Ammonium nitrate promotes sulfate formation through uptake kinetic regime

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Abstract. Although the anthropogenic emissions of SO2 have decreased significantly in China, the decrease in SO42− in PM2.5 is much smaller than that of SO2. This implies an enhanced formation rate of SO42− in the ambient air, and the mechanism is still under debate. This work investigated the formation mechanism of particulate sulfate based on statistical analysis of long-term observations in Shijiazhuang and Beijing supported with flow tube experiments. Our main finding was that the sulfur oxidation ratio (SOR) was exponentially correlated with ambient RH in Shijiazhuang (SOR = 0.15 + 0.0032 × exp(RH/16.2)) and Beijing (SOR = −0.045 + 0.12 × exp(RH/37.8)). In Shijiazhuang, the SOR is linearly correlated with the ratio of aerosol water content (AWC) in PM2.5 (SOR = 0.15 + 0.40 × AWC/PM2.5). Our results suggest that uptake of SO2 instead of oxidation of S(IV) in the particle phase is the rate-determining step for sulfate formation. NH4NO3 plays an important role in the AWC and the change of particle state, which is a crucial factor determining the uptake kinetics of SO2 and the enhanced SOR during haze days. Our results show that NH3 significantly promoted the uptake of SO2 and subsequently the SOR, while NO2 had little influence on SO2 uptake and SOR in the presence of NH3.

1 Introduction

Atmospheric particulate matter (PM) is a worldwide concern due to its adverse effect on human health, such as its association with respiratory and cardiovascular diseases, lung cancer, and premature death (WHO, 2013; Lelieveld et al., 2015). The Chinese government has made great efforts to improve the air quality (Cheng et al., 2019). For example, the annual PM2.5 concentration in Beijing decreased from 89.5 µg m−3 in 2013 to 58 µg m−3 in 2017 due to the stringent reduction of local and regional emissions (Cheng et al., 2019; Ji et al., 2019). However, the PM2.5 concentrations in most regions of China (Cheng et al., 2019; Q. Chen et al., 2019; Huang et al., 2019; Tian et al., 2019) are still significantly higher than the PM2.5 standard recommended by...
the World Health Organization (WHO) (WHO, 2006). Haze events also occur with high frequency, especially, in autumn and winter.

Secondary inorganic aerosol (SIA) including sulfate (SO$_4^{2-}$), nitrate (NO$_3^-$), ammonium (NH$_4^+$) and secondary organic aerosol (SOA) usually contribute to ≈70% of PM$_{2.5}$ mass concentration in different regions (Huang et al., 2014; An et al., 2019). SIA often accounts for more than a half of PM$_{2.5}$ mass in severe pollution events (Zheng et al., 2015; Wang et al., 2016). Even SO$_4^{2-}$ exceeds more than 20% of PM$_{2.5}$ mass (Guo et al., 2014; Wang et al., 2016; Xie et al., 2015; He et al., 2018). Interestingly, the anthropogenic emissions of SO$_2$ in 2017 reduced by ~90% when compared with 2000 in Beijing (Cheng et al., 2019; Lang et al., 2017). However, the decrease rate of particulate SO$_4^{2-}$ concentration (Lang et al., 2017; C. Li et al., 2017) is much smaller than SO$_2$ (Lang et al., 2017; Zhang et al., 2020; Liu et al., 2021). For example, the annual mean concentration of SO$_4^{2-}$ decreased by 0.1 µg m$^{-3}$ yr$^{-1}$ from 2000 to 2013, followed by 1.9 µg m$^{-3}$ yr$^{-1}$ from 2013 to 2015 in Beijing, while it decreased by 3.8 µg m$^{-3}$ yr$^{-1}$ for SO$_2$ (Lang et al., 2017). This implies an enhanced oxidation rate of SO$_2$ in the atmosphere (Lang et al., 2017). However, the mechanisms and kinetics of particulate SO$_4^{2-}$ formation in the real atmosphere are still open questions in many regions of China although they have been extensively discussed (Ervens, 2015; Warneck, 2018).

Particulate SO$_4^{2-}$ can be formed through homogeneous oxidation of SO$_2$ by hydroxyl radicals (OH) and stabilized Criegee intermediates (SCIs) in the gas phase and subsequent uptake onto particles, while the OH pathway is the dominant gas-phase oxidation pathway (Seinfeld and Pandis, 2006; L. Liu et al., 2019). Modeling studies greatly underestimated (~54%) SO$_4^{2-}$ concentration in serve pollution events in Beijing if only considering gas-phase oxidation of SO$_2$, while the normalized mean bias (NMB) decreased significantly after heterogeneous oxidation of SO$_2$ being considered (Zheng et al., 2015). Several heterogeneous and/or multiphase oxidation pathways, such as oxidation of SO$_2$ or sulfite by H$_2$O$_2$ (Huang et al., 2015; Maalõ et al., 1999; P. Liu et al., 2020; Ye et al., 2021; Liu et al., 2021), HONO (J. F. Wang et al., 2020), and O$_3$ (Maalõ, 1983) or photo-chemical oxidation of SO$_2$ (Yu et al., 2017; Xie et al., 2015), catalytic oxidation of SO$_2$ by transition metal ions (TMI) (Warneck, 2018; Martin and Good, 1991; Wang et al., 2021) and oxidation of SO$_2$ by NO$_2$ (He et al., 2014; Clifton et al., 1988; Wang et al., 2016; Cheng et al., 2016; Wu et al., 2019; Spindler et al., 2003) in aqueous phase and heterogeneous oxidation of SO$_2$ on black carbon (Zhao et al., 2017; Zhang et al., 2020; Yao et al., 2020), have been proposed based on field measurements and laboratory and modeling studies. However, the relative contribution of these pathways to the SO$_4^{2-}$ production is still controversial. For example, the contribution of heterogeneous oxidation to SO$_4^{2-}$ production had been evaluated to be (48 ± 5)% based on oxygen isotopic measurements (He et al., 2018), while it was 31% even in the nighttime calculated by an observation-based modeling (OBM) (Xue et al., 2016). Gas-phase oxidation by OH could explain 33%–36% of SO$_4^{2-}$ production in the Beijing–Tianjin–Hebei region (L. Liu et al., 2019), while it was negligible based on isotopic measurements (He et al., 2018) and OBM simulations (Xue et al., 2016). As for the oxidation of S(IV) species, which includes SO$_2$, HSO$_3^-$ and SO$_3^{2-}$, in aqueous phase, oxidation by H$_2$O$_2$ (T. Liu et al., 2020; P. Liu et al., 2020; Ye et al., 2021), NO$_2$ (J. F. Wang et al., 2020; Wang et al., 2016; Cheng et al., 2016), O$_3$ (Fang et al., 2019), or TMI (Mn$^{2+}$) (Wang et al., 2021) was proposed as the most important pathway by different researchers. However, the relative importance of these oxidation paths varied greatly among different researches. For instance, TMI-catalyzed oxidation could explain ~69% of aqueous sulfate formation in North China Plain based on isotopic measurements and modeling (Shao et al., 2019), while oxidation by NO$_2$ or O$_3$ was the dominant oxidation path (66%–73%) based on isotopic measurements in another study (He et al., 2018). It should be noted that some reaction mechanisms mentioned above were proposed based on case studies in short-term observations. Thus, long-term observations at different environments are required to verify whether these mechanisms are statistically important. In addition, the previous studies mainly focused on oxidation process of SO$_2$ in particle phase, while it is unclear what are the controlling factors of the S(IV)-to-S(VI) conversion from the gas phase to the particle phase. In particular, it has been found that the mass fraction of NO$_3^-$ and NH$_4^+$ is increasing gradually (Lang et al., 2017; Li et al., 2018). This will modify its physical properties, such as morphology, phase-state and so on. The feedback between aerosol physics and aerosol chemistry is still poorly understood.

