respectively. The alkane, alkene and aromatic EFs from maize stover are 550 ± 220 mg kg⁻¹, 490 ± 54 mg kg⁻¹, and 550 ± 170 mg kg⁻¹, respectively. Most of EFs of individual NMHC species from wheat straw and maize stover are similar.

Table 1 also presents speciated NMHC EFs from the literature. Most of the NMHC species EFs from Andreae and Merlet [13] are higher than in the findings of this study. Jenkins et al. reported EFs for benzene and toluene from four types of

### Table 1. Emission factors of speciated NMHCs from open burning (mg kg⁻¹).

| No | Species | Wheat straw (n = 3) | Maize stover (n = 3) | EFs in references |
|----|---------|---------------------|---------------------|------------------|
| 1  | Alkanes | 690 ± 170          | 550 ± 220           |                  |
| 2  | 1-Propene | 100 ± 50            | 170 ± 33            | 520⁺             |
| 3  | 2-Butene | 2.9 ± 3.4           | 7.4 ± 5.6           | 15⁰              |
| 4  | 3-Butene | 16 ± 15             | 25 ± 6.3            | 60⁰              |
| 5  | 1-Pentene | 2.0 ± 2.0           | 12 ± 11             |                  |
| 6  | Cyclopentane | 2.9 ± 5.0         | 0.25 ± 0.23         |                  |
| 7  | n-Hexane | 5.8 ± 8.0           | 4.6 ± 7.9           | 70⁰              |
| 8  | Cyclohexane | 2.2 ± 2.2          | 7.8 ± 12            |                  |
| 9  | n-Heptane | 4.8 ± 3.3           | 8.3 ± 2.2           |                  |
| 10 | n-Octane | 3.8 ± 2.4           | 9.0 ± 3.4           |                  |
| 11 | n-Decane | 1.9 ± 2.2           | 10 ± 7.1            |                  |
| 12 | 2,3-Dimethylbutane | 160 ± 100     | 62 ± 95             |                  |
| 13 | 2,4-Dimethylbutane | 0.60 ± 0.92       | 0.27 ± 0.24         |                  |
| 14 | 2-Methylpentane | 170 ± 92          | 57 ± 90             |                  |
| 15 | 3-Methylpentane | 12 ± 3.2           | 11 ± 4.3            |                  |
| 16 | 2-Methylketone | 2.8 ± 2.1          | 4.6 ± 2.6           |                  |
| 17 | 2,4-Dimethylpentane | 1.9 ± 0.94      | 2.3 ± 1.7           |                  |
| 18 | 2,3-Trimethylpentane | ND               | 0.27 ± 0.47         |                  |
| 19 | Propene  | 120 ± 41            | 210 ± 59            | 1000⁺             |
| 20 | 1-buten  | 40 ± 21             | 36 ± 16             | 80⁺               |
| 21 | 1-butene | 68 ± 36             | 61 ± 27             | 130⁺              |
| 22 | 2,3-Butadiene | 26 ± 15            | 26 ± 11             | 40⁰               |
| 23 | 1,3-butadiene | 4.6 ± 1.8          | 2.2 ± 1.5           |                  |
| 24 | 1,3-Butadiene | 19 ± 11            | 20 ± 8.9            | 50⁰               |
| 25 | 1-pentene | 19 ± 15             | 16 ± 10             | 8⁰                |
| 26 | Isoprene | 55 ± 18             | 49 ± 11             | 50⁰               |
| 27 | 2,3-Pentene | 17 ± 9.5            | 14 ± 7.1            |                  |
| 28 | 2,4-Pentene | 14 ± 9.8            | 16 ± 5.7            |                  |
| 29 | 3-Methyl-1-butene | 0.75 ± 0.52       | 10 ± 11             |                  |
| 30 | 2-Methyl-1-butene | 13 ± 9.1           | 11 ± 5.6            |                  |
| 31 | 2-Methyl-2-butene | 6.1 ± 3.2          | 4.7 ± 2.3           |                  |
| 32 | Aromatics | 590 ± 230          | 550 ± 170           |                  |
| 33 | Benzene  | 310 ± 82            | 190 ± 100           | 140⁺, 6–301⁴     |
| 34 | Toluene  | 140 ± 63            | 130 ± 59            | 26⁺, 2–371⁴      |
| 35 | Ethylbenzene | 22 ± 10            | 20 ± 8.5            | 30⁰               |
| 36 | m, p-xylene | 49 ± 28            | 55 ± 17             |                  |
| 37 | o-xylene | 18 ± 9.5            | 22 ± 5.6            |                  |
| 38 | i-Propylbenzene | 2.5 ± 1.4          | 2.4 ± 1.2           |                  |
| 39 | n-Propylbenzene | 5.2 ± 4.1          | 8.6 ± 2.0           |                  |
| 40 | 1,3,5-Trimethylbenzene | 8.6 ± 5.1        | 15 ± 3.6            |                  |
| 41 | 1,2,4-Trimethylbenzene | 14 ± 8.7          | 44 ± 23             |                  |
| 42 | 1,2,3-Trimethylbenzene | 8.5 ± 5.1          | 21 ± 8.3            |                  |
| 43 | m-Ethyltoluene | 10 ± 6.2            | 22 ± 7.3            |                  |
| 44 | p-Ethyltoluene | 5.6 ± 3.4          | 15 ± 5.1            |                  |
| 45 | o-Ethyltoluene | 3.2 ± 2.0          | 8.7 ± 2.9           |                  |
| 46 | Phenol   | 5.4 ± 4.1           | 8.6 ± 2.0           |                  |
| 47 | Anisole  | 5.4 ± 3.1           | 8.6 ± 2.0           |                  |
| 48 | Aldehydes | 15 ± 10             | 24 ± 8.5            |                  |
| 49 | Benzaldehyde | 5.6 ± 3.1          | 8.6 ± 2.0           |                  |
| 50 | Phenol   | 5.4 ± 4.1           | 8.6 ± 2.0           |                  |
| 51 | Anisole  | 5.4 ± 3.1           | 8.6 ± 2.0           |                  |
| 52 | Aldehydes | 15 ± 10             | 24 ± 8.5            |                  |

