

**Fertilization-driven Pulses of Atmospheric Nitrogen Dioxide Complicate Air Pollution
in Early Spring over the North China Plain**

Tian Feng,¹ Guohui Li,^{2,*} Shuyu Zhao,³ Naifang Bei,⁴ Xin Long,⁵ Yuepeng Pan,⁶ Yu Song,⁷ Ruonan
Wang,² Xuexi Tie,² Luisa T. Molina⁸

¹ Department of Geography & Spatial Information Techniques, Ningbo University, Ningbo, China

² KLACP, State Key Laboratory of Loess and Quaternary Geology, Institute of Earth Environment,
Chinese Academy of Sciences, Xi'an, China

³ Ningbo Meteorological Bureau, Ningbo, China

⁴ School of Human Settlements and Civil Engineering, Xi'an Jiaotong University, Xi'an, China

⁵ Research Center for Atmospheric Environment, Chongqing Institute of Green and Intelligent Technology,
Chinese Academy of Sciences, Chongqing, China

⁶ State Key Laboratory of Atmospheric Boundary Layer Physics and Atmospheric Chemistry (LAPC),
Institute of Atmospheric Physics, Chinese Academy of Sciences, Beijing, China

⁷ State Key Joint Laboratory of Environmental Simulation and Pollution Control, Department of
Environmental Science, Peking University, Beijing, China

⁸ Molina Center for Energy and the Environment, Boston, MA, USA

*Corresponding author. Email: liqh@ieecas.cn

Abstract

Atmospheric nitrogen dioxide (NO₂) has shown periodic conspicuous pulses in tropospheric column in March over the North China Plain during the past two decades. However, these repetitive pulses have never been reported and its underlying causes remain unclear. Here, we present robust evidence to demonstrate that agricultural fertilization drives the early-spring NO₂ column increases. The fertilization-driven soil NO_x (=NO+NO₂) emissions, comparable to anthropogenic sources, exert complicated influences on regional air quality. They significantly reduce nocturnal and diurnal O₃ concentrations in agricultural areas in early spring, distinct from the scenarios in summer, but increase fine particulate matter (PM_{2.5}) concentrations via strongly enhancing nitrate aerosol formation. The impact also extends to urban areas, approximately half that of agricultural areas. These findings are with increasing implications for coordinated control of PM_{2.5} and O₃ under global warming. We thus suggest that reducing NO_x emissions in croplands is essential to achieve better air quality in agricultural countries and regions.

Short Summary

Impacts of agricultural fertilization on nitrogen oxide and air quality are becoming more pronounced with continuous reductions in fossil fuel sources in China. We report that atmospheric nitrogen dioxide pulses driven by agricultural fertilizations largely complicate air pollution in the North China Plain, highlighting the necessity of agricultural emission control.

Keywords: Nitrogen Dioxide, Fertilization, Air Pollution, Ozone, PM_{2.5}

1 Introduction

Nitrogen oxide (NO_x = nitric oxide (NO) + nitrogen dioxide (NO_2)) is a major air pollutant in the troposphere and a key precursor to ozone (O_3) and fine particulate matter ($\text{PM}_{2.5}$) due to its photochemical properties (Seinfeld and Pandis, 2006; Zhang et al., 2015). It is also a short-lived climate forcer regulated by both China and the United States (IPCC, 2023). Understanding the NO_x budget is crucial for addressing these issues. Globally, atmospheric NO_x is mainly produced by fossil fuel combustion (Crippa et al., 2023; Janssens-Maenhout et al., 2015; Yan et al., 2005), with smaller contributions from wildfires and lightning (Bauwens et al., 2020; Murray et al., 2012). Additionally, soil generates substantial NO_x through nitrification and denitrification processes (Bouwman et al., 2002; Cárdenas et al., 1993; Davidson, 1992; Yan et al., 2005), particularly after the rewetting of dry soils (Galbally and Roy, 1978; Huber et al., 2020; Yienger and Levy, 1995). On a regional scale, soil NO_x emissions may even exceed those from fossil fuel sources in summer (Almaraz et al., 2018; Sha et al., 2021). Model- and satellite-based studies estimate that global annual soil NO_x emissions, with the largest contributor of cultivated croplands, range from 9 to 27 Tg N (Hudman et al., 2012; Steinkamp and Lawrence, 2011; Vinken et al., 2014; Yan et al., 2005), accounting for about 15% of total NO_x emissions (Hudman et al., 2012). This wide range is due to the complex response of soil NO_x emissions to driving factors like fertilization, temperature, and soil moisture (Huber et al., 2020; Oikawa et al., 2015), making accurate estimation challenging.