In this work, 1-year field observations have been performed in Shijiazhuang and Beijing, synchronously. The formation mechanism of particulate sulfate has been statistically investigated to identify the controlling factors. The role of mass transfer of SO$_2$ and the oxidation of S(IV) in particle phase have been discussed based on flow tube experiments and field measurements. The conversion ratio of SO$_2$ to sulfate is statistically and linearly correlated to the aerosol water content (AWC), which is strongly modulated by particulate ammonium nitrate. The reaction kinetics and other factors affecting sulfate production have also been discussed.

2 Material and methods

2.1 Field measurements

Field measurements were performed at Shijiazhuang University (SJZ; 38.0281° N, 114.6070° E) and the west campus of Beijing University of Chemical Technology (BUCT; 39.9428° N, 119.2966° E) from 15 March 2018 to
15 April 2019. The SJZ station is on a rooftop of the main teaching building (5 floors, ~ 23 m above the surface), which is around 250 m from the Zhuijiang Road of Shijiazhuang. The BUCT station is on a rooftop of the main building (5 floors, ~ 18 m above the surface), which is around 550 m from the 3rd Ring Road of Beijing. The distance between the two stations, which are the representative cities of Beijing-Tianjin-Hebei (BTH), is 260 km (Fig. S1). Both stations are surrounded by traffic and residential emissions, thus, are typical urban observation sites. The details about the observation stations have been described in our previous work (Y. Liu et al., 2020c, b, a).

Ambient air was drawn from the roof of the corresponding building. At the SJZ station, the mass concentration of PM$_{2.5}$ was measured by a beta attenuation mass monitor (BAM-1020, Met One Instruments, USA) with a smart heater (model BX-830, Met One Instruments Inc., USA) to control the RH of the incoming air to 35% and a PM$_{2.5}$ inlet (URG) to cut off the particles with diameter larger than 2.5 μm. Particle-phase total concentrations of iron and manganese were measured using a heavy-metal analyzer (EHM-X100, Skyray Instrument). Water-soluble ions (Na$^+$, K$^+$, Mg$^{2+}$, Ca$^{2+}$, NH$_4^+$, SO$_4^{2-}$, Cl$^-$ and NO$_3^-$) in PM$_{2.5}$ and gas pollutants (HCl, HONO, HNO$_3$, SO$_2$ and NH$_3$) were measured using an analyzer for monitoring aerosols and gases (MARGA, ADI 2080, Applikon Analytical B.V., the Netherlands) with 1 h of time resolution. At the BUCT station, the mass concentration of PM$_{2.5}$ was the mean concentration obtained from four surrounding monitoring stations (including Wanliu, Gucheng, Wanshouxigong and Guanyuan) of China Environmental Monitoring Centre (http://www.cnemc.cn, last access: 31 December 2020). The chemical composition of PM$_{2.5}$ was measured using a time-of-flight aerosol chemical speciation monitor (ToF-ACSM, Aerodyne) after the ambient air went through a PM$_{2.5}$ inlet (URG) and a Nafion dryer (MD-700-24, Perma Pure). The configuration and the operation protocol of ToF-ACSM have been described in previous work (Fröhlich et al., 2013). The ionization efficiency (IE) calibration for ACSM was performed using 300 nm dry NH$_4$NO$_3$ every month. Ambient air was drawn from the roof using a Teflon sampling tube (BMET-S, Beijing Saak-Mar Environmental Instrument Ltd.) with the residence time <10 s for gas-phase pollutant measurements. Trace gases including NO$_x$, SO$_2$, CO and O$_3$ were measured with the corresponding analyzer (Thermo Scientific, 42i, 43i, 48i and 49i) at both the SJZ and BUCT stations. Meteorological parameters including temperature, pressure, relative humidity (RH), wind speed and wind direction were measured using weather stations (WXT 520 at HAS/SJZ station and AWS 310 at AHL/BUCT station, Vaisala).

2.2 Uptake kinetics of SO$_2$ on dust internally mixed with NH$_4$NO$_3$

To understand the influence of RH on uptake kinetics ($\gamma_{SO_2}$), the $\gamma_{SO_2}$ on dust internally mixed with NH$_4$NO$_3$ was measured using a coated-wall flow tube reactor. The configuration of the reactor and data process protocol have been described in detail previously (Han et al., 2013; Liu et al., 2015). The $\gamma$ parameter, presenting the mass transfer kinetic of gas-phase SO$_2$ to particle phase, is defined by the net loss rate of SO$_2$ per collision onto the surface (Ravishankara, 1997; Usher et al., 2003), namely,

$$\gamma_{obs} = \frac{-dc}{dt} = \frac{2k_{obs}r_{tube}}{<c>},$$

where $-dc/dt$ is the net loss rate of SO$_2$ when the surface is exposed to SO$_2$ (molecules s$^{-1}$); $\omega$ is the collision frequency (s$^{-1}$); $k_{obs}$, $r_{tube}$, and $<c>$ are the first-order rate constant of SO$_2$, the flow tube radius, and the average molecular velocity of SO$_2$, respectively. A correction for gas-phase diffusion limitations was considered for $\gamma_{obs}$ calculations using the Cooney–Kim–Davis (CKD) method (Cooney et al., 1974; Murphy and Fahey, 1987). The Brunauer–Emmett–Teller ( BET) uptake coefficient ($\gamma_{SO_2,BET}$) was obtained from the mass dependence of $\gamma_{obs}$ as follows (Han et al., 2013; Liu et al., 2015):

$$\gamma_{SO_2,BET} = \left[ \frac{\text{slope}}{A_g} \right] \frac{A_g}{S_{BET}},$$

where [slope] is the slope of the plot of $\gamma_{obs}$ versus the sample mass in the linear regime (mg$^{-1}$), $A_g$ is the inner surface area of the sample tube (cm$^2$) and $S_{BET}$ is the specific surface area of the particle sample (cm$^2$ mg$^{-1}$).

