Significant source of secondary aerosol: formation from gasoline evaporative emissions in the presence of SO_2 and NH_3

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Abstract

Gasoline evaporative emissions have become an important anthropogenic source of urban atmospheric volatile organic compounds (VOCs) and secondary organic aerosol (SOA). These emissions have a significant impact on regional air quality, especially in China where car ownership is growing rapidly. However, the contribution of evaporative emissions on the secondary aerosol (SA) is not clear in air pollution complex in which high concentration of SO_2 and NH_3 was present. In this study, the effects of SO_2 and NH_3 on SA formation from unburned gasoline vapors were investigated in a 30 m^3 indoor smog chamber. It was found that increase in SO_2 and NH_3 concentrations (0−151 ppb and 0−200 ppb, respectively) could promote linearly the formation of SA, which could be enhanced by a factor of 1.6−2.6 and 2.0−2.5, respectively. Sulfate was most sensitive to the SO_2 concentration, followed by organic aerosol, which was due not only to the acid catalytic effect, but also related to the formation of organic sulfur-containing compounds. In the case of increasing NH_3 concentration, ammonium nitrate increased more significantly than organic aerosol, and nitrogen-containing organics were also enhanced, as revealed by the results of positive matrix factorization (PMF) analysis. New particle formation (NPF) and particle size growth were also significantly enhanced in the presence of SO_2 and NH_3. This work indicates that gasoline evaporative emissions will be a significant source of SA, especially in the presence of high concentrations of SO_2 and NH_3. Meanwhile, these emissions might also be a potential source of sulfur- and nitrogen-containing organics. Our work provides a scientific basis for the synergistic emission reduction of secondary aerosol precursors, including NO_x, SO_2, NH_3 and particularly VOCs, to mitigate PM pollution in China.

Keywords

36 Secondary inorganic aerosol; Secondary organic aerosol; Sulfur dioxide; Ammonia; Sulfur-
37 containing organics; Nitrogen-containing organics

1 Introduction

Many areas in China such as the Beijing - Tianjin - Hebei region (BTH), Yangtze River Delta (YRD), Sichuan Basin and Pearl River Delta (PRD) are suffering from severe haze events (Li et al., 2017; Sun et al., 2016; Shen et al., 2015; He et al., 2014; Huang et al., 2014; Guo et al., 2014; Tan et al., 2009). Haze pollution has attracted widespread attention in recent years because of its adverse effects on human health, climate change and visibility (Thalman et al., 2017; Davidson et al., 2005; Pöschl, 2005).

During the haze events, high concentrations of SO_2, NH_3, and volatile organic compounds (VOCs) have always been observed (Zou et al., 2015; Liu et al., 2013; Meng et al., 2011; Yang et al., 2009), which are the precursors of secondary aerosol. Although the emission of SO_2 has decreased continuously since 2005 (Lu et al., 2010), China is still the largest contributor of SO_2 emissions in the world, mainly owing to the great demand for coal combustion (Bauduin et al., 2016). Also, high concentrations of SO_2 of more than 100 ppb (parts per billion) have been observed in northern China, especially during the heating period (Hou et al., 2016; Tong et al., 2016; Yang et al., 2009). As for atmospheric NH_3, as an alkaline inorganic gas, its main emission source is agricultural practices in China (Zhang et al., 2018; Fu et al., 2015). Vehicles equipped with three-way catalytic converters also contributes to NH_3 emission in the urban areas (Sun et al., 2017). Sometimes, high concentrations of NH_3 of up to 100 ppb have been observed in Beijing, China (Ianniello et al., 2010), which mainly derived from the regionally transportation of agricultural activity and fertilizer use, while could not exclude the influence by traffic emissions at local Beijing (Pan et al., 2016; Kang et al., 2016). With respect to VOCs, aromatics from anthropogenic sources (especially vehicle-related sources in urban areas) are critical secondary organic aerosol (SOA) precursors (Liu et al., 2015a; Gordon et al., 2014; Platt et al., 2013; Calvert et al., 2002). These aromatics could react with oxidants (e.g., O_3, OH, and NO_3 radicals), and undergo multi-step oxidative processes to form multifunctional products, which have

sufficiently low volatility to contribute to SOA via gas-particle partitioning (Hallquist et al., 2009; Atkinson and Arey, 2003).

Researches have shown that secondary aerosol (SA) makes a significant contribution (30−77%) to $PM_{2.5}$ (particles with diameter less than 2.5 μm) during the severe haze events in China (Huang et al., 2014; Guo et al., 2014; Jimenez et al., 2009). However, there still exists a significant gap between the predicted SA derived from the current atmospheric quality models and that observed in field observations (Zhao et al., 2018; Yang et al., 2018; Zheng et al., 2015). Therefore, considering the characteristics of complex pollution in China, it is crucial to study the synergistic effects of SO_2 and NH_3 on the formation of SA, which have been considered an important potential source of SA formation (Zhao et al., 2018; Chu et al., 2016; Liu et al., 2016; Santiago et al., 2012; Na et al., 2007).

A few studies have focused on the influence of SO_2 or NH_3 on SA formation. Jang and Kamens (2001) first reported the acid-catalytical effect of acidic H_2SO_4 on the oxidation of atmospheric carbonyls. And the promotion effect of SO_2 were further found on the SA formation from typical biogenic (e.g., isoprene and α-pinene) (Lin et al., 2013; Jaoui et al., 2008; Kleindienst et al., 2006; Edney et al., 2005) and anthropogenic (e.g., toluene, o-xylene, 1,3,5-trimethylbenzene, and gasoline vehicle exhaust) precursors (Chu et al., 2016; Liu et al., 2016; Santiago et al., 2012) through acid-catalyzed heterogeneous reactions (Jang et al., 2002; Jang et al., 2003a, b; Czoschke et al., 2003), which promote the reactive uptake process of organic species or enhance the formation of high-molecular-weight compounds (Liggio and Li, 2008; Liggio et al., 2007; Liggio and Li, 2006). With regard to the role of NH_3 in SA formation, knowledge is still limited. In previous studies, inconsistent impacts of NH_3 on SA formation have been reported under different precursor systems. For example, NH_3 could elevate SA formation in the α-pinene/ozone oxidation system through acid-base reactions (Na et al., 2007), while the effects of NH_3 neutralization were masked by other multiple factors

and did not show significant influence on isoprene-derived SOA formation (Lin et al., 2013), and addition of NH_3 even significantly reduced the SA formation in the styrene/ozone system, which was caused by nucleophilic attack from the NH_3 molecule leading to rapid decomposition of the major aerosol products (Na et al., 2006). For the photo-oxidation of aromatic VOCs (e.g., toluene, o-/m-/p-xylene), the presence of NH_3 could facilitate new particle formation (NPF) and particle growth, subsequently leading to increased SA formation (Li et al., 2018; Liu et al., 2015b).

At the present time, the effects of SO_2 and NH_3 on SA formation have rarely been studied under highly complex pollution conditions (Chu et al., 2016). Vehicular evaporative emissions have been reported to be non-negligible contributors (39.20 %) to ambient VOCs from anthropogenic sources compared with vehicular tailpipe emissions (Liu et al., 2017a). In addition to short-chain alkanes, a certain proportion of aromatics and alkanes (C6 to C12) were also contained in the evaporative emissions (Liu et al., 2008; Zhang et al., 2013). Previous studies have reported that aromatics and long-chain (C6 to C19) alkanes, which are intermediate volatility organic compounds (IVOCs) (Donahue et al., 2006), could contribute to SOA formation (Pye and Pouliot, 2012; Tkacik et al., 2012; Lim and Ziemann, 2005). Therefore, it is necessary to study the influence of SO_2 and NH_3 on SA formation from evaporative emissions.

In this study, unburned gasoline vapors were used as a substitute for evaporative emissions, and the roles of SO_2 and NH_3 on SA formation from the photo-oxidation of unburned gasoline vapors were investigated in a 30 m^3 indoor smog chamber, in order to understand the formation potential of SA from oxidation of gasoline vapor in the cocktail of pollutants in Beijing. The respective influences of SO_2 and NH_3 on both the microphysics and chemistry of SA formation were examined. Meanwhile, the chemical compositions of the formed SOA in the presence of SO_2 and NH_3 were further explored by applying positive matrix factorization (PMF) analysis. The formation potentials of SA, sulfur- and nitrogen-containing

organics from vehicular evaporative emissions in the presence of SO_2 and NH_3 were evaluated and discussed.