The emission rates from fertilized croplands are 1 to 2 orders of magnitude higher than nearby grasslands and forest soils (Almaraz et al., 2018; Anderson and Levine, 1987; Guo et al., 2020; Yienger and Levy, 1995). Recent studies show significant NO_x emissions from croplands post-fertilization, exceeding pre-fertilization rates by an order of magnitude (Almaraz et al., 2018; Hickman et al., 2017; Laville et al., 2011; Liu et al., 2005; Oikawa et al.,

2015; Zhao et al., 2015). Despite these robust evidences of strong NO_x emissions from agricultural fertilization, the lack of extensive *in-situ* measurements hinders accurate estimation of these emissions and their environmental impacts. Additionally, the effect of agricultural fertilization on air quality has not received sufficient global attention, although some pioneering studies have pointed the implications for air quality since the 1990s (Davidson et al., 1998; Hall et al., 1996). In recent years, studies have reported that agricultural soil emissions significantly increase atmospheric NO_x levels (Almaraz et al., 2018; Hickman et al., 2017; Huang et al., 2018; Oikawa et al., 2015) and enhance O₃ formation in summer in California (Oikawa et al., 2015) or during the growing season of crops in sub-Saharan Africa (Hickman et al., 2017; Huang et al., 2018).

The North China Plain (NCP) is one of the major grain-producing regions in China. Winter wheat-maize double cropping is a typical rotation system mainly practiced in this region (Liu et al., 2003; Zhu et al., 1994). China has been the world's largest consumer of N-fertilizer since 2000 (Liu et al., 2013), with annual usage peaking at approximately 31.2 Tg N in 2014 (Yu et al., 2022). About half of this fertilizer is lost to the environment (Liu et al., 2013), indicating a significant potential source for NO_x emissions from China's croplands. The agricultural management in the NCP has been known for incorporating high fertilization rates with excessive N fertilization (Sun et al., 2022; Vitousek et al., 2009; Zhao et al., 2006). Thus, this region is the largest consumer of agricultural fertilizer N in China (Yu et al., 2022) and has shown substantial soil NO_x emissions (Liu et al., 2010; Tang et al., 2020; Zhang et al., 2011). The emissions significantly increase ambient NO_x levels and enhance O₃ formation in summer (Huang et al., 2023; Lu et al., 2021; Wang et al., 2022a). These concerns typically focus on the warm season when higher temperatures favor NO_x emissions from soils. However, frequent agricultural activities and N-fertilizer use also occur during transitional seasons, and how

periodic agricultural fertilization affects soil NO_x emission and regional air quality remains unclear.

In this study, we present a pulse of atmospheric NO₂ column in early spring during the past two decades over the NCP. However, this phenomenon has not been previously reported in this region. Combining agricultural fertilization records, surface NO₂ and NH₃ observations, long-term satellite observations of NO₂ and NH₃, and a flexible scheme of soil NO_x emission, we explain successfully the underlying cause for the NO₂ column peaks using a regional atmospheric transport model online coupled with chemistry, and further assess the impacts of the pulsing NO_x emission on regional air quality.

2 Materials and Methods

2.1 Model and configurations

The Weather Research and Forecasting model fully coupled with atmospheric chemistry (WRF-Chem, version 3.6.1) we used is a modified model by Li et al. (2011a; 2012; 2010; 2011b) and Feng et al (2021), in which we implement the BDSNP mechanism by Hudman et al. (2012) to calculate soil NO_x emission related to agricultural fertilization and the influences on regional air quality in the NCP. The model is configured with grid spacing of 6 km × 6 km (240 × 280 grid cells) with the center at 38°N and 116°E (Figure 1). Thirty-five vertical levels are employed in the stretched vertical grid with spacing ranging from 50 m near surface, to 500 m at 2.5 km and 1 km above 14 km. Meteorological initial and boundary conditions use the National Centers for Environmental Prediction (NCEP) FNL 1°×1° analysis data, and the chemical initial and boundary conditions are interpolated from the CAM-Chem 6-h output (Buchholz et al., 2019; Emmons et al., 2020). The non-soil emission inventory is developed by Zhang et al. (2009) and the biogenic emissions are calculated online using the Model of Emissions of Gases and Aerosol from Nature (MEGAN) model (Guenther et al., 2006).