Similar to a previous work (Zhang et al., 2019), dust internally mixed with NH$_4$NO$_3$ was used in the kinetics study, because it was difficult to deposit enough real ambient particles onto the inner surface of the sample holder. Although the composition of the model particles is much simpler than that of ambient particles, it is still meaningful, because we mainly focused on the influence of RH or aerosol water content (AWC) on uptake kinetics of SO$_2$. The mixture (mass ratio $= 2 : 1$) of A1 ultrafine test dust (Powder Technology Inc.) and NH$_4$NO$_3$ (AR, Sinopharm Chemical Reagent Co. Ltd, China) were suspended in the mixture of ethanol and water ($v : v = 1 : 3$). The inner surface of the Pyrex quartz tube (sample holder) was uniformly coated by the above mixture and dried overnight in an oven at 393 K. The sample mass was calculated according to the weighted mass of the dry tube before and after coating. NH$_4$NO$_3$ in the mixture was further confirmed using an ion chromatograph (Ω Metrohm 940, Applikon Analytical B.V., the Netherlands). Around 50% of the NH$_4$NO$_3$ remained in the mixture even after heating and potential evaporation. To avoid the wall loss of SO$_2$ on the sample holder, all the inner surface of the
sample holder was covered with particles. The wall loss of \( \text{SO}_2 \) on the remained surface (the inner surface of the outside tube and the outside surface of the sample holder) was subtracted at a steady-state condition at the corresponding RH before the uptake experiment as done in our previous work (Liu et al., 2015). The mean concentrations of \( \text{SO}_2 \), \( \text{NO}_2 \) and \( \text{NH}_3 \) were 8.3 ± 5.2 (0.4–49.1), 31.5 ± 13.2 (2.5–85.1) and 41.0 ± 18.4 (0.3–126.4) ppb, respectively, in polluted events with the PM\(_{2.5}\) concentration higher than 75 µg m\(^{-3}\) and the RH less than 90% in Shijiazhuang. The initial concentrations of \( \text{SO}_2 \), \( \text{NO}_2 \) and \( \text{NH}_3 \) in the reactor were 190 ± 2.5, 100 ± 2.5 and 50 ± 2.5 ppb, respectively. The initial concentrations of \( \text{NO}_2 \) and \( \text{NH}_3 \) were close to their ambient concentrations, while a high initial \( \text{SO}_2 \) concentration was used here to obtain a good signal to noise ratio for \( \gamma_{\text{SO}_2} \) measurements. In this work, we aimed to understand the influence of AWC on the uptake kinetics of \( \text{SO}_2 \). Therefore, we fixed the initial concentrations of pollutants and the temperature at 300 K. \( \text{SO}_2 \) and \( \text{NO}_2 \) were measured using the corresponding analyzer (Thermo 43i and 42i), and \( \text{NH}_3 \) was measured by an ammonia analyzer (EAA-22, LGR, USA). The specific surface area of the mixture of A1 dust and \( \text{NH}_3\text{NO}_3 \) was 0.813 m\(^2\) g\(^{-1}\), measured by a nitrogen BET physisorption analyzer (Quantachrome Autosorb-1-C). RH from 0 to 80% was adjusted by varying the ratio of dry to wet zero air (water bubbler) and measured by a RH sensor (HMP110, Humicap). Control experiments demonstrate that adsorption of \( \text{SO}_2 \) on the quartz tube is negligible. It should be noted that the wall loss of \( \text{SO}_2 \) in the presence of \( \text{NH}_3 \) and/or \( \text{NO}_2 \) would be larger in the absence of seed aerosols. Additional control experiments in the presence of \( \text{NO}_2 \) and \( \text{NH}_3 \) demonstrate that the contribution of wall loss of \( \text{SO}_2 \) should be less than 3% to the measured \( \gamma \).

### 2.3 Calculations of AWC, aerosol pH and production rates of sulfate in aerosol liquid water

The AWC and aerosol pH in Shijiazhuang were calculated using the ISORROPIA II model using the measured concentrations of \( \text{SO}_2^{2-} \), \( \text{NH}_4^+ \), \( \text{NH}_3 \), \( \text{NO}_3^- \), \( \text{HNO}_2 \), \( \text{HCl}^- \), \( \text{HCl} \), \( \text{Na}^+ \), \( \text{Ca}^{2+} \), \( \text{K}^+ \) and \( \text{Mg}^{2+} \), as well as RH and temperature as input. The particles were assumed in metastable phase using a forward method (Song and Osada, 2020; Shi et al., 2019). The dataset with RH lower than 35% was excluded (Pye et al., 2020) due to large uncertainties of aerosol pH (Ding et al., 2019; Guo et al., 2016; Pye et al., 2020). The pH was then calculated according to Pye et al. (2020) and Ding et al. (2019):

\[
\text{pH} = -\log_{10}(\gamma_{\text{H}^+} m_{\text{H}^+}) = -\log_{10} \left( \frac{1000 \gamma_{\text{H}^+} c_{\text{H}^+}}{\text{AWC}} \right),
\]

where \( \gamma_{\text{H}^+} \) is the activity coefficient of \( \text{H}^+ \), and \( m_{\text{H}^+} \) is the molality of \( \text{H}^+ \). The deliquescence curves of inorganic salts were calculated at 298.5 K using the E-AIM model (Clegg et al., 1998). Then, the AWC attributed to individual salt was calculated with the mass of the salt and the mass-based growth factor at the corresponding RH. The AWC of model particles for laboratory studies was also calculated with the known composition, while the aerosol pH values in Beijing were not calculated, because the concentrations of \( \text{Na}^+ \), \( \text{Ca}^{2+} \), \( \text{K}^+ \) and \( \text{Mg}^{2+} \) were unavailable.

Similar to previous studies (P. Liu et al., 2020; Cheng et al., 2016), four oxidation pathways of \( \text{S(IV)} \) in aqueous-phase were accounted for, i.e., oxidation by \( \text{O}_3 \), \( \text{H}_2\text{O}_2 \), \( \text{NO}_2 \) and TMI (\( \text{Fe}^{3+} \) and Mn\(_2\)\(^{2+} \)), according to the following equations (Seinfeld and Pandis, 2006; Cheng et al., 2016; P. Liu et al., 2020):

\[
\frac{d[S(IV)]}{dt}_{\text{O}_3} = (k_0 [\text{SO}_2,\text{aq}]) + k_1 [\text{HSO}_3^-],
\]

\[
\frac{d[S(IV)]}{dt}_{\text{H}_2\text{O}_2} = k_2 [\text{H}^+] [\text{HSO}_3^-] [\text{H}_2\text{O}_2,\text{aq}],
\]

\[
\frac{d[S(IV)]}{dt}_{\text{TMI}} = k_3 [\text{Fe}^{3+}] [\text{Mn}^{2+}] [\text{S(IV)}],
\]

