The formation and mitigation of nitrate pollution: Comparison between urban and suburban environments

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Abstract. Ambient nitrate has been of increasing concern in $PM_{2.5}$, while there are still large uncertainties in quantifying the formation of nitrate aerosol. The formation pathways of nitrate aerosol at an urban site and a suburban site in the Pearl River Delta (PRD) are investigated using an observation-constrained box model. Throughout the campaigns, aerosol pollution episodes were constantly accompanied with the increase of nitrate concentrations and fractions at both urban and suburban sites. The simulations demonstrate that chemical reactions in the daytime and at night both contributed significantly to formation of nitrate in the boundary layer at the two sites. However, nighttime reactions predominately occurred aloft in the residual layer at the urban site and downward transport from the residual layer in the morning is an important source (53%) for surface nitrate at the urban site, whereas similar amounts of nitrate were produced in the nocturnal boundary layer and residual layer at the suburban site, which results in little downward transport of nitrate from the residual layer to the ground at the suburban site. We show that nitrate formation was in the volatile organic compounds (VOCs)-limited regime at the urban site, and in the transition regime at the suburban site, identical to the response of ozone at both sites. The reduction of VOCs emissions can be an efficient approach to mitigate nitrate in both urban and suburban areas through influencing hydroxyl radical (OH) and N_2O_5 production, which will also be beneficial for the synergistic control of regional ozone pollution. The results highlight that the relative importance of nitrate formation pathways and ozone can be site-specific, and the quantitative understanding of various pathways of nitrate formation will provide insights for developing nitrate and ozone mitigation strategies.

Keywords: nitrate, ozone, volatile organic compounds, N_2O_5, formation pathways, urban and suburban sites

1 Introduction

Particulate nitrate is a substantial chemical component of fine particles, which plays a significant role in the acid deposition, visibility reduction, hygroscopic properties, and radiative forcing (Li et al., 1993;Watson, 2002;Pathak et al., 2009;Xu and Penner, 2012;Zhang et al., 2017;Liu et al., 2020). Due to the larger emission reduction of SO_2 than NO_x and little change of NH_3 since the implementation of the clean air actions in China (Guo et al., 2018;Liu et al., 2019a;Zhai et al., 2021), a considerable increase in the nitrate fractions in aerosols has been observed in haze periods in the northern China Plain (Wen et al., 2018;Li et al., 2018;Lu et al., 2013;Fu et al., 2020), southern China (Pathak et al., 2009;Pathak et al., 2011) and eastern China (Griffith et al., 2015;Tao et al., 2018;Yun et al., 2018b;Li et al., 2018), which indicates the growing significance of nitrate in the formation of haze events. In addition, the photolysis of particulate nitrate can increase the production of sulfate and nitrous acid (HONO), implying the importance of nitrate in the synergetic enhancement of the atmospheric oxidizing capability in haze events (Gen et al., 2019;Zhang et al., 2020;Ye et al., 2016;Ye et al., 2017) , although the photolysis of particulate nitrate to produce HONO still remains highly uncertain (Romer et al., 2018). Hence, identifying and understanding the driving factors of nitrate formation are essential to establishment of optimized mitigation policies for fine particles.

Particulate inorganic nitrate is primarily produced through two processes: the photochemical reaction of hydroxyl radical (OH) and NO_2 during daytime (R1) and the heterogeneous uptake of N_2O_5 (R2–R5) during nighttime. The gaseous nitric acid (HNO_3) is produced by the reaction of OH and NO_2, and then reacts with ammonia (NH_3) to form particulate nitrate (Stelson and Seinfeld, 1982). The partitioning process of HNO_3 between the gas and particle phase is regulated by ambient temperature (*T*), relative humidity (RH) (Mozurkewich, 1993), aerosol pH and the abundance of NH_3 (R7) (Xue et al., 2014a;Yun et al., 2018b;Franchin et al., 2018). The pH value within a certain range plays an important role in the gas-particle partitioning of nitrate, which significantly impacts the nitrate formation (Guo et al., 2018;Lawal et al., 2018;Nenes et al., 2020).

$$OH + NO_2 \xrightarrow{k_1} HNO_3(g) \quad \text{R1}$$

$NO_2 + O_3 \xrightarrow{k_3} NO_3 + O_2$ R2

$NO_3 + NO_2 \overset{k_4}{\leftrightarrow} N_2O_5$ R3

$NO_3 + VOCs \rightarrow products$ R4

$N_2O_5 + (H_2O \text{ or } Cl^-) \xrightarrow{k_5} \varphi ClNO_2 + (2-\varphi)NO_3^-(p)$ R5

$k_5 = \frac{\omega 1 * \gamma * Sa}{4}$ (Eq.1)

$ClNO_2 \xrightarrow{k_6} Cl\cdot + NO_2$ R6

$k_6 = J_{ClNO_2}$ (Eq.2)

$HNO_3(g) + NH_3(g) \overset{k_7}{\leftrightarrow} NO_3^-(p) + NH_4^+(p)$ R7

The heterogeneous uptake reaction of N_2O_5 can occur on the surface of water or chlorine-containing particle (R5); and the reaction constant (k_5) is described by Eq.1, where φ is the production yield of $ClNO_2$ in R5, ω_1 is the average molecular speed of N_2O_5 (m s^{-1}), γ is the uptake coefficient of N_2O_5 and S_a (m^2 m^{-3}) is the aerosol surface area concentration in Eq.1. The nitryl chloride ($ClNO_2$) produced by the heterogeneous uptake reaction of N_2O_5 at night would photolysis in the next morning, which would produce chlorine atom and NO_2 (R6). Here the reaction rate k_6 was denoted as the photolysis rate of $ClNO_2$ (J_{ClNO_2}). The heterogeneous uptake reaction of N_2O_5 is affected by the uptake coefficient (γ) and the production yield of $ClNO_2$ (φ), which cannot be directly measured and are significantly impacted by the aerosol components and ambient RH (Bertram and Thornton, 2009;Bian et al., 2017;McDuffie et al., 2018a;McDuffie et al., 2018b). Thus, the nocturnal contribution to nitrate formation still has great uncertainty.

With the radiative cooling in the afternoon, the mixed layer decoupled into a steady, near surface nocturnal boundary layer (NBL) and a residual layer (RL), which is a neutral layer and formed aloft during the turbulence attenuation process (Prabhakar et al., 2017). The heterogeneous uptake of N_2O_5 in the nocturnal boundary layer is greatly disturbed in the presence of fresh NO emissions, which titrate the NO_3 radical within the stagnant boundary layer (Geyer and Stutz, 2004;Li et al., 2020;Chen et al., 2020). However, aircraft observations in California and Utah in the US have revealed that active uptake of N_2O_5 in the residual layer contributed a major portion of the near-surface nitrate accumulation during the morning transport from aloft (Brown et al., 2006;Chow et al., 2006;Prabhakar et al., 2017;McDuffie et al., 2019;Womack et al.,

2019). Similarly, ground- and tower-based field observations also pointed out the important contribution of this pathway to the rapid increase of near-surface nitrate concentrations in Beijing, China (Wang et al., 2018a;Chen et al., 2020). However, under different atmospheric conditions, the relative importance of nitrate production varies significantly within the residual layer (McDuffie et al., 2019;Tang et al., 2021), giving widely varying relative contributions of the major chemical pathways to nitrate pollution among different sites (Wang et al., 2018a;Womack et al., 2019;Chen et al., 2020;Lin et al., 2020). A comprehensive understanding of the nitrate production in the residual layer is required to quantify the contributions of different formation pathways to nitrate pollution.

The nitrate production from the reaction of OH and NO_2 pathway during daytime is well-understood, and the control of NO_x emission is commonly considered as an effective strategy to reduce ambient nitrate. However, several studies reported that the efficiency of NO_x reduction in nitrate control is limited, and it may enhance nitrate production under some conditions (Womack et al., 2019;Dong et al., 2014;Hou et al., 2019). The study by Womack et al. (2019) showed that both nitrate and ozone were VOCs-limited in Salt Lake City, suggesting that VOCs control would effectively reduce nitrate. Similarly, modeling studies also found that the nitrate formation was more sensitive to the change in VOCs concentrations over the northern and eastern China (Dong et al., 2014;Lu et al., 2019;Fu et al., 2020). The sensitivity of nitrate production to both NO_x and VOCs in different regions should be comparatively evaluated, which could provide helpful implications in formulating effective control strategies for the mitigation of aerosol pollution.

In recent years, the nitrate formation in haze episodes has been studied in northern China (Liu et al., 2015;Wang et al., 2017a;Wen et al., 2018;Fu et al., 2020;Chen et al., 2020), eastern China (Tao et al., 2016;Lin et al., 2020) and southern China (Qin et al., 2017;Tao et al., 2018;Yun et al., 2018b;Su et al., 2020), and the important contribution of the heterogeneous uptake of N_2O_5 in the nighttime has been discussed (Wang et al., 2017b;Yun et al., 2018b;Yun et al., 2018a;Chen et al., 2020). However, these ground-based observations rarely considered the potential contributions of reactive uptake of N_2O_5 aloft in the residual layer, which could be an important source of near-surface nitrate concentrations. In addition, few studies have comprehensively evaluated the relative influence of NO_x and VOCs reductions on nitrate production in the urban and suburban areas (Hou et al., 2019).