Specifically, monthly ammonia (NH_3) emissions are incorporated from a high-resolution NH_3 emission inventory developed by Huang et al., (2012), which includes emissions from fertilizer application, livestock, and other sources. The model spin-up time is 2 days (Table 1).

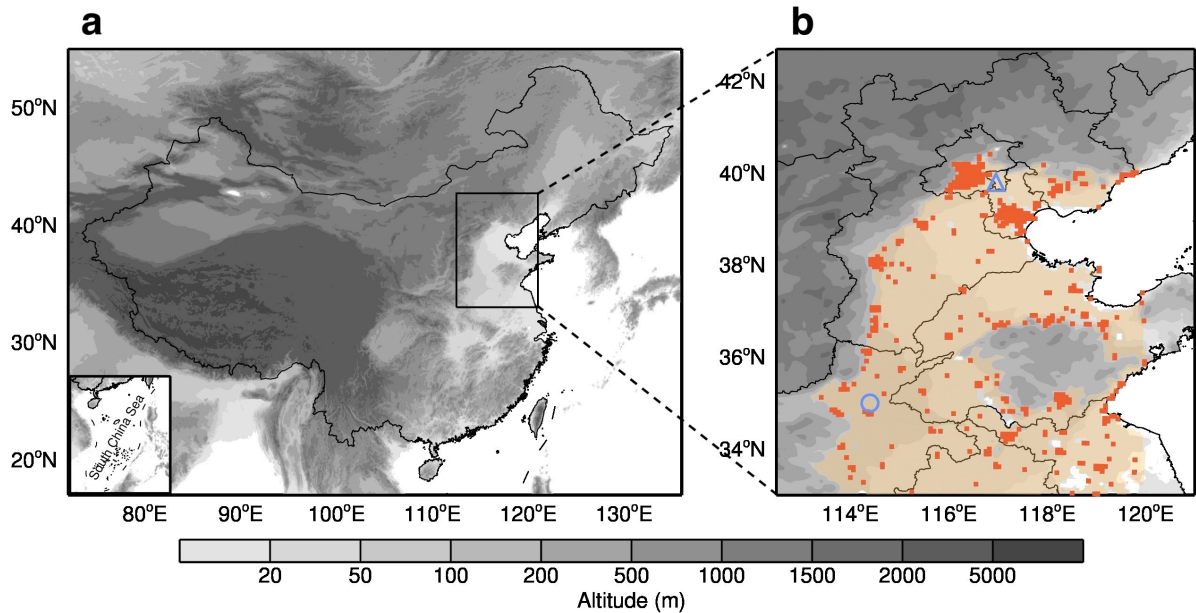


Figure 1. Domain overview. (a) Geographic location of the NCP, which is predominantly characterized by plains at an elevation less than 100 m, and known for a major agricultural zone. (b) Extensive cultivated croplands distribute in the NCP, marked by the orange shadings, while urban areas are marked by red shadings. The graphic markers denote locations of field observation sites, of which the blue circle represents the Fengqiu cropland ecological station, Chinese Academy of Sciences, with a long-term record on agricultural fertilization, and the triangle represents the rural Xianghe station with ambient NH_3 measurements. The agricultural areas in orange within the NCP are defined as croplands with altitude less than 100 m, and the urban areas in red are defined as built-up areas within the NCP.

Table 1. Model configuration for the simulation domain, meteorological schemes, initial and boundary conditions, and emission inventories.

Item	Configuration
Period	February through April, 2020
Region	The NCP and surrounding areas
Domain center	116°E, 38°N
Domain size	1440 km × 1680 km
Horizontal resolution	6 km × 6 km
Vertical resolution	35 vertical levels with a stretched vertical grid with spacing ranging from 50 m near surface, to 500 m at 2.5 km and 1 km above 14 km
Microphysics scheme	WRF Single-Moment 6-class scheme (Hong and Lim, 2006)
Boundary layer scheme	MYJ TKE scheme (Janjić, 2002)
Surface layer scheme	MYJ surface scheme (Janjić, 2002)
Land-surface scheme	Noah land surface model (Chen and Dudhia, 2001)
Longwave radiation scheme	New Goddard scheme (Chou et al., 2001)
Shortwave radiation scheme	New Goddard scheme (Chou and Suarez, 1999)
Meteorological boundary and initial condition	NCEP FNL 1° × 1° analysis data
Chemical boundary and initial condition	CAM-Chem 6-h output (Buchholz et al., 2019; Emmons et al., 2020)
Anthropogenic emission inventory	MEIC emission inventory (Li et al., 2017b; Zhang et al., 2009), except for NH ₃
NH ₃ emission inventory	NH ₃ emission inventory in China (Huang et al., 2012)
Biogenic emission inventory	MEGAN model (2006)
NO _x emission from various types of soils	Soil NO _x emission mechanism (2012)
Spin-up time	2 days

