



Enhanced atmospheric oxidation and particle reductions driving changes to nitrate formation mechanisms across coastal and inland regions of north China

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Abstract. Nitrate (NO_3^-) has surpassed sulfate as the dominant secondary inorganic ion, posing a significant challenge to air quality improvement measures in China. We utilized the WRF–CMAQ (Weather Research and Forecasting and Community Multiscale Air Quality) model and isotope analysis to investigate the nitrate formation mechanisms driving regional changes in inland and coastal cities in north China during the winters of 2013 and 2018. Among the nitrate formation pathways, the oxidation reaction of OH radicals with NO_2 ($\text{OH} + \text{NO}_2$) and the heterogeneous reaction of N_2O_5 (het N_2O_5) were determined to be the dominant pathways (88 %–95.5 %), whereas the other pathways contributed less than 12.4 % to the total amount of nitrate formation. In inland cities, 63.7 %–85.6 % of nitrate was formed via $\text{OH} + \text{NO}_2$ and 8.3 %–27.7 % was formed from het N_2O_5 . In coastal cities, approximately half of the nitrate (48.2 %–56.5 %) was produced from $\text{OH} + \text{NO}_2$, whereas het N_2O_5 contributed 37.0 %–45.7 % due to higher N_2O_5 concentrations and longer NO_3 radical lifetimes. Compared with that in 2013, the $\text{OH} + \text{NO}_2$ contribution in 2018 increased by 7.6 % in inland cities and 3.6 % in coastal cities due to the increased atmospheric oxidizing capacity. Scenario simulations indicated that a 60 % reduction in NO_x emissions led to a 4.5 % decrease in nitrate concentrations in Beijing. The reduction reached 32.4 % reduction in Qingdao. A 60 % combined reduction in NH_3 , NO_x , and volatile organic compounds (VOCs) yielded 44.2 % and 60.0 % reductions in nitrate in Beijing and Qingdao, respectively, underscoring the necessity of multipollutant control strategies.

1 Introduction

As a key component of fine particulate matter ($\text{PM}_{2.5}$), nitrate (NO_3^-) exacerbates health risks (Tie et al., 2009; Sun et al., 2014) and affects the physical and chemical properties of particulates, such as their hygroscopicity, light absorption, and acidity (Cao et al., 2013; Wang et al., 2023). These properties directly influence the atmospheric radiation balance (Ramanathan and Feng, 2009; Tegen et al., 2000), atmospheric visibility, and air quality. Furthermore, nitrate serves as a key source of cloud condensation nuclei, affecting cloud

formation and precipitation patterns which subsequently influence the global water cycle and climate regulation (Yu et al., 2020; Kalkavouras et al., 2019). Moreover, the photolysis process of nitrate in atmospheric boundary layers is highly active, serving as an important source of NO_x and regulating the atmospheric oxidation capacity, thus influencing the formation of secondary pollutants such as sulfate (SO_4^{2-}) and brown carbon (BrC) (Ye et al., 2016; Xue et al., 2019; Zheng et al., 2020; Yang et al., 2021). Therefore, nitrate is closely related to regional haze pollution occurrence (Zhai et al., 2021; Zhang et al., 2021; Xu et al., 2019a, b; Fu et

al., 2020). Understanding the formation mechanism of nitrate is essential not only for advancing the atmospheric chemical processes but also for mitigating regional haze pollution.

In tropospheric atmospheres, nitrate formation primarily follows two pathways (Fig. 1). During the daytime, NO_2 is oxidized by hydroxyl radicals (OH) to produce gaseous HNO_3 (Reaction R1). Conversely, at night, O_3 oxidizes NO_2 , leading to the formation of the NO_3 radical, which combines with NO_2 to form N_2O_5 . This compound can subsequently be adsorbed onto particles via heterogeneous reactions, resulting in the formation of HNO_3 (Reaction R2) (Hallquist et al., 1999; Pathak et al., 2011). These dynamics highlight the intrinsic dependence of atmospheric nitrate formation on NO_x and oxidants, especially O_3 and OH radicals. The generation of OH and O_3 is intricately linked to the photochemical reactions of NO_x and volatile organic compounds (VOCs) (Atkinson, 2000). OH radicals and hydroperoxyl radicals (HO_2) are produced via the photolysis of ozone (O_3), nitrous acid (HONO), oxygenated volatile organic compounds (OVOCs), and hydrogen peroxide (H_2O_2) and via reactions between O_3 and VOCs. OH subsequently reacts with VOCs to generate organic peroxy radicals (RO_2) and HO_2 , which are then recycled back to OH via their interaction with nitrogen monoxide (NO). During this cycle, NO is transformed into NO_2 , which yields O_3 upon photolysis (Fu et al., 2020).

Nitrate formation mechanisms exhibit significant regional differences, especially in winter. Some studies based on isotope analysis have shown that the OH pathway is important for nitrate formation in inland cities. For example, in Beijing, the OH pathway contributes 66 %–92 % to winter nitrate formation, whereas the heterogeneous reaction of the N_2O_5 (het N_2O_5) pathway contributes 8 %–34 % to nitrate (Chen et al., 2020). In Xi'an, the contribution of the het N_2O_5 pathway to winter nitrate formation ranges from 13 % to 35 % (Wu et al., 2021). In coastal areas, which are influenced by high humidities, high sea salt levels, and the combined effects of marine emissions and air masses (Zhong et al., 2023; Athanasopoulou et al., 2008; Zhao et al., 2024), the contribution of the OH pathway is smaller than that in inland cities, whereas the het N_2O_5 pathway plays a more significant role. For example, isotope analysis revealed that the contribution of the $\text{OH} + \text{NO}_2$ pathway to winter nitrate formation is 48 %–74 % in Shanghai (He et al., 2020), whereas the contributions of the OH , het N_2O_5 , and $\text{NO}_3 + \text{HC}$ pathways to winter nitrate formation are 20.2 %, 38.2 %, and 21.6 %, respectively, in Xiamen (Li et al., 2022). Broad coastal studies have revealed similar significant regional differences. Michalski et al. (2003) noted that in coastal California, approximately 90 % of winter nitrate originates from the het N_2O_5 pathway. In the polar environment of Greenland (Kunasek et al., 2008), nearly all nitrate is formed via the $\text{NO}_3 + \text{HC/DMS}$ and het N_2O_5 pathways in winter, with contributions of 60 % and 40 %, respectively. Furthermore, the impacts of continental and marine air masses on nitrate

formation mechanisms differ. In the south Yellow Sea, under the dominance of marine air masses, the contributions of the $\text{OH} + \text{NO}_2$, $\text{NO}_3 + \text{HC/DMS}$, and het N_2O_5 pathways to nitrate formation are 43.9 %, 22.4 %, and 33.6 %, respectively, whereas the contributions (12.6 %, 59.0 %, and 28.4 %, respectively) in the Bohai and north Yellow seas are affected by continental pollution (Zhao et al., 2024). These studies have confirmed substantial regional differences in nitrate formation mechanisms.

In addition to isotope analysis, model simulation studies have provided significant support for elucidating nitrate formation mechanisms. Global models such as GEOS-Chem have revealed significant land–ocean differences in nitrate formation pathways. Alexander et al. (2020), using the GEOS-Chem model at a global scale, reported that the $\text{OH} + \text{NO}_2$ reaction (41 %–42 %) and N_2O_5 hydrolysis (28 %–41 %) were the main pathways involved below 1 km altitude, with N_2O_5 hydrolysis dominating in winter in the mid- to high-latitude northern continents; however, in remote marine regions, XNO_3 hydrolysis emerged as the primary pathway. However, owing to their relatively low resolution (typically $0.5^\circ \times 0.625^\circ$ or coarser) and simplified treatment of aerosol–surface heterogeneous reactions, global models clearly have limitations in capturing regional differences (Heald et al., 2012). Consequently, the average outputs from global models may not accurately reflect the nitrate formation mechanisms of specific regions (Alexander et al., 2020), leading to potential overestimations or underestimations of local nitrate formation, particularly in terms of ammonia emissions and nitrate production rates. Compared with global models, regional models, such as WRF-Chem (Weather Research and Forecasting model coupled with Chemistry) and WRF-CMAQ (Weather Research and Forecasting and Community Multiscale Air Quality model), can offer greater accuracy in capturing local chemical processes. For example, Sun et al. (2022) used the WRF-CMAQ model to study the Yangtze River Delta (YRD) and reported that during winter, the $\text{OH} + \text{NO}_2$ reaction contributed 69.3 % of nitrate formation in Shanghai (a coastal area) and 66.9 % in Hefei (an inland area), whereas the contribution of het N_2O_5 in Hefei was 27.1 %. Similarly, Li et al. (2021) used the WRF-Chem model to investigate the North China Plain (NCP) and YRD and reported that near the surface, the $\text{OH} + \text{NO}_2$ reaction dominated nitrate formation (60 %–92 %), with het N_2O_5 contributing 8 %–40 %. Moreover, Kim et al. (2014b) applied the WRF-CMAQ model to the Great Lakes region of North America and reported that het N_2O_5 accounted for 57 % of nitrate formation, highlighting the regional variability. For targeted regions, especially heavily polluted urban areas or those with unique ecosystems, employing high-resolution regional simulations and more detailed chemical mechanisms allows for a more precise characterization of nitrate formation processes.

The North China Plain (NCP) region is a focal point for studying air pollution due to its severe winter haze issues,

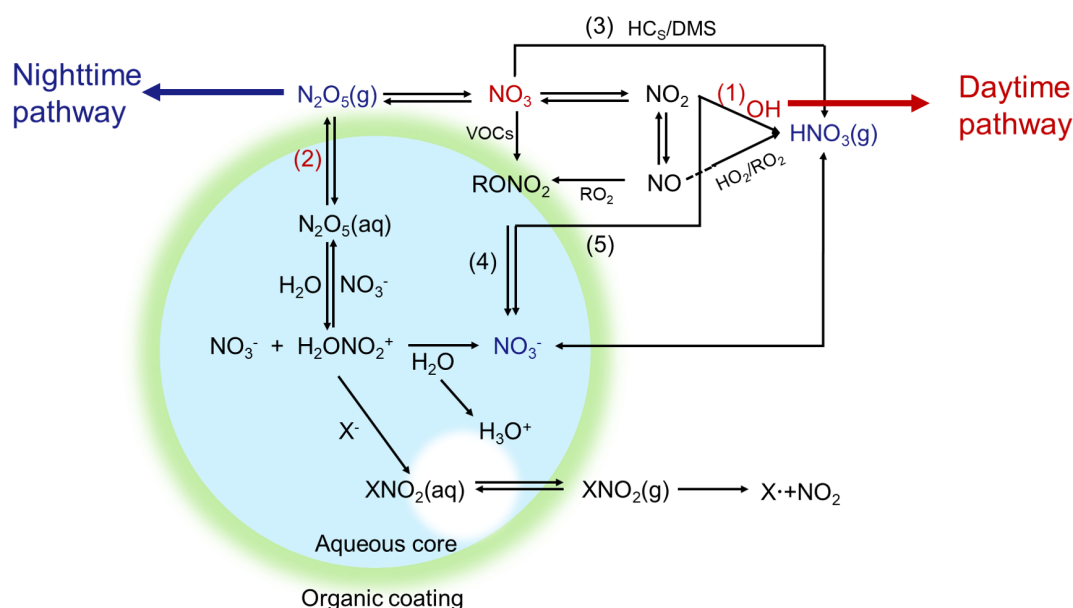


Figure 1. Formation mechanisms of nitrate in the atmosphere (Wang et al., 2023).

which consistently rank among the areas with the highest $\text{PM}_{2.5}$ concentrations globally (Chen et al., 2016). Before 2013, sulfate was the dominant pollutant during haze events in the NCP, accounting for a significant proportion of the total $\text{PM}_{2.5}$ (Li et al., 2019). However, following the implementation of the Clean Air Action Plan (CAAP) in 2013, which aimed to reduce $\text{PM}_{2.5}$ concentrations and improve air quality (Zhang et al., 2023a), the composition of winter haze pollution shifted from being sulfate-dominated to nitrate-dominated (Li et al., 2019; Xu et al., 2019a). Data from the Multiresolution Emission Inventory for China (MEIC) (Zheng et al., 2018) show that from 2013 to 2017, anthropogenic emissions of SO_2 , NO_x , CO, and NH_3 decreased by 59 %, 21 %, 23 %, and −3 %, respectively. Correspondingly, the national annual average $\text{PM}_{2.5}$ level decreased by 30 %–50 % from 2013–2018 (Zhai et al., 2019). While these emission reduction measures led to significant declines in CO and SO_2 levels and in particulate sulfate concentrations, the reduction in particulate nitrate levels was smaller than expected. In fact, nitrate mass concentrations have increased in some major cities (Xu et al., 2019b; Shao et al., 2018; Zhou et al., 2019; Fu et al., 2020). This divergence in the responses of sulfate and nitrate to emission controls underscores the complex chemistry of nitrate formation and its precursors. Most studies have focused on nitrate formation mechanisms in inland cities such as Beijing–Tianjin–Hebei (Fan et al., 2020b; Fu et al., 2020; Yang et al., 2024; Chen et al., 2020); thus, the formation mechanisms in coastal cities remain poorly understood. The coastal regions of China experience monsoon transitions, where marine air masses are transported inland under the influence of sea and land breezes, affecting the formation of nitrate over terrestrial areas.