In this study, we present the results from the ground- and tower-based measurements in both urban and suburban areas in southern China. An observation-constrained box model was used to simulate the production rates of nitrate from different formation pathways, and to compare the effects of reducing NO_x and VOCs emissions in both urban and suburban areas. This work provides new insights into the synergetic mitigation of particle and ozone pollution, which can guide development of the most effective nitrate control strategies.

2 Method and data

2.1 Field observation

The ground-based field measurements were conducted at both an urban site in Guangzhou and a suburban site in Heshan. The tower-based measurements were conducted at an urban site in Guangzhou. The ground-based study in Guangzhou was carried out from late September to mid-November in 2018 at the Institute of Geochemistry (GIG), Chinese Academy of Sciences (23.1°N, 113.2°E), which is a typical urban site surrounded by a residential area and traffic avenues (Fig. 1). The instruments were deployed on the top of the 25-m building at GIG site. The ground-based measurement at the suburban site was performed from late September to mid-November in 2019 at the supersite of Heshan county (22.7°N, 112.9°E), which is approximately 50 km southwest to Foshan and 80 km southwest to Guangzhou, and is frequently influenced by anthropogenic emissions from upwind Guangzhou-Foshan mega-city areas. The tower-based measurements were conducted simultaneously at the ground and 448 m on the Canton Tower from late September to mid-November in 2018 concurrent with the measurements at the GIG site, which are approximately 5.7 km apart each other (Fig. 1).

The chemical components of PM_1, trace gases, and non-methane hydrocarbons (NMHC), and particle BC content and particle size distribution were both measured at the GIG and Heshan sites, whereas only trace gases (NO_x and O_3) and meteorological parameters were measured at the Canton Tower site. The non-refractory chemical compositions of PM_1 (NR-PM_1), including organics (Org), sulfate (SO_4^{2-}), nitrate (NO_3^-), ammonium (NH_4^+), and chloride (Cl^-) were measured using a high-resolution time-of-flight aerosol mass spectrometer (HR-ToF-AMS, Aerodyne Research Inc., US) (Hu et al., 2016;Chen et al., 2021). Black carbon (BC) was measured using an

aethalometer (AE33, Magee Scientific Co., US). Particle number size distribution was measured using a scanning mobility particle sizer with an aerodynamic diameter ranging from 10 to 650 nm (SMPS, TSI, US) and aerosol particle sizer ranging from 500 nm to 20 μm (APS, TSI, US). Details on the limit of detection and accuracy of the instruments are presented in Table S1~ Table S3.

HNO_3, N_2O_5, and $ClNO_2$ were measured using iodide-time-of-flight chemical ionization mass spectrometry (Iodide-TOF-CIMS, Aerodyne Research Inc., US) (Wang et al., 2020b;Ye et al., 2021). The non-methane hydrocarbons (NMHC) were measured using online GC-MS-FID (Wuhan Tianhong Co., Ltd, China) (Yuan et al., 2012) (Table S4). The concentrations of oxygenated VOCs (OVOCs), including formaldehyde (HCHO) and acetaldehyde (CH_3CHO), the sum of methyl vinyl ketone (MVK) and methacrolein (MACR) were measured via a high-resolution proton transfer reaction time-of-flight mass spectrometry (PTR-ToF-MS, Ionicon Analytik, Austria) (Wang et al., 2020a;Wu et al., 2020). HONO was detected using a long path absorption photometer (LOPAP) at the GIG site (Yu et al., 2021), and was measured by the gas and aerosol collector (GAC) instrument at the Heshan site (Dong et al., 2012;Yang et al., 2014). NH_3 was also measured by two sets of instruments: a cavity ring-down spectroscopy (CRDS, Picarro, US) was used at the GIG site and the GAC instrument was used at the Heshan site (von Bobrutzki et al., 2010).

In addition, trace gases (O_3 (49i), NO_x (42i), CO (48i) and SO_2 (43i)) (Thermo Scientific, US) and meteorological parameters (i.e., wind speed (WS), wind direction (WD), temperature (*T*), relative humidity (RH) and pressure (*P*)) (Vantage Pro 2, Davis Instruments Co. , US) were simultaneously measured during these campaigns. The photolysis frequencies of O_3, NO_2, HCHO, and HONO (PFS-100, Focused Photonics Inc., China) were also measured during the campaigns. Considering the integrity and temporal coverage of the measurements, we mainly focus on the investigated periods from October 7 to 29, 2018, at the GIG site and from October 16 to November 16, 2019, at the Heshan site.

2.2 Box Model description

A zero-dimensional observation-based box model (F0AM) (Wolfe et al., 2016) was used to simulate the production of nitrate in this study. The F0AM box model uses a subset of the Master Chemical Mechanism (MCM) v3.3.1 (Saunders et al.,

2003;Jenkin et al., 2003;Bloss et al., 2005), which explicitly describe chemical reactions of VOCs, RO_x radicals (including OH, HO_2 and RO_2), ozone and nitrate, and was widely used in laboratory and theoretical researches (Edwards et al., 2017;Anderson et al., 2017;D'Ambro et al., 2017;Womack et al., 2019).

In this study, the box model was constrained by observations of NMHCs, HCHO, CH_3CHO, NO, CO, CH_4, HONO, and meteorological parameters (i.e., photolysis rates, RH, *T* and *P*) measured at the GIG and Heshan sites. To investigate the convection of nitrate between the residual layer and the surface, the box model was split into two boxes at night (from 17:00 to 6:00 of the following morning) to separately represent the nocturnal boundary layer and the residual layer, respectively (Womack et al. (2019) (Fig. S1).

The simulation of the residual layer at the GIG site was constrained by the observation data from 488 m at the Canton Tower, while the simulation of the residual layer at the Heshan site was freely evolved from sunset time using the ground observation data of Heshan. The detailed model settings are described in Text S1, and the agreement between the observation data and simulations at the GIG and Canton Tower sites supports the use of similar simulation of the residual layer at the Heshan site. The model was operated in a time-dependent mode with a 5-min resolution. It was run for a 72-hour spin-up time to build steady-state concentrations for secondary pollutants that were not constrained during simulation. To prevent the build-up of long-lived species to unreasonable levels, an additional physical dilution process was applied in the model(Lu et al., 2017;Decker et al., 2019;Novak and Bertram, 2020;Liu et al., 2021;Yun et al., 2018b). To achieve agreement with the observation, a life time of 24 h and 8 h were used at the GIG and Heshan site, respectively. The sensitivity tests with different dilution constant at the GIG and Heshan site were shown in Fig.S2 and Fig.S3, respectively. The background concentrations for ozone and CH_4 were set as 30 ppb and 1.8 ppm, respectively (Wang et al., 2011).

The nocturnal production of nitrate from N_2O_5 hydrolysis and the subsequent reactions (R5 and R6) are added to the box model. γ and φ are calculated using the observation-based empirical parameterization method from Yu et al. (2020), where the impacts of nitrate, chloride, and aerosol liquid water content (ALWC) were evaluated to better represent the observed γ. The average values of γ were 0.018±0.01 and 0.019±0.01 at the GIG and Heshan sites, respectively, which were comparable with the

observed mean data of γ (0.020±0.019) at the Heshan site in 2017. The φ used in this study were 0.18±0.15 and 0.20±0.23 at the GIG and Heshan sites, which were slightly lower than the observed mean data of φ at the Heshan site (0.31 ± 0.27) in 2017 (Yu et al., 2020). The chemical compositions of fine particle were not measured at the Canton Tower site, thus values of γ and φ in the residual layer were assigned equal to those of the nocturnal boundary layer. The γ and φ exhibited complicated nonlinear dependence on aerosol composition, aerosol liquid water and RH (Bertram and Thornton, 2009;McDuffie et al., 2019;Yu et al., 2020), such that γ and φ has positive and negative dependence with RH, respectively. There was higher RH, and lower chloride at the 488 m site, compared to the ground site of Canton Tower. The nitrate concentration was comparable at the 488 m site to the ground site in the study of Zhou et al. (2020). Combined with the higher RH and lower $PM_{2.5}$ concentrations in the residual layer in this study (as shown in Fig.S4), we inferred the negative deviations for γ and positive deviations for φ in the residual layer. The dry aerosol surface area concentration (S_a) was calculated from the particle number size distribution and calibrated to the actual atmospheric S_a using the RH-dependent hygroscopic growth factor (*f*(RH)). The f(RH) was estimated from the aerosol composition measured by AMS and the aerosol liquid water content, which included the inorganic-associated and organic-associated water. The sum of inorganic-associated water estimated from ISORROPIA thermodynamic model and organic-associated water estimated from the dry organic aerosol mass, was used to calculate the growth of wet matter contributions, as described in the study of McDuffie et al. (2018a). J_{ClNO_2} was scaled from measured NO_2 photolysis frequencies divided by a factor of 30 (Riedel et al., 2014).