2.2 Soil NO_x emission scheme

We implement a soil NO_x emission scheme, the Berkeley-Dalhousie Soil NO Parameterization (BDSNP) by Hudman et al. (2012), into the WRF-Chem model. The scheme comprehensively considers various factors, including available soil nitrogen content (N_{avail} , ng N m⁻²) from the fertilizer application and nitrogen deposition, in which the soil NO_x emission (E_{soil} , ng N m⁻² s⁻¹) is a function of N_{avail} , climate, and edaphic conditions:

$$E_{soil} = A'_{biome}(N_{avail}) \times f(T) \times g(\theta) \times P(l_{dry}) \quad (1)$$

where N_{avail} is available soil nitrogen mass, and A'_{biome} (ng N m⁻² s⁻¹) represents the biome-dependent emission factor. $f(T)$ (dimensionless) and $g(\theta)$ (dimensionless) are parameters

regulated by soil temperature and moisture, respectively. $P(l_{dry})$ (dimensionless) denotes the pulsed soil emission from wetting of dry soils. The product by $f(T)$ and $g(\theta)$ is calculated following:

$$f(T) \times g(\theta) = e^{0.103T} \times a\theta e^{-b\theta^2} \quad (2)$$

where T ($0 \leq T \leq 30^\circ\text{C}$) is soil temperature and θ ($0 \leq \theta \leq 1$, dimensionless) is water-filled pore space, defined as the ratio of the volumetric soil moisture content to the porosity. According to the laboratory and field measurements (Hudman et al., 2012), the constants a and b are determined so that $g(\theta)$ maximizes when $\theta = 0.2$ for arid soils and $\theta = 0.3$ elsewhere.

The pulsing term $P(l_{dry})$, following Yan et al. (2005), describes the magnitude of the peak flux relative to the pre-wetting flux, which is parameterized as:

$$P(l_{dry}) = [13.01 \ln(l_{dry}) - 53.6] \times e^{-ct} \quad (3)$$

where l_{dry} (hours) represents the length of the antecedent dry period, and c ($c = 0.068 \text{ h}^{-1}$) is a constant rate denoting the rise/fall time of the pulse. Fertilizer applications data are interpolated from the global gridded chemical fertilizer and manure application inventory at $0.5^\circ \times 0.5^\circ$ (Potter et al., 2010; Yan et al., 2005). The chemical and manure fertilizers are obtained from the International Fertilizer Association (IFA) and the Food and Agriculture Organization of the United Nations (FAO). The Chinese chemical fertilizer application (straight N application) from IFA is about 19.6 Tg N a^{-1} for 2000, quite close to the amount of 19.9 Tg N a^{-1} for 2020 from the China Statistical Yearbook (<https://www.stats.gov.cn/sj/ndsj/2021/indexch.htm>). More details of the scheme are found in related studies elsewhere (Hudman et al., 2012; Lu et al., 2021). It should be noted that we use the default BDSNP fertilizer and global emission

assumptions, and these assumptions (primarily the 75/25 split and not applying the topdressing in a single application) may not accurately reflect the fertilizer applications in China. The BDSNP tuning, which was done for year 2000 agricultural emissions estimated by Stehfest and

Bouwman (2006), is close to recent estimates of global emissions, possibly on the high side (Gong et al., 2025; Wang et al., 2022b).

2.3 Emission inventories

We employ two emission inventories in this study, of which the Hemispheric Transport of Air Pollution Version 3 (HTAP v3, 2005-2018) emission inventory includes soil and non-soil emissions (Li et al., 2017a), and the Multi-resolution Emission Inventory for China (MEIC v1.3, 2007-2018) has no soil emission (Li et al., 2017a). In the HTAP inventory, the non-soil emission inventory includes energy, industry, ground transport, residential, waste, shipping and aviation sources in the HTAP inventory, with a spatial resolution of $0.1^{\circ} \times 0.1^{\circ}$ and a temporal resolution of one month. Agricultural emissions are involved in the latest HTAP v3 inventory, which includes soil NO_x emissions (Crippa et al., 2023). Nevertheless, the soil emissions in this inventory are calculated using traditional “bottom-up” method (Kurokawa and Ohara, 2020), rather than estimated by a process-based emission module. The monthly MEIC emission inventory with a spatial resolution of $0.25^{\circ} \times 0.25^{\circ}$ is incorporated in parallel with the HTAP emission inventory. Here, we focus on NO_x and NH_3 emissions from croplands with fertilization, and adopt NH_3 emission inventory by Huang et al. (2012) because they explicitly distinguish NH_3 produced by agricultural fertilization from other NH_3 sources.