\[
\frac{d[S(IV)]}{dt}_{\text{NO}_2} = k_5 [\text{NO}_2,\text{aq}] [\text{S(IV)}],
\]

where \( k_0 = 2.4 \times 10^4 \text{ M}^{-1} \text{s}^{-1}, k_1 = 3.7 \times 10^5 \text{ M}^{-1} \text{s}^{-1}, k_2 = 1.5 \times 10^9 \text{ M}^{-1} \text{s}^{-1}, k_3 = 7.45 \times 10^7 \text{ M}^{-1} \text{s}^{-1}, k_4 = 3.72 \times 10^7 \text{ M}^{-1} \text{s}^{-1}, k_5 = 1.24 \times 10^7 \text{ M}^{-1} \text{s}^{-1} \), and \( \gamma_{\text{H}^+} = -0.74 \) (for \( \text{pH} \leq 4.2 \)), \( \gamma_{\text{H}^+} = 2.51 \times 10^{13} \text{ M}^{-1} \text{s}^{-1} \), \( \gamma_{\text{H}^+} = 0.67 \) (for \( \text{pH} > 4.2 \)), and \( \gamma_{\text{H}^+} = (1.24–1.67) \times 10^7 \text{ M}^{-1} \text{s}^{-1} \) (for 5.3 ≤ \( \text{pH} \) ≤ 8.7; the linear interpolated values were used for \( \text{pH} \) between 5.3 and 8.7) at 298 K (Clifton et al., 1988; P. Liu et al., 2020; Tilgner et al., 2021; Liu et al., 2021). \( [\text{O}_3,\text{aq}], [\text{H}_2\text{O}_2,\text{aq}] \) and \( [\text{NO}_2,\text{aq}] \) were calculated according to the Henry’s constants, which are \( 1.1 \times 10^{-2}, 1.0 \times 10^5 \) and \( 1.2 \times 10^{-2} \text{ M atm}^{-1} \) at 298 K for \( \text{O}_3 \), \( \text{H}_2\text{O}_2 \) and \( \text{NO}_2 \) respectively (Seinfeld and Pandis, 2006). \( \text{H}_2\text{O}_2 \) concentrations were unavailable during our observations. It was fitted based on temperature like in previous work (Fang et al., 2019). Figure S2 shows the derived \( \text{H}_2\text{O}_2 \) concentrations and the diurnal curves of \( \text{H}_2\text{O}_2 \) in winter in Shijiazhuang. The \( \text{H}_2\text{O}_2 \) concentrations varied from 0.05 to 3.7 ppbv, with a mean value of 0.62 ± 0.52 ppbv. Overall, the wintertime \( \text{H}_2\text{O}_2 \) concentrations derived in this work are comparable with those reported in the literature (Ye et al., 2018). The concentrations of \( \text{Fe}^{3+} \) and Mn\(_2\)\(^{2+} \) were calculated according to the measured total Fe and Mn concentrations assuming 18% of total Fe and 30% of total Mn were soluble (Wang et al., 2014; Cui et al., 2008) and the precipitation equilibriums of \( \text{Fe(OH)}_3 \) and Mn\(_2\)\((\text{OH})_2 \) depending on pH. The concentrations of Fe and Mn before December 2018 were estimated according to their mean ratios to PM\(_{2.5}\) mass concentration (Wang et al., 2014), because the instrument was unavailable.
3 Results and discussion

3.1 Variation of sulfate in PM$_{2.5}$

Figure 1a shows the hourly mean mass concentration of PM$_{2.5}$ measured at SIZ and BUCT stations from 15 March 2018 to 15 April 2019. The mass concentration of PM$_{2.5}$ in Shijiazhuang generally coincided with that in Beijing. This highlights the regional characteristic of air pollution in BJHH. However, Shijiazhuang usually showed significantly higher PM$_{2.5}$ concentration than that in Beijing. The hourly mean PM$_{2.5}$ concentration varied in the range of 0–650 µg m$^{-3}$ with an annual mean concentration of 86.4 ± 77.8 µg m$^{-3}$. The corresponding values in Beijing were 1.5–556 and 55.0 ± 51.0 µg m$^{-3}$. Particularly, the wintertime mass concentration of PM$_{2.5}$ in Shijiazhou was around 2.4 times as that in Beijing. This is consistent with previous results that Shijiazhuang is suffering from more serious air pollution (D. Chen et al., 2019) because of its larger density of heavy industries and more intensive emissions than in Beijing (D. Chen et al., 2019a).

Like the mass concentration of PM$_{2.5}$, both the mass concentration (Fig. 1b) and the fraction of sulfate in PM$_{2.5}$ (Fig. 1c) in Shijiazhuang were usually higher than those in Beijing. The annual mean sulfate concentrations in Shijiazhuang and Beijing were 117.7 ± 12.7 and 5.4 ± 6.9 µg m$^{-3}$, which annually contributed 15.3 ± 8.7 % and 10.7 ± 7.3 % to the PM$_{2.5}$ mass concentrations, respectively. However, the molar ratio of NO$_3^-$ to SO$_4^{2-}$ (3.37 ± 3.05) was significantly lower than that in Shijiazhuang. This shows a high proportion of gas phase oxidation processes, which may indicate that the conversion of SO$_2$ to sulfate is promoted in pollution days when high RH is present.

Previous studies have found that severe pollution events are frequently accompanied with high RH (Zhang et al., 2018; Tang et al., 2016; Wu et al., 2018; Y. Liu et al., 2019; Clifton et al., 1988; Maahs, 1983; Martin and Good, 1991). As shown in Fig. 2d, the high concentration of sulfate positively correlated with high RH in most cases, which are shaded in grey columns. However, some pollution events (shaded in purple columns) also occurred under high RH, but the sulfate concentration or sulfate fraction in soluble PM was not so high. This means that high RH is a necessary but not a sufficient condition for sulfate formation in severe haze pollution events. Thus, it is difficult to fully understand the general regularity behind the dataset or overemphasize the importance of a specific process in the atmosphere based on

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case studies. This might be the reason why contrary conclusions about the formation path of sulfate were drawn by different researchers. We statistically analyzed the relationship between the SOR and the RH. All the hourly mean data of the SOR and RH have been binned into $100 \times 100$ boxes. Then, the density of data points, which statistically indicates the occurrence of the events at given values of RH and SOR, was calculated using a bivariate kernel density estimator (Wand and Jones, 1993).