2 Materials and Methods

2.1 Gasoline fuel

The utilized gasoline fuel with grade 92# was collected (refer to the standard Method for manual sampling of petroleum liquids (GB/T 4756-2015)) from a gas station located in Beijing. The gasoline complies with the China V gasoline fuel standard. It contains 65.1 % (v/v) alkanes (C6 to C12), 22.8 % (v/v) aromatics (mainly including benzene, toluene, xylene, trimethylbenzene) and 12.1 % (v/v) alkenes. The composition of the gasoline is similar to the gasoline collected in North China reported by Tang et al. (2015) and could represent the gasoline used in most areas of China for studying SA formation potential. Details of the gasoline composition are given in Table S1.

2.2 Smog chamber facility

A series of photochemical experiments with unburned gasoline vapors in the absence or presence of SO_2 or NH_3 were performed in a 30 m^3 indoor smog chamber at the Research Center for Eco-Environmental Sciences, Chinese Academy of Sciences (RCEES-CAS). The detailed schematic structure of the indoor smog chamber is given in Fig. S1 in the Supplement and described elsewhere (Chen et al., 2019a, b). Briefly, the cuboid chamber reactor (L × W × H = 3.0 × 2.5 × 4.0 m, S/V = 1.97 m^{-1}) was irradiated by 120 UV lamps (Philips) with peak intensity at 365 nm, providing a NO_2 photolysis rate of 0.55 min^{-1}. The interior was coated with 125 μm-thick FEP100 film (DuPont™, US) and the chamber was located in a temperature-controlled room, in which the temperature (T) and relative humidity (RH) could be controlled mechanically. A three-wing stainless-steel fan coated with Teflon was installed inside the reactor to guarantee that the gas and particle phase species mix sufficiently before photochemical reaction.

The chamber was also equipped with a series of gas- and particle-phase monitoring instruments. For

126 gaseous NO_x, O_3 and SO_2, a chemiluminescence analyzer (Model 42i-TL, Thermo Fisher Scientific, USA),
127 a UV photometric analyzer (Model 49i, Thermo Fisher Scientific, USA) and a pulsed fluorescence analyzer
128 (Model 43i, Thermo Fisher Scientific, USA) were used to monitor the concentrations in real time,
129 respectively. The VOC species in gasoline were measured with a gas chromatograph (7890B GC, Agilent,
130 USA) equipped with a DB-624 column (60 m × 0.25 mm × 1.40 μm, Agilent, USA) and a mass spectrometry
131 detector (5977A MS, Agilent, USA) (GC-MS). In addition, proton-transfer-reaction time of flight mass
132 spectrometry (PTR-TOF) (Ionicon Analytik GmbH, Austria) was also used for the measurement of gas-
133 phase hydrocarbons and their intermediate products (Yuan et al., 2017). The size distribution and number
134 concentration of the formed particulate matter (PM) were measured using a scanning mobility particle sizer
135 (SMPS, TSI, USA), which was composed of a differential mobility analyzer (DMA, 3080 Classifier, TSI,
136 USA) coupled with a condensation particle counter (CPC, 3776, TSI, USA). The mass concentration was
137 estimated based on the volume concentration and the density of PM calculated from the equation $\rho = d_{va}/d_m$,
138 where d_{va} is the mean vacuum aerodynamic diameter measured by an Aerodyne high-resolution time-of-
139 flight aerosol mass spectrometer (HR-ToF-AMS) and d_m is the mean electrical mobility diameter measured
140 by SMPS (DeCarlo et al., 2004). The calculated density of PM ranged from 1.5 to 1.6 g cm^{-3} in the different
141 reaction systems, which was in the range of density of SOA derived from aromatic hydrocarbons (1.24−1.48
142 g cm^{-3}) (Sato et al., 2010) and ammonium nitrate (NH_4NO_3, 1.72 g cm^{-3}) (Bahreini et al., 2005) and could
143 be comparable with the previous studies (Li et al., 2018). The mass concentration and chemical composition
144 of PM were simultaneously monitored using a high-resolution time-of-flight aerosol mass spectrometer (HR-
145 ToF-AMS, Aerodyne Research Inc. USA). For all experiments, the HR-ToF-AMS operated in a cycle
146 including two modes, 3 min V mode and 2 min W mode. Specifically, V mode (higher signal) can obtain the
147 mass concentrations of the aerosols and W mode (higher resolution) can obtain high resolution mass spectral

data. The inlet flow rate, ionization efficiency (IE), and particle sizing were calibrated according to the standard protocols (Drewnick et al., 2005; Jimenez et al., 2003; Jayne et al., 2000), using the size-selected pure ammonium nitrate (AN) particles. All HR-ToF-AMS data were analyzed with ToF-AMS analysis toolkit SQUIRREL 1.57I/PIKA 1.16I version, in Igor Pro Version 6.37. HR-ToF-AMS results were also corrected using the mass concentration derived from SMPS according to the same method as Gordon et al. (2014), details of this correction are shown in the Supplement. As for the RH control system, it is achieved by vaporizing Milli-Q ultrapure water contained in a 5.0 L high pressure resistant container and the water vapor is flushed with purified dry zero air into the chamber. T and RH were monitored real-time using a hydro-thermometer (Vaisala HMP110) during the entirety of each experiment.

2.3 Wall loss corrections

The measured particle concentration was corrected in accordance with the relationship between the deposition rate (k_{dep}) and particle diameter (D_p, nm) (i.e., $k_{dep} = 4.15 \times 10^{-7} \times D_p^{1.89} + 1.39 \times D_p^{-0.88}$), which was described by Takekawa et al. (2003). The wall loss rates of NO_2, NO, O_3, SO_2 and VOC species were determined to be $(1.67 \pm 0.25) \times 10^{-4}$, $(1.32 \pm 0.32) \times 10^{-4}$, $(3.32 \pm 0.21) \times 10^{-4}$, $(4.52 \pm 0.11) \times 10^{-4}$ and $(2.20 \pm 0.39) \times 10^{-4}$ min^{-1}, respectively. Therefore, the wall loss of gas phase species was evaluated to be less than 5% of their maximum concentration in this study.

Wall losses of semi-volatile organic compounds (SVOCs) and low-volatility organic compounds (LVOCs) would lead to a substantial underestimation of SA formation (Krechmer et al., 2016; Ye et al., 2016; Zhang et al., 2015; Zhang et al., 2014), which is caused by the competition between these vapors condensing onto particles versus onto chamber walls. This competition could be evaluated by the corresponding timescales associated with reaching gas-to-particle partitioning equilibrium ($\bar{\tau}_{g\text{-}p}$) and vapor wall loss ($\tau_{g\text{-}w}$) (Zhang et al., 2014), and this underestimation of SA formation could be approximately quantified by the

ratio of these two timescales (i.e., $\bar{\tau}_{g\text{-}p}/\tau_{g\text{-}w}$). According to the methods described by Zhang et al. (2014), $\bar{\tau}_{g\text{-}p}$ and $\tau_{g\text{-}w}$ could be estimated assuming an upper bound and a lower bound of the molecular mass of organic vapors (MW) (100−300 g mol^{-1}) (as discussed in the Supplement). In order to accurately quantify the SA formation, the underestimation caused by the loss of SVOCs and LVOCs (include sulfuric acid gas) to the chamber walls was taken into account in this study. In this study, the SA yields were underestimated by a factor of 1.97−2.82 fold when considering the ratio of these two timescales (i.e., $\bar{\tau}_{g\text{-}p}/\tau_{g\text{-}w}$), which showed a decreasing trend with the increase of the SO_2 and NH_3 initial concentrations, suggesting that an increasing proportion of vapors is partitioned onto the suspended particle surface rather than the chamber wall.

2.4 Experimental conditions

Prior to each experiment, the chamber reactor was flushed by purified and dry zero air for about 24−36 h at a flow rate of 100 L min^{-1} until almost no gas-phase species (i.e., NO_x, O_3 and SO_2) could be detected (< 1 ppb) and the particle number concentration was < 10 cm^{-3}. Before the experiments, the chamber was humidified to ~50 % RH by passing purified zero air through ultra-pure water (18.2 MΩ, Millipore Milli-Q). After that, a known volume of liquid gasoline (100 μL) was injected into the chamber through a heated Teflon line system (~100 °C) carried by purified dry zero air to ensure that all were evaporated into the chamber. Subsequently, NO, SO_2 or/and NH_3 were successively injected into the chamber from standard gas cylinders using mass flow controllers. The initial VOCs/NO_x ratio (ppbC ppb^{-1}) was kept constant (Table 1). In order to reduce the adsorption of NH_3 in the pipeline, the NH_3 flow in a bypass line was balanced for about 30 min before it was injected into the chamber. The concentrations of NO and SO_2 were continuously monitored until they were stable, ensuring that the gaseous species mixed well in the chamber. For the concentration of NH_3, the value was estimated according to the amount of NH_3 introduced and the volume of the reactor chamber. The experiment was then conducted for about 8 h after turning off the fan and turning

on the UV lights. All the experiments were performed at a temperature of 26 ± 1 °C and wet conditions (RH = 50 ± 3 %). The detailed experimental conditions are listed in Table 1. The letters in the abbreviations represent the reactants introduced into the chamber reactor for each experiment. For example, SGN is an experiment with the presence of sulfur dioxide (S), gasoline vapor (G), and nitrogen oxides (N). Four experiments (Exps. SGN1, SGN2, SGN3, and SGN4) were carried out at different SO_2 initial concentrations. AGN is an experiment with the presence of ammonia (A), gasoline vapor (G), and nitrogen oxides (N). Two experiments (Exps. AGN1 and AGN2) were carried out at different NH_3 initial concentrations.