To understand the factors controlling nitrate concentration variations in inland and coastal cities, especially during periods dominated by different primary pollutants, we selected the winters of 2013, when the CAAP was initiated, and 2018, after the project had concluded, to explore the nitrate formation mechanisms in five inland cities and two coastal cities on the NCP via a regional multiscale model combined with isotope analysis. The primary objectives of this study were to quantify the contributions of different nitrate formation pathways, such as $\text{OH} + \text{NO}_2$ and hetN_2O_5 , in inland and coastal cities, investigate the impacts of emission control measures on nitrate formation, and provide insights into regional differences in nitrate control.

2 Methods

2.1 Model configuration

The Community Multiscale Air Quality (CMAQ) model (version 5.3.3) was used to simulate the chemical reactions and physical processes that contributed to NO_3^- and TNO_3 ($\text{HNO}_3 + \text{NO}_3^-$) formation in the NCP region. Two distinct periods were considered, i.e., one period from 1 December 2013 to 28 February 2014, representing the period when the CAAP was initiated, and one period from 1 December 2018 to 28 February 2019, representing the period after the CAAP was completed. As shown in Fig. 2, a dual-layer grid nesting method was implemented in our simulations. Domain 1 (D01) encompassed the majority of China with a 36 km horizontal resolution, whereas domain 2 (D02) covered the NCP with a 12 km horizontal resolution. The major cities within the NCP included

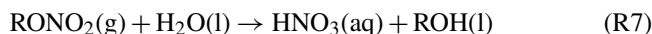
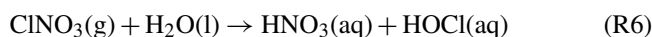
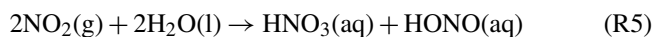
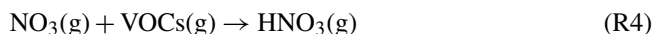
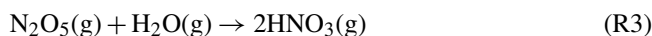
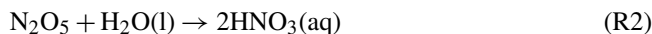
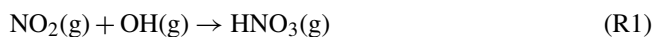
Beijing (BJ), Tianjin (TJ), Shijiazhuang (SJZ), Jinan (JN), Zhengzhou (ZZ), Qingdao (QD), and Yantai (YT). The terrain elevation data used in Fig. 2 came from the GEBCO 2024 Grid provided by the GEBCO Bathymetric Compilation Group (2024, <https://doi.org/10.5285/1c44ce99-0a0d-5f4f-e063-7086abc0ea0f>). The model was vertically segmented into 14 layers, stretching from the Earth's surface to the troposphere, with the first layer height at approximately 31 m. The CB6 chemical mechanism was chosen to simulate gas-phase chemistry (Luecken et al., 2019; Yan et al., 2021) and the aerosol mechanism. Aerosol module version 7 (AERO7; Appel et al., 2021) was used to simulate particulate-phase chemistry. The initial and boundary conditions used as the inputs of D01 were provided via the GEOS-Chem global simulation results, and D02 was provided via D01. To minimize the impact of the initial conditions, we initiated the model run 7 d before the analysis period. Anthropogenic emission data were sourced from the Multiresolution Emission Inventory for China (MEIC) source emission inventory maintained by Tsinghua University (<http://meicmodel.org/>, last access: 17 June 2024), while biogenic emission estimates were derived from the Model of Emissions of Gases and Aerosols from Nature version 2.0.4 (MEGANv2.0.4, <https://bai.ess.uci.edu/megan>, last access: 3 June 2024).

The required meteorological data were generated with the Weather Research and Forecasting (WRF) model version 3.7, a system for predicting mesoscale weather patterns. The selection of physics options for this model conformed with the methodologies applied in our earlier studies (Liu et al., 2021; Chen et al., 2021). The WRF simulations in this study depended on the final operational global analysis (FNL) datasets, which are global reanalysis data with temporal and spatial resolutions of 6 h and $1^\circ \times 1^\circ$, respectively, provided by the U.S. Environmental Prediction Center and the National Center for Atmospheric Research (<https://rda.ucar.edu/datasets/ds083-2/>, last access: 12 March 2024). The WRF model outputs were subsequently processed via Model-3/CMAQ Modeling System Interface Processor (MCIP) version 4.3, ensuring compatibility with the CMAQ model format.

2.2 Process analysis

Within the CMAQ modeling framework, the integrated reaction rate (IRR) module of the process analysis (PA) tool was used to simulate the formation reactions of TNO₃ (HNO₃ + NO₃⁻). The reaction rates of the chemical reactions at each moment were provided by the IRR module, enabling the quantitative evaluation of the target reactions. The complex chemical formation of TNO₃ involved seven reaction pathways, which were categorized into three main pathways on the basis of their significance: OH + NO₂ (Reaction R1), hetN₂O₅ (Reaction R2), and other (Reactions R3 to R7). The chemical equations for these seven production reaction path-

ways could be expressed as follows.



2.3 Observation data

In this study, the simulation results, including meteorological parameters and atmospheric pollutants, for the major cities in the NCP were validated in depth. The meteorological parameters included the 2 m temperature (T2), relative humidity (RH), 10 m wind speed (WS10), and 10 m wind direction (WD10). The atmospheric pollutants included the maximum 8 h moving average ozone (MDA8 O₃), NO₂, PM_{2.5}, and PM_{2.5} components.

2.3.1 Meteorological data and sources

Meteorological data, comprising T2, WS10, WD10, and RH, were sourced from the National Climatic Data Center (NCDC) of the National Oceanic and Atmospheric Administration (NOAA) (<https://www.ncdc.noaa.gov/cdo-web/>, last access: 17 October 2023). Data were collected from a total of 68 stations, with a temporal resolution of either 3 or 1 h.

2.3.2 Atmospheric pollutant concentrations

Data on O₃, NO₂, and PM_{2.5} concentrations were retrieved from the real-time national urban air quality dataset disseminated by the China Environmental Monitoring Center (<https://air.cnemc.cn:18007/>, last access: 13 February 2024). The SO₄²⁻, NH₄⁺, and NO₃⁻ concentration data were sourced from our own observations in Qingdao and published data (detailed information can be found in Table S3).

In the city of Qingdao, we collected total suspended particle (TSP) samples during two winter seasons. The TSP samples were collected on preheated quartz fiber filters with a high-volume ($1.05 \text{ m}^3 \text{ min}^{-1}$) aerosol sampler (Qingdao Laoshan Electronics Co., Ltd., China). The sampling site was situated on the roof of Darwin Hall ($36^\circ 06' \text{ N}$, $120^\circ 33' \text{ E}$; 16 m) at the Ocean University of China, approximately 1 km from the beach. Detailed information can be found in Ding et al. (2019). Water-soluble inorganic ions, such as NO₃⁻, SO₄²⁻, and NH₄⁺, were extracted from the samples using ultrapure water ($> 18.2 \text{ M}\Omega \text{ cm}$), and their concentrations were measured with a Thermo Scientific Dionex ICS-1100 IC system, as described in a previous study (Qi et al., 2020). The

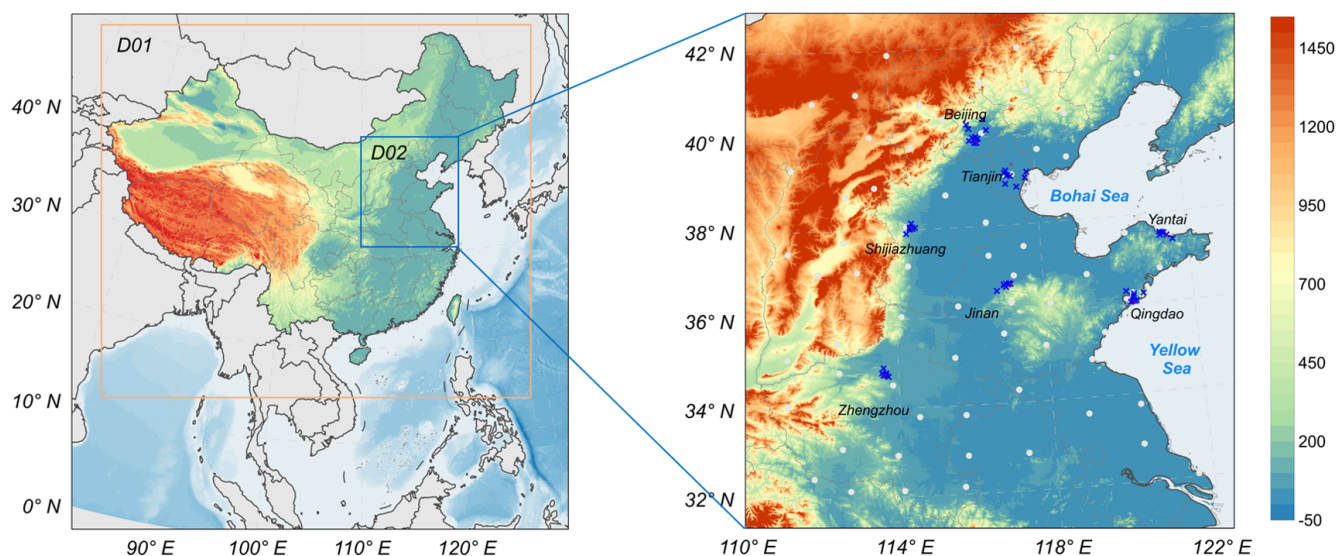


Figure 2. Terrain heights of the NCP region and locations of the major cities (Beijing, Tianjin, Shijiazhuang, Jinan, Zhengzhou, Qingdao, and Yantai) within the study domain. The white dots indicate meteorological observation stations, and the blue crosses indicate air quality monitoring stations. Publisher's remark: please note that the above figure contains disputed territories.

ionic concentrations in the TSP samples were calibrated by subtracting the concentrations in the blanks. The $\delta^{18}\text{O}$ and $\delta^{15}\text{N}$ values of NO_3^- in the TSP samples were determined via the bacterial denitrifier method (Casciotti et al., 2002; Sigman et al., 2001) with the Gasbench-IRMS system (Delta V model, Thermo Scientific). For analytical calibration, four international reference standards were employed, namely USGS32, USGS34, USGS35, and IAEA- NO_3^- . The standard deviations of the replicates were $\pm 0.2\text{‰}$ for $\delta^{15}\text{N}-\text{NO}_3^-$ and $\pm 0.5\text{‰}$ for $\delta^{18}\text{O}-\text{NO}_3^-$. The analytical procedures for determining $\delta^{15}\text{N}-\text{NO}_3^-$ and $\delta^{18}\text{O}-\text{NO}_3^-$ were detailed in a previous study (Luo et al., 2021). The blank contributed less than 5 % to the sample results. Additional details are provided in Sect. S1 in the Supplement.

2.4 Dual-isotopic Bayesian mixing model

The Bayesian isotope mixing model (Stable Isotope Analysis in R, SIAR) and dual-isotopic compositions ($\delta^{15}\text{N}-\text{NO}_3^-$ and $\delta^{18}\text{O}-\text{NO}_3^-$) determined in Qingdao and adapted from a reference study in Beijing (Zong et al., 2020; Fan et al., 2020b) were employed to estimate the contributions of OH radicals and the hetN_2O_5 pathway to particulate NO_3^- . Equations (S1)–(S7) for calculating the isotopic compositions of nitrogen oxides in the atmosphere are provided in the Supplement. The atmospheric $\delta^{15}\text{N}-\text{NO}_3^-$ and $\delta^{18}\text{O}-\text{NO}_3^-$ can be expressed by Eqs. (S1) and (S2), respectively. The endmembers of $[\delta^{15}\text{N}-\text{HNO}_3]_{\text{OH}}$, $[\delta^{15}\text{N}-\text{HNO}_3]_{\text{N}_2\text{O}_5}$, $[\delta^{18}\text{O}-\text{HNO}_3]_{\text{OH}}$, and $[\delta^{18}\text{O}-\text{HNO}_3]_{\text{N}_2\text{O}_5}$ are expressed by Eqs. (S3), (S4), (S5), and (S6), respectively (Walters and Michalski, 2016). This study was focused on the seasonal scale, where observations of $\delta^{18}\text{O}-\text{NO}_3^-$ did not reveal a

significant diurnal difference (Feng et al., 2020; Zhang et al., 2022), as shown in Fig. S1 in the Supplement. Therefore, we did not account for the diurnal variation in $\delta^{18}\text{O}-\text{NO}_2$ in the calculation of endmembers (Walters et al., 2018; Zhang et al., 2025; Albertin et al., 2021, 2024). The $\delta^{15}\text{N}$ values of tropospheric NO_x and the $\delta^{18}\text{O}$ values of tropospheric $\text{H}_2\text{O}(\text{g})$, NO_x , O_3 , and $\cdot\text{OH}$ occurred within certain ranges, as described in Sect. S2 and Table S1 in the Supplement. Therefore, the endmembers of $\delta^{15}\text{N}$ and $\delta^{18}\text{O}$ for the two pathways could be estimated according to $f\text{NO}_2$ (the molar ratio of NO_2 and NO_x) and the isotope fractionation values of nitrogen and oxygen, i.e., $\alpha_{\text{NO}_2/\text{NO}}$, $\alpha_{\text{OH}/\text{H}_2\text{O}}$, and $\alpha_{\text{N}_2\text{O}_5/\text{NO}_2}$. The nitrogen and oxygen isotope fractionations were temperature-dependent and could be estimated via Eq. (S7) and Table S2. On the basis of the endmembers of $[\delta^{15}\text{N}-\text{HNO}_3]_{\text{OH}}$, $[\delta^{15}\text{N}-\text{HNO}_3]_{\text{N}_2\text{O}_5}$, $[\delta^{18}\text{O}-\text{HNO}_3]_{\text{OH}}$, and $[\delta^{18}\text{O}-\text{HNO}_3]_{\text{N}_2\text{O}_5}$, the contribution of the OH radical formation pathway (γ) was estimated via the dual-isotopic Bayesian mixing model (Luo et al., 2021). The uncertainty of the Bayesian model mainly stemmed from the standard deviations in the endmember calculations for each formation pathway and the posterior probability distributions output by the model (Moore and Semmens, 2008).