*Dry fuel mass basis. The results are given as means ± one standard deviations (x ± 1σ). Reference [13]. Less than background level. Reference [9].
agricultural residues burned in wind tunnel varied about two orders of magnitude [9]. Our values fall with their ranges.

The emission factors showed high run-to-run variations. The three repeated tests give 26–173% coefficient of variation (CV, the standard deviation divided by the mean) for wheat straw and 20–173% CV for maize stover. The high CVs could be associated with the variations of burning conditions and errors from the sampling and analysis. The errors appeared to be large for those species with low concentrations. High run-to-run variations were also found in VOC measurements from biomass burning in cookstoves in China [23, 24]. We will conduct systematic measurements in the future and investigate possible sources of the variability in hopes of reducing errors.

3.2. Source profiles of NMHCs

Source profiles of 52 NMHC species from two types of agricultural wastes burning in the field are given in table 2. The data are expressed as percentage of the mass concentration of one species relative to the sum of all quantified NMHCs.

The result demonstrates that NMHC profiles for wheat straw and maize stover are very similar. The primary NMHC species are propane, n-pentane, 2,3-dimethylbutane, 2-methylpentane, propene, benzene, and toluene, accounting for 55.3%–68.0% of total NMHCs. Of total NMHC from wheat straw burning, alkanes are the biggest class, accounting for 42.1%, followed by aromatics (34.6%) and alkenes (23.3%). The proportions of alkanes, alkenes and aromatics from maize stover are 33.8%, 31.5% and 34.7%, respectively.

There is little information available on VOC source profiles for field burning of agricultural residues. Considering the similarity between agricultural residues burned in the field and burned as fuel in residential stoves, we compared our results with previous studies on agricultural residues burned in residential stoves. Both Tsai et al and Liu et al observed that benzene, ethene, ethylene, acetylene, propane, propene, and propene were the main VOC species from biofuel combustion in typical Chinese residential stoves [24, 25]. Unfortunately, our analysis did not measure C2 hydrocarbons such as ethene, ethylene, and acetylene. Other than C2 hydrocarbons, the main VOC species in the cited studies agreed with our results.