3 Results and discussion

3.1 Effect of SO_2 and NH_3 on the gas-phase species

Time-resolved concentrations of inorganic and organic gas-phase species during the photo-oxidation of gasoline/NO_x in the absence or presence of SO_2 and NH_3 are shown in Fig. S2 and Fig. S3 in the Supplement, respectively. After turning on the UV lights, NO was rapidly converted to NO_2. At the same time, O_3 was gradually generated, with a maximum concentration of up to 350 ppb (Fig. S2). As shown in Fig. S2, there was no obvious difference in the variation of NO_x and O_3 in the presence of SO_2 or NH_3. Additionally, the decay of typical VOC precursors (e.g., benzene, toluene, methylcyclopentane, methylcyclohexane) measured by PTR-TOF and GC-MS are given in Fig. S3, which traced very closely with each other (Fig. S4, in the Supplement). There were also no observable differences in these precursors VOCs among these experiments. According to the decay curves of aromatic hydrocarbons, the OH radical concentrations were estimated to be (7.54−8.40) × 10^6 molecules cm^{-3}, which were also similar among these experiments. This was consistent with the previous study conducted by Chu et al. (2016), who found that the presence of SO_2 and NH_3 did not significantly impact the OH concentration during the photo-oxidation of toluene in the presence of NO_x.

However, as for the gas-phase intermediates formed during the photo-oxidation of gasoline/NO_x under different conditions, such as small molecule oxygenated VOCs (OVOCs), which could also be measured by PTR-TOF. The time series of OVOCs concentration would vary with the concentration of SO_2 and NH_3. For example, we observed that acetic acid concentration decreased with the increased concentration of SO_2 (Fig. S5, in the Supplement), suggesting that the uptake of acetic acid may be enhanced. This phenomenon was consistent with those reported by Liggio and Li (2006), who observed that the uptake of organic compounds under acidic conditions would be enhanced significantly. Moreover, the presence of high concentrations of SO_2 would generate gaseous H_2SO_4, which would contribute to the formation of particle phase, as discussed in the next section. Similarly, the concentration of acetic acid also shown an obviously decreased trend in the presence of NH_3 (Fig. S5, in the Supplement), which could be caused by the reaction of acid-base reaction or the uptake of acetic acid in the presence of NH_3 (Liu et al., 2015c).

3.2 Role of SO_2 in secondary aerosol formation

To investigate the effects of SO_2 on SA formation from the photo-oxidation of gasoline/NO_x, smog chamber experiments with different SO_2 initial concentrations were carried out (Table 1). As shown in Fig. 1, compared to the experiments without the addition of SO_2, the SA concentration was enhanced to different degrees (1.6−2.6 times) in the presence of different SO_2 concentrations (35−151 ppb, i.e., 100−431 μg m^{-3}). As for each chemical species (i.e., organics, nitrate, sulfate, and ammonium), they all showed a trend of linear increase with the increase of SO_2 concentration (Fig. 2), especially for the sulfate ($k = 8.4\times10^{-2}$) and organic aerosol ($k = 2.9\times10^{-2}$). Previous studies have also revealed its promoting role on SA formation from different precursors (Zhao et al., 2018; Liu et al., 2017b; Díaz-de-Mera et al., 2017; Liu et al., 2016; Chu et al., 2016).

Additionally, the particle number concentrations and size growth were greatly enhanced by the presence

of SO_2. As evident from Fig. 3, the corresponding maximal particle number concentrations (5.82×10^4−1.91×10^5 cm^{-3}) were significantly enhanced by a factor of 2.9−3.3 in the presence of SO_2. This universal phenomenon has been reported by many studies (Díaz-de-Mera et al., 2017; Liu et al., 2017b; Liu et al., 2016; Chu et al., 2016). For example, the maximal particle number concentrations were enhanced by the presence of SO_2 (~130 ppb) to one order of magnitude in the photo-oxidation of high concentration toluene/NO_x (Chu et al., 2016). For complex precursor systems (gasoline vehicle exhaust), Liu et al. (2016) have also found that under high SO_2 concentration (~150 ppb) conditions, the maximum particle number concentrations increased by 5.4−48 times compared to those without SO_2 during the photo-oxidation of gasoline vehicle exhaust. This higher magnification of SO_2 might be related to the different VOCs composition between evaporative emissions and gasoline vehicle exhaust, especially the aromatic and IVOCs (Liu et al., 2017). Our recent study demonstrated that SOA formation could be significantly enhanced by the increase of aromatic content (Chen et al., 2019b). Those unspeciated organic emissions (e.g., IVOCs) from gasoline vehicle exhaust would also have a significant contribution to SOA formation (Jathar et al., 2014; Gordon et al., 2014). Moreover, a small amount of POA was present in the initial reaction systems in Liu et al. (2016). These enhanced SOA formation and the pre-existing POA would provide larger surface areas for the condensation and heterogeneous uptake of low-volatility vapors (e.g., gaseous H_2SO_4), then promoting a higher magnification in particle number concentrations in the presence of SO_2. The higher initial mixing ratios of precursors (2.2−4.3 ppm) was also present in the reaction systems conducted by Liu et al. (2016), which would further be beneficial to the SOA formation. In addition, size distributions of generated SA in smaller size ranges (4−160 nm) were also determined using another SMPS equipped with a nanometer differential mobility analyzer (Nano-DMA), indicating that the new particle formation (NPF) phenomenon was enhanced significantly when the SO_2 concentration increased (Fig. S6). The presence of high

concentrations of SO_2 would generate sulfuric acid (H_2SO_4), which would contribute to nucleation and increase the total particle number concentrations (Zhao et al., 2018; Sipilä et al., 2010). As the SO_2 concentration increased from 35 ppb to 151 ppb, the maximal particle diameters (144−172 nm) became larger, which will have a direct impact on the scattering and absorption of light (Seinfeld and Pandis, 2016). An enhancement effect of SO_2 on the surface area of particles was also observed. As shown in Table 1, the surface area of aerosol particles at the end of each experiment increased from 1.12×10^3 to 2.46×10^3 μm^2 cm^{-3} when the SO_2 concentration increased from 0 to 151 ppb. The larger surface area would be beneficial to the condensation and heterogeneous uptake of low-volatility vapors (Chapleski et al., 2016), consequently leading to higher SA yield in the presence of SO_2 (Table 1) (Santiago et al., 2012). Additionally, it is worth noting that there was a discrepancy between the magnification of particle number concentrations, surface areas and SO_2 concentrations. On one hand, there might be some particles, especially nanoclusters, were lost to the chamber wall and not be detected; on the other hand, the initial size of nanoclusters contributed from gaseous H_2SO_4 was small enough (sub-3 nm) (Chu et al., 2019; Sipilä et al., 2010) and couldn't be detected by our general SMPS. That is to say, the particle number concentrations and surface areas measured by our SMPS might be the particles after growing up by collision. This could be supported by the enhancement in the particle diameters (144−172 nm) and sulfate concentrations (13−38 $\mu g\ m^{-3}$) in the presence of SO_2. After considering the underestimation of particles formation (factor of 1.97−2.82, seen in Section 2.3), the sulfate concentrations will be enhanced by a factor of 5.8 when comparing between experiments SGN 1 and SGN 4.

In order to further investigate the role of SO_2 in the chemistry of SOA formation, the particle acidities were estimated using the E-AIM model (Model II: H^+ - NH_4^+ - SO_4^{2-} - NO_3^- - H_2O) (Clegg and Brimblecombe, 2005; Wexler and Clegg, 2002; Clegg et al., 1998). The concentrations of chemical components (i.e., NH_4^+,

SO_4^{2-}, and NO_3^-) at the time when the SOA formation rate reached its peak were used as the inputs of the model. As shown in Fig. 4, the H^+ concentration was increased from 8.5 to 32.5 nmol m^{-3} with the increase of SO_2 concentration under moderate humidity conditions (RH = 50 %) and the higher SOA concentration and SOA yield could be well explained by the enhancement of the particle acidities (R^2 = 0.960 and R^2 = 0.986, respectively). The higher SOA concentration and SOA yield were related to the acid-catalyzed reactions of multifunctional aldehydes (e.g., glyoxal and methylglyoxal), which were the products of aromatic hydrocarbons in the gasoline vapors through the gas-phase photo-oxidation. Hemiacetals, acetals and alcohols could be generated through the acid-catalyzed heterogeneous reactions of glyoxal (Czoschke et al., 2003; Jang et al., 2002). These low-vapor-pressure products generated from heterogeneous reactions preferentially contribute to the SOA formation (Kroll and Seinfeld, 2008; Cao and Jang, 2007; Casale et al., 2007; Jang et al., 2002).