2.5 Model evaluation

The following statistical indicators were used to evaluate the simulation effect: the mean deviation (MB), normalized mean deviation (NMB), normalized mean error (NME), correlation coefficient (R), root mean square error (RMSE), and index of agreement (IOA). The definitions and standards of all the statistical indicators are provided in Table S3.

To evaluate model performance, Emery et al. (2001) provided benchmarks of the above statistical indicators for major meteorological parameters such as T2, WS10, and WD10, which are presented in Table 1. Similarly, Emery et al. (2017) and Huang et al. (2021) proposed benchmarks for the concentrations of major air pollutants, including PM_{2.5}, NO₂, MDA8 O₃, and NO₃⁻, as comprehensively detailed in Table 2.

For the comparison between the model simulations and observational data, the model grid cell corresponding to the geographic coordinates of each observation station was first identified. At each station, the observed values were compared with the simulated values from the corresponding grid cell on a point-by-point, hour-by-hour basis. These paired comparisons were then aggregated across all stations, and statistical evaluation metrics were generated from the entire dataset to assess model performance comprehensively. All data processing and comparisons were conducted on an hourly basis.

2.6 Emission reduction scenario simulation design

Notably, emission reduction scenarios were specifically designed to investigate how reductions in NH₃, NO_x, and VOC emissions affected PM_{2.5} nitrate concentrations in Beijing and Qingdao, which are typical inland and coastal cities, respectively, in northern China. The simulations included single-pollutant reduction strategies for NH₃, NO_x, and VOCs and combined reduction scenarios. For each pollutant, emissions were reduced by 20 %, 40 %, and 60 %, with a focus on assessing the resulting influence on nitrate formation. The combined reduction scenarios entailed simultaneous reductions in NH₃, NO_x, and VOCs to evaluate synergistic effects. The nitrate concentration responses to these reductions were analyzed to determine the most effective strategies for controlling winter nitrate levels.

3 Results and discussion

3.1 Model evaluation

3.1.1 Evaluation of meteorological parameters

The statistical model performance for meteorological parameters such as T2, WS10, WD10, and RH in the NCP is summarized in Table 1, covering the winter months from December 2013 to February 2014 (winter 2013) and from December 2018 to February 2019 (winter 2018). The simulated T2 and RH values exhibited satisfactory reproducibility, with temperature simulations exhibiting MB values slightly higher than the recommended threshold ($MB \leq \pm 0.5$). The simulated wind speeds during both winters were slightly overestimated, with the RMSE slightly exceeding the criterion (2 m s^{-1} ; Emery et al., 2001), whereas the wind direction results fully satisfied the criterion. The observed wind speed overestimation by the WRF model could be attributed to its

inability to accurately capture the impact of high aerosol loadings on shortwave radiation in winter, which likely reduced the near-surface wind speed (Tan et al., 2017; Jacobson and Kaufman, 2006). In general, the meteorological field simulations of the WRF model were reliable and could effectively reveal the changes in various meteorological elements in the NCP.

3.1.2 Evaluation of atmospheric pollutants

As indicated in Table 2, the model exhibited a favorable simulation ability for the concentrations of NO₂, MDA8 O₃, and PM_{2.5} pollutant in the NCP region. Specifically, the simulations accurately captured NO₂ concentrations, with NMB values ranging from -30% to 32% . Notably, during the winter of 2018, the simulated NO₂ concentration in Beijing was slightly higher than the recommended range (by 2 %), whereas that in other cities remained within this range. The MDA8 O₃ simulations generally exhibited slight underestimation, whereas in Tianjin, Shijiazhuang, and Zhengzhou, the simulations slightly exceeded the benchmark. This difference was primarily due to the uncertainty in anthropogenic VOC emissions (Wang et al., 2014), making accurate ozone simulations particularly challenging (Sun et al., 2022; Yang et al., 2024). The PM_{2.5} simulations were generally accurate, although the simulated concentrations in Tianjin during the winter of 2013 were slightly elevated, with an NMB of 32 %, exceeding the standard by 2 %; however, those in the other cities met the standard requirements.

Figure 3 shows a comparison between the simulated and observed NO₃⁻ concentrations in the six NCP cities during the winters of 2013 and 2018. Yantai was excluded from the comparison because of the inadequacy of the observation data. The data sources are listed in Table S4. The comparison results indicated that the simulated NO₃⁻ trends in the NCP were accurate during the winters of 2013 and 2018, with R values of 0.61 and 0.72, respectively, meeting the benchmark of 0.60. However, the simulation for 2013 yielded slightly overestimated values, with an MB of 1.27, corresponding to $NMB = 7\%$ and $NME = 51\%$. In contrast, the simulation results for 2018 were closer to the observed values, with $NMB = 5\%$ and $NME = 51\%$. Overall, the model demonstrated suitable stability and accuracy in simulating atmospheric pollutants in the NCP region, providing a solid foundation for future analysis.

3.2 Variations in the concentrations of PM_{2.5} and its components between 2013 and 2018

During the winter of 2013, the average PM_{2.5} concentration in the NCP reached $80.52 \pm 77.13 \mu\text{g m}^{-3}$, with high-concentration areas mainly concentrated in the provinces of Beijing, Tianjin, southern Hebei, Henan, Shandong, and Anhui (Fig. 4). By 2018, the average PM_{2.5} concentration had decreased to $50.74 \pm 49.26 \mu\text{g m}^{-3}$, representing a significant

Table 1. Statistical performance of the modeled meteorological parameters in the NCP during the winters of 2013 and 2018 (68 sites).

| Parameters | Winter 2013 | | | | Winter 2018 | | | | Benchmark | | | |
|---------------------------|-------------|--------|------|------|-------------|--------|------|------|----------------|----------------|----------------|---|
| | MB | RMSE | IOA | R | MB | RMSE | IOA | R | MB | RMSE | IOA | R |
| T2 (°C) | 0.60 | 3.25 | 0.93 | 0.88 | 1.15 | 3.14 | 0.94 | 0.90 | $\leq \pm 0.5$ | / | $\geq \pm 0.8$ | / |
| RH (%) | −3.67 | 17.53 | 0.82 | 0.69 | −5.35 | 19.36 | 0.81 | 0.68 | / | / | / | / |
| WS10 (m s ^{−1}) | 1.55 | 2.79 | 0.61 | 0.47 | 1.47 | 2.72 | 0.60 | 0.46 | $\leq \pm 0.5$ | $\leq \pm 2.0$ | ≥ 0.6 | / |
| WD10 (°) | 5.44 | 115.02 | 0.70 | 0.43 | −1.65 | 121.00 | 0.69 | 0.42 | $\leq \pm 10$ | / | / | / |

* T2 denotes the 2 m temperature. RH denotes the relative humidity. WS10 denotes the 10 m wind speed. WD10 denotes the 10 m wind direction.

Table 2. Model performance for the major air pollutants in typical cities of the NCP during the winters of 2013 and 2018.

| City | Pollutants ($\mu\text{g m}^{-3}$) | 2013 winter | | | | 2018 winter | | | |
|--------------------|--|-----------------|-------------|--------|----------|-------------|------|--------|------|
| | | NMB | NME | MB | R | NMB | NME | MB | R |
| Beijing (BJ) | MDA8 O ₃ | −5 % | 25 % | −2.04 | 0.76 | −13 % | 25 % | −6.80 | 0.68 |
| | NO ₂ | 1 % | 19 % | 0.08 | 0.89 | 32 % | 46 % | 13.85 | 0.73 |
| | PM _{2.5} | −16 % | 28 % | −17.41 | 0.88 | −10 % | 45 % | −5.11 | 0.64 |
| Tianjin (TJ) | MDA8 O ₃ | −4 % | 30 % | −1.23 | 0.73 | −29 % | 33 % | −14.19 | 0.63 |
| | NO ₂ | 8 % | 22 % | 5.53 | 0.76 | 26 % | 39 % | 14.16 | 0.68 |
| | PM _{2.5} | 32 % | 41 % | 34.73 | 0.81 | 2 % | 51 % | 1.75 | 0.62 |
| Shijiazhuang (SJZ) | MDA8 O ₃ | −15 % | 29 % | −5.49 | 0.64 | −25 % | 35 % | −11.25 | 0.58 |
| | NO ₂ | −5 % | 27 % | −3.66 | 0.73 | 12 % | 32 % | 7.40 | 0.61 |
| | PM _{2.5} | −25 % | 34 % | −48.32 | 0.80 | −20 % | 39 % | −23.13 | 0.59 |
| Jinan (JN) | MDA8 O ₃ | 2 % | 25 % | 0.87 | 0.77 | −11 % | 31 % | −6.41 | 0.54 |
| | NO ₂ | −14 % | 20 % | −10.20 | 0.73 | −8 % | 30 % | −5.01 | 0.61 |
| | PM _{2.5} | −7 % | 21 % | −9.54 | 0.85 | −10 % | 33 % | −8.82 | 0.70 |
| Zhengzhou (ZZ) | MDA8 O ₃ | −6 % | 49 % | −2.65 | 0.03 | 1 % | 34 % | 0.16 | 0.63 |
| | NO ₂ | 12 % | 24 % | 7.52 | 0.69 | 11 % | 34 % | 6.10 | 0.61 |
| | PM _{2.5} | 24 % | 33 % | 30.63 | 0.85 | −9 % | 36 % | −10.34 | 0.63 |
| Qingdao (QD) | MDA8 O ₃ | −1 % | 21 % | −0.66 | 0.82 | −3 % | 22 % | −1.91 | 0.47 |
| | NO ₂ | 11 % | 23 % | 7.52 | 0.69 | −4 % | 28 % | −1.99 | 0.76 |
| | PM _{2.5} | 4 % | 21 % | 3.70 | 0.89 | −3 % | 38 % | −1.84 | 0.70 |
| Yantai (YT) | MDA8 O ₃ | 5 % | 17 % | 2.48 | 0.79 | −10 % | 22 % | −6.50 | 0.27 |
| | NO ₂ | −30 % | 32 % | −15.34 | 0.81 | −9 % | 33 % | −3.31 | 0.78 |
| | PM _{2.5} | −6 % | 25 % | −4.09 | 0.84 | −19 % | 33 % | −10.84 | 0.78 |
| Benchmark | MDA8 O ₃ | $\leq \pm 15$ % | < 25 % | / | > 0.50 | / | / | / | / |
| | NO ₂ | $\leq \pm 30$ % | ≤ 75 % | / | / | / | / | / | / |
| | PM _{2.5} | $\leq \pm 30$ % | < 50 % | / | > 0.40 | / | / | / | / |

* The benchmarks for these pollutants, including NO₂ according to US-EPA (2007), MDA8 O₃, and PM_{2.5}, were proposed by Emery et al. (2017) and Huang et al. (2021).

reduction of 37 ± 10 %. Previously, scholars reported similar trends in PM_{2.5} reduction across China (Li et al., 2023; Zhai et al., 2019). We found that the SO₄^{2−} concentration decreased by 53 ± 5 %, whereas the NO₃[−] and NH₄⁺ concentrations decreased by 13 ± 20 % and 33 ± 13 %, respectively (refer to Fig. S2 for city-specific changes). Our analysis revealed a pronounced difference in nitrate concentration. Specifically, inland cities experienced a more pronounced reduction in nitrate concentrations (4.6 % to 41.7 %), whereas

coastal cities showed relatively small changes, with Yantai experiencing a 4.8 % decrease and Qingdao exhibiting a 3.9 % increase. The substantial reduction in the three inorganic components (SO₄^{2−}, NH₄⁺, and NO₃[−]) led to significant changes in the PM_{2.5} composition, with the proportion of nitrate increasing relative to that of sulfate (Fig. 5). Nitrate concentrations demonstrated significantly different patterns of reduction between coastal and inland regions, necessitat-

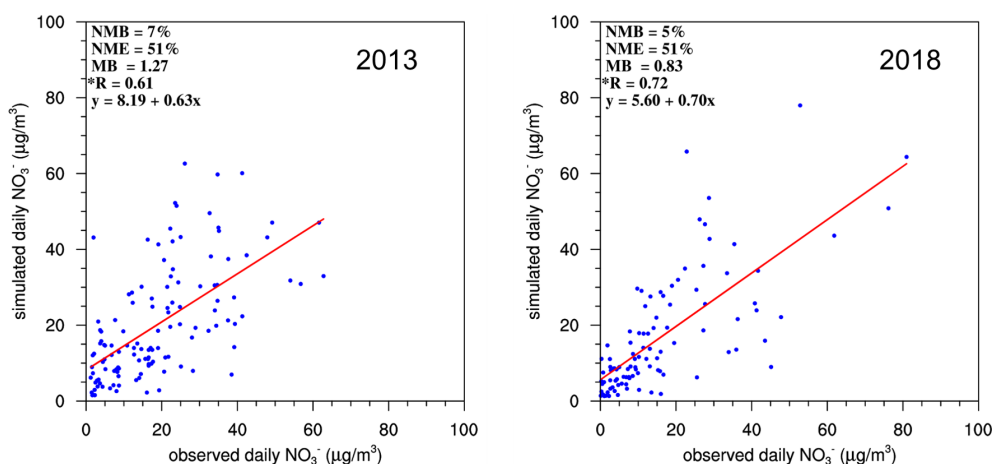


Figure 3. Scatter plots of the simulated and observed daily mean NO_3^- concentrations in the NCP during the winter seasons of 2013 and 2018; the * symbol indicates R values that are statistically significant ($P < 0.05$).

ing the identification of their dominant controlling factors to facilitate effective regional pollution control strategies.