The equilibrium coefficient between HNO_3 and particulate nitrate is incorporated into the box model as a pseudo-first-order reaction (Eq.3 and 4) through the equilibrium absorption partitioning theory (Jacob, 2000;Yuan et al., 2016):

$$HNO_3\ (g) \underset{k_{8b}}{\overset{k_{8f}}{\Leftrightarrow}} NO_3^-\ (p) \qquad \text{R8}$$

$$k_{8f} = \left(\frac{R_a}{D_g} + \frac{4}{\omega * \alpha}\right)^{-1} * S_a \qquad \text{(Eq.3)}$$

$$k_{8b} = \left(\frac{R_a}{D_g} + \frac{4}{\omega * \alpha}\right)^{-1} \frac{S_a}{K_{eq}} \qquad \text{(Eq.4)}$$

where R_a is the radius of nitrate particles (m), D_g is the gas-phase molecular diffusion coefficient ($m^2\ s^{-1}$), ω is the mean molecular speed of HNO_3 ($m\ s^{-1}$), α is the mass

accommodation coefficient of HNO_3, and K_{eq} represents the equilibrium constant of HNO_3 and nitrate. These coefficients are the same as those in the chemical aqueous-phase radical mechanism (CAPRAM) (Ervens et al., 2003;Wen et al., 2015).

The empirical kinetic modeling approach (EKMA) is used here to identify the sensitivity of ozone and nitrate to the variations of NO_x and VOCs. The observed diurnal average conditions are used as the input for the base simulation. Sensitivity tests are conducted by increasing and decreasing initial anthropogenic VOCs (AVOCs) and NO_x concentrations by a ratio ranging from 0.1 to 2.0 with 20 equal-distance steps without changing other parameters in the model (Tan et al., 2018;Lyu et al., 2019;Womack et al., 2019). The maximum concentration of ozone and nitrate in each scenario are plotted to generate the contour plots of the respective isopleths. Isoprene was included in the simulation as biogenic VOC (BVOC). Reducing BVOCs such as isoprene is impractical, so it is not scaled with AVOCs concentrations in the sensitivity simulations on control of precursors.

Since the N_2O_5 is affected by the chemistry between ozone and VOCs, constraining N_2O_5 concentrations with the change in NO_x ratio arbitrarily during the isopleth simulations is improper. Thus, we set the simulation of base case (S0) without N_2O_5 constrained. To evaluate the results of the base case, we design another simulation with N_2O_5 constrained (S1) and compare the two simulated nitrate with the observation in Fig. S5. The model scenarios were described in Table S5 in detail. The base case simulation (S0) was comparable to the observation. The simulated nitrate with N_2O_5 constrained (S1) during October 9 to 10, 2018 was observed to be much higher compared to both the observations and base case simulation (S0) at the GIG site, which suggest that high concentrations of ambient N_2O_5 measured during this short period may not contribute significantly to nitrate formation (Fig. S6). Overall, the simulated nitrate of base case without N_2O_5 constrained agreed well with the observation suggesting the robustness of the model simulations.

Gaussian error propagation was used to evaluate the uncertainties about measurement parameters and reaction rates in the model, as described in Lu et al. (2012). The uncertainties of various measurement parameters (VOCs, trace gases, meteorological parameters, etc.) ranged from 0 to 20%, and uncertainties of reaction rates are in the order of ~20% (Lu et al., 2012).

3 Results and discussion

3.1 Overview of nitrate concentrations during the campaign

The temporal variations of mass concentrations of the major chemical components in PM_1 are shown in Fig. 2. The mean concentration of PM_1 was 41.7±23.1 μg m^{-3} at the GIG site during the investigated period, which was comparable with that at the Heshan site (40.6 ±15.5 μg m^{-3}). The aerosol composition differed between sites, with inorganic ions (sulfate, nitrate, and ammonia) higher and organic matter lower at the GIG site compared to the Heshan site.

Although the mass concentrations at the two sites were comparable, the mass fraction of nitrate in PM_1 at the GIG site increased from 10% to 33% as the mass concentration of PM_1 increased from 20 to 130 μg m^{-3} (Fig. 3), while the fraction of nitrate increased from 10% to 20% at the Heshan site, suggesting that nitrate plays a more important role in the increase in PM_1 at the urban site than that at the suburban site. The significant increasing ratio of nitrate fraction from clean condition to polluted condition (~ 43%) was also revealed in the airborne observation in Utah Valley, US (Franchin et al., 2018). In addition, although the concentration of sulfate was higher than that of nitrate during most of the sampling periods, as PM_1 increased the mass concentration ratio of nitrate/sulfate increased from 0.5 to 2.0 at the GIG site and from 0.5 to 1.5 at the Heshan site. The higher ratios of nitrate/sulfate during the polluted periods implies that reducing nitrate may be essential for reducing the occurrence of PM pollution in southern China. The increasing contributions of nitrate to PM_1 in this study were similar with those observed in northern China during haze pollution (Yang et al., 2017;Fu et al., 2020;Wen et al., 2015;Liu et al., 2015), suggesting the significance of nitrate mitigation to further reduce mass concentrations of fine particles in China.

The diurnal patterns of mean nitrate, NH_3, NO_2 and HNO_3 concentrations observed at the GIG and Heshan sites are shown in Fig. 4. The highest nitrate concentration was observed in the morning at the GIG site and during nighttime at the Heshan site, suggesting differences in the processes that dominated the formation of nitrate at the two sites. At the GIG site, nitrate rapidly increased from 4:00 to 9:00, but the concentrations of NH_3 and HNO_3 increased slowly, which suggests the minor contribution of direct production of HNO_3 from the reaction of OH and NO_2. The increase of nitrate during this period might be associated with the downward transport from the residual layer to the ground. The diurnal variations in O_3 and NO_x measured

at the GIG and Canton Tower sites are shown in Fig. 5. The ground-based observations at the Canton Tower showed similar variation patterns of O_3 and NO_x to the GIG site. However, the average concentration of O_3 at 488 m of Canton Tower site was 2.4 times higher than that at the GIG site during nighttime, and the lower nocturnal concentrations of NO (nearly zero) at the 488 m site would enhance the production of NO_3 and N_2O_5 (Wang et al., 2018b;McDuffie et al., 2019). Therefore, heterogeneous uptake of N_2O_5 during nighttime may be active at 488 m at urban site, which will be further investigated in Section 3.2. At the Heshan site, nitrate increased sharply in the early nighttime (before midnight), which may be attributable to the shallow nocturnal boundary layer or the enhanced nocturnal N_2O_5 heterogeneous uptake reactions. Subsequently, there was a significant increase in nitrate from 7:00 to 9:00. The concentration of NH_3 showed variation pattern that was similar with that of nitrate and increased after 7:00, while the concentrations of HNO_3 and NO_2 showed a decreasing trend from 7:00 to 9:00 at the Heshan site. The different growth characteristics of nitrate and the variation patterns of precursors at the two sites may be related to different formation processes, which will be discussed in detail later.

In this study, the wind speeds in the investigated periods at the GIG and Heshan sites were generally below 2 m s^{-1} (Table S6), which suggests that regional transport may have limited contributions to the abundance of nitrate at the observation sites. Therefore, the discussion of the chemical formation process of nitrate in this study focuses on local production.

The molar ratios of $[NH_4^+]$ to the sum of $2\times[SO_4^{2-}]+[NO_3^-]$ are calculated (Fig. S7) to determine whether there was enough NH_4^+ to neutralize nitrate. The molar ratios were approximately 1.0 at both GIG and Heshan sites, suggesting both NH_3 and HNO_3 were crucial precursors for nitrate formation. Based on these discussions, we will discuss the NH_3 effect on the nitrate partitioning firstly by thermodynamic ISORROPIA II model. The nitrate chemical formation pathways, which is mainly attributable to the production of HNO_3 and/or heterogeneous uptake of N_2O_5 combining the box model, will be discussed in Sec. 3.2.

The ISORROPIA II model setting is described in Test S2 in detail. The ISORROPIA II modeled results of nitrate, ammonium, HNO_3, and NH_3 at the GIG and Heshan site were displayed in Fig.S8 ~ Fig.S9. The particle-phase nitrate and ammonium at the GIG site showed a bit overestimation, while the gas-phase HNO_3, and NH_3 showed overestimation at the Heshan site. Overall, the simulated components

showed good correlations with the observed concentrations at both sites. We use the ISORROPIA II model results to evaluate the particle fraction of nitrate in the sum of HNO_3+nitrate ($\varepsilon(NO_3^-)$) against aerosol pH. Aerosol pH, which depends on the aerosol acidity and water content, is calculated by the following equation:

$$\mathrm{pH} = -\log_{10}\frac{1000\,H^+_{air}}{\mathrm{ALWC}} \quad \text{(Eq.5)}$$

where H^+_{air} ($\mu g\ m^{-3}$) is the hydronium concentration of the equilibrium particle and ALWC ($\mu g\ m^{-3}$) is the aerosol water content from ISORROPIA II simulation.