In addition, the sulfur-containing organics formed in the presence of SO_2 might be another reason for the increase of SOA yield (Kundu et al., 2013; Liggio et al., 2005). Jaoui et al. (2008) have reported that the acidic aerosol generated in the presence of SO_2 could lead to sulfur-incorporating reactions in the particle phase during the photo-oxidation of α-pinene/toluene/NO_x mixtures. Sulfur-containing organics could be generated via reactions of organic species (e.g., polycyclic aromatic hydrocarbons (PAHs), C10−C12 alkanes, alcohols, epoxides) with sulfate, bisulfate or sulfuric acid, especially under high relative humidity and acidity conditions (Riva et al., 2015, 2016; Huang et al., 2015; Hatch et al., 2011; Surratt et al., 2007; Liggio et al., 2005). Huang et al. (2015) have revealed that sulfur-containing organics with R-O-SO_3^- functional groups will yield S-bearing organic fragments ($C_xH_yO_zS$) during ionization, which subsequently could be detected by HR-ToF-AMS and used as marker ions to quantify them. In our gasoline/NO_x experiments in the presence of SO_2, the ions CSO^+, $CH_3SO_2^+$ and $CH_3SO_3^+$ could be separated (Fig. S7), although uncertainty might be

induced in the peak-fitting of the highly abundant ions $C_2H_4O_2^+$, $C_6H_7^+$, and $C_5H_3O_2^+$. These characteristic ions (i.e., CSO^+, $CH_3SO_2^+$ and $CH_3SO_3^+$) also have been observed from sulfur-containing organics in previous field measurements (Huang et al., 2015; Farmer et al., 2010). According to the estimation method for sulfur-containing organics mentioned in Huang et al. (2015), we found that the signal of these ions and the concentrations of sulfur-containing organics increased with the SO_2 initial concentration (Fig. 5). The conservative lower-bound estimated concentrations of sulfur-containing organics (13−26 ng m^{-3}) were comparable to those (~ 20 ng m^{-3}) observed in the mid-Atlantic United States, which were derived from biogenic and anthropogenic hydrocarbons (Meade et al., 2016). Additionally, it should be noted that the sulfur-containing organics concentration in this study might be underestimated by the HR-ToF-AMS when considering one cannot resolve all the sulfur-containing fragments that may exist, and some of the sulfur-containing organics might fragment into masses that do not contain sulfur and thus are quantified as organic. Furthermore, the relative ionization efficiency (RIE) for the sulfur-containing organics fragments was assumed to be equivalent to the remainder of the organics (1.3), since a RIE value for sulfur-containing organics is unknown. This may introduce an additional uncertainty to the quantitation of sulfur-containing organics. Therefore, photo-oxidation of gasoline vapor in the presence of SO_2 might be a noteworthy source of sulfur-containing organics, although the concentration was very low compared to that of generated SO_4^{2-} (~ 0.1% of SO_4^{2-}).

3.3 Role of NH_3 in secondary aerosol formation

Similarly, the role of NH_3 in SA formation was examined. It is worth noting that ammonium aerosols were formed without the addition of gaseous NH_3 (Fig. S8, in the Supplement), which signified that some NH_3 was present in the background air in the chamber or introduced during the humidification process of the chamber (Liu et al., 2015c). Unfortunately, appropriate instruments are unavailable to measure the exact

concentration of background NH_3 in the chamber. According to the concentration of generated ammonium aerosols, the concentration of background NH_3 was estimated to be ~15 ppb using the E-AIM model (Clegg and Brimblecombe, 2005; Wexler and Clegg, 2002; Clegg et al., 1998). Therefore, for the experiments with the presence of NH_3, the concentration of injected NH_3 (150−200 ppb) was much higher than this value to identify the effect of NH_3 on SA formation. The SA concentration was enhanced by a factor of 2.0−2.5 in the presence of NH_3, as shown in Fig. S9a. The formation of SOA, NO_3^- and NH_4^+ was enhanced to varying degrees. The increase of NO_3^- and NH_4^+ could be attributed to the formation of inorganic NH_4NO_3 in the presence of NH_3. The NO^+/NO_2^+ ratio, which could be derived from HR-ToF-AMS, has often been used as a proxy for identification of inorganic nitrate and organic nitrogen compounds (Farmer et al., 2010; Sato et al., 2010; Rollins et al., 2009). Generally, the NO^+/NO_2^+ ratio of inorganic nitrate (1.08−2.81) is lower than that of organic nitrogen compounds (3.82−5.84) (Liu et al., 2016). In this study, the NO^+/NO_2^+ ratio became substantially lower (~ 2.00) in the presence of NH_3 compared with that in the absence of NH_3 (~ 5.46). Therefore, NH_4NO_3 was the dominant nitrate species in the presence of NH_3. As for the reason for SOA enhancement, the presence of NH_3 could react with some organic acids and subsequently contribute to SOA formation (Na et al., 2007; Na et al., 2006), which could be supported by the increase of N/C (from 0.016 to 0.033) with increasing NH_3 concentration at similar concentrations of NO_x. In addition, we have found that the presence of NH_3 readily increased the particle diameter and number concentration of SA generated in the photo-oxidation of gasoline (Figs. S9b and S9c), which revealed that NH_3 played an important role in new particle formation (NPF). These are consistent with the simulation results finding that NH_3 promotes atmospheric NPF and also the conversion of SO_2 and NO_2 (Jiang and Xia, 2017). The increased surface area of particles was also observed (Table 1, 2.07×10^3 and 2.48×10^3 μm^2 cm^{-3}) as the NH_3 concentration increased from 0 to 150 and 200 ppb. Similarly, the larger surface area would favor the partitioning of low-

346 volatility vapors to the particle phase, leading to the higher SA yield (Table 1).

347 Previous studies have reported that the reaction of carbonyl compounds (e.g., glyoxal) could be
348 catalyzed by NH_4^+ ions through a Bronsted acid pathway or an iminium pathway, which could generate N-
349 containing products and oligomers (Nozière et al., 2009), and then contribute a substantial fraction to SOA
350 (Liu et al., 2015c; Farmer et al., 2010; Cheng et al., 2006). Researchers have identified the characteristic
351 fragments of nitrogen-containing organics as $C_xH_yN_n$ and $C_xH_yO_zN_n$ using HR-ToF-AMS (Lee et al., 2013;
352 Farmer et al., 2010; Galloway et al., 2009). In this study, the typical normalized mass spectrum of N-
353 containing fragments in SOA after 480 min of photo-oxidation reaction at different concentrations of NH_3
354 are given in Fig. 6. The prominent peaks in the $C_xH_yN_n$ family were at *m/z* 27 (CHN^+), 30 (CH_4N^+),
355 40($C_2H_2N^+$), 41(CHN_2^+, $C_2H_3N^+$), 42($C_2H_4N^+$), 43($C_2H_5N^+$), 54($C_2H_2N_2^+$, $C_3H_4N^+$), 55($C_3H_5N^+$), and
356 68($C_3H_4N_2^+$, $C_4H_6N^+$); and the $C_xH_yO_zN_n$ fragments were dominated by 45(CH_3ON^+), 46(CH_4ON^+),
357 59($C_2H_5ON^+$), 63($CH_5O_2N^+$), 73($C_2H_5ON_2^+$, $C_3H_7ON^+$), 86($C_3H_4O_2N^+$, $C_3H_6ON_2^+$), 91($C_3H_9O_2N^+$),
358 97($C_4H_5ON_2^+$), and 104($C_3H_6O_3N^+$, $C_4H_{10}O_2N^+$). The N-containing fragments observed in the experiment
359 without added NH_3 could be attributed to the reactions between organic peroxy (RO_2) radicals and NO_x
360 (Arey et al., 2001) or uptake of background NH_3 by SOA. Additionally, it was obvious that the signal
361 intensities of most N-containing fragments became significantly stronger as the NH_3 concentration increased
362 (150−200 ppb). Therefore, a considerable amount of nitrogen-containing organics (the ratio of nitrogen-
363 containing organics to SOA was about 6.7−7.7%) was formed during the photo-oxidation of gasoline vapor
364 in the presence of NH_3. This was consistent with the previous study conducted by Liu et al. (2015c), who
365 observed the formation of organic nitrogen compounds in the SOA generated from the OH oxidation of m-
366 xylene. The promoting role of NH_3 in the formation of N-containing species was also observed in the reaction
367 system of ozonolysis and photo-oxidation of α-pinene (Babar et al., 2017).