3.3 Rates, contributions, and diurnal variations in nitrate formation reactions in the major cities

The NO_3^- concentration slightly decreased from 2013 to 2018 due to a decrease in the TNO_3 production rate ($0\text{--}0.05\text{ ppb h}^{-1}$), which could be attributed to the production rate of the hetN_2O_5 pathway ($0\text{--}0.03\text{ ppb h}^{-1}$). According to Table 3, the contribution increased in Zhengzhou from 2013 to 2018, whereas it decreased in six cities, including Beijing (-4.4%), Tianjin (-4.6%), Shijiazhuang (-5.9%), Jinan (-7.8%), Qingdao (-4.5%), Yantai (-2.2%), and Zhengzhou ($+2.1\%$). For the $\text{OH} + \text{NO}_2$ reaction pathway, the rate remained unchanged in all but four cities (Beijing, Jinan, Zhengzhou, and Qingdao); it varied between -0.02 and 0.01 ppb h^{-1} in the other cities. However, the contribution of the $\text{OH} + \text{NO}_2$ reaction pathway to TNO_3 formation increased overall due to the decreased rate of the hetN_2O_5 reaction pathway, although the production rate remained almost unchanged. Moreover, the contribution increase was greater in inland cities than in coastal cities. Among the five inland cities (Beijing, Tianjin, Shijiazhuang, Jinan, and Zhengzhou), the $\text{OH} + \text{NO}_2$ reaction pathway was the main nitrate formation mechanism, with its contribution ratio ranging from 63.7% – 77.8% in 2013 and increasing to 72.9% – 85.6% in 2018. In contrast, in the two coastal cities (Qingdao and Yantai), the $\text{OH} + \text{NO}_2$ reaction pathway contributed to 48.2% – 56.5% of nitrate formation. However, the contribution of the hetN_2O_5 pathway ranged from 37.0% – 45.7% . Therefore, both pathways were equally important in nitrate formation in coastal cities. These findings highlighted notable differences in nitrate formation mechanisms between inland cities and coastal cities, demonstrating that nitrate formation mechanisms were strongly affected by the ocean.

Since the $\text{OH} + \text{NO}_2$ and hetN_2O_5 reactions were the major pathways affecting nitrate formation, we comprehensively analyzed the diurnal variations in these two TNO_3 formation reactions in the NCP, considering the inland city of Beijing and the coastal city of Qingdao as examples. Figure 6 shows the average diurnal variations in the TNO_3 production rates via the different pathways in Qingdao and Beijing. We found that the $\text{OH} + \text{NO}_2$ reaction pathway exhibited similar diurnal characteristics in these two cities. Specifically, from night to the early morning (00:00 to 08:00 CST (UTC+8)), the reaction rate under this pathway remained almost zero because of the lack of photochemical processes that generated OH radicals; these processes were limited by solar radiation (Liu et al., 2020; Sun et al., 2022; Tan et al., 2021). Observational data from the Pearl River Delta region supported this result, showing that OH radicals only began to accumulate significantly after 06:00 CST (UTC+8) (Hofzumahaus et al., 2009; Lu et al., 2012). After approximately 08:00 CST (UTC+8), the reaction rate rapidly increased, reaching a peak between 11:00 and 13:00 CST (UTC+8) at approximately 0.4 ppb h^{-1} ($\sim 2.8\text{ }\mu\text{g m}^{-3}\text{ h}^{-1}$), resembling the findings for the NCP and Shanghai (Tan et al., 2021; Sun et al., 2022; Liu et al., 2020). The reaction rate decreased with decreasing sunlight during the afternoon and returned to low levels at night due to a decrease in the number of photochemical processes that occurred. Compared with those during the winter of 2013, the peak reaction rates of the OH pathway decreased from 0.43 to 0.42 ppb h^{-1} and from 0.45 to 0.44 ppb h^{-1} in 2018 in Qingdao and Beijing, respectively. The contributions of the $\text{OH} + \text{NO}_2$ reaction pathway to TNO_3 formation were in the ranges of 90.2% – 95.1% and 75.8% – 82.2% in Beijing and Qingdao, respectively, during the daytime. However, the other cities in the NCP exhibited similar diurnal variations. Therefore, the $\text{OH} + \text{NO}_2$ reaction pathway dominated during the daytime in both the inland and coastal cities.

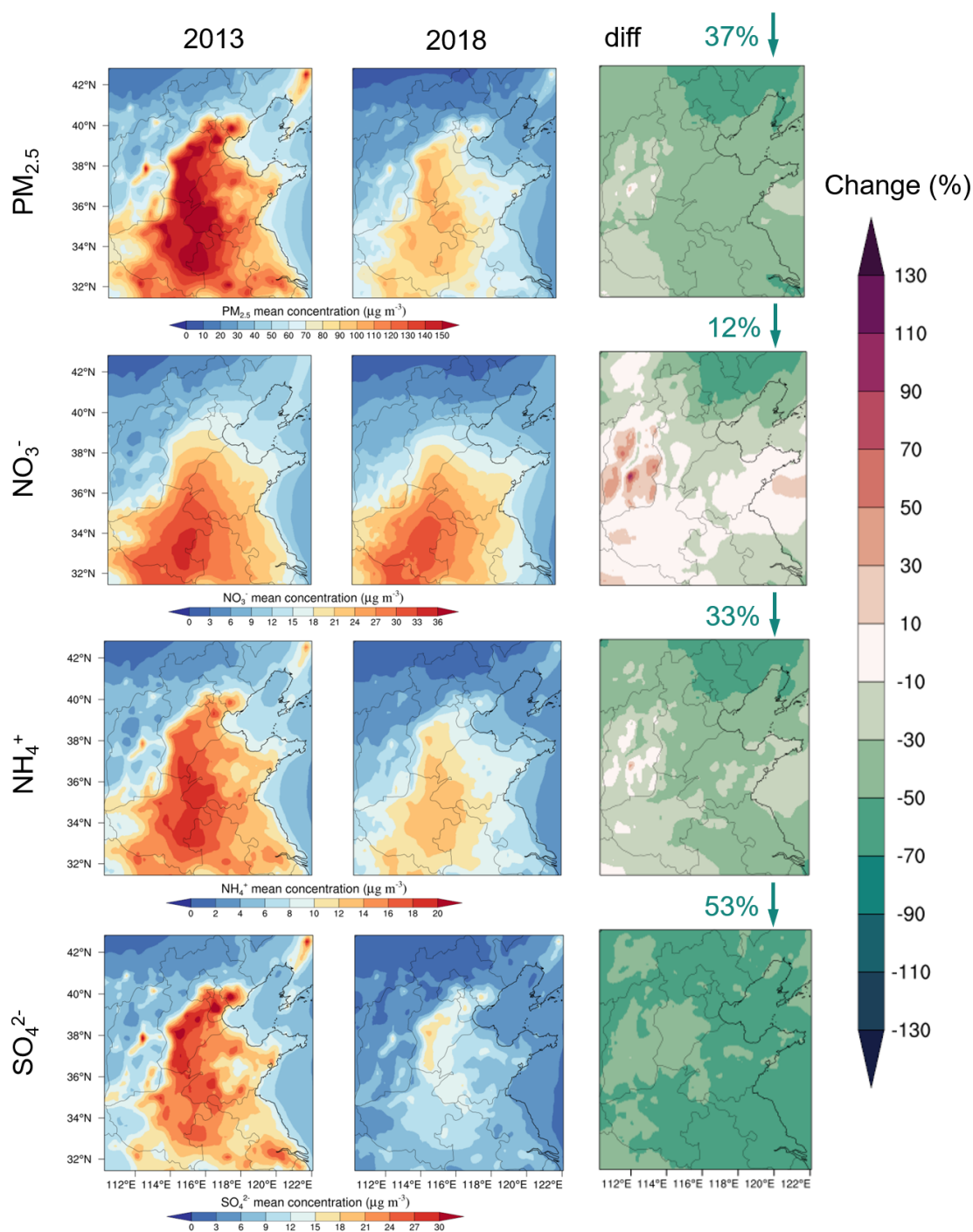


Figure 4. Spatial distribution and changes in $\text{PM}_{2.5}$ and its components in the NCP during the winters of 2013 and 2018. The up arrows indicate increases, and the down arrows indicate decreases.

At night, the hetN_2O_5 reaction pathway notably affected TNO_3 formation. The rate of the hetN_2O_5 reaction pathway showed similar bimodal diurnal variations in Qingdao and Beijing. However, the peak values occurred at 02:00 and 20:00 CST (UTC+8) in Qingdao, with much higher peak values than those in Beijing, where the peak values

occurred from 04:00–05:00 and 18:00 CST (UTC+8). The hetN_2O_5 reaction rate was relatively high at night but decreased rapidly by early morning, reaching its lowest point during the daytime (09:00 to 16:00 CST (UTC+8)). This phenomenon occurred because the N_2O_5 hydrolysis reaction depended heavily on NO_3 radicals and N_2O_5 , which

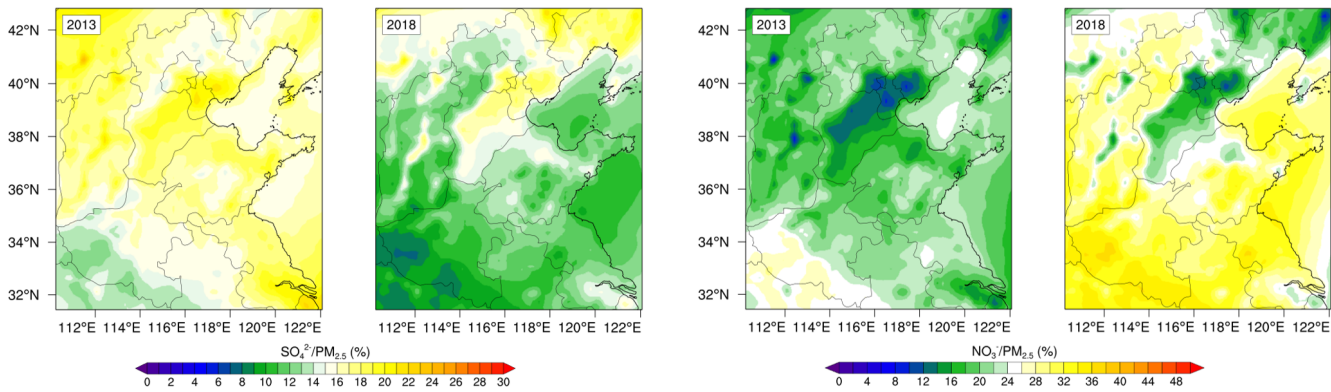


Figure 5. Spatial distributions of the nitrate and sulfate proportions in PM_{2.5} in the NCP region during the winters of 2013 and 2018.