The benzene to toluene ratio (B/T) was often used to identify VOC sources. Higher B/T ratios (> 1) were thought to be associated with biomass burning, coal burning and industrial emissions, while B/T ratios of approximately 0.5 are reported to characterize vehicular emissions [13, 25–27]. B/T ratios in our study are 1.9 ± 0.62.

3.3. Assessment of the ozone forming potential of NMHC emissions

The large amounts of NMHCs generated from the field burning of agricultural residues are important precursors of ozone (O₃). Evidence indicates that the field burning of agricultural residues significantly affects ground-level O₃ concentrations [7, 28]. Open burning of wheat straw in North China in June 2006 caused the O₃ concentration in Beijing to rise to levels of up to 187 ppb [7], significantly

### Table 2. Source profiles of NMHCs from open burning (wt%).

| No | Species                  | Wheat straw (n = 3) | Maize stover (n = 3) |
|----|--------------------------|---------------------|----------------------|
| 1  | n-Butane                 | 0.83 ± 0.72         | 0.73 ± 0.81          |
| 2  | 2-Methylpropene          | 0.14 ± 0.13         | 0.92 ± 0.87          |
| 3  | Cyclopentene             | 0.13 ± 0.22         | 0.02 ± 0.02          |
| 4  | n-Pentane                | 10.39 ± 15.96       | 4.26 ± 4.6           |
| 5  | 2-Methylpentane          | 0.38 ± 0.48         | 0.22 ± 0.38          |
| 6  | Cyclohexane              | 0.12 ± 0.14         | 0.4 ± 0.55           |
| 7  | n-Heptane                | 0.26 ± 0.14         | 0.56 ± 0.28          |
| 8  | n-Octane                 | 0.21 ± 0.07         | 0.62 ± 0.37          |
| 9  | n-Nonane                 | 0.13 ± 0.13         | 0.57 ± 0.27          |
| 10 | 2,3-Dimethylbutane       | 9.05 ± 2.93         | 3.11 ± 4.49          |
| 11 | 2-Methylbutane           | 0.04 ± 0.06         | 0.02 ± 0.02          |
| 12 | 2,3-Dimethylpentane      | 9.53 ± 2.84         | 2.81 ± 4.28          |
| 13 | 2,3,4-Trimethylpentane   | 0.13 ± 0.07         | 0.17 ± 0.13          |
| 14 | Methylcyclopentane       | 0.17 ± 0.13         | 0.32 ± 0.25          |
| 15 | 2-Methylcyclopentane     | 0.05 ± 0.05         | 0.21 ± 0.17          |
| 16 | 1-Methylcyclopentane     | 0.05 ± 0.04         | 0.35 ± 0.37          |
| 17 | 2,3,4-Trimethylpentane   | 0.20 ± 0.08         | 0.25 ± 0.31          |
| 18 | 2-Methylpentane          | 0.21 ± 0.14         | 0.15 ± 0.16          |
| 19 | 2-Methylheptane          | 0.07 ± 0.07         | 0.19 ± 0.15          |
| 20 | 2-Methylhexane           | 0.17 ± 0.21         | 0.36 ± 0.29          |
| 21 | Alkenes                  | 23.3 ± 3.4          | 31.5 ± 5.2           |
| 22 | Propene                  | 6.85 ± 10.0         | 14.66 ± 6.88         |
| 23 | i-Butene                 | 2.26 ± 0.50         | 2.21 ± 0.39          |
| 24 | 1-Butene                 | 3.81 ± 0.85         | 3.73 ± 0.65          |
| 25 | cis-2-Butene             | 1.44 ± 0.35         | 1.59 ± 0.30          |
| 26 | 1-Octene                 | 1.02 ± 0.47         | 0.95 ± 0.32          |
| 27 | Isobutene                | 3.25 ± 0.41         | 3.08 ± 0.27          |
| 28 | 2-Methylpentane          | 0.93 ± 0.22         | 0.83 ± 0.21          |
| 29 | 3-Methyl-1-butenes       | 0.58 ± 0.12         | 0.49 ± 0.09          |
| 30 | 2-Methyl-1-butenes       | 0.70 ± 0.27         | 0.68 ± 0.15          |
| 31 | 2-Methyl-2-butenes       | 0.34 ± 0.07         | 0.28 ± 0.07          |
| 32 | 2-Methylcyclopentane     | 36.4 ± 2.9          | 34.7 ± 1.2           |
| 33 | Benzene                  | 18.46 ± 2.29        | 11.37 ± 3.77         |
| 34 | Toluene                  | 7.97 ± 1.07         | 7.96 ± 1.66          |
| 35 | Ethylbenzene             | 1.23 ± 0.20         | 1.20 ± 0.20          |
| 36 | m, p-xylene              | 2.71 ± 0.73         | 3.44 ± 0.12          |
| 37 | o-xylene                 | 1.02 ± 0.22         | 1.37 ± 0.02          |
| 38 | 1-Propylbenzene          | 0.14 ± 0.04         | 0.14 ± 0.04          |
| 39 | n-Propylbenzene          | 0.14 ± 0.04         | 0.14 ± 0.04          |
| 40 | 1,3,5-Trimethylbenzene   | 0.48 ± 0.13         | 0.97 ± 0.42          |
| 41 | 1,2,4-Trimethylbenzene   | 0.77 ± 0.23         | 3.09 ± 2.25          |
| 42 | 1,2,3-Trimethylbenzene   | 0.47 ± 0.14         | 1.47 ± 0.88          |
| 43 | m-Ethyltoluene           | 0.56 ± 0.17         | 1.47 ± 0.79          |
| 44 | p-Ethyltoluene           | 0.31 ± 0.09         | 1.03 ± 0.57          |
| 45 | o-Ethyltoluene           | 0.18 ± 0.05         | 0.59 ± 0.33          |