In addition, elemental analysis was also carried out to elucidate the SOA chemical composition and SOA formation mechanisms (Chhabra et al., 2011; Heald et al., 2010) at different concentrations of NH_3. The time evolution of H/C and O/C in SOA formed from the photo-oxidation of gasoline vapor at different concentrations of NH_3 is shown in Fig. 7. As evident from Fig. 7, all data points are located in the triangular area for slope between -1 and 0, which suggests that SOA formation from the photo-oxidation of gasoline vapor is a combination of carboxylic acid and alcohol/peroxide (Heald et al., 2010). Moreover, in the presence of NH_3, as shown in Fig. 8, N/C increased as reaction proceeded in the initial oxidation stage (0−120 min), accompanied by a rapid increase of O/C (0.12−0.67), a decrease of H/C (2.12−1.61), and a rapid formation of SOA. During this stage, the photo-oxidation of VOC precursors leads to a rapid increase in O/C and a rapid decrease in H/C. The termination chemistry of NO_x with free radicals and the NH_3 uptake result in a rapid increase in N/C. As the reaction proceeded further (120−300 min), an increase of H/C which should be caused by NH_3 uptake resulted in an almost constant oxidation state of SOA in the continuous photo-oxidation, accompanied by an increase in the SOA concentration. Nozière et al. (2009) have reported that N-containing products would be generated from carbonyl compound (e.g., glyoxal) self-reactions catalyzed by NH_4^+ ions, which will have a dramatic impact on the volatility of oxidation products and the yield of SOA (Ortiz-Montalvo et al., 2014). In the last stage of the reaction (360−480 min), NH_3 uptake might reach saturation; therefore, H/C and N/C are almost constant. Comparing experiments with different concentrations of NH_3, the average H/C shows an obvious increase (1.53−1.70) while the average O/C (0.70−0.78) shows a slight increase with the increase of NH_3 concentration (0−200 ppb), seen in Fig. S10. The slope in the Van Krevelen diagram shows a trend from slope = -1 to slope = 0 (Fig. S10), indicating that the formed carboxylic acid would further react with NH_3 via acid-base reaction to generate an ammonium salt of a carboxylate anion in the presence of NH_3 (Na et al., 2007). Xu et al. (2018) recently found that

390 imidazole products containing multiple oxygen atoms could be generated through heterogeneous reactions
391 between NH_3 and carbonyl compounds (e.g., glyoxal), which might also contribute to the increase in the
392 O/C of the SOA.

393 **3.4 Different roles of SO_2 and NH_3 in SOA chemical properties**

394 The chemical properties of the SOA generated under the different concentration of SO_2 or NH_3 were
395 further compared by applying positive matrix factorization (PMF) analysis to the HR-ToF-AMS data,
396 respectively (Chu et al., 2016; Liu et al., 2014). The details of PMF analysis are given in the Supplement.
397 For the experiments under different SO_2 concentration conditions (i.e., Exps. GN, SGN1, SGN2, SGN3 and
398 SGN4), two factors (Factor 1-S and Factor 2-S, Fig. S11a) were identified from the PMF analysis, and the
399 difference mass spectra (*m/z* 12−170) between the two factors and the time series of the mass concentrations
400 are shown in Fig. 9. The intensity of C_xH_y and S-bearing organic fragments ($C_xH_yO_zS$) in Factor 1-S was
401 obviously stronger than that in Factor 2-S. Meanwhile, fragments in the high *m/z* range (> 110 Da) were
402 more abundant in Factor 1-S (Fig. 9a, marked in red box). By contrast, the fragments containing oxygen in
403 Factor 2-S were more abundant than in Factor 1-S, such as the typical fragment CO_2^+ (*m/z* 44). Therefore,
404 Factor 1-S was tentatively assigned to the less-oxygenated organic aerosol and oligomers, while Factor 2-S
405 was more-oxygenated organic aerosol (Ulbrich et al., 2009). Similarly, for the experiments at different NH_3
406 concentration (i.e., Exps. GN, AGN1 and AGN2), two factors (Factor 1-N and Factor 2-N, Fig. S11b) were
407 also identified in the same way. According to Fig. 10, Factor 1-N was tentatively assigned to the less-
408 oxygenated organic aerosol and oligomers, while Factor 2-N was more-oxygenated organic aerosol and
409 nitrogen-containing organics.

410 As shown in Fig. 9b and Fig. 10b, these two factors both had different time series during the entire
411 reaction. With respect to Exps. GN, SGN1, SGN2, SGN3 and SGN4, Factor 1-S was formed later (~ 30 min)

than Factor 2-S, and then continuously increased during the entire reaction. Comparing experiments with different SO_2 concentrations, the maximum concentration of Factor 1-S, which was related to the less-oxygenated organic aerosol and oligomers, was enhanced with increased SO_2 concentration (R^2 = 0.881, Fig. 9c). This suggested that the presence of SO_2 was prone to decrease the oxidation state of organic aerosol via acid-catalyzed reactions and enhance the formation of oligomers (Liu et al., 2016), which was consistent with the evolution of O/C vs. H/C shown in Fig. S12. Moreover, the gradually increasing concentration of Factor 1-S was related to the formation of sulfur-containing organics in the presence of SO_2 (Blair et al., 2017). By contrast, Factor 2-S was first gradually increased with the progress of the reaction and then decreased after reaching a peak (i.e., inflection point). And the time to reach the inflection point was affected by the SO_2 concentration (Fig. 9b). As the initial concentration of SO_2 increased from 0 ppb to 151 ppb, the time corresponding to the inflection point decreased, which indicated that the adverse influence of acid catalysis on Factor 2-S was gradually enhanced. In addition, the maximum concentration of Factor 2-S was negatively related with SO_2 concentration (R^2 = 0.987, Fig. 9c); this suggested that the presence of SO_2 and acid catalysis was adverse to the formation of more-oxygenated organic aerosol, leading to the decrease of the oxidation state of organic aerosol (Fig. S12).

By contrast, for Exps. GN, AGN1 and AGN2, Factor 1-N was first increased with the progress of the reaction and then gradually decreased after reaching a peak (Fig. 10b); while Factor 2-N was formed later (~ 30 min) than Factor 1-N, and then continuously increased during the entire reaction. This phenomenon was consistent with the expected behavior, that less-oxidized organic aerosol would be further oxidized to form more-oxidized organic aerosol. When comparing experiments with different NH_3 concentrations, it was observed that the concentration of Factor 2-N increased with increasing NH_3 concentration. Meanwhile, Factor 2-N, which was related to the more-oxidized organic aerosol and nitrogen-containing organics, was

a dominant factor in the presence of NH_3, and its maximum concentration was enhanced with the increase in NH_3 concentration (R^2 = 0.988, Fig. 10c). Thence, the formation of more-oxygenated organic aerosol and nitrogen-containing organics will be enhanced with the increase of NH_3 concentration. In contrast, a negative correlation was observed between the maximum concentration of Factor 1-N and NH_3 concentration (R^2 = 0.876, Fig. 10c); this revealed that less-oxygenated organic aerosol was gradually transformed to more oxidized species and nitrogen-containing organics in the presence of NH_3.

4 Conclusions

In this study, SA formation from the photo-oxidation of gasoline/NO_x in the presence of SO_2 or NH_3 was investigated. Our experimental results demonstrated that SA was enhanced by a factor of 1.6−2.6 or 2.0−2.5, respectively, with the increase of SO_2 or NH_3 concentration (0−151 ppb and 0−200 ppb, respectively). Meanwhile, both secondary organic aerosol (SOA) and secondary inorganic aerosol (SIA) were increased by varying degrees. In the presence of SO_2, SO_4^{2-} was the most sensitive linear increase with the increase of SO_2 concentration, and SOA was also greatly enhanced due to the acid catalytic effect and the formation of sulfur-containing organics. In the presence of NH_3, NH_4NO_3 was most enhanced, following by SOA. The formation of nitrogen-containing organics was also promoted by the presence of NH_3. Meanwhile, conspicuous new particle formation (NPF) and particle size growth were enhanced in the presence of SO_2 or NH_3.