The $\varepsilon(NO_3^-)$ against pH at the GIG and Heshan site are shown in Fig.6. The pH data are colored by relative humidity and fit to an "s-curve" as in Guo et al. (2018). The clustering of pH data mainly located between 1~ 3, and the $\varepsilon(NO_3^-)$ are sensitive to the change of pH. To further evaluate the sensitivity of NH_3 and sulfate on this effect, the input of total ammonium (NHx, ammonium + NH_3) and sulfate were reduced from 10% to 90% relative to the ISORROPIA II base model, respectively, while keeping other parameters constant. The response of simulated nitrate concentration and aerosol pH to changes in NHx and SO_4^{2-} are shown in Fig.7. The nitrate concentration decreased with the reduction of NHx, and had little variation with the reduction of SO_4^{2-} (Fig.7 (a ~ b)) at both sites. Along with the reduction of NHx, the pH values decreased significantly (Fig.7 (c ~ d)), which caused the further decrease of $\varepsilon(NO_3^-)$. The pH values showed a bit increase with the reduction of SO_4^{2-}, which may be caused by that there would be more available ammonium neutralized the hydronium. It is consistent with the study of Guo et al. (2018) and Nenes et al. (2020), suggesting the partitioning of nitrate was also affected by the NH_3 in the pH values between 1~3. Thus, the control of NH_3 is effective for the reduction of nitrate by affecting the partitioning process of nitrate at both GIG and Heshan site in this study. The partitioning of nitrate increased with the reduction of sulfate suggests the limited role of sulfate reduction on the mitigation of nitrate.

3.2 Contributions of different pathways to nitrate formation

To further investigate the chemical formation pathways of nitrate, which related to the photochemical and heterogeneous reactions, we adopt the box model results to simulate the contribution of different pathways to nitrate formation. The temporal variations in simulated and observed nitrate concentrations at the GIG and Heshan sites are presented in Fig. 8; simulated and observed nitrate showed similar concentrations and variation patterns. The diurnal variation of simulated nitrate is compared with the

observation in Fig.S10. The diurnal simulated nitrate was comparable with the observation at the GIG site, especially when considering the vertical transport from the residual layer in the morning. Unlike the GIG site, the diurnal simulated nitrate performed higher in the daytime, and little bit lower in the late nighttime, compared with the observation. It may be related to the lack of quantitative transport in the box model. The box model performance was evaluated using the mean bias (MB), index of agreement (IOA), and correlation coefficient (*r*) (Table S7) (Liu et al., 2019b;Lyu et al., 2017;Wang et al., 2019;Curci et al., 2015). The IOA was larger than 0.7 and r was larger than 0.5 at both sites, indicating good agreements between simulated and observed nitrate concentrations. The temporal variations in simulated N_2O_5 and $ClNO_2$ concentrations were higher than the observations at the Heshan site as shown in Fig. S6 (c, d), the simulated results at the GIG site from October 9 to 10 were significantly lower than the observations (Fig.S6 (a, b)). The abnormally high observed concentrations of N_2O_5 and $ClNO_2$ that lasted for short periods (10-30 minutes) at the GIG site may be caused by transported air masses from upwind regions or vertical transport without well-mixed with fresh urban NO emissions. Simulation of these near-instantaneous processes transported to the site using a box model is difficult, as box model is more suitable to simulate the well-mixed airmass with little transport effects. However, the simulated nitrate concentrations without observed N_2O_5 constrained was adequately comparable with the observations as shown in Fig. S5, implying the influence of the instantaneously high concentrations of N_2O_5 on nitrate formation was negligible at the GIG site.

Based on these simulation results, we calculated the daily-averaged contributions of the two different reaction pathways to the nitrate concentration - the daytime production from OH + NO_2 reaction and the nighttime production from N_2O_5 uptake reaction in the nocturnal boundary layer and in the residual layer. The nitrate produced in the residual layer is only gradually mixed to the surface as the boundary layer develops during the following morning, while the nitrate contributed to the boundary layer column concentration always included the N_2O_5 uptake in the residual layer during the whole nighttime (Wang et al., 2018a;Womack et al., 2019). The calculation methods to determine contribution to the boundary layer column concentrations and to ground-level nitrate concentrations should be distinguished.

To calculate the contribution to the boundary layer column concentration, the integral of the nitrate production rate from N_2O_5 uptake from both the nocturnal surface

layer and the residual layer directly contribute to nitrate column concentrations layer during the whole nighttime, weighted as 0.4 and 0.6 based on their altitude fractions of the two layers, respectively. This calculation for the contributions to column concentration is the same as the methods presented by Wang et al. (2018a) and Womack et al. (2019). However, to quantify the contribution of nitrate produced from the residual layer to the ground nitrate concentration, one must account for the dynamic exchange between the residual layer and the surface-based boundary layer that develops during daytime. The integral time for this dynamic exchange was assumed from 6:00 to 10:00 in the morning. Detailed descriptions of the calculations are provided in Text S3 in Supplementary Materials. The calculation about partitioning process from OH and NO_2 reaction in the daytime was the same in the two methods mentioned above, which was the partition part of the integral of the OH and NO_2 reaction during the daytime.

The contributions of nitrate to the boundary layer column concentration (i.e. average from ground to 1000 m) are shown in Fig. 9a. The contribution of nitrate production rate from N_2O_5 uptake in the residual layer was 17.9 μg m^{-3} day^{-1} at the GIG site, which was much greater than the N_2O_5 uptake in the nocturnal boundary layer (0.4 μg m^{-3} day^{-1}). This may be caused by the fresh NO surface emissions, which titrate the NO_3 radical and ozone in the nocturnal boundary layer, as the mean NO concentration during the nighttime at the GIG site was 12.1 ppb. The contribution from nocturnal nitrate production in the boundary layer was comparable with the contribution from OH and NO_2 reaction (13.2 μg m^{-3} day^{-1}) during the daytime. In contrast to the GIG site, the contribution of nitrate production rate from N_2O_5 uptake in the nocturnal boundary layer (6.2 μg m^{-3} day^{-1}) was comparable with that in the residual layer (4.4 μg m^{-3} day^{-1}) at the Heshan site. The similar nitrate concentration and production rate from N_2O_5 uptake between the nocturnal boundary layer and residual layer in Fig.S12 (c, d) was due to smaller NO emissions at the Heshan site. The results demonstrate that nocturnal nitrate production plays an important role in nitrate production in the boundary layer, with nighttime contributions of 58% at the urban site and 35% at the suburban site.

The relative magnitudes of the contributions to the daily-averaged surface nitrate differ somewhat from the contributions to the entire boundary layer. The contributions from the three major pathways to surface nitrate concentrations at the two sites are compared in Fig. 9b. At the GIG site the nitrate production rate from the OH and NO_2

reaction and downward transport from the residual layer were 13.2 μg m^{-3} day^{-1} and 16.6 μg m^{-3} day^{-1}, contributing 43% and 53% of ground-level nitrate concentrations, with a minor contribution (1.1 μg m^{-3} day^{-1}) from the production of N_2O_5 uptake in the nocturnal boundary layer. This is similar with the results in Fig.9a, implying a large nitrate contribution from N_2O_5 uptake in the residual layer, but not in the nocturnal boundary layer at the urban site.

However, at the suburban Heshan site (Fig.9b), downward transport from the residual layer made no contribution to the surface nitrate concentration, which was smaller than the contribution of nitrate from the residual layer in Fig. 9a. This is due to the similar nitrate production rate from N_2O_5 uptake between the nocturnal boundary layer and residual layer (see Fig. S12), inducing negligible convection between the two layers as the result of small concentration gradient (Brown et al., 2003;Baasandorj et al., 2017;Prabhakar et al., 2017). The nitrate production rate from OH and NO_2 reaction (19.9 μg m^{-3} day^{-1}) and nocturnal N_2O_5 uptake (15.6 μg m^{-3} day^{-1}) were the major nitrate formation pathways, which contributed 56% and 44% to the surface total nitrate production, respectively. Therefore, the importance of residual layer contribution to the surface nitrate can vary significantly and should be comprehensively evaluated in different environments. In addition, the nitrate contributions to the surface concentrations and boundary layer column concentrations can also be different in different regions, which should be clarified and distinguished in future studies.

In summary, the N_2O_5 uptake reaction was active in the residual layer both at urban and suburban sites, the downward transport from the residual layer was a significant contributor to surface nitrate at the urban site, but not at the suburban site. This is attributable to the titration of the NO_3 radical and ozone by fresh NO emissions during the stagnant boundary layer at the urban site, resulting in the large difference of nitrate production between the residual and nocturnal boundary layers. In contrast, at the suburban site, lower NO emissions favored NO_3 production and heterogeneous uptake of N_2O_5 both in the nocturnal boundary layer and the residual layer. The horizontal transport in the residual layer from nocturnal jets may contribute to the different nitrate production at urban and suburban sites, which has been discussed in the research of Chow et al. (2006) and Brown et al. (2006). Due to the limitation of box model, this issue could be studied by the chemistry transport model in further research.

3.3 Control of NOx and VOCs as mitigation strategies of nitrate

Overall, the contributions of nitrate from the three major pathways, all involving NO_x and ozone, suggest that nitrate formation depends not only on the reactions of NO_x but also is closely associated with the VOCs-NO_x-O_3 chemistry. Therefore, the influence of both NO_x and VOCs reduction on nitrate production should be considered in formulating policies to control aerosol pollution.