Figure 3a and b show the 2D kernel density graphs between the SOR and the RH in Shijiazhuang and Beijing. The color bar shows the density of data points. Although the SOR varied obviously at a certain RH, the most probable distribution of SOR could be exponentially fitted as a function of RH in Shijiazhuang (Fig. 3a); i.e., $\text{SOR} = 0.15 + 0.0032 \times \exp(RH/16.2)$ ($R = 0.79$). This is consistent with the dependence of SOR on RH based on previous studies (Tian et al., 2019; Wu et al., 2019). It should be noted that both SOR and
RH showed obvious diurnal variation (Fig. S4). Their diurnal variations were somewhat similar, but a 4 h time lag was observed between their minimum values. This means that the diurnal variations of SOR and RH might also contribute to the strong dependency of SOR on RH (Fig. 3a and b). However, the exponential dependency of SOR on RH was still observable in the night or in the day (Fig. S5a and b). It did so in winter or summer (Fig. S5c and d). This means that aqueous reactions are important for sulfate formation even if the influence of diurnal and seasonal variations are ruled out (Wang et al., 2016; Cheng et al., 2016).

In Fig. 3a, 72.5% of the data points of Shijiazhuang (6509 over 8980 effective points) were in the domain with the RH range of 10%–70% and the SOR range of 0.05–0.42, while 10.1% of data points were in the region with the RH greater than 70% and the SOR greater than 0.42. The first region corresponded to a lower mean PM$_{2.5}$ concentration, sulfate concentration and SOR (76.1 ± 62.78 µg m$^{-3}$, of 8.1 ± 6.3 µg m$^{-3}$, and 0.21 ± 0.09, respectively) compared with the second one (115.7 ± 96.7 µg m$^{-3}$, 22.4 ± 20.4 µg m$^{-3}$ and 0.62 ± 0.14, respectively). As shown in Fig. 3b, the SOR also exponentially increased as a function of RH in Beijing. A percentage of 74.6% of 8169 data points were in the first region. The mean PM$_{2.5}$ concentration, sulfate concentration and SOR were 48.2 ± 44.8 µg m$^{-3}$, 2.9 ± 3.0 µg m$^{-3}$ and 0.21 ± 0.10, respectively, in the low RH region, while they were 69.9 ± 50.9 µg m$^{-3}$, 9.4 ± 8.5 µg m$^{-3}$ and 0.83 ± 0.15 in the high RH region. The most probable distribution of SOR in Beijing could also be exponentially fitted as a function of RH (SOR = −0.045 + 0.12 × exp(RH/37.8), R = 0.92). However, the SOR was more sensitive to RH in Beijing than that in Shijiazhuang. This might be explained by the increase of the formation rates of sulfate on RH in the range of 35%–100%. In Shijiazhuang, the dataset for RH below 35% was omitted due to the large uncertainty in aerosol pH calculations (Ding et al., 2019; Guo et al., 2016; Pye et al., 2020). The relative contributions of different oxidation paths of S(IV) also varied obviously as a function of RH (P. Liu et al., 2020; T. Liu et al., 2020). If oxidation of S(IV) was the most important path followed by TMI. In winter, however, NO$_2$, O$_3$ or H$_2$O$_2$ could contribute to the major oxidation path. This might be the reason why these oxidation pathways showed inconsistent relative importance of among different studies even using the same method, such as isotopic measurements (Shao et al., 2019; He et al., 2018). Figure 4b and c show the dependence of the formation rates of sulfate on RH in the range of 35%–100% in Shijiazhuang. The dataset for RH below 35% was omitted due to the large uncertainty in aerosol pH calculations (Ding et al., 2019; Guo et al., 2016; Pye et al., 2020). The relative contributions of different oxidation pathways of S(IV) also varied obviously as a function of RH. NO$_2$ and O$_3$ played important role in aqueous S(IV) oxidation when RH was from 35% to 45%, while TMI became the dominant when RH ranged from 45% to 70%. Above 70% RH, the contribution of H$_2$O$_2$ was dominant, which is consistent with several recent studies (P. Liu et al., 2020; T. Liu et al., 2020). However, the total formation rate of sulfate in aerosol liquid phase slightly decreased as RH increased. A weak downward trend of the d(SO$_4^{2-}$/dl rate with RH was also observable in the 2D kernel density graphs as shown in Fig. 4c. This is opposite to the dependencies of the SOR and the $\gamma_{SO_2}$ on RH as discussed above, which means the RDS for sulfate formation should be the uptake of SO$_2$ instead of oxidation of S(IV) in aqueous phase. We further calculated the production rate of sulfate through uptake of SO$_2$ (mass transfer to aerosol
Figure 3. Relationship between SOR and $\gamma_{SO_2,BET}$ on dust internally mixed with NH$_4$NO$_3$ (2:1) and RH in (a) Shijiazhuang and (b) Beijing and the correlation of (c) SOR in Shijiazhuang and (d) $\gamma_{SO_2,BET}$ with AWC/PM$_{2.5}$. The initial concentrations of SO$_2$, NO$_2$ and/or NH$_3$ in the flow tube reactor were 190 ± 2.5, 100 ± 2.5 and/or 50 ± 2.5 ppb, respectively. The grey lines are the fitting curves for the most probable SOR, and the white lines are the fitting curves for the $\gamma_{SO_2,BET}$.

particles) according to

$$\frac{d[SO_4^{2-}]}{dt} = 3600 \cdot \frac{96}{64} \cdot \gamma_{SO_2,A,BET}SO_2 \cdot \alpha_{SO_2} \cdot \omega \cdot \gamma_{SO_2},$$

(9)

where $A_s$ is the surface area concentration of PM$_{2.5}$, $\omega$ is the mean molecular velocity of SO$_2$ and $\gamma_{SO_2}$ is the mass concentration of SO$_2$. As shown in Fig. 4c, the probability-weighted production rate of sulfate through uptake of SO$_2$ (the grey line) is lower than that through aqueous oxidation of SO(IV), in particular, when RH is lower than 70%. It should be noted that the mass transfer of SO$_2$ was not assumed to be the RDS using a large mass accommodation coefficient of SO$_2$ ($\alpha = 0.11$) (Cheng et al., 2016). According to the relationship between the mass accommodation coefficient ($\alpha$) and the uptake coefficient ($\gamma$) of SO$_2$ (Kulmala and Wagner, 2001), the $\alpha_{SO_2}$ on particles is on the same order of the $\gamma_{SO_2}$. This means that mass transfer rate might have been overestimated by Cheng et al. (2016).