Table 3. TNO₃ production rates (ppb h^{−1}) in the surface layer and contributions (%) of the major production pathways in the surface atmosphere in seven representative cities of the NCP in 2013 and 2018.

| City | Year | Production rates (ppb h ^{−1}) | | | Daytime contribution (%) | | | Nighttime contribution (%) | | | Total contribution (%) | | |
|------|------|---|-------------------|----------------------------------|--------------------------|----------------------------------|-------|----------------------------|----------------------------------|--------|------------------------|----------------------------------|--------|
| | | TNO ₃ | OHNO ₂ | hetN ₂ O ₅ | OHNO ₂ | hetN ₂ O ₅ | Other | OHNO ₂ | hetN ₂ O ₅ | Others | OHNO ₂ | hetN ₂ O ₅ | Others |
| BJ | 2013 | 0.19 ± 0.15 | 0.15 ± 0.16 | 0.03 ± 0.02 | 90.2 % | 5.2 % | 4.6 % | 33.4 % | 46.9 % | 19.7 % | 77.8 % | 14.3 % | 7.9 % |
| | 2018 | 0.17 ± 0.15 | 0.15 ± 0.16 | 0.02 ± 0.01 | 95.1 % | 2.6 % | 2.3 % | 48.4 % | 39.4 % | 12.2 % | 85.6 % | 9.9 % | 4.5 % |
| TJ | 2013 | 0.22 ± 0.17 | 0.15 ± 0.19 | 0.04 ± 0.03 | 85.7 % | 6.5 % | 7.8 % | 15.2 % | 58.8 % | 25.9 % | 67.8 % | 19.7 % | 12.4 % |
| | 2018 | 0.17 ± 0.15 | 0.13 ± 0.17 | 0.03 ± 0.02 | 92.2 % | 3.9 % | 3.9 % | 22.4 % | 57.1 % | 20.4 % | 77.5 % | 15.1 % | 7.4 % |
| SJZ | 2013 | 0.19 ± 0.16 | 0.14 ± 0.17 | 0.03 ± 0.01 | 87.8 % | 5.5 % | 6.6 % | 21.2 % | 46.8 % | 32.0 % | 73.8 % | 14.2 % | 12.0 % |
| | 2018 | 0.18 ± 0.17 | 0.15 ± 0.18 | 0.02 ± 0.01 | 93.8 % | 2.0 % | 4.2 % | 30.3 % | 41.1 % | 28.6 % | 83.6 % | 8.3 % | 8.1 % |
| JN | 2013 | 0.26 ± 0.14 | 0.16 ± 0.18 | 0.07 ± 0.05 | 85.1 % | 8.9 % | 6.0 % | 21.1 % | 65.4 % | 13.5 % | 63.7 % | 27.7 % | 8.5 % |
| | 2018 | 0.22 ± 0.15 | 0.16 ± 0.18 | 0.04 ± 0.03 | 90.7 % | 4.4 % | 4.9 % | 27.6 % | 59.5 % | 12.9 % | 72.9 % | 19.9 % | 7.1 % |
| ZZ | 2013 | 0.24 ± 0.21 | 0.19 ± 0.23 | 0.03 ± 0.01 | 88.9 % | 4.7 % | 6.4 % | 26.8 % | 37.4 % | 35.8 % | 77.6 % | 10.6 % | 11.8 % |
| | 2018 | 0.24 ± 0.20 | 0.19 ± 0.22 | 0.03 ± 0.02 | 91.9 % | 3.1 % | 5.0 % | 32.8 % | 47.5 % | 19.6 % | 79.2 % | 12.7 % | 8.2 % |
| QD | 2013 | 0.27 ± 0.09 | 0.14 ± 0.14 | 0.11 ± 0.07 | 75.8 % | 17.8 % | 6.4 % | 16.5 % | 75.6 % | 7.9 % | 51.5 % | 41.5 % | 7.0 % |
| | 2018 | 0.25 ± 0.09 | 0.14 ± 0.15 | 0.09 ± 0.06 | 82.2 % | 12.0 % | 5.8 % | 17.5 % | 74.8 % | 7.6 % | 56.5 % | 37.0 % | 6.5 % |
| YT | 2013 | 0.24 ± 0.06 | 0.12 ± 0.12 | 0.11 ± 0.07 | 76.0 % | 18.1 % | 5.9 % | 14.7 % | 78.9 % | 6.4 % | 48.2 % | 45.7 % | 6.1 % |
| | 2018 | 0.21 ± 0.06 | 0.11 ± 0.11 | 0.09 ± 0.07 | 79.9 % | 14.7 % | 5.4 % | 14.9 % | 78.4 % | 6.7 % | 50.4 % | 43.5 % | 6.0 % |

were only stable in weakly illuminated environments (Tie et al., 2003a, b; Atkinson et al., 2004).

High concentrations of NO₂ and O₃, precursors for N₂O₅, were considered key in this process (Dentener and Crutzen, 1993).



However, owing to high rates of photolysis during the day, NO₃ radicals and N₂O₅ could accumulate only at night (Zhao et al., 2023; Atkinson et al., 2004). In Beijing, the hetN₂O₅ reaction rate was relatively low, with a peak value of only 0.03 ppb h^{−1} and a nighttime contribution rate of 46.9 %. In contrast, Qingdao presented a peak hetN₂O₅ reaction rate of 0.2 ppb h^{−1} (approximately 1.4 μg m^{−3} h^{−1}) at night, with an

average contribution of 75.2 %. This difference arose primarily due to the higher N₂O₅ concentrations and RH levels in Qingdao than in Beijing, as detailed in Sect. 3.5.3. The other inland cities exhibited diurnal variations similar to those in Beijing, whereas the coastal cities presented trends similar to those in Qingdao. Compared with those during the winter of 2013, the reaction rates of the hetN₂O₅ pathway in both cities during the winter of 2018 decreased from 0.19 to 0.18 ppb h^{−1} in Qingdao and from 0.07 to 0.03 ppb h^{−1} in Beijing. A comparative analysis revealed significant variations in the nitrate formation pathways between the coastal and inland areas, particularly in the hetN₂O₅ reaction rate (Fig. 7). The factors influencing these variations are discussed in Sect. 3.5.3, which specifically addresses the under-

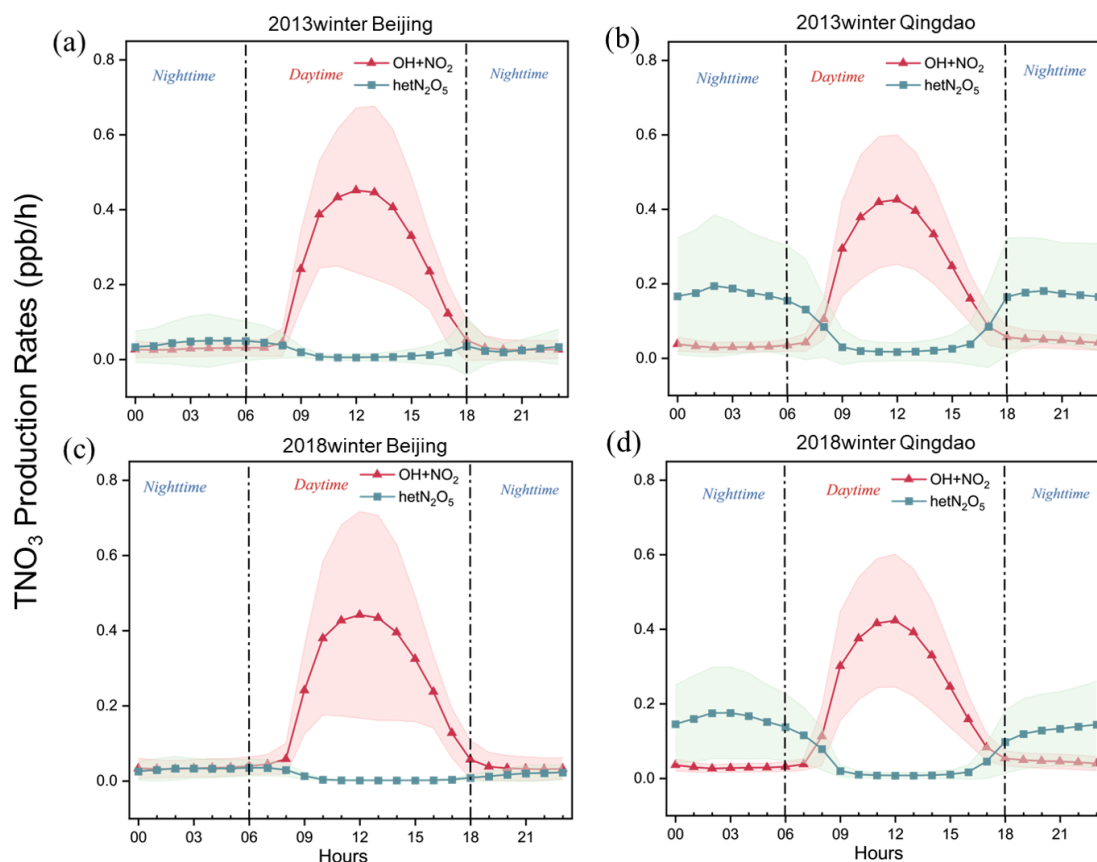


Figure 6. Average diurnal variation in the TNO_3 production rate under different pathways (daytime: 06:00–18:00 CST (UTC+8); nighttime: 18:00–06:00 CST (UTC+8)): (a) Beijing 2013, (b) Qingdao 2013, (c) Beijing 2018, and (d) Qingdao 2018.

lying mechanisms responsible for the differences in hetN_2O_5 reaction rates between coastal and inland regions.

3.4 Assessment of the simulated nitrate formation reactions via the isotopic method

To verify the accuracy of nitrate formation reaction simulations, we conducted a comprehensive isotope analysis in Qingdao and Beijing. The isotope analysis conducted in Qingdao and Beijing for the 2013 and 2018 data (Fig. 8) revealed that the model-simulated and isotope-based contributions of NO_3^- formation pathways generally exhibited consistent trends, although some differences were observed. In 2013 in Qingdao, the CMAQ model yielded contribution values of 72.9 % ($\text{OH} + \text{NO}_2$) and 18.7 % (hetN_2O_5), whereas the isotope analysis suggested 62.4 ± 22.5 % and 37.6 ± 22.5 %, respectively. By 2018 in Beijing, the model yielded 89.5 % vs. an isotope-derived 76.5 ± 15.8 % for $\text{OH} + \text{NO}_2$ and 5.4 % vs. 23.5 ± 15.8 % for hetN_2O_5 . Overall, although the differences persisted, the model captured the general trends, and the deviations fell within an acceptable range, indicating that the model simulations generally agreed with the isotope analysis. Several factors could explain these

discrepancies. First, the isotope analysis carried intrinsic uncertainties, commonly approximately 10 %–30 %, due to the standard deviations (SDs) in each pathway's endmember calculations and the posterior probability distributions generated by the Bayesian model (Moore and Semmens, 2008). Second, the isotope analysis considered only two principal pathways: $\text{OH} + \text{NO}_2$ and hetN_2O_5 . Although the other pathways only contributed to a small fraction of NO_3^- formation, their omission could cause biases, especially under specific meteorological or pollution conditions. Third, the IRR module did not consider long-range transport effects, which could further contribute to the discrepancies in the isotope results. In brief, the model simulation results align reasonably well with the isotope analysis findings despite some discrepancies between the two methods.

Notably, both the model and isotope results indicated an increasing trend in the $\text{OH} + \text{NO}_2$ pathway contribution in Beijing from 2013 to 2018 (Fig. 9), with the model and isotope analysis showing increases of 5.1 % and 9.7 %, respectively. This trend reflected the increased atmospheric oxidation capacity in the region. Regional comparative analysis revealed significant differences in nitrate formation pathways between Qingdao and Beijing (Fig. 9). In 2018, the Qingdao

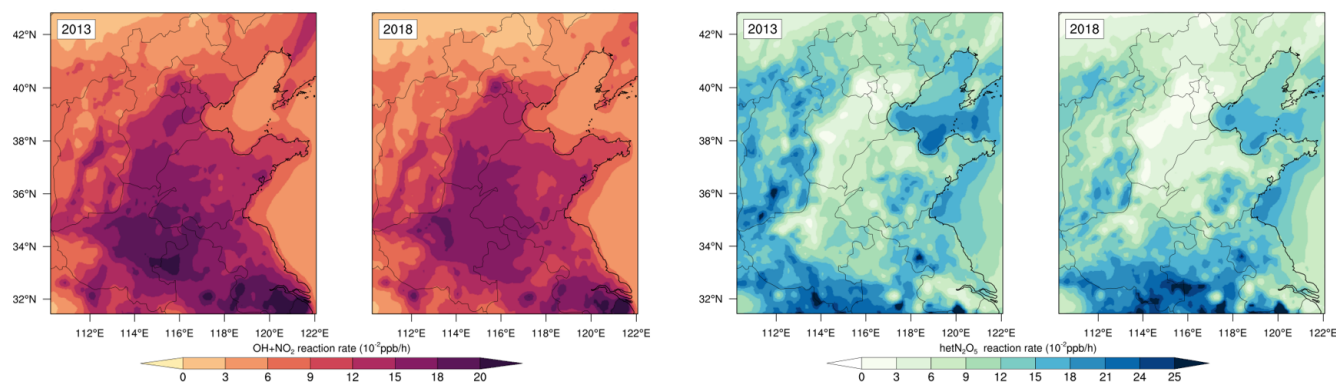


Figure 7. Spatial distributions of the mean reaction rates for the nitrate formation pathways ($\text{OH} + \text{NO}_2$ and hetN_2O_5) in the NCP during the winters of 2013 and 2018.