In this study, we adopted the widely used EKMA approach, generally used for ozone sensitivity analysis (Edwards et al., 2014;Mazzuca et al., 2016;Xue et al., 2014b;Wang et al., 2015) to investigate the response of nitrate formation in changing emissions of VOCs and NO_x. The dependence of simulated nitrate concentrations with changing of VOCs and NO_x concentration allow to construct isopleths of nitrate and ozone production at the GIG and Heshan sites, as displayed in Fig. 10. The production of nitrate and ozone were in the VOCs-limited regime at the GIG site, and in the transition regime at the Heshan site, where nitrate and ozone are sensitive to both VOCs and NO_x reduction. As shown in Fig. 11, the reduction of NO_x emissions from 0 ~ 70% would increase nitrate and ozone concentrations at the GIG site, but decrease those concentrations at the Heshan site. The decrease in VOCs concentrations would decrease nitrate and ozone concentrations at both sites. These results suggest that control of VOCs emissions will efficiently reduce nitrate and ozone production in both urban and suburban areas, but control of NO_x emissions will give different responses between urban and suburban area for both ozone and nitrate. Fig. 11 show that the nitrate sensitivity to the reduction of VOCs and NO_x emissions was identical to the response of ozone at both sites. These results demonstrate the possibility of synergetic control for nitrate and ozone at both urban and suburban sites through VOCs control.

The accuracy of the isopleth plots in Fig. 10 depends on several variables and parameters included in the box model. Figs S13 ~14 show the results of simulation experiments on the dependence of the isopleths upon changing various parameterization for estimating HONO concentrations, N_2O_5 uptake coefficient, and $ClNO_2$ yields as described in Text S4. The sensitivity regime of nitrate and ozone did not change, although the peak concentrations of ozone and nitrate did change, which supports the reliability of the results discussed above.

As nitrate and ozone exhibit similar sensitivity to the reduction of NO_x and VOCs, different VOCs/NO_x ratios may point to different control strategies. In the cases of the

Heshan and GIG sites, the reduction of NO_x can adequately control nitrate production with a VOCs/NO_x ratio of 1.8 at the Heshan site, while a contrary result can be found at the GIG site (with a VOCs/NO_x ratio of 0.8) with a less than 70% reduction of NO_x emission. The simulated results at the GIG site agree well with those reported in the urban areas of Shanghai in China (Dong et al., 2014) and the Salt Lake City and San Joaquin Valley in the US (Betty and Christian, 2001;Womack et al., 2019), which all emphasized the decrease of nitrate production with the reduction of VOCs emissions, and the enhanced nitrate production with NO_x reduction. The results at the Heshan site were consistent with the simulations at the suburban site of northern China, where a higher VOCs/NO_x ratio was found (Wen et al., 2018;Lu et al., 2019). The synergetic reduction of NO_x and VOCs is necessary to effectively mitigate the nitrate production in consideration of the different VOCs/NO_x ratios in the urban and suburban areas.

The above discussions revealed that direct reduction of NO_x may not lead to a decrease in nitrate production. Meanwhile, the reduction of VOCs is effective to mitigate nitrate production, though they were not the direct precursors of nitrate. To illustrate these findings, the impacts of changing VOCs and NO_x on the production rate of the OH radical, and the rate of OH plus NO_2, and the N_2O_5 uptake reaction were evaluated. During daytime nitrate production involves OH production and its subsequent reaction with NO_2. As shown in Fig. 12, the NO_x-saturated condition at the GIG site provided sufficient NO_2 to quench the OH radical during daytime. A less than 70%reduction of NO_x will increase ozone production and thereby drive more production of OH, leading to increase in the OH and NO_2 reaction rates. When NO_x is lower than 30% of the base case emissions, ozone production would decrease and lead to the decrease of OH production and its reaction with NO_2, which in turn bring about a decrease in nitrate production. In contrast, at the Heshan site, the base case NO_x concentrations are lower, giving a production rate of OH that is already sensitive to both NO_x and VOCs reductions. The model results indicates that further emission reductions in both NO_x and VOCs will simultaneously mitigate the production of nitrate and ozone.

During nighttime, the initial ozone concentration participated the nocturnal chemistry increased/decreased with the reduction of NO_x at the GIG/Heshan site. In addition, the decrease in NO_x will reduce the titration effect of NO on NO_3 radical and ozone at the GIG site, which enhances production of N_2O_5 and promotes nitrate production in both the nocturnal boundary layer and the residual layer (Fig.13).

However, at the Heshan site, the reduction of NO_x cuts down the sources of NO_2 and NO_3, decreasing the formation of N_2O_5 and thus its heterogeneous uptake to produce nitrate. The reduction of VOCs decreases ozone formation during daytime, thus attenuating the nocturnal formation of NO_3, N_2O_5 and nitrate at both the GIG and Heshan sites.

In summary, nitrate and ozone show similar responses to the reduction of NO_x and VOCs for both daytime and nighttime chemical processes, as the result of the coupling between the formation reactions of ozone and nitrate. The results of this study emphasize the complex effects of reductions of NO_x emissions on nitrate concentrations in the urban and suburban areas. In addition, the reduction of VOCs emissions would be effective in the concurrent mitigation of ozone and nitrate, suggesting that the reduction of VOCs at present is an effective method for the synergistic control of ozone and $PM_{2.5}$ at present. As there are limitations of box modeling, a comprehensive three-dimensional model assessment is needed on a regional scale.

4 Conclusions

In this study, we use an observation-constrained box model to explore the nitrate formation pathways and implications for nitrate mitigation strategies at urban and suburban sites. At both sites, the mass fraction of nitrate in PM_1 increased as the absolute PM_1 levels increased (from 10% to 33% at the urban site and from 10% to 20% at the suburban site), suggesting the important role played by nitrate in increasing particle concentrations in the PRD.

Both HNO_3 and NH_3 are important precursors for nitrate formation. Combined with the ISORROPIA II thermodynamic model, the reduction of NH_3 is effective for the nitrate reduction by affecting the partitioning process of nitrate at both GIG and Heshan site. The box model simulations demonstrate that chemical reactions in the daytime and at night both contributed significantly to formation of nitrate in the boundary layer at the two sites, with nighttime contributions of 58% at the urban site and 35% at the suburban site. However, nighttime reactions predominately occurred aloft in the residual layer at the urban site and downward transport from the residual layer in the morning are important source (53%) for surface nitrate at the urban site, whereas similar amounts of nitrate were produced in the nocturnal boundary layer and residual layer at the suburban site, which results in little downward transport of nitrate from the residual layer to the ground at this region. The spatial differences of nocturnal

reactions and the different contributions from downward transport of the residual layer to surface nitrate at urban and suburban sites were attributed to different fresh emissions and concentration levels of NO_x at the two sites during the night time, suggesting that nitrate production under different NO_x conditions should be explored to better understand the its formation pathways.

The non-linear relationships between nitrate and NO_x, VOCs was developed to investigate the nitrate mitigation strategies. The simulations demonstrated that the formation processes of both nitrate and ozone were in the VOCs-limited region at the urban site and in the transition region at the suburban site. The same sensitivity regimes of nitrate and ozone at two sites was caused by the similar chemical processes that account to produce nitrate and ozone. These results suggest that control of VOCs emissions would effectively mitigate nitrate in both urban and suburban areas.

Overall, the formation processes of nitrate are systematically investigated in both urban and suburban areas in this study, which provides the opportunity to identify different influencing factors of nitrate production in different environments and offers insights into the comprehensive mitigation of nitrate pollution in regional scale. NO_x emission controls alone might not be an effective strategy for reducing the nitrate production, while the reduction of VOCs emissions would take effect in the concurrent mitigation of ozone and nitrate. Thus, an emission control policy focusing on VOCs will be an effective means for the synergistic control of ozone and $PM_{2.5}$ at present. In the long-term, multi-pollutant control should be implemented to achieve better control strategies for ozone and $PM_{2.5}$. As the result of limitation for the 0-D box model, vertical transport and horizontal transport cannot be considered explicitly in this study. Given the limitations of the box model, three-dimensional models should be used to further investigate the synergistic control of ozone and particles on the regional scale.

Data availability

The observational data used in this study are available from corresponding authors upon request (byuan@jnu.edu.cn)

Author contributions

BY and MS designed the research. SXY, YWP, SH, WC, WWH, CLP, CMW, ZLW, TGL, EZ, MFC, XBL, SHW, CHW, WWJ, CSY, WS and PC contributed to data collection. SXY performed the data analysis, with contributions from JZ， DD. Parrish,

XJH, CCL, XYY, YS, HCW, DHC, XMW, ZYZ, JYZ and XMW. SXY and BY prepared the manuscript with contributions from the other authors. All the authors reviewed the manuscript.

Competing interests

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgments

This work was supported by the National Key R&D Plan of China (grant No. 2019YFE0106300, 2018YFC0213904, 2016YFC0202206), the National Natural Science Foundation of China (grant No. 41877302), the National Natural Science Foundation of China (grant No. 41905111), Guangdong Natural Science Funds for Distinguished Young Scholar (grant No. 2018B030306037), Key-Area Research and Development Program of Guangdong Province (grant No. 2019B110206001), Guangdong Soft Science Research Program (2019B101001005), and Guangdong Innovative and Entrepreneurial Research Team Program (grant No. 2016ZT06N263). This work was also supported by Special Fund Project for Science and Technology Innovation Strategy of Guangdong Province (Grant No.2019B121205004).