In this study, a linear relationship between the SA yield and SO_2 or NH_3 concentration was also obtained (Fig. S13). Considering the typical concentrations of SO_2 and NH_3 of 40 ppb and 23 ppb in haze pollution in the north China plain (Cheng et al., 2016), and the lower aromatics content (~ 10%) in vehicular evaporative emissions (Zhang et al., 2013), the SA yield is roughly estimated to be about 0.20. Recently, an updated emission inventory of vehicular evaporative emissions was reported to be 1.65 Tg yr^{-1} (Liu et al.,

2017a). Then, the SA formed from the photo-oxidation of VOCs emitted by vehicular evaporation in the presence of SO_2 and NH_3 is roughly estimated to be 0.33 Tg yr^{-1}, which is about 1.5 times as much as the primary $PM_{2.5}$ emissions from transportation (0.21 Tg yr^{-1}) in China (Jing et al., 2015; Zhang et al., 2007) and accounting for about 21 % of the SOA production (1.6 Tg yr^{-1}) from anthropogenic precursors estimated by global chemical transport model (Farina et al., 2010). In addition, the photo-oxidation of long-chain alkanes (> C6, IVOCs) contained in evaporative emissions would also contribute to SOA formation (Pye and Pouliot, 2012; Tkacik et al., 2012; Presto et al., 2009; Lim and Ziemann, 2005; Zhao et al., 2016). This estimate suggests that vehicular evaporative emissions will be a significant source of SA in the presence of SO_2 and NH_3, although the estimate might have a high uncertainty due to the fact that SA yield might vary considerably under different atmospheric conditions. Meanwhile, in the presence of NO_x, SO_2 and NH_3, vehicular evaporative emissions may be a potential source of sulfur- and nitrogen-containing organics, according to the results obtained from our study. Sulfur- and nitrogen-containing organics will have an adverse influence on the climate by light absorption and/or by affecting aerosol hygroscopicity (Staudt et al., 2014; Nguyen et al., 2012), and they also have a significant contribution to SOA and nitrogen or sulfur budgets (Lee et al., 2016; Shang et al., 2016).

Therefore, under the compound pollution conditions of SO_2 and NH_3, synergistic emission reduction of vehicular evaporative emissions, SO_2 (e.g., coal-fired flue gas) and NH_3 (e.g., emitted from agricultural non-point source and traffic emissions) should be taken into consideration by policy makers for future management, which will contribute to reducing the burden of $PM_{2.5}$, and then cut the environmental, economic and health costs caused by PM pollution. Our work will provide a scientific basis for taking corresponding control measures to relieve haze events in China. Additionally, there might be some differences between the VOCs composition of gasoline vapors directly injected to the smog chamber and

vehicular evaporative emissions. Thus, further work should be focused on SA formation directly from vehicular evaporative emissions under coexisting SO_2 and NH_3 conditions to shed light on the formation mechanism of SA under more atmospherically relevant conditions.

Author contributions

TZC and YCL contributed equally to this work and should be considered as co-first authors. HH, QXM, YCL, and TZC proposed the initial idea. YCL and TZC designed and led the study. YCL, BWC, QXM, PZ, and TZC conducted the data analyses. TZC, YCL, BWC, PZ, CGL, and JL interpreted the data. TZC, YCL, JL, and QXM wrote the manuscript, with inputs from all coauthors.

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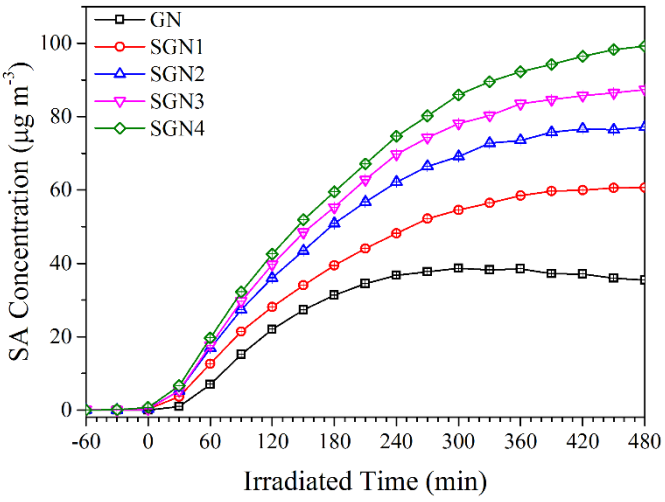
879 Table 1. Summary of experimental conditions in this study.

Exp. [a]	RH (%)	T (°C)	SO_2 (ppb)	NH_3 [b] (ppb)	HC_0 (ppb)	$NO_{x,0}$ (ppb)	$HC_0/NO_{x,0}$ (ppbC ppb^{-1})	Surface [c] ($\mu m^2\ cm^{-3}$)	ΔHC ($\mu g\ m^{-3}$)	Δ*M* ($\mu g\ m^{-3}$)	SA yield [d]
GN	50±3	26±1	−	−	411.0	128.4	20.61	1.12×10^3	747.8	34.6	0.130
SGN1	50±3	26±1	35	−	419.8	121.0	22.34	1.73×10^3	871.6	58.0	0.155
SGN2	50±3	26±1	74	−	412.0	121.3	21.88	2.06×10^3	866.2	77.8	0.193
SGN3	50±3	26±1	116	−	383.6	119.8	20.62	2.23×10^3	791.1	87.1	0.226
SGN4	50±3	26±1	151	−	394.4	125.9	20.17	2.46×10^3	810.7	106.3	0.258
AGN1	50±3	26±1	−	150	413.8	120.4	22.12	1.79×10^3	700.6	47.6	0.158
AGN2	50±3	26±1	−	200	411.5	122.6	21.61	2.23×10^3	749.1	58.3	0.166

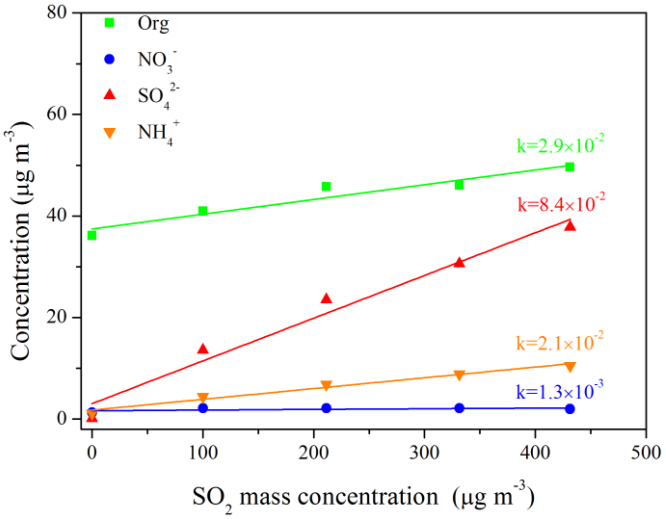
880 [a] Letters in abbreviations represent the reactants introduced into the chamber reactor, i.e., "G" represents
881 gasoline, "N" represents nitrogen oxides, "S" represents sulfur dioxide, "A" represents ammonia.
882 [b] The concentration of NH_3 is estimated by the amount of NH_3 added and the volume of the smog chamber.
883 [c] The surface area of aerosol particles measured by SMPS after 480 min of each experiment.
884 [d] SA yield was calculated after taking vapor and particle wall loss into account.



885

886 Fig. 1. Time series of secondary aerosol concentrations during the photo-oxidation experiments with different SO_2

887 concentrations (Exps. GN, SGN1, SGN2, SGN3, and SGN4).



888

889 Fig. 2. Linear relationship between the concentration of chemical species (i.e., organic (green), nitrate (blue), sulfate (red), and

890 ammonium (orange)) and SO_2 under different SO_2 initial concentration conditions (Exps. GN, SGN1, SGN2, SGN3, and

891 SGN4). Each line represents a linear fitting and the k values are the corresponding slopes for each chemical species.

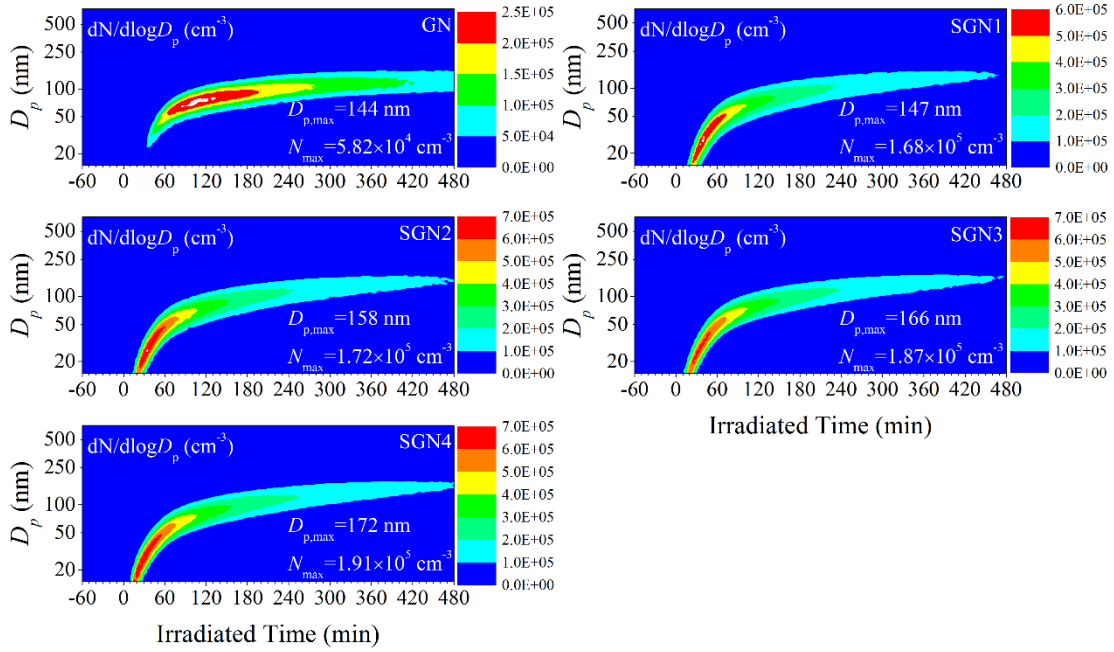


Fig. 3. Time series of the size distributions for the generated secondary aerosol during the photo-oxidation experiments with different SO_2 initial concentrations (Exps. GN, SGN1, SGN2, SGN3, and SGN4). $D_{p,max}$ and N_{max} represent the maximal diameter and number concentration of generated secondary aerosol, respectively, during each photo-oxidation experiment.