a The results are given as means ± one standard deviations (x ± 1σ). b Less than background level.
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figure 1. spatial distribution of fire points from the open burning of agricultural residues in china in 2007 based on modis fire products. (fire points are shown in black.)

exceeding china’s national ambient air quality standards for ozone (160 mg N\(^{-1}\) m\(^{-3}\), about 75 ppb) [29].

different nmhc species react at different rates and with different reaction mechanisms of ozone formation. in other words, the nmhaps have significantly different effects on ozone formation. maximum incremental reactivity (mir) was introduced to quantify ozone impacts of vocs [29]. for example, saturated organic compounds such as alkanes have lower mir values than alkenes and aromatics that are more active in term of ozone formation. we estimate ozone formation potentials (ofps) using measured emission factors and mir values developed by carter [30]. we realize that ofp results are rough because the mir values used in our estimation are derived from the united states condition, which may be significantly different the chinese condition. however, the oft results still provide useful information for comparing various emission sources or ranking different nmhc species in term of o\(_3\) production. the average offps for total nmhaps are 5.9 ± 2.7 g O\(_3\) kg\(^{-1}\) of dry fuel and 7.0 ± 0.85 g O\(_3\) kg\(^{-1}\) of dry fuel for wheat straw and maize stover, respectively. we compare them with the published offps from various emissions. chang et al reported vehicular offps for total 56 nmhaps are 0.97 g O\(_3\) km\(^{-1}\) vehicle for liquefied petroleum gas-powered vehicles and 1.83 g O\(_3\) km\(^{-1}\) vehicle for gasoline-powered vehicles [31]. burning 1 kg agricultural residues in the field would produce ofp equivalent to driving a liquefied petroleum gas-powered vehicle for about 6.6 km, and a gasoline-powered vehicles for about 3.5 km, respectively. tsai et al estimated offps for total 44 nmhaps ranging from 9 mg to 11 g O\(_3\) kg\(^{-1}\) of dry fuel from 16 fuel–stove combinations tested [24]. burning 1 kg agricultural residues in the field would produce ofp equivalent to burning about 3.8 kg solid biomass fuel in chinese cookstoves. our estimated results indicate that propene, 1-butene, isoprene, toluene, and m, p-xylene have high ofp values, accounting for about 50% of total offps for 52 nmhaps in this study. alkenes played the most important role in ozone formation, accounting for 61.5% for wheat straw and 62.1% for maize stover, followed by aromatics (25.2% for wheat straw and 30.3% for maize stover) and alkanes (13.3% for wheat straw and 7.6% for maize stover). biomass combustion yields high emission levels of ethylene—a species with a high mir value [13, 30]. if ethylene was quantified in this study, it might be near the top of high ofp species. moreover, alkenes effect on ozone formation would be more prominent.