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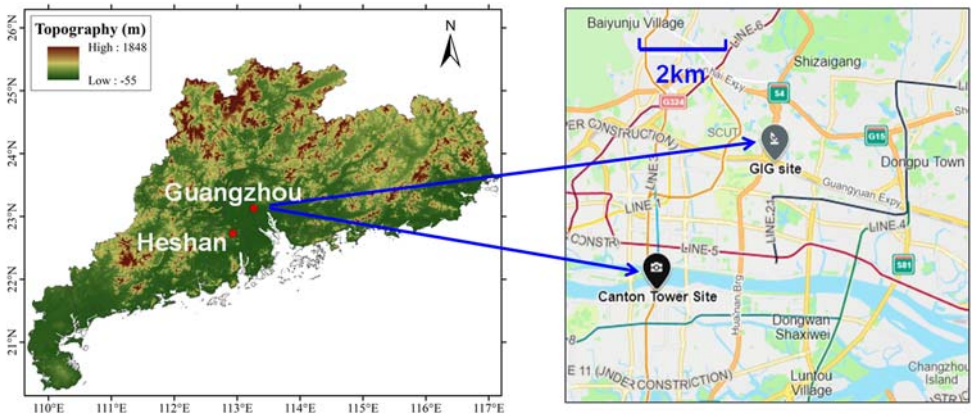

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1085 **Figure 1.** Sampling site at Guangzhou Institute of Geochemistry, Chinese Academy of
1086 Sciences (GIG), Heshan and Canton Tower. Note that the map is extracted from
1087 Microsoft Bing maps by the authors.
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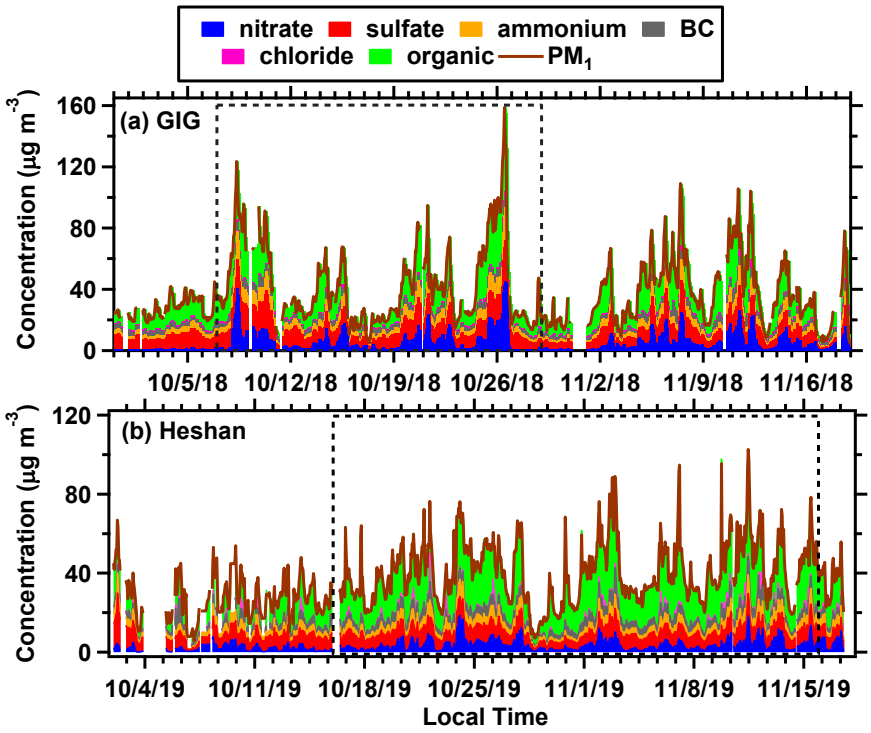


Figure 2. Temporal variations of the mass concentration of the major chemical components in PM_1 including nitrate (NO_3^-), sulfate (SO_4^{2-}), ammonium (NH_4^+), black carbon (BC), chloride (Cl^-) and organics at (a) GIG site and (b) Heshan site. The black dashed rectangle represents the investigated period which had complete set of data.

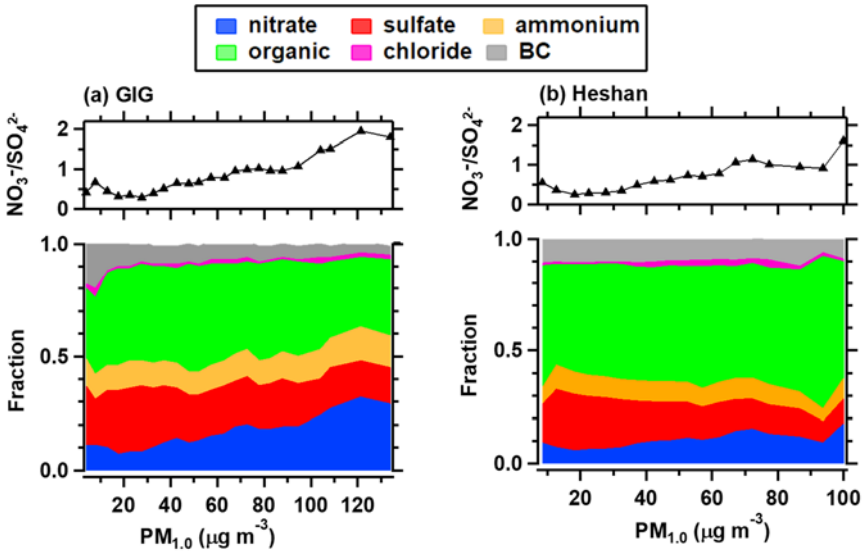


Figure 3. The mass concentration ratio of NO_3^-/SO_4^{2-} (top) and fractions of major chemical components (bottom) in PM_1 at (a) GIG site and (b) Heshan site.

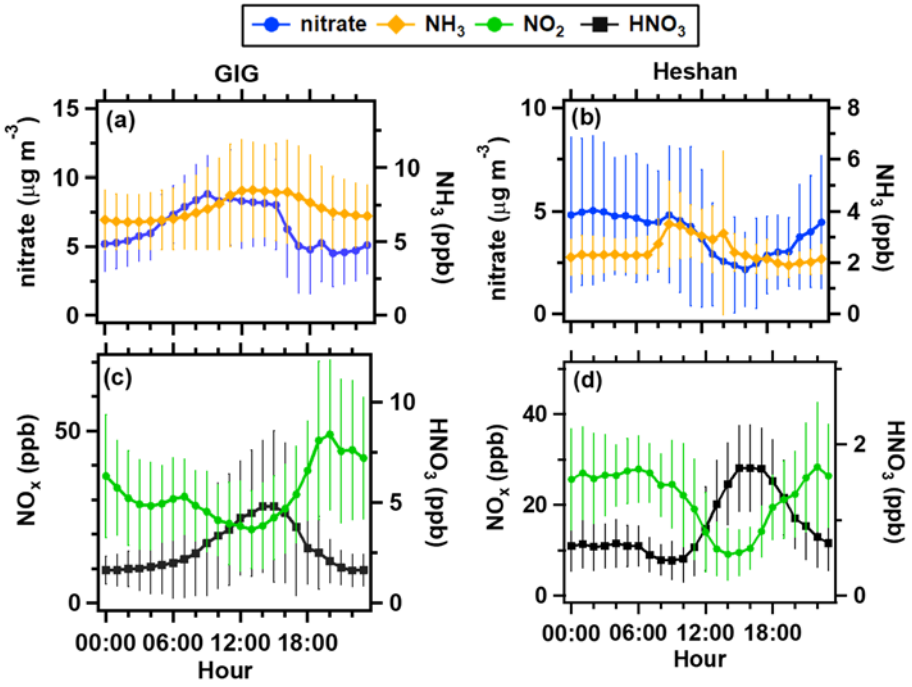


Figure 4. Diurnal variations of mean concentrations of nitrate and related pollution species at (a) GIG site and (b) Heshan site. The error bars represent the standard deviation of the means.

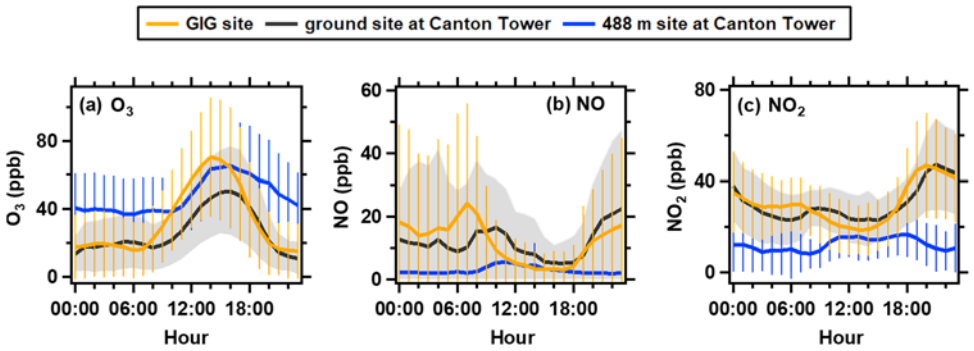


Figure 5. Diurnal variation of mean concentrations of (a) O_3, (b) NO, (c) NO_2 at GIG (orange lines), and the ground and 488m sites of Canton Tower (black and blue lines, respectively). The orange and blue error bars represent the standard deviations of the mean concentrations at the GIG site and the 488m site of Canton Tower, the grey areas show one standard deviation of the mean concentration at ground site of Canton Tower.