Phase state is a crucial factor determining the mass transfer of pollutants from gas phase to particle phase (Davis et al., 2015; Marshall et al., 2018; Shiraiwa et al., 2011; Liu et al., 2014), while the AWC or RH greatly affects the phase state of aerosol particles (Duan et al., 2019; Y. Liu et al., 2019; Shiraiwa et al., 2017). For example, ambient particles were found to change from semisolid to liquid state when the RH was above ~60% (Y. Liu et al., 2019, 2017), corresponding to the AWC higher than ~15 µg m$^{-3}$ (Y. Liu et al., 2017) under the typical urban environment in Beijing based on rebound fractions measurements. It was also confirmed that haze particles displayed a solid–aqueous equilibrium state when the RH was around 60%–80% using an individual particle hygroscopicity system (Sun et al., 2018). As shown in Fig. S6, the most probable distribution of the AWC exponentially increased with the RH (AWC = $-5.76 + 5.15 \times \exp(RH/36.1)$, $R = 0.98$) in Shijiazhuang. An obvious transition region of the RH between 60% and 80% was also observed. These results indicate that the liquid-phase aerosol should appear when the RH is higher than ~60% (Y. Liu et al., 2019, 2017) and subsequently promote the conversion of SO$_2$ to sulfate. The SOR increased as a power function of AWC (SOR = 0.072 + 0.043 × AWC$^{0.53}$, $R = 0.78$), while it was linearly correlated with the ratio of AWC/PM$_{2.5}$ (SOR = 0.15 + 0.40 × AWC/PM$_{2.5}$, $R = 0.78$) as shown in Fig. 3c. Similarly, the AWC of dust internally mixed with NH$_4$NO$_3$ was also calculated using the ISORROPIA II model. The $\gamma_{SO_2,BET}$ also showed a similar trend as a function of AWC/PM$_{2.5}$ ($\gamma_{SO_2,BET} = 3.08E - 5 \times \text{AWC/PM}_{2.5}$, $R = 0.95$) (Fig. 3d), although the ranges of AWC/PM$_{2.5}$ were different due to the difference in aerosol composition. This means that the fraction of aerosol liquid water governs both the conversion of SO$_2$ to sulfate and uptake kinetics of SO$_2$.

It should be noted that although the SOR showed a similar RH dependence as the SO$_2$, a deviation was observed in both Shijiazhuang and Beijing (Fig. 3). The $\gamma_{SO_2}$ was mea-
The relative importance of oxidation paths of S(IV) in aqueous phase, the dependence of sulfate formation rates and the probability-weighted sulfate formation rates on RH in Shijiazhuang.

Figure 4. (a) The relative importance of oxidation paths of S(IV) in aqueous phase, the dependence of (b) sulfate formation rates and (c) the probability-weighted sulfate formation rates on RH in Shijiazhuang.

sured at a fixed temperature and initial SO$_2$ concentration. In the atmosphere, both of them varied obviously. This might lead to the observed deviation. On the other hand, the coexisted components such as organic aerosol and black carbon in atmospheric particles should have complicated the influence on the hygroscopicity and the phase change of particles. The difference between the model particles and the real ambient aerosol particles might also partially lead to the deviations of the RH dependence between the SOR and the $\gamma$$_{SO_2,BET}$. In addition, it also implies that besides the reaction in aerosol liquid phase, other reaction paths such as oxidation of SO$_2$ by gas-phase oxidants should also play an important role in sulfate formation (Duan et al., 2019).

3.3 Influence of particle composition on AWC and sulfate formation

Besides RH, particle composition is another important factor to affect the AWC. According to the ions balance (Fig. S7a), ammonia was adequate to neutralize the anions in PM$_{2.5}$, which is consistent with the results in the literature (Y. Wang et al., 2020b). In addition, (81.5 ± 15.9) % (with the median of 87.1 %) of ionic anions (nitrate, chloride and sulfate) were neutralized by ammonium (Fig. S7b). This means NH$_4$NO$_3$, (NH$_4$)$_2$SO$_4$ and NH$_4$Cl should be the dominant form of nitrate, sulfate and chloride in PM$_{2.5}$, respectively. We further reconstructed the molecular composition from the ions based on the principles of aerosol neutralization and molecular thermodynamics (Kortelainen et al., 2017). The molecular concentrations were estimated according to the molar ratio of NH$_4^+$ to SO$_4^{2-}$ ($R_{NH_4^+/SO_4^{2-}}$) according to the following rules: (i) if 0<$R_{NH_4^+/SO_4^{2-}}$<1, NH$_4^+$ existed as the chemical forms of H$_2$SO$_4$ and NH$_4$HSO$_4$; (ii) 1<$R_{NH_4^+/SO_4^{2-}}$<2, NH$_4^+$ existed as (NH$_4$)$_2$SO$_4$ and NH$_4$HSO$_4$; (iii) if $R_{NH_4^+/SO_4^{2-}}$>2, then the fraction NH$_4^+$ corresponding to twice the amount of SO$_4^{2-}$ existed as (NH$_4$)$_2$SO$_4$, and the remaining fraction of NH$_4^+$ was associated with NO$_3^-$ and Cl$^-$; and (iv) the rest of NO$_3^-$, which was not neutralized by NH$_4^+$, was from NaNO$_3$. Figure 5a and b show the variation of the molecular composition as a function of RH in Shijiazhuang. Obviously, NH$_4$NO$_3$ and (NH$_4$)$_2$SO$_4$ were the major molecular components. Both of them showed upward trend as the RH increased. In particular, the fraction of NH$_4$NO$_3$ increased gradually from ∼10 % to ∼50 % when the RH increased from ∼30 % to 90 %. Correspondingly, the fraction of (NH$_4$)$_2$SO$_4$ decreased as the RH increased.

It should be noted that the deliquescence RH (DRH) of NH$_4$NO$_3$ (61.8 %) (Onasch et al., 1999) is lower than those values of NH$_4$Cl (78 %) (Hu et al., 2011) and (NH$_4$)$_2$SO$_4$ (80 %) (Lightstone et al., 2000). We further calculated the AWC attributed to the individual molecular component based on the growth factors and mass concentrations. As shown in Fig. 5c, the sum of the AWC of individual salts is underestimated by around 13 % compared to that calculated using the ISORROPIA II model (the grey line), because the mixing state was not considered in the former method. However, we can still draw a conclusion that NH$_4$NO$_3$ and (NH$_4$)$_2$SO$_4$ are the major contributors to the AWC. Especially NH$_4$NO$_3$
dominated the AWC when the RH ranged from 60 % to 80 %, in which the SOR and the $\gamma_{SO_2}$ were very sensitive to RH. These results suggest that NH₄NO₃ should be the most important mediator to AWC and subsequently the uptake of SO₂ in the transition regime of RH in Fig. 3a. It should be noted that (NH₄)HSO₄ has a lower DRH than NH₄NO₃ (Y. J. Li et al., 2017). However, 98.4 % of the data points showed an $R_{NH_4+/SO_2^-}$ higher than 2.0 in Shijiazhuang. This means that the contribution of (NH₄)HSO₄ to PM₂.₅ should be negligible because of the abundance of atmospheric NH₃ in northern China. In previous work, it has been found that SO₂ oxidation can be promoted by particulate nitrate through the accumulation of proton (Zhang et al., 2019) and the formation of NO⁻NO₃⁻ (Kong et al., 2014). Our results further showed the importance of NH₄NO₃ in the AWC, which possibly determines the phase state of particles and subsequently the uptake kinetics of SO₂ and the SOR as discussed above. To further confirm the role of NH₄NO₃ in the uptake of SO₂, uptake experiment of SO₂ on pure dust has been carried out at 2 % and 80 % RH. The corresponding $\gamma_{SO_2,BET}$ was $1.10 \pm 1.05 \times 10^{-7}$ and $1.66 \pm 0.28 \times 10^{-7}$ on pure dust sample in the presence of NH₃ and NO₂. However, as discussed above, it was 0 and $1.12 \pm 0.15 \times 10^{-5}$ on dust internally mixed with 33 % NH₄NO₃. This directly confirmed the role of NH₄NO₃ in SO₂ uptake via aerosol liquid water.