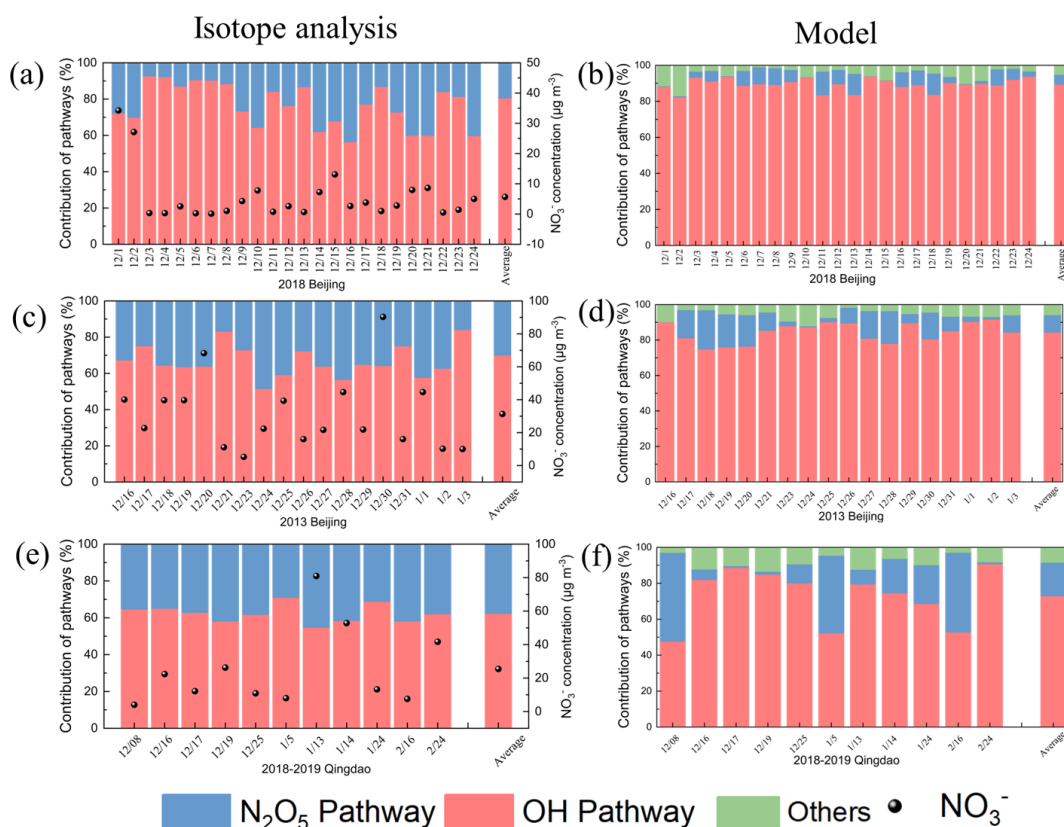


Figure 8. Time series of the contributions of the atmospheric NO_3^- formation pathways: (a, b) Beijing 2018, (c, d) Beijing 2013, and (e, f) Qingdao 2018, based on (a, c, e) dual-isotope analysis and (b, d, f) model simulations.

model results showed contributions of 72.9 % and 18.7 % to the total nitrate formation for the $\text{OH} + \text{NO}_2$ and hetN_2O_5 pathways, respectively, compared with the isotope results of 62.4 ± 22.5 % and 37.6 ± 22.5 %, respectively. In contrast, the Beijing model results indicated contributions of 89.5 % and 5.4 % to the total nitrate formation for the $\text{OH} + \text{NO}_2$ and hetN_2O_5 pathways, respectively, whereas the isotope results were 76.5 ± 15.8 % and 23.5 ± 15.8 %, respectively. Both methods demonstrated that the $\text{OH} + \text{NO}_2$ pathway domi-

nated nitrate formation in Beijing, while the hetN_2O_5 pathway played a substantial role in Qingdao, highlighting regional differences in the mechanism of nitrate formation and resulting in different changes in the nitrate concentration.

As discussed above, field observation data could include contributions from long-range transport (Walters et al., 2024); however, the CMAQ model's IRR module confined nitrate formation to individual grid cells. To quantify this discrepancy, we employed a backward trajectory analy-

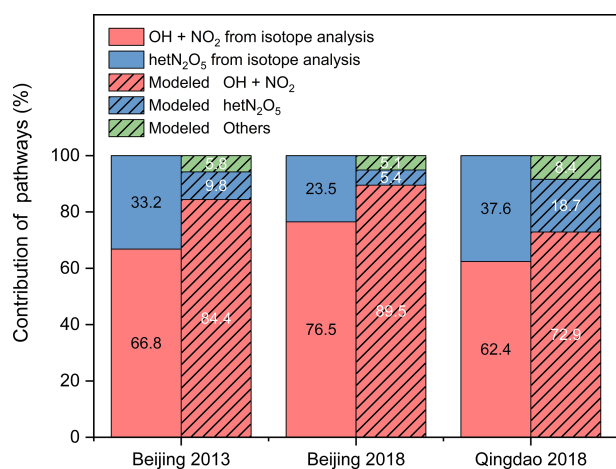


Figure 9. Comparison of the contributions of the atmospheric NO_3^- formation pathways based on the dual-isotope results and model simulations for Beijing in 2013 and 2018 and for Qingdao in 2018.

sis method incorporating temporal and spatial constraints to calculate the source regions and their contributions to the nitrate formation pathway.

The backward trajectory correction significantly altered the estimated nitrate formation pathways (Fig. 10). In Qingdao, the corrected model results revealed an average $\text{OH} + \text{NO}_2$ pathway contribution of 33.3 %, which was substantially lower than the isotope analysis result of 62.4 ± 22.5 %, whereas the hetN_2O_5 pathway contribution was 58.4 %, which was significantly greater than the isotope result of 37.6 ± 22.5 %. In Beijing, the corrected model in 2013 indicated an $\text{OH} + \text{NO}_2$ pathway contribution of 48.3 % (isotope: 66.8 ± 26.1 %) and a hetN_2O_5 pathway contribution of 38.5 % (isotope: 33.2 ± 26.1 %). In 2018, the corrected model showed an $\text{OH} + \text{NO}_2$ pathway contribution of 57.2 % (isotope: 76.5 ± 15.8 %) and a hetN_2O_5 pathway contribution of 35.1 % (isotope: 23.5 ± 15.8 %). Compared with the uncorrected model, the backward trajectory correction significantly improved the estimation of the hetN_2O_5 pathway contribution (Qingdao: from -18.9 % to $+20.8$ %; Beijing: from -18.1 % to $+11.6$ %) but exacerbated the discrepancy in the $\text{OH} + \text{NO}_2$ pathway contribution (Qingdao: from $+10.5$ % to -29.1 %; Beijing: from $+13.0$ % to -19.3 %). The effects of this correction exhibited regional differences: the improvement in the hetN_2O_5 pathway was more pronounced in Qingdao, where regional transport had a greater impact, whereas the overadjustment of the $\text{OH} + \text{NO}_2$ pathway was more evident in Beijing, where local processes were dominant.

Remarkably, Walters et al. (2024) achieved relatively accurate results using the unmodified IRR module without considering transport effects, reporting an overall RMSE of 2.6 ‰ for $\Delta^{17}\text{O}(\text{HNO}_3)$ between modeled and observed values. This finding indicated that if the trends in nitrate formation pathways were similar across regions and vertical lay-

ers, the impact of transport on the overall simulation accuracy could be limited (Walters et al., 2024). Our results highlighted that in regions with significant spatial heterogeneity in formation pathways, such as Qingdao and Beijing, incorporating transport effects through backward trajectory analysis remained crucial for improving model performance.

These results demonstrated that backward trajectory correction played a positive role in improving the estimation of the hetN_2O_5 pathway, particularly in regions significantly influenced by regional transport. However, the overadjustment of the $\text{OH} + \text{NO}_2$ pathway highlighted the need for further optimization of the correction method. Key areas for improvement would include refining OH radical concentration estimates and optimizing the quantification of regional transport. Currently, the overly broad consideration of transport regions could obscure local formation signals, suggesting a need for increasingly precise spatial delineation in transport modeling. These enhancements would be essential for improving the model's applicability across different regions and under varied meteorological conditions. Currently, owing to the lack of successful transport correction methodologies, we discuss the influencing factors without applying correctional procedures.

3.5 Factors influencing nitrate formation pathways

To explore the reasons for the observed coastal–inland differences in nitrate formation, we further investigated the influences of several key factors, including NH_3 , NO_2 , OH, O_3 , RH, NO, and N_2O_5 , on these pathways.

3.5.1 Influence of NH_3 on nitrate formation

The reaction of NH_3 with nitric acid (HNO_3) to form ammonium nitrate (NH_4NO_3) was considered an important process in the formation of atmospheric aerosols. In the absence of sufficient NH_3 , the material would first react with H_2SO_4 to form $(\text{NH}_4)_2\text{SO}_4$ and then with HNO_3 to form NH_4NO_3 (Zhai et al., 2021). The availability of NH_3 was considered a critical factor governing the partitioning of HNO_3 to particulate nitrate (pNO_3). The gas ratio (GR) (Fu et al., 2020; Ansari and Pandis, 1998) was used to study whether NH_3 limits nitrate formation. The GR could be calculated as follows.

$$\text{GR} = \frac{([\text{NH}_3] + [\text{NH}_4^+]) - 2 \times [\text{SO}_4^{2-}]}{[\text{NO}_3^-] + [\text{HNO}_3]}$$

When the GR exceeded 1, the NH_3 concentration in the atmosphere was considered sufficient. A GR value between 0 and 1 indicated NH_3 -neutral conditions, whereas a value less than 0 suggested NH_3 -poor conditions, thus limiting NO_3^- formation due to insufficient NH_3 . In the NCP region, the GR values were generally greater than 2 in 2013 and 2018 (Fig. 11), indicating that the region mainly exhibited

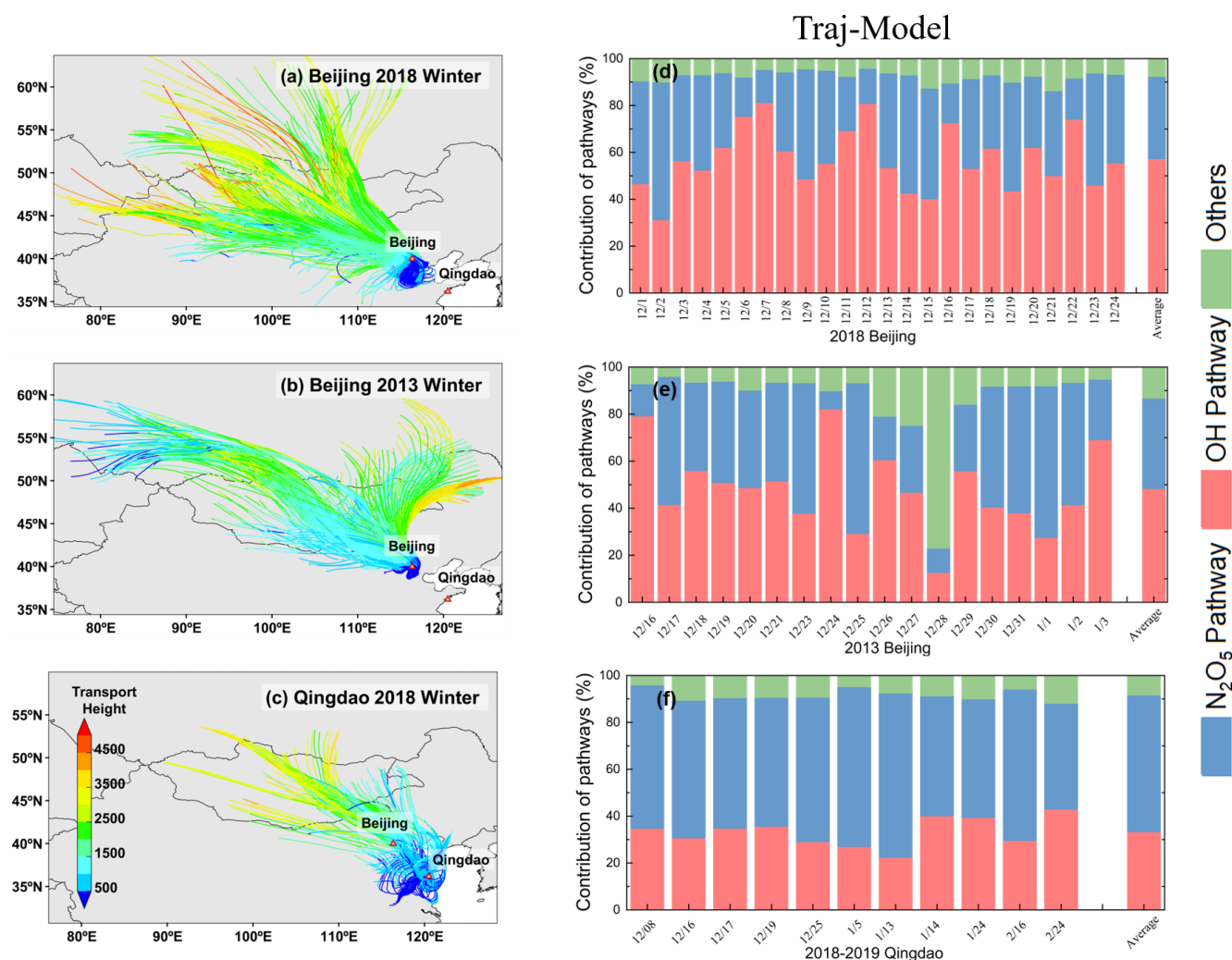


Figure 10. Backward trajectories and nitrate formation pathways: (a–c) backward trajectories for (a) Beijing 2018, (b) Beijing 2013, and (c) Qingdao 2018; time series of the contributions of the atmospheric NO_3^- formation pathways based on backward-trajectory-constrained model simulations for (c) Beijing 2018, (d) Beijing 2013, and (e) Qingdao 2018.

a state of sufficient NH_3 . Similar phenomena were observed by other researchers in the NCP region (Zhai et al., 2021; Xu et al., 2019c; Li et al., 2018). Therefore, in the NCP region, the formation of NH_4NO_3 was not limited by NH_3 in terms of supply.