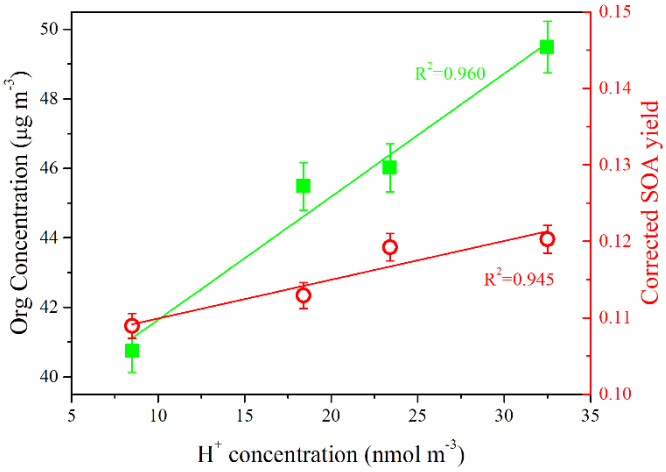
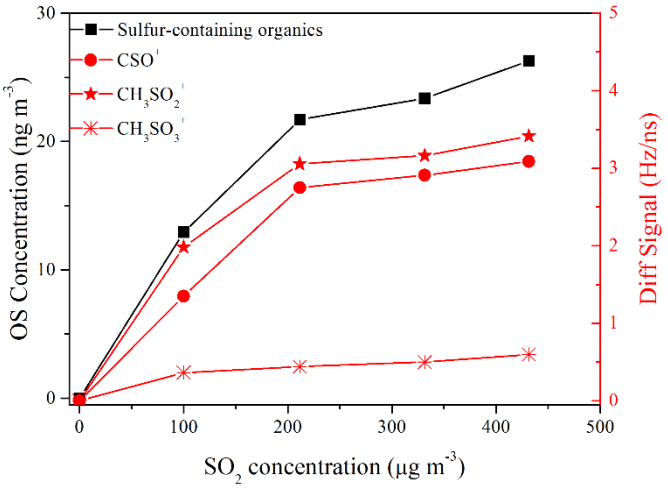


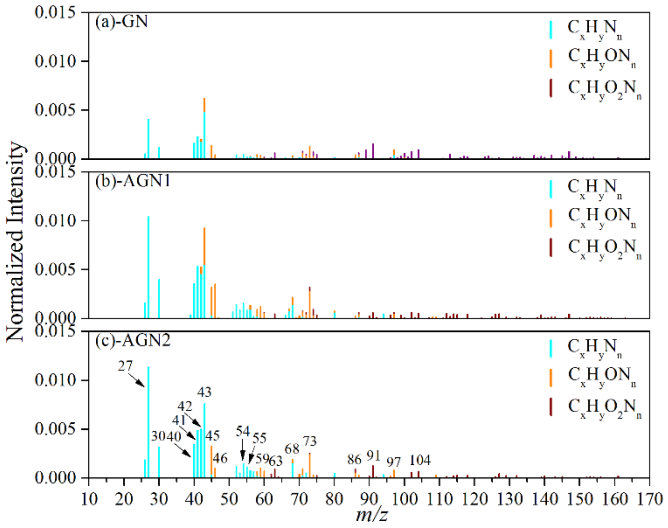
Fig. 4. Relationship between SOA concentration (left y axis), corrected SOA yield (right y axis) and H^+ concentration, which was used to characterize the particle acidities. The H^+ concentration presented in this plot was the value when the SOA formation rate reached the peak during each experiment (Exps. SGN1, SGN2, SGN3, and SGN4).



900

901 Fig. 5. Signal of fitted peaks, i.e., CSO^+, $CH_3SO_2^+$, $CH_3SO_3^+$ (right y axis) and sulfur-containing organics concentration (left

902 y axis) as a function of SO_2 initial concentration.



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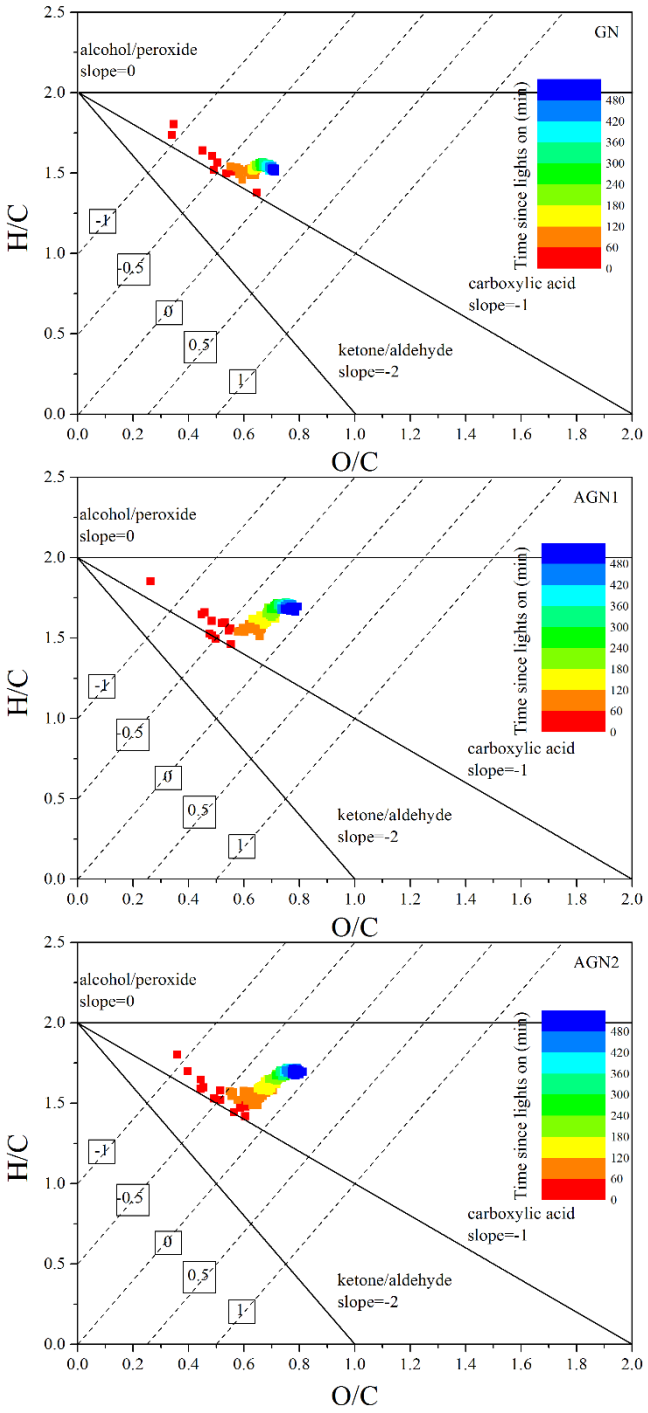
904 Fig. 6. Typical normalized mass spectra of N-containing fragments in SOA formed from the photo-oxidation of gasoline vapor

905 at different concentrations of NH_3 (Exps. GN, AGN1 and AGN2).