Figure S8 shows the dependencies of the AWC/PM₂.₅ and SOR on the fraction of the individual molecular component. Both the AWC/PM₂.₅ and SOR statistically increased as the fraction of NH₄NO₃ increased (Fig. S8a and d). A weak increase followed by a decrease was observed for the AWC/PM₂.₅ as the fraction of (NH₄)₂SO₄ increased, while a negative correlation between the AWC/PM₂.₅ and the fraction of NH₃Cl was observed. It did so for the SOR and the fraction of NH₃Cl. These phenomena were overall consistent with the sequence of their hygroscopicity. In addition, chloride is a primary pollutant mainly from coal combustion and biomass burning (Bi et al., 2019). Besides chloride, other primary particles from combustion such as soot, which were not accounted for in this work, might also decrease the uptake capability of water and subsequently be unfavorable for SO₂ uptake.

To assess the relative importance of sulfate and nitrate (the major SIA component) to AWC, the sensitivity of their fraction to AWC in Shijiazhuang was tested using the ISORROPIA II model and shown in Fig. S9. The base case means the AWC was calculated using the measured concentration of the ions. Then, we reduced the fraction of NH₄NO₃ or (NH₄)₂SO₄ from 0 to 80 % individually compared with the base case. Figure S9a shows the time series of the calculated AWC after reducing 50 % of NH₄NO₃ or (NH₄)₂SO₄. Reduction of either NH₄NO₃ or (NH₄)₂SO₄ resulted into obvious decrease of AWC during pollution events. In most cases, the reduction amplitude of AWC was larger when reducing 50 % of NH₄NO₃ than (NH₄)₂SO₄. Figure S9b shows the mean ratio of AWC at a certain reduction fraction of NH₄NO₃ or (NH₄)₂SO₄ to that under the base case. When NH₄NO₃ was reduced from 0 % to 80 %, the AWC linearly reduced from 0 % to 61.1 ± 0.1 % with a slope of 0.48 %.
It varied from 0 % to 66.0 ± 0.2 % for (NH₄)₂SO₄ (with a slope of 0.41 %). This means that the AWC is more sensitive to the fraction of NH₄NO₃ than (NH₄)₂SO₄ in Shijiazhuang. This also implies the importance of NH₄NO₃ in the observed high AWC in haze days. On the other hand, reducing 10 % of NH₄NO₃ can lead to a reduction of 5.2 ± 1.0 % AWC during haze days. Subsequently, we can roughly estimate that the SOR might be reduced by ~ 4 % through a linear interpolation according to the equation of the SOR and the AWC/PM₂.₅ (SOR = 0.15 ± 0.40 × AWC/PM₂.₅) fitted in Fig. 3c. This means reduction of NOₓ and NH₃ should lead to additional reduction of particulate sulfate.

3.4 Influence of other factors on sulfate formation

Several studies have proposed out that NO₂ can promote the oxidation of SO₂ on particle surfaces and in aqueous phase. For example, laboratory studies have found that parts per million (ppm) levels of NO₂ can promote sulfate formation on the surface of dust through NO₂+NO₃ which is disproportionated from N₂O₄ intermediate (He et al., 2014; Liu et al., 2012; Ma et al., 2008) or parts per million levels of NO₂ can promote the oxidation of SO₂ in the deliquesced oxalic acid (Duan et al., 2019). This is supported by the evidence that a high fraction of sulfate in PM₂.₅ is positively correlated with NO₂ concentration (Xie et al., 2015) and that a high PM₂.₅ concentration is accompanied with high ratio of NO₂/SO₂ in several case studies (He et al., 2014). The importance of the SO₂ oxidation by NO₂ in aqueous phase has also been confirmed in modeling studies (Cheng et al., 2016; Xue et al., 2016). However, this reaction path is still under debate because of the following reasons. (1) The concentration of NO₂ in laboratory studies was about 2 orders of magnitude higher than that in ambient air. This will affect the surface concentration of the intermediate (N₂O₄) and the concentration of solved NO₂ in aqueous phase. (2) The dissolved NO₂ concentration is highly sensitive to pH. The pH value in aerosol was 5.6–6.2 estimated in modeling study (Cheng et al., 2016). However, a recent work found that it varied from 3.8 to 4.5 at RH > 30 % and showed a moderate acidity because of the thermodynamic equilibrium between NH₃⁺ and NH₃ (Ding et al., 2019). (3) The previous calculations were conducted using a high reaction rate constant of the NO₂ reaction with dissolved S(IV) (Clifton et al., 1988; Cheng et al., 2016), while a smaller value was reported in the more recent study (Spindler et al., 2003; Tilgner et al., 2021). (4) The relative importance of each path depends on the concentration of the relevant pollutants including H₂O₂ and TMI (P. Liu et al., 2020). Therefore, it is necessary to verify the importance of this process by long-term observation at different environments.

Figure 6 shows the 2D kernel density graph of the sulfate fraction in soluble PM and the SOR in Shijiazhuang as a function of the concentration of different gas-phase pollutants. It should be pointed out that the SOR or the γSO₄ should be positively correlated to NO₂ concentration if it can promote the conversion of SO₂ to sulfate or the uptake of SO₂. However, both sulfate fraction and SOR were negatively correlated with the concentration of NO₂ from the point of view of statistics. A same trend was observed in Beijing (Fig. S10). This is similar to recent studies that observed the opposite correlation between SOR and NO₃ concentration in the Sichuan Basin (Tian et al., 2019) and in Beijing (Fang et al., 2019). This means that NO₂ concentration is statistically not a determining factor for sulfate formation in the atmosphere. This is well supported by the uptake kinetics of SO₂ measured using a flow tube reactor. As shown in Fig. 3a and b, when 50 ± 2.5 ppb of NH₃ presenting in the reactant gases, no difference was observable for the γSO₂,BET between the presence (red squares) and absence of 100 ± 2.5 ppb of NO₂ (white triangles). This is consistent with these previous studies that found NO₂ to be having no influence on SO₂ uptake when NH₃ was abundant in the atmosphere (Wu et al., 2019; Wang et al., 2021). In addition, it is consistent with the fact that H₂O₂ dominated the oxidation of Si(IV) in aerosol liquid water when RH was higher than 60 % (Fig. 3a). It should be pointed out that the γSO₂ at 80 % RH was 1.7 ± 0.3 × 10⁻⁶ on the mixture of dust and NH₄NO₃ in the absence of NH₃ and NO₂ (Fig. 3). It increased to 3.7 ± 0.2 × 10⁻⁶ in the presence of NO₂. This is consistent with the promotion effect of NO₂ for converting SO₂ to sulfate in the absence of NH₃ as observed in both a smog chamber (Wang et al., 2016) and a bubbling reactor (T. Chen et al., 2019). However, the enhanced uptake of SO₂ induced by NO₂ might be too low to be measured in the presence of NH₃. Therefore, the weak promotion effect by NO₂ alone cannot explain the negative correlation between the SOR and the concentration of NO₂ in Fig. 6f.