The GR in 2018 was greater than that in 2013, indicating an increasing surplus of NH_3 . We believed that the variation in GR was caused by the significant reduction in sulfate and slight reduction in nitrate in $\text{PM}_{2.5}$ (Fig. 12a) because of the minimal changes in NH_3 emissions. This hypothesis could be verified from an emission perspective. The total ammonia emissions in China increased from 9.64 to 9.75 Tg from 2013–2015 and then gradually decreased to 9.12 Tg by 2018 (Liao et al., 2022). Additionally, there were coastal–inland differences in NH_3 surplus levels. Notably, the NH_3 surplus ratio was consistently greater than 3.4 in inland regions, whereas it ranged from 1.7 to 2.5 in coastal cities. This

difference was attributed primarily to intensive industrial activities related to fertilizer production in inland areas, which resulted in increased NH_3 emissions. Data showed that NH_3 emissions from fertilizer production in the inland regions of the NCP ranged from 1.0 to 40.0 Gg yr^{-1} , whereas emissions in coastal areas were significantly lower, ranging only from 0 to 0.1 Gg yr^{-1} (Chen et al., 2022).

In summary, NH_3 played a critical role in influencing particulate nitrate concentrations by affecting the gas-to-particle conversion of HNO_3 in the NCP region, although its availability was sufficient. Furthermore, variations in NH_3 emissions and distinct land–ocean transport patterns significantly affected the formation and spatial distribution of particulate nitrate.

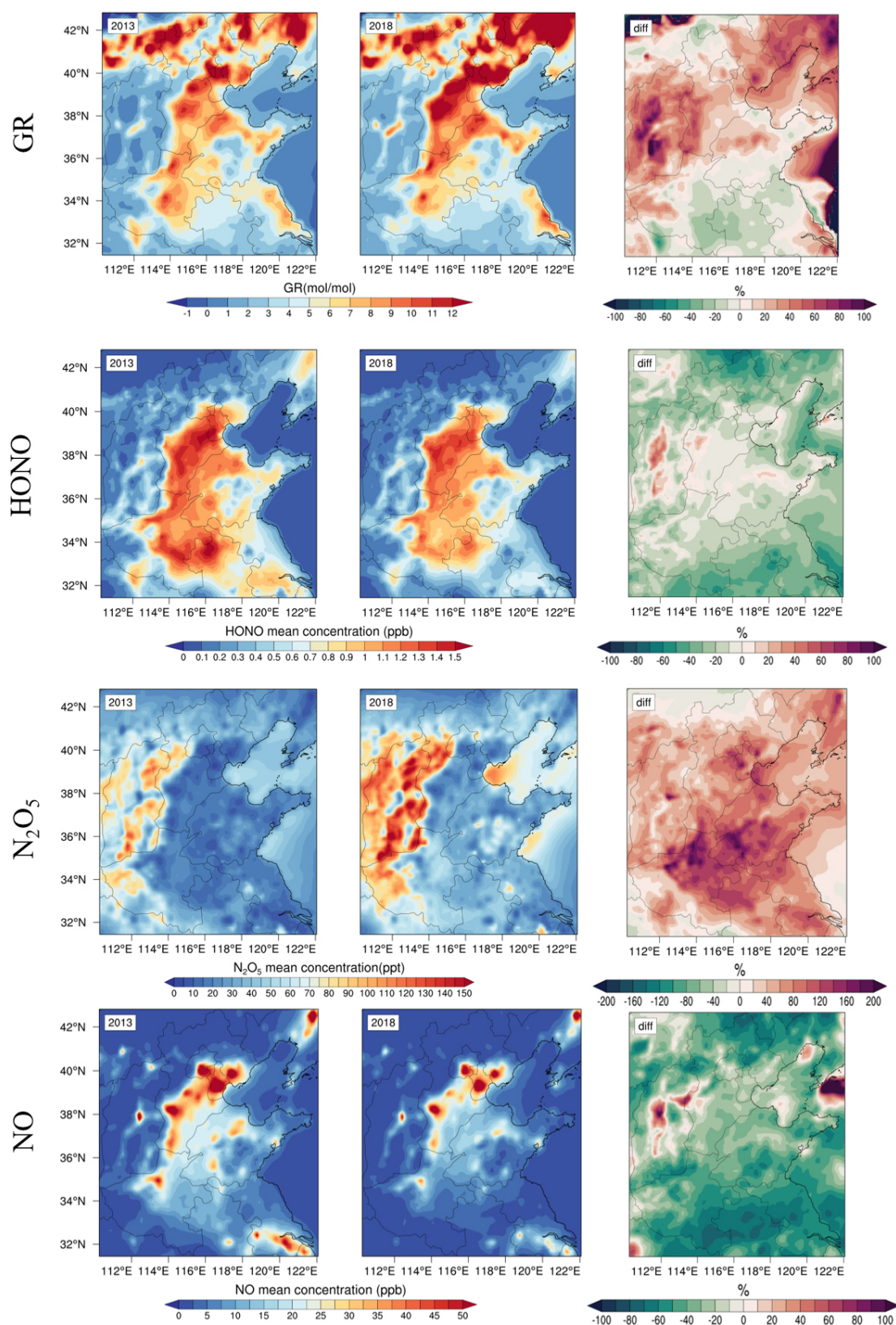


Figure 11. Spatial distributions and interannual variations in the GR, HONO, N_2O_5 , and NO_x concentrations over the North China Plain during the winters of 2013 and 2018. The percentage changes (diff) represent the relative differences between 2018 and 2013.

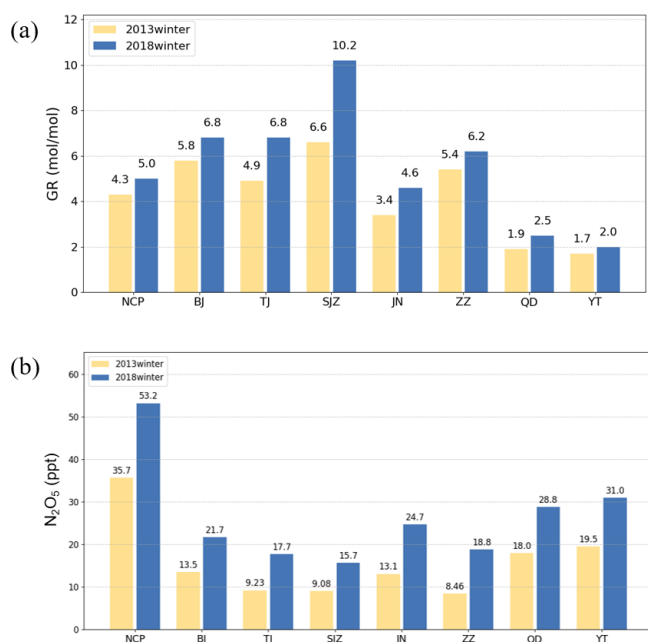


Figure 12. (a) GR and (b) N₂O₅ concentrations across the North China Plain and seven major cities during the winters of 2013 and 2018.

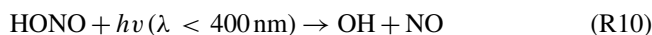
3.5.2 Factors influencing the OH + NO₂ reaction rate

As the OH + NO₂ reaction significantly contributed to TNO₃ formation, the OH and NO₂ concentrations served as key factors. The NO₂ concentration in the NCP was 30.3 μg m⁻³ during the winter of 2013 and decreased to 24.7 μg m⁻³ during the winter of 2018, and its concentration remained several orders of magnitude greater than that of OH radicals. Figure S3 shows that the NO₂/OH molar ratio in the NCP region generally exceeded 10⁸, indicating that the NO₂ concentration was in excess under emission reduction measures. Therefore, the reaction primarily depended on the OH radical concentration. Next, we examined the impact of the OH radical concentration on the reaction rate. OH radicals were produced primarily via photochemical reactions, such as by the decomposition of HONO into OH and NO under sunlight (Song et al., 2023), which could account for 20 % to 90 % of the total primary production of OH radicals (Song et al., 2023; Xue et al., 2020; Kim et al., 2014a). Additionally, OH radicals could be produced indirectly via O₃ photolysis in the presence of water vapor (Fu et al., 2020; Kim et al., 2014a).

We compare the simulated and observed OH radical concentrations in Table 4. The simulated daily average OH radical concentrations were relatively close to the observed values. However, the peak concentrations of OH radicals were lower than those observed. Underprediction of the OH peak concentration was determined to be a common issue in current model simulations (Czader et al., 2013; Stone et al., 2012; Xue et al., 2020). Although the CMAQ model

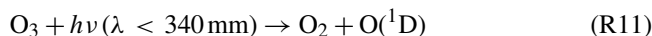
often underestimated high daytime concentrations (Czader et al., 2013), it satisfactorily captured the diurnal variation in OH radicals, likely because it underestimated HONO concentrations (Zhang et al., 2023b). The data (Fig. 13a–c) revealed that the peak value of OH radicals in Beijing significantly increased by 0.98×10^6 molec. cm⁻³ in 2018 compared with that in 2013. A similar trend was observed in Qingdao, with an increase of 0.58×10^6 molec. cm⁻³. The GEOS–Chem simulations (Zhang et al., 2023a) indicated a stable upward trend in OH radical concentrations across the NCP from 2014 to 2017, ranging from 0.05 to 0.17×10^6 molec. cm⁻³ yr⁻¹. Our study data suggested that despite China's emission reduction measures, these efforts did not effectively control OH radical concentrations. The continuous increase in OH radicals accelerated the nitrate formation rate via the OH + NO₂ pathway.

One of the primary pathways for the production of OH radicals was the photolysis of nitrous acid (HONO) in the tropospheric atmosphere. This photolytic reaction can be expressed as follows.



However, the data indicated that the HONO concentration decreased between 2013 and 2018 (Fig. 11), suggesting that this production pathway did not significantly contribute to the increase in the OH radical concentration.

In the atmosphere, another source of OH was the reaction between water vapor (H₂O) and excited oxygen atoms (O(¹D)). Notably, excited oxygen atoms would typically be produced via O₃ photolysis (Kim et al., 2014a; Tan et al., 2019).



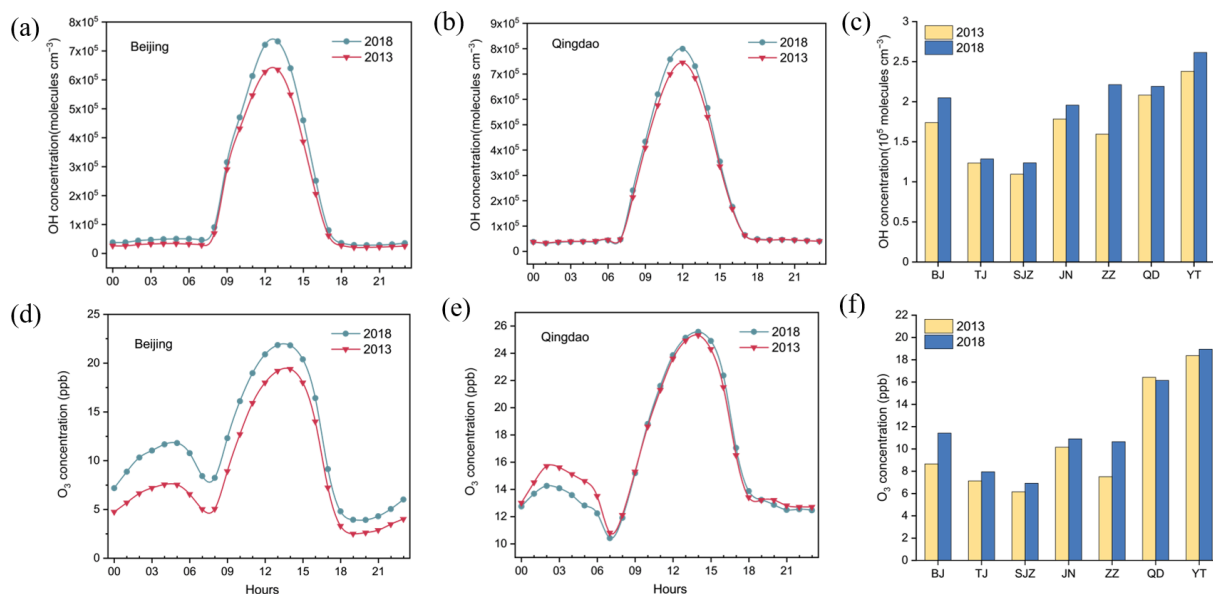
The excited oxygen atoms (O(¹D))) subsequently reacted with water vapor to produce OH radicals.