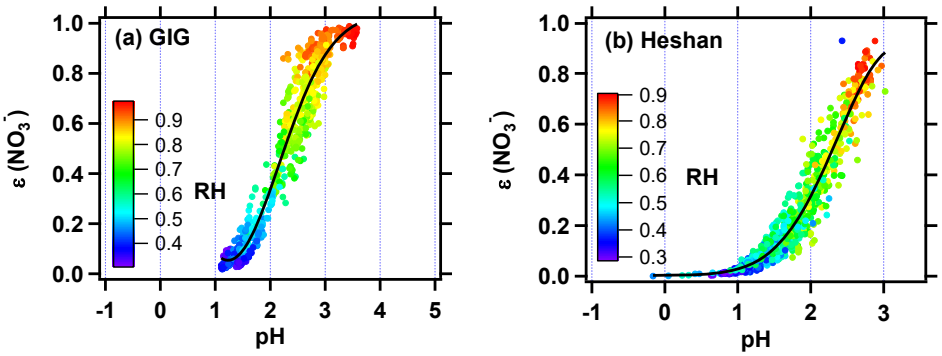


Figure 6. The particle fraction of nitrate in the sum of HNO_3+nitrate ($\varepsilon(NO_3^-)$) against aerosol pH. The pH data are colored by relative humidity and fit to an "s-curve" in black line, as shown in Guo et al. (2018).

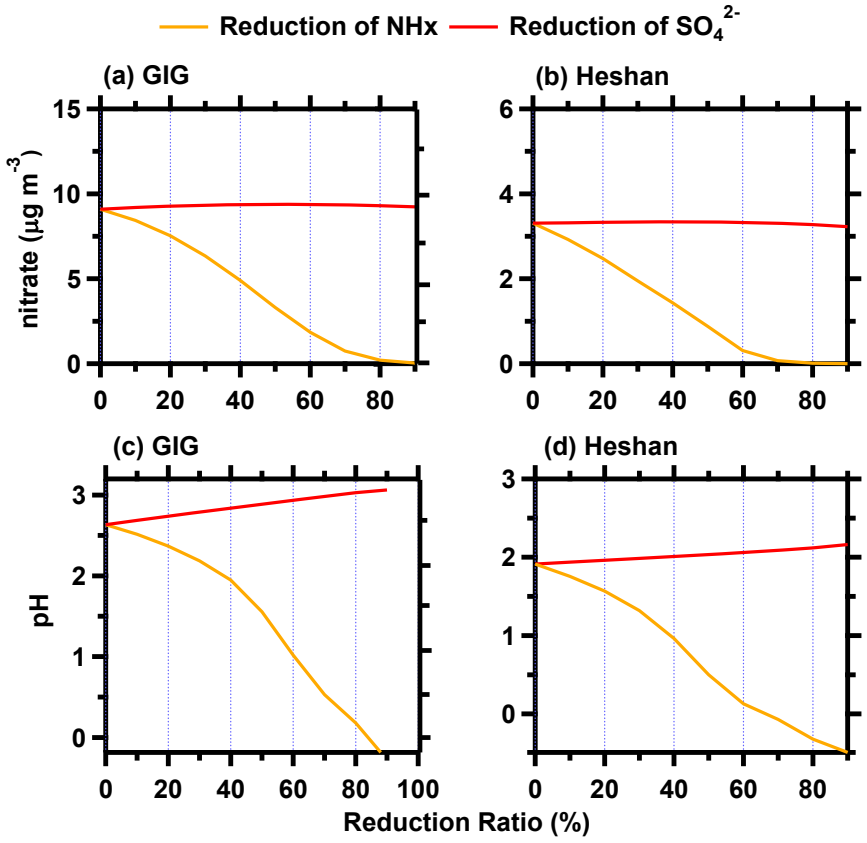


Figure 7. ISORROPIA-predicted average nitrate (a, b) and pH (c, d) as a function of changes in NHx (ammonium + NH_3, orange line) and SO_4^{2-} (red line) at the GIG and Heshan site during the study period.

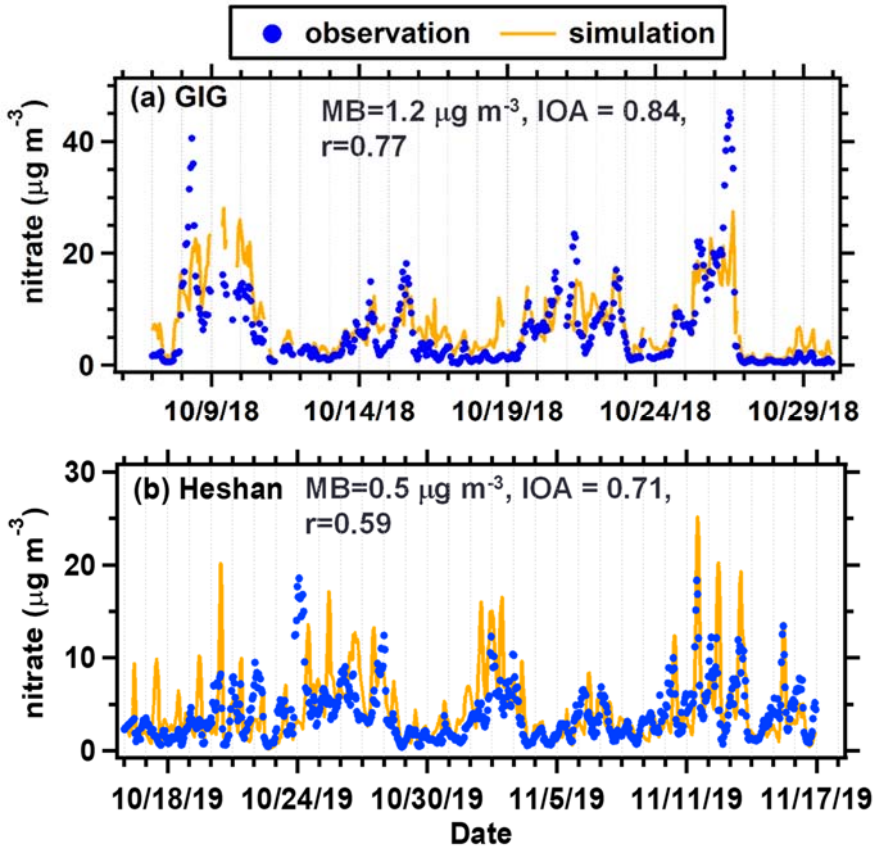


Figure 8. Comparison of the temporal box model simulated and observed nitrate at the (a) GIG site and (b) Heshan site.

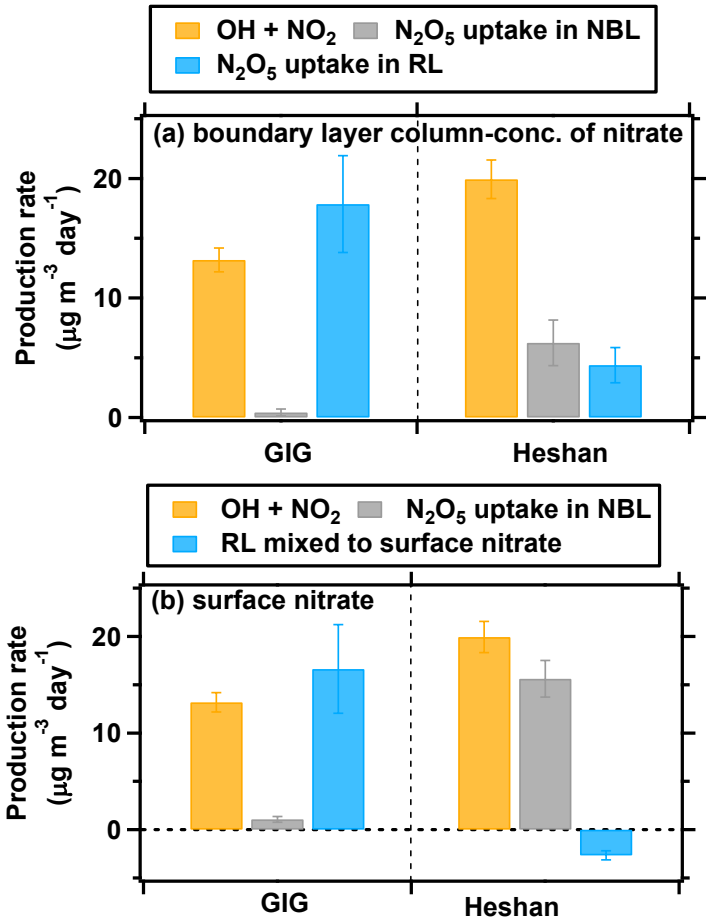


Figure 9. The daily-averaged contribution (a) to boundary layer column concentration and (b) to surface nitrate from three pathways (OH +NO_2 reaction, N_2O_5 uptake in NBL, and N_2O_5 uptake in RL/N_2O_5 uptake from RL mixed process) at the GIG and Heshan sites. The error bars represent the standard deviations of the mean production rate**.**