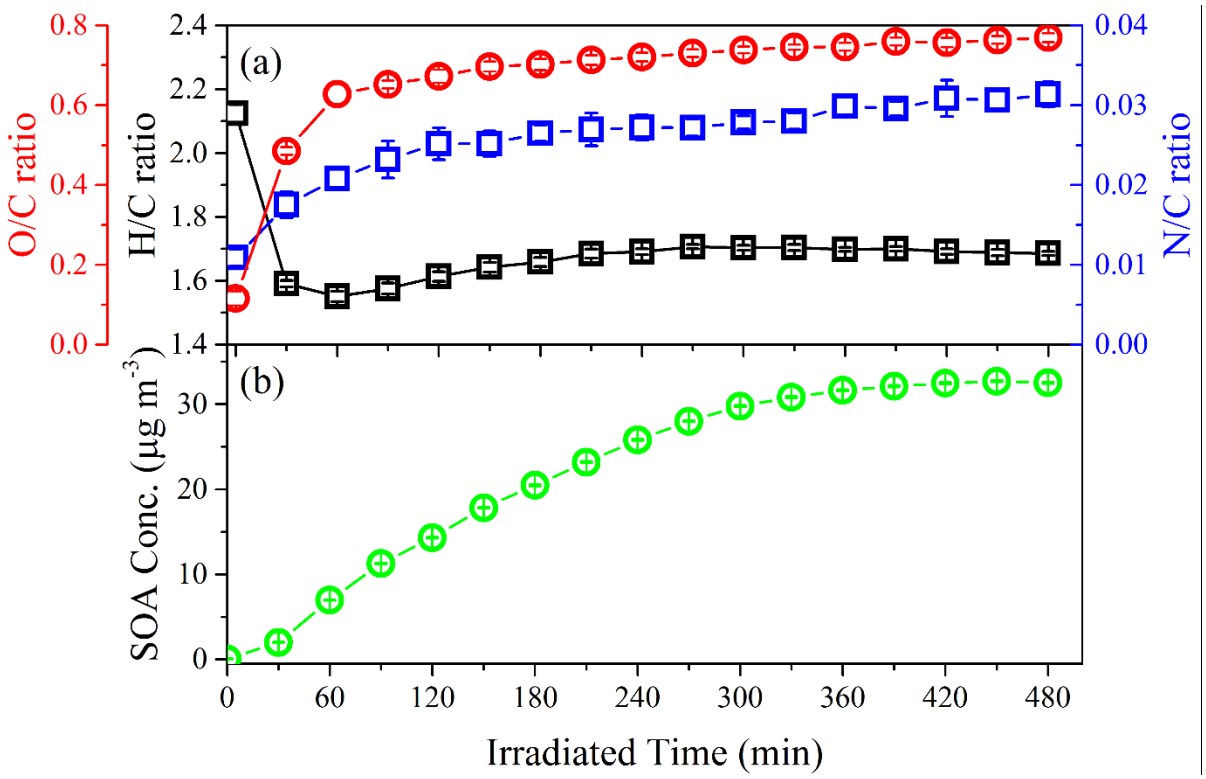
906

907 Fig. 7. Time evolution of H/C and O/C in SOA formed from the photo-oxidation of gasoline vapor at different concentrations

908 of NH_3 (Exp. GN, AGN1 and AGN2). The numbers (i.e., -1, -0.5, 0, 0.5, and 1) labeling the dashed lines show the average

909 carbon oxidation state (OSc $=$ 2×O/C−H/C) (Kroll et al., 2011). The black lines represent the addition of functional groups

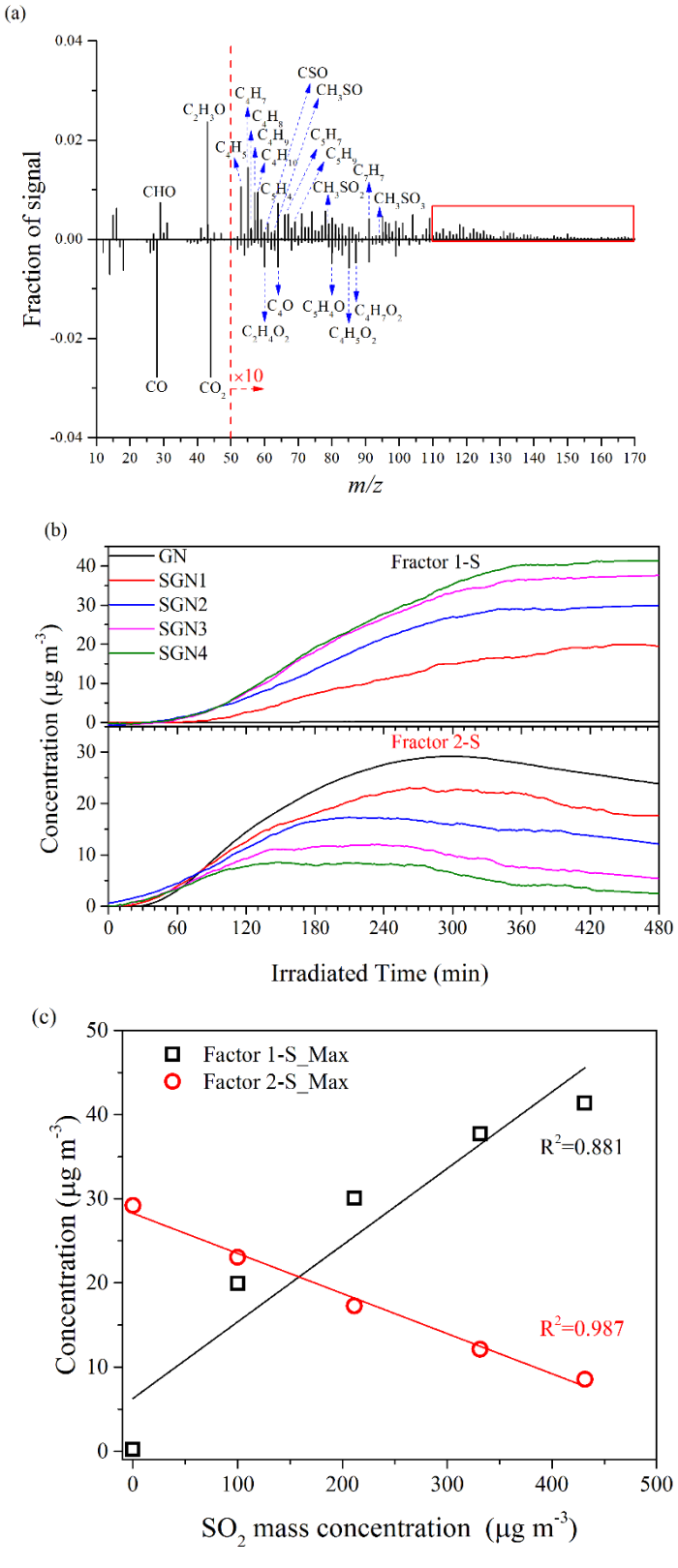
910 to an aliphatic carbon (Heald et al., 2010).



911

912 Fig. 8. Time evolution of (a) O/C, H/C and N/C and (b) SOA concentration in the photo-oxidation of gasoline vapor in the

913 presence of 150 ppb NH_3 (Exp. AGN1).



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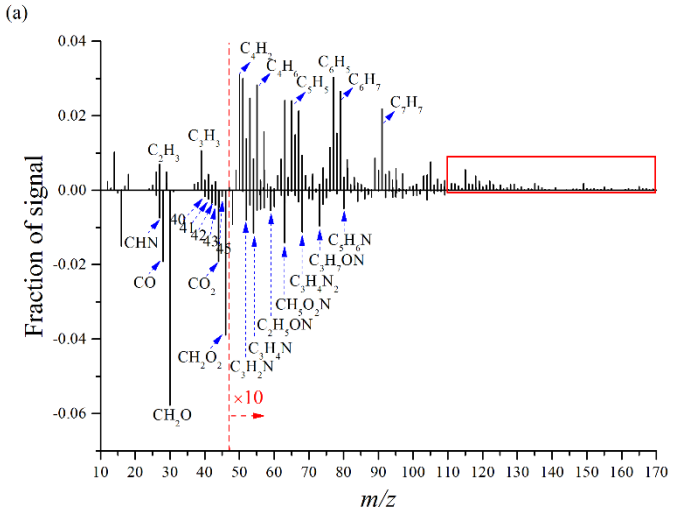
917 Fig. 9. (a) Difference mass spectra (Factor 1-S−Factor 2-S) between the two factors, (b) Time series of the mass concentration,

918 and (c) Relationship between the concentration of SO_2 and the maximum concentration of the two factors identified by

919 applying PMF analysis to the AMS data derived from the experiments at different concentrations of SO_2 (Exps. GN, SGN1,

920 SGN2, SGN3 and SGN4).



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924 Fig. 10. (a) Difference mass spectra (Factor 1-N−Factor 2-N) between the two factors, (b) Time series of the mass

925 concentration, and (c) Relationship between the concentration of NH_3 and the maximum concentration of the two

926 factors identified by applying PMF analysis to the AMS data derived from the experiments at different

927 concentrations of NH_3 (Exps. GN, AGN1 and AGN2).