Therefore, O₃ photolysis played an indirect but crucial role in the formation process. O₃ could oxidize NO to NO₂, which then could react with OH radicals to form HNO₃, significantly influencing the OH + NO₂ pathway. As shown in Fig. 13d–f, the increase in the O₃ concentration indirectly increased the OH + NO₂ reaction rate for nitrate formation. This effect was greater in inland cities, where the increase in the O₃ concentration was greater than that in coastal cities, leading to a greater contribution of the OH + NO₂ pathway. Thus, the OH and O₃ concentrations affected the formation rate and contribution of the OH + NO₂ pathway to nitrate formation due to the excess NO₂ concentration, although the NO₂ emissions decreased. Therefore, inland cities should prioritize controlling the emissions of VOCs and NO_x. These pollutants could generate OH radicals through photochemical reactions, which would subsequently dominate nitrate

Table 4. Observations and CMAQ model simulation results for the winter OH radical concentration.

| Species | Location | Obs. period | Obs. average | Sim. average | Reference |
|---------------------------------------|--|--------------------|-----------------------------------|-------------------------------|-------------------|
| OH (10^6 molec. cm^{-3}) | Huairou_Beijing (40.41° N, 116.68° E) | January–March 2016 | Peak 2.4 | Peak 0.8 | Tan et al. (2018) |
| | Beijing PKU (40° N, 116.3° E) | Winter 2017 | Peak 1.5–2.0 Daily average 0.3 | Peak 0.8 Daily average 0.2 | Ma et al. (2019) |

**Figure 13.** Diurnal variations in OH radicals and O₃ in Beijing and Qingdao and average concentrations across seven major cities in the NCP (2013, 2018): (a) diurnal variation in OH radicals in Beijing, (b) diurnal variation in OH radicals in Qingdao, (c) average OH radical concentration in the seven cities, (d) diurnal variation in O₃ in Beijing, (e) diurnal variation in O₃ in Qingdao, and (f) average O₃ concentration in the seven cities.

formation via the OH + NO₂ pathway. By reducing precursor emissions (VOCs and NO_x), the production of OH radicals could be suppressed, thereby slowing the reaction kinetics and mitigating atmospheric nitrate accumulation.

3.5.3 Factors influencing hetN₂O₅ reaction rates

The heterogeneous N₂O₅ reaction was considered a major pathway for nitrate formation influenced by many factors, including the N₂O₅ concentration, aerosol surface area, and uptake coefficient of N₂O₅ ($\gamma_{\text{N}_2\text{O}_5}$). The value of $\gamma_{\text{N}_2\text{O}_5}$ varied widely, ranging from an extremely low value of 10^{-4} to a relatively high value of 0.1 (Wang et al., 2023). Experimental results indicated that environmental factors such as temperature and humidity; the composition of aerosols such as NO₃[−], Cl[−], and SO₄^{2−}; the liquid water content in aerosols; the presence of organic compounds; and the mixing state of aerosols were critical factors influencing $\gamma_{\text{N}_2\text{O}_5}$ (Folkers et al., 2003; Mitroo et al., 2019; Thornton and Abbatt, 2005; Wahner et al., 1998; Wang et al., 2020). These factors collec-

tively influenced the reaction rate and yield of heterogeneous N₂O₅ reactions.

With increasing concentrations of atmospheric O₃ (Fig. 13f), the generation rate of N₂O₅ significantly increased. Consequently, the average N₂O₅ concentration in the NCP was 35.7 ppt in 2013 and increased to 53.2 ppt in 2018 (Fig. 12b). However, the N₂O₅ concentrations were generally higher in ocean areas and western mountainous regions than in central inland areas. We found that this spatial distribution of N₂O₅ was caused by NO. In urban areas, especially during severe haze nighttime periods, the rapid titration of NO with O₃ led to near-zero concentrations of surface O₃ (Zang et al., 2022), thereby inhibiting the in situ generation of NO₃ radicals and N₂O₅ (Zhao et al., 2023). As shown in Fig. 11, the NO concentration exhibited a spatial distribution inversely correlated with that of N₂O₅, where higher NO concentrations corresponded to lower N₂O₅ levels and vice versa. This inverse relationship explains the low N₂O₅ concentrations in inland cities. Additionally, observations have showed that the lifetime of NO₃ radicals in marine atmospheres could reach 30 min because of the low NO_x mixing

ratio in marine air masses, which was longer than 1 min in inland atmospheres (Crowley et al., 2011) and contributed to the high N_2O_5 concentrations in coastal cities. Observations in Hebei Province revealed low N_2O_5 concentrations in summer, with lifetimes ranging from 0.1 to 10 min (Tham et al., 2018). Therefore, the spatial differences in hetN_2O_5 reaction rates depended on NO and the lifetime of NO_3 radicals.

Moreover, we speculated that the decreased rate of heterogeneous N_2O_5 was influenced not only by N_2O_5 but also by other factors, such as RH and the surface area of aerosol particles, due to the interannual variations in RH and PM. Compared with 2013, the RH in the NCP decreased by 3.1 % in 2018 (Fig. S4). The sensitivity experiments revealed that both the N_2O_5 reaction rate and its contribution to total nitrates remained essentially unchanged when the humidity decreased by 10 %. This finding indicated that humidity changes were not the main factor influencing the decrease in the reaction rate of hetN_2O_5 in the NCP. The total dry surface area of the particles in the CMAQ model was calculated as the sum of the modal dry surface areas, corresponding to the particles in the Aitken, the accumulation levels, and coarse modes (Bergin et al., 2022). During the winter of 2013 in the NCP, the average surface reaction factor (SRF) value was $689 \mu\text{m}^2 \text{cm}^{-3}$, which decreased to $425 \mu\text{m}^2 \text{cm}^{-3}$ during the winter of 2018, indicating a reduction of 38.3 %. This significant reduction in the aerosol surface area likely contributed to the observed decrease in the reaction rate of heterogeneous N_2O_5 . Air quality monitoring data revealed that PM_{10} decreased by 20 % and $\text{PM}_{2.5}$ decreased by 28 % in China from 2014 to 2018 (Fan et al., 2020a). Therefore, we considered that the reduction in the $\text{PM}_{2.5}$ concentration was a major factor contributing to the decrease in the hetN_2O_5 generation rate.

In brief, hetN_2O_5 reaction rates were affected by three key factors: N_2O_5 concentrations, aerosol surface area, and the spatial distribution of NO. The decline in hetN_2O_5 reaction rates across the NCP between 2013 and 2018 was driven primarily by a substantial reduction in aerosol surface area, particularly due to decreased $\text{PM}_{2.5}$ concentrations. Furthermore, regional variations in NO concentrations and NO_3 radical lifetimes significantly influenced N_2O_5 concentrations, resulting in distinct spatial patterns. To further mitigate nitrate pollution, coastal cities should prioritize reducing particulate matter concentrations, which would reduce hetN_2O_5 reaction rates and enhance air quality.

3.6 Impact of reducing NO_x and VOC emissions on the nitrate concentration

To reduce the nitrate content in $\text{PM}_{2.5}$, effective control strategies must target key factors, such as OH radicals, O_3 , and the photochemical reactions of NO_x and VOCs. However, a previous study revealed that even with a 30 % reduction in VOC and NO_x emissions, the winter nitrate concentration in $\text{PM}_{2.5}$ in the NCP decreased by only 8.6 % (Fu et

al., 2020). NH_3 played a crucial role in the gas-to-particle conversion process of HNO_3 in the NCP region. Therefore, we designed single and combined pollutant reduction strategies for NH_3 , NO_x , and VOCs to examine the effects of the emission of these species on nitrate in $\text{PM}_{2.5}$.

The effectiveness of emission reduction scenarios notably differed between Beijing and Qingdao (Fig. 14). In Beijing, single-pollutant reduction strategies yielded outcomes dependent on target factors. The city showed limited sensitivity to NO_x reduction, with nitrate concentrations initially increasing before showing a 4.5 % decrease in nitrate concentration until reaching a 60 % NO_x reduction threshold. A 60 % reduction in VOC emissions led to a 10.5 % decrease in nitrate concentration, whereas NH_3 reduction led to a 38.4 % decrease at the same level. In contrast, Qingdao demonstrated relatively strong responsiveness to NO_x controls, with a 60 % reduction resulting in a 32.4 % decrease in nitrate concentration. However, Qingdao showed relatively weak sensitivity to VOC controls, with a 60 % reduction in VOC emissions resulting in an 11.1 % decrease in nitrate concentration. Qingdao exhibited the strongest response to NH_3 reduction measures, where a 60 % decrease in NH_3 emissions led to a substantial 51.7 % reduction in nitrate concentration.

The comprehensive reduction scenarios revealed more pronounced intercity differences. Beijing showed a relatively weak response to combined NO_x and VOC reductions, with only a 13.0 % decrease in nitrate concentration at a 60 % reduction in both pollutants. Qingdao, however, achieved a significantly greater reduction of 34.7 % under the same conditions. The most effective strategy involved simultaneous reductions in NH_3 , VOCs, and NO_x , with Qingdao achieving a 60.0 % reduction and Beijing achieving a 44.2 % reduction at 60 % emission cuts. These findings demonstrated that effective nitrate control measures would require the coordinated reduction of multiple precursors. While multipollutant strategies yielded optimal results, NH_3 reductions were determined to be the most effective single-pollutant control measure for both cities.

4 Conclusion

In this study, nitrate formation mechanisms and influencing factors in seven cities across the NCP during the winters of 2013 and 2018 were investigated. Despite inherent uncertainties, the model simulations and isotopic measurements showed remarkable agreement, demonstrating the reliability of the modeling approach in elucidating nitrate formation mechanisms. Our findings revealed that nitrate formation was significantly influenced by NH_3 , NO_2 , OH radicals, O_3 , NO, and N_2O_5 , with distinct differences between regions. In inland cities, the $\text{OH} + \text{NO}_2$ reaction pathway dominated, contributing 63.7 %–85.6 % of nitrate formation. This phenomenon was largely driven by the increased concentra-

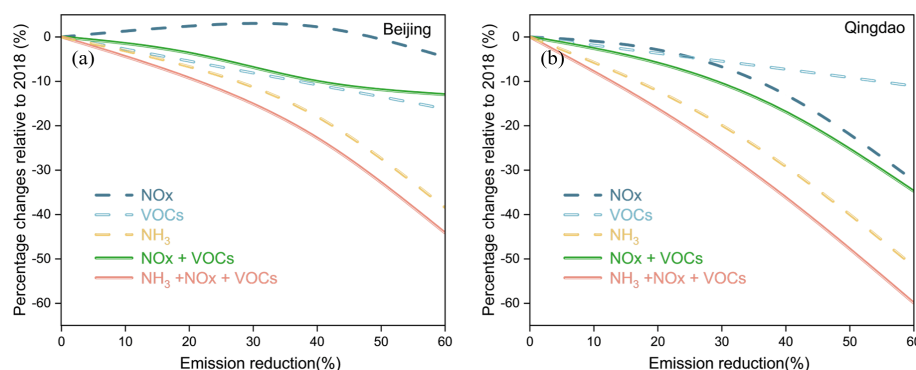


Figure 14. Relative changes in the PM_{2.5} nitrate contribution in response to the emission reduction scenarios in (a) Beijing and (b) Qingdao, compared with the winter 2018 baseline conditions

tions of OH radicals and O₃, resulting in a 7.6 % greater contribution of this pathway in inland cities in 2018 than in 2013. In contrast, coastal cities presented a greater contribution of the hetN_2O_5 pathway (37.0 % to 45.7 %) because of the higher N₂O₅ concentrations and longer NO₃ radical lifetimes. High NO concentrations in inland areas facilitated O₃ titration, inhibiting N₂O₅ formation and further differentiating nitrate formation processes between the two regions.

Our emission reduction experiments revealed distinct regional responses and highlighted the need for tailored, multipollutant strategies to effectively control nitrate pollution. Among the single-pollutant control strategies, NH₃ reduction demonstrated the highest effectiveness, with a 60 % emission reduction leading to a 38.4 % decrease in nitrate concentrations in Beijing and a substantial 51.7 % reduction in Qingdao. The combined emission control strategy, which involved simultaneous reductions in multiple pollutants, was more effective than NH₃ reduction alone, achieving a 44.2 % nitrate reduction in Beijing and a 60.0 % decrease in Qingdao. This study advanced our understanding of regional variations in atmospheric nitrogen chemistry and provided essential insights for developing targeted measures to mitigate nitrate pollution and improve air quality in northern China. To further advance research in this field, it is recommended that researchers in the future should focus on developing robust methodologies to comprehensively account for the influence of atmospheric transport on nitrate formation.

Data availability. All datasets supporting the findings of this study are available through the following channels. Primary datasets and analysis results are available from the corresponding authors upon reasonable request. All plotting data and essential research data have been deposited in a publicly accessible repository on Zenodo at <https://doi.org/10.5281/zenodo.16717577> (Liu and Qi, 2025).

Supplement. The supplement related to this article is available online at <https://doi.org/10.5194/acp-25-8719-2025-supplement>.

Author contributions. ZL: visualization, formal analysis, writing (original draft). XL: project administration, writing (review and editing). YN: validation. LX: resources. JQ: conceptualization, methodology, writing (review and editing).

Competing interests. The contact author has declared that none of the authors has any competing interests.

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