Both the fraction of sulfate and the SOR in Shijiazhuang statistically decreased as a function of SO₂ and CO concentration, respectively (Fig. 6b, c, g and h). This might be explained by the high concentration of primary aerosol components when pollution events occurred with high concentrations of primary gas-phase pollutants. However, the fraction of sulfate increased as a function of O₃ (Fig. 6d). When the O₃ concentration was greater than 50 ppb, the SOR slightly increased with the O₃ concentration (Fig. 6i). A more obvious positive dependence of sulfate fraction on O₃ concentration was observed in Beijing (Fig. S10e). This means oxidation capacity also plays an important role in sulfate formation, especially in Beijing. This is consistent with the recent finding that O₃ plays an important role in SO₂ oxidation at different locations (Fang et al., 2019; Tian et al., 2019; Duan et al., 2019). As shown in Fig. 6j, the SOR positively correlated with the concentration of NH₃ in Shijiazhuang. This means that NH₃ can promote the conversion of SO₂ to sulfate. This is well in agreement with laboratory studies that observed the promotion effect by NH₃ to the heterogeneous reaction of SO₂ on different mineral oxides (Yang et al., 2016). In addition, flow tube experi-

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ments were also carried out by exposing the internal mixing sample (2:1 dust and NH₄NO₃) to 200 ± 2.5 ppb SO₂ in the absence of NH₃ and NO₂ at 2% and 80% RH, respectively. As shown in Fig. 3a and b, the γSO₂,BET was zero regardless of the reactants under dry condition (2% RH), while it increased to (1.66 ± 0.28) × 10⁻⁶ at 80% RH. However, it was significantly smaller than the γSO₂,BET ((1.13 ± 0.21) × 10⁻⁵) in the presence of 50 ± 2.5 ppb NH₃ with or without 100 ± 2.5 ppb NO₂. These results further confirm that NH₃ can promote the uptake of SO₂ at high RH, possible through enhancing the solubility of SO₂ in water (T. Chen et al., 2019; Cheng et al., 2016; Wang et al., 2016), because the effective solubility of SO₂ can be enhanced due to the increase of the aerosol pH.

Aerosol acidity is one of the important factors affecting the sulfate formation and the partitioning of semivolatile gases in the atmosphere (Liu et al., 2021). As shown in Fig. S11, when aerosol pH is lower than 4.5, the oxidation rate of S(IV) in aerosol liquid phase decreases with decreasing pH, because the oxidation of S(IV) by transition metals is the dominant path and is decreasing with aerosol pH. However, the oxidation rate of S(IV) increases when the aerosol pH is higher than 4.5. This can be explained by the fact that the solubility and effective Henry’s law constant of SO₂ are positively dependent on pH (Cheng et al., 2016; Liu et al., 2021; P. Liu et al., 2020), which is consistent with the promotion effect of sulfate formation by NH₃.

4 Conclusions and atmospheric implications

Based on 1 year of observations, we confirmed that high PM₂.₅ mass concentration in pollution events usually coincided with the high sulfate concentration, the fraction of sulfate, and the SOR in both Beijing and Shijiazhuang. In Shijiazhuang, the SOR exponentially increased as a function of RH from the point of view of statistics, which was similar to the RH dependence of the γSO₂ on the model particles containing 33% NH₄NO₃ in the presence of NH₃. The SOR and γSO₂ linearly increased as a function of the fraction of aerosol water content in PM₂.₅. The enhanced uptake coefficient of SO₂ at high RH after the liquid-phase aerosol appeared might explain the increased SOR, because uptake of SO₂ was the rate-determining step for the conversion of SO₂ to particulate sulfate. NH₄NO₃ played an important role in the AWC, the phase state of aerosol particles and subsequently the uptake kinetics of SO₂ in haze days under high RH conditions.

The contribution of nitrate to PM₂.₅ is increasing in China (Li et al., 2018; Tian et al., 2019) due to the intensive emissions of NOₓ from steel production and cement manufacturing (Wu et al., 2018; Qi et al., 2017) as well as the increasing NOₓ emissions from traffic (Liu et al., 2007; Wang et al., 2011). The mean fraction of nitrate in PM₂.₅ was 21.4 ± 12.4% in Shijiazhuang and 15.8 ± 13.4% in Beijing. They were close to the reported values in PM₁.₀ during the summer of Beijing (24%) (Li et al., 2018) and in PM₂.₅ during the winter of Chengdu (23.3%) and Chongqing (17.5%) (Tian et al., 2019). It has been found that the fraction of nitrate and ammonium usually increases as a function of PM₂.₅ mass concentration (Li et al., 2018). Therefore, NOₓ should be an urgent air pollutant in the future in China even from the point view of its contribution to PM₂.₅ mass.

As observed in this work, NH₄NO₃ has an important contribution to PM₂.₅ mass concentration, the aerosol water content and subsequently the phase state of particles in the RH range of 60%–80%. Reduction of NOₓ emissions should lead to a decrease in NH₄NO₃ concentration and subsequently the AWC during serve pollution events. This will lead to an additional reduction of SO₂ uptake and the formation of particulate sulfate through aqueous reactions. Based on our rough estimation, 4% of sulfate might be reduced due to...
to aqueous reaction in Shijiazhuang if the mass concentration of NH$_4$NO$_3$ was reduced by 10%. More work is required to quantitatively assess the contribution of nitrate to sulfate formation from aqueous reactions in the future. It should be noted that ozone pollution becomes more and more important in China (Z. Chen et al., 2019; Ziemke et al., 2019). This requires us to harmoniously reduce NO$_x$ and volatile organic compounds in the near future. It is also important to take actions on NH$_3$ emission control in the future as NH$_3$ can significantly promote the uptake of SO$_2$ in liquid-phase aerosol.

Data availability. The experimental data are available upon request to the corresponding authors.

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Author contributions. YoL and XB designed the experiments. YoL and YuL wrote the paper. ZF, FZ, YZ, XF, CY, BC, YW, WD and JC carried out measurements at BUCT. XB and TJ carried out measurements at SJZ. YG, YZ and YuL carried out flow tube experiments. PL, YM and YoL performed sulfate formation calculations. YuL, FB, TP, YM, HH and MK revised the paper.

Competing interests. The authors declare that they have no conflict of interest.

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