3.4. discussions

vocs emissions from field burning of agricultural residues, contributing 4.8–5.5% to the total national emissions [10, 32], do not play a significant role on a national scale. however, on a regional scale and during the burning season, vocs emissions from agricultural residues burning in the field may have a considerable impact on the local and regional air quality.

fire-count data from satellite observation provide a useful proxy for spatial and temporal variability of biomass burning [33]. based on the moderate resolution imaging spectroradiometer (modis) fire product, the chinese environmental satellite center analyzed the temporal and spatial distribution of fire points from open burning of agricultural residues in china [34, 35]. figure 1 demonstrates spatial distribution of fire points from open burning of agricultural residues in china in 2007. open burning of agricultural residues appeared in most regions, except tibet.
Autonomous Region and Qinghai province. Open burning of agricultural residues were mainly located in Jiangsu, Anhui, Henan, Hebei and Shandong provinces. It was estimated that the burned areas during 2002–2005 covered 5.6–17.0 thousand km². In North China, the open burning of agricultural residues (mainly wheat straw and corn stover) mainly occurs in Summer and Autumn. In South China, the open burning of agricultural residues (mainly rice straw and sugar-cane stalk) mainly occurs in Spring and Winter. June, the main season for wheat harvesting, is the period when open burning of agricultural residues happens most seriously [34, 35].

Previous studies pointed out that wheat straw field burning in June, the most typical biomass burning practice around Beijing in each year, has a serious impact on air quality over Beijing [6, 7, 34–37]. In 2005, based on MODIS fire images and backward air mass trajectories, we concluded that air quality in Beijing from 19 to 22 June was affected by wheat straw field burning. Fire points around the Beijing area and backward trajectories on 19 June 2005 are given in figures 2 and 3, respectively.

4. Conclusions

The research team conducted field measurements to determine NMHCs emissions from two major types of agricultural residues (e.g., wheat straw and maize stover) in China.

The resulting emission factors of 52 NMHC species are presented. The emission factors of total NMHCs from wheat straw and maize stover are 1690 ± 580 mg kg⁻¹ and 1590 ± 430 mg kg⁻¹, respectively. By category, the emission factors of alkanes, alkenes and aromatics from wheat straw are 690 ± 170 mg kg⁻¹, 410 ± 190 mg kg⁻¹ and 590 ± 230 mg kg⁻¹, respectively. From maize stover, they are 550 ± 220 mg kg⁻¹, 490 ± 54 mg kg⁻¹ and 550 ± 170 mg kg⁻¹, respectively.

The analysis of NMHC chemical composition indicated that propane, n-pentane, 2,3-dimethylbutane, 2-methylpentane, propene, benzene, toluene are the primary species, together accounting for 55.3% to 68.0% of the total NMHCs. For wheat straw, alkanes are the biggest category among all NMHCs from wheat straw burning, accounting for 42.1%, followed by aromatics (34.6%) and alkenes (23.3%). For maize stover, the proportions of alkanes, alkenes and aromatics are very close—33.8%, 31.5% and 34.7%, respectively.

The estimated OFPs indicate that propene, 1-butene, isoprene, toluene, m, p-xylene are species with high OPT values, accounting for about 50% of total OFPs for the 52 quantified NMHCs quantified in this study. Alkenes played the most important role in ozone formation, accounting for over 60%, followed by aromatics. Alkanes contributed least to ozone formation.

Rice straw, another major agricultural residue, is also often burned in the field after harvesting in southern China, thus more work is needed to investigate its emissions.

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