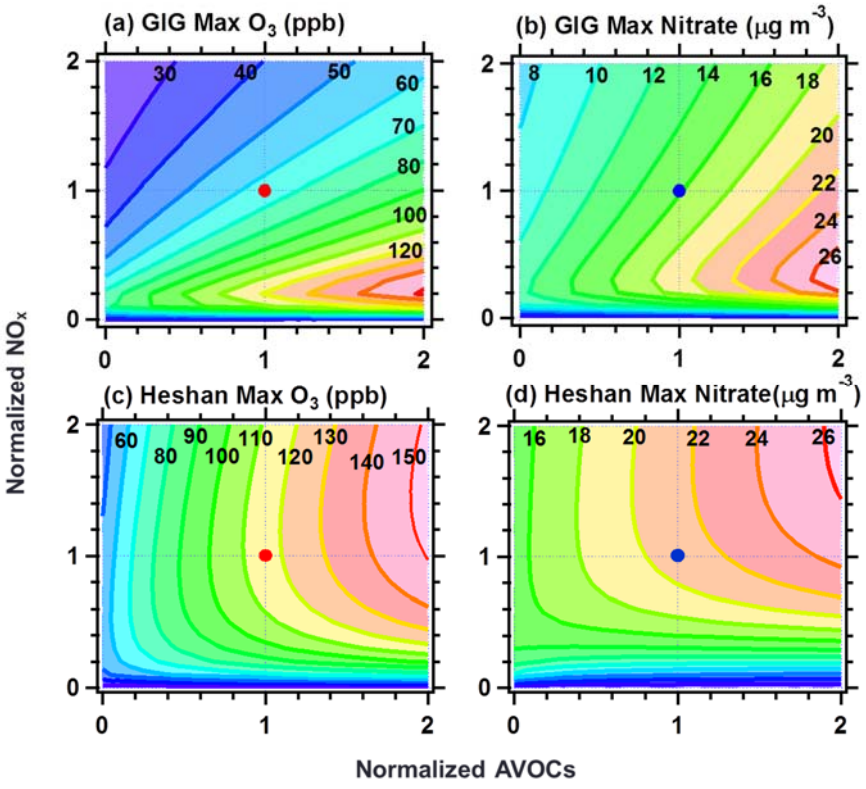
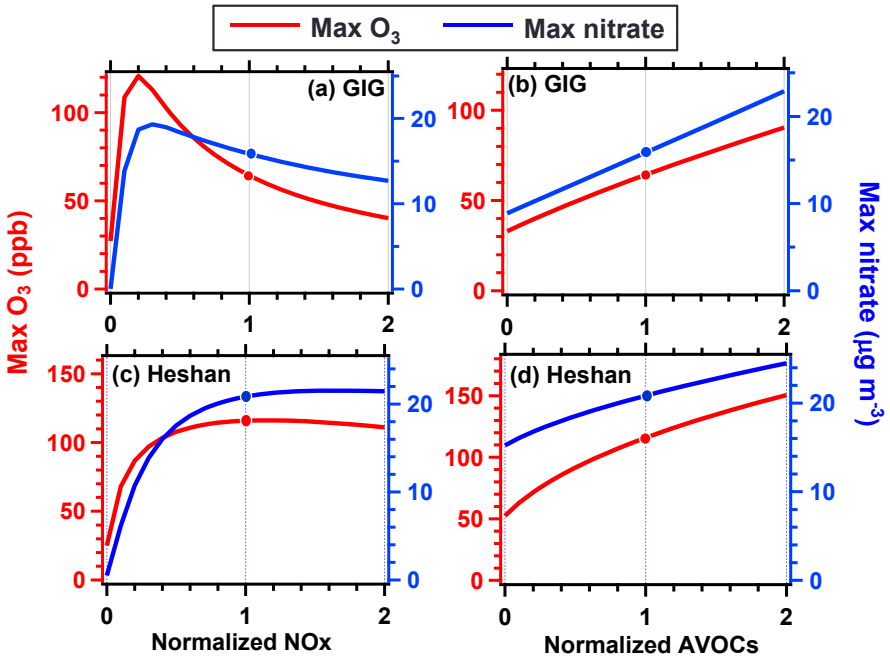


Figure 10. The simulated isopleths of ozone and nitrate with normalized NO_x and AVOCs concentration at the (a, b) GIG site and (c, d) Heshan site, each isopleth represents the maximum ozone and nitrate in the simulation, and the red and blue circles represent the base cases.



1142

1143 **Figure 11.** Simulated maximum ozone and nitrate concentration with normalized
1144 NOx and AVOCs at the (a, b) GIG site and (c, d) Heshan site, cutting through the
1145 simulated isopleth in Figure 8 with normalized AVOCs and NOx ratio at 1, respectively.
1146 The red and blue circles represent the base cases.
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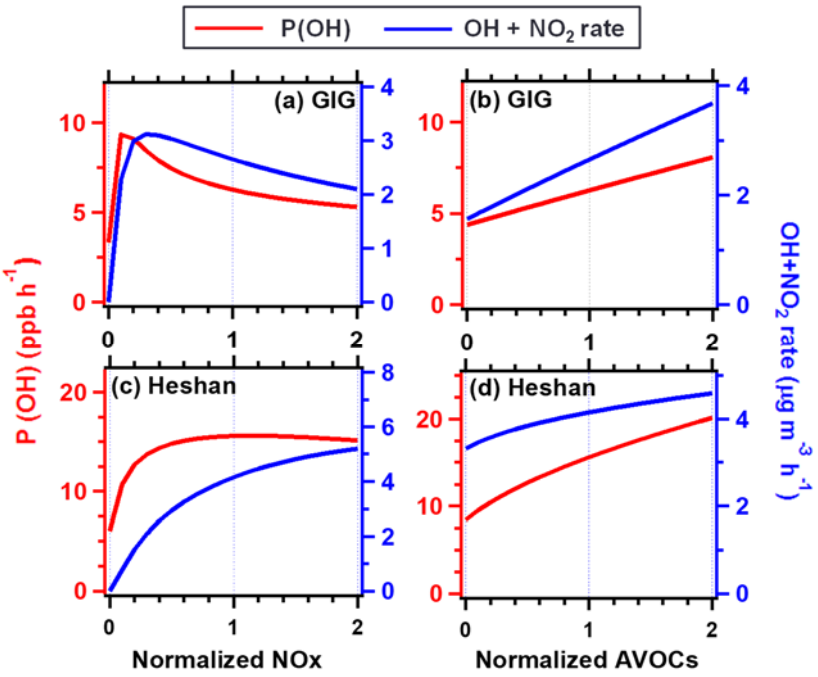

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1149 **Figure 12.** Simulated average production rates of OH (P (OH)) and the reaction rate of
1150 OH and NO_2 with the normalized changes of NO_x and AVOCs emissions at the (a, b)
1151 GIG site and (c, d) Heshan site.

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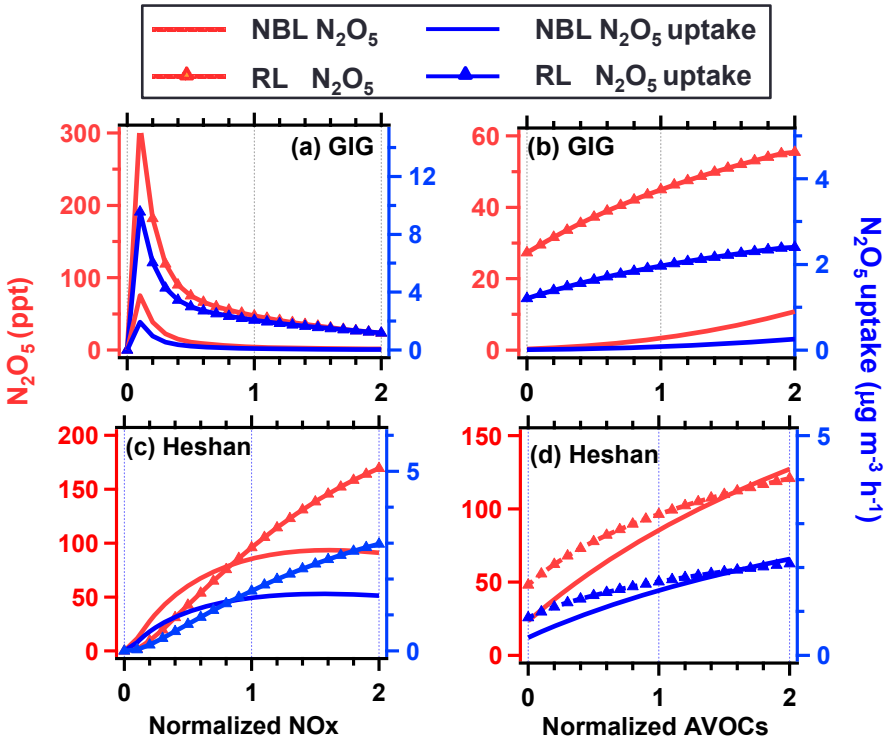

1153

1154 **Figure 13.** Simulated average concentration of N_2O_5 and nitrate production rate from
1155 N_2O_5 uptake with the normalized changes of NO_x and AVOCs emissions at the (a, b)
1156 GIG site and (c, d) Heshan site in the NBL and RL.
1157