2.4 Air pollutant measurements

Satellite-derived tropospheric NO_2 columns are from OMI hosted by the Aura satellite that is launched by the National Aeronautics and Space Administration (NASA). The Level-3 product, where pixel level data of good quality are binned and “averaged” into $0.25^{\circ} \times 0.25^{\circ}$ grids, was retrieved and analyzed in the present study. The satellite operates in a sun-synchronous polar orbit and has a local overpass time of around 13:45 (local time, LT) in North China. The dataset

is for all atmospheric conditions, and for sky conditions with cloud fraction less than 30% (https://cmr.earthdata.nasa.gov/search/concepts/C1266136111-GES_DISC.html). The dataset has a spatial resolution of $13\text{ km} \times 24\text{ km}$, with a temporal coverage of 2005-2022 (Lamsal et al., 2021). Note that the number of pixels included in NO_2 retrievals changes over time because of the increase in the number of pixels affected by the row anomaly issue, making the data unsuitable for trend analysis and possibly introducing uncertainty in seasonal averages. The Level-2 product of NH_3 columns is employed, which is from the Space Administration and the Infrared Atmospheric Sounding Interferometer (IASI) hosted on the MetOp series of satellites. The satellite also operates in a sun-synchronous polar orbit and has a local overpass time of around 9:30 am and 9:30 pm in North China (twice a day). We construct a $0.25^\circ \times 0.25^\circ$ mesh grid and calculate the average of the NH_3 columns from IASI within each grid cell (Clarisse et al., 2023). Low-quality satellite data are filtered out due to the interference of clouds. To cover all the domain (Figure 1), the data used in this study are averaged into seven-day mean datasets of NO_2 and NH_3 columns with a non-overlapping 7-day window during 2007-2021. The data are interpolated into the model grids using bilinear interpolation.

Ambient surface NO_2 , O_3 , and $\text{PM}_{2.5}$ mass concentrations at 141 sites in the NCP are from the China National Environmental Monitoring Centre (CNEMC, Figure S1). These *in-situ* measurements are performed by the Thermo ScientificTM ambient particulate monitor and gas analyzers, of which NO_2 and O_3 are measured by Model 42i Chemiluminescence $\text{NO-NO}_2\text{-NO}_x$ Analyzer, Model 49i UV Photometric Ozone Analyzer, respectively. $\text{PM}_{2.5}$ is measured by Model 5030 Synchronized Hybrid Ambient, Real-time Particulate (SHARP) Monitor, which uses proprietary digital filtering to continuously calibrate mass to obtain an accurate, precise and real-time mass concentration. The sampling time is 1 min for these monitoring devices. Agricultural NH_3 concentration is monitored by a Picarro analyzer based on the principle of

cavity ring-down spectroscopy (CRDS) at the rural Xianghe station (Figure 1), with a sampling frequency of 1 Hz. Hourly data are derived by averaging the high-frequency measurements.

3 Results and Discussion

3.1 Satellite-retrieved NO₂ column pulses

During the past two decades, seven-day mean tropospheric column of NO₂ measured by the Ozone Monitoring Instrument (OMI) in the NCP exhibits a significant temporal variation, with the magnitude varying from less than 100 $\mu\text{mol m}^{-2}$ to more than 680 $\mu\text{mol m}^{-2}$ (Figure 2a). The annual cycle is highly prominent and its seasonal variation is remarkable, with significantly higher levels in cold seasons than those in warm seasons. Throughout the year, the pattern of NO₂ column looks like a rhinoceros horn, which is characterized by a major peak in winter and multiple noticeable sub-peaks in other seasons. These sub-peaks often occur at fixed times, such as in March, June and October, of which the sub-peak is the most noticeable in March, with the highest magnitude (Figures 2a and S1). We examine the monthly variation in anthropogenic NO_x emission rates over the NCP in global and regional emission inventories, and find that the monthly variation is more evident in the regional emission inventory, with a significantly higher emission than that in the global emission inventory. Nevertheless, neither of them reveals any discernible sub-peaks of NO_x emission rates from fossil fuel combustion to coincide with the sub-peaks of the NO₂ column (Figure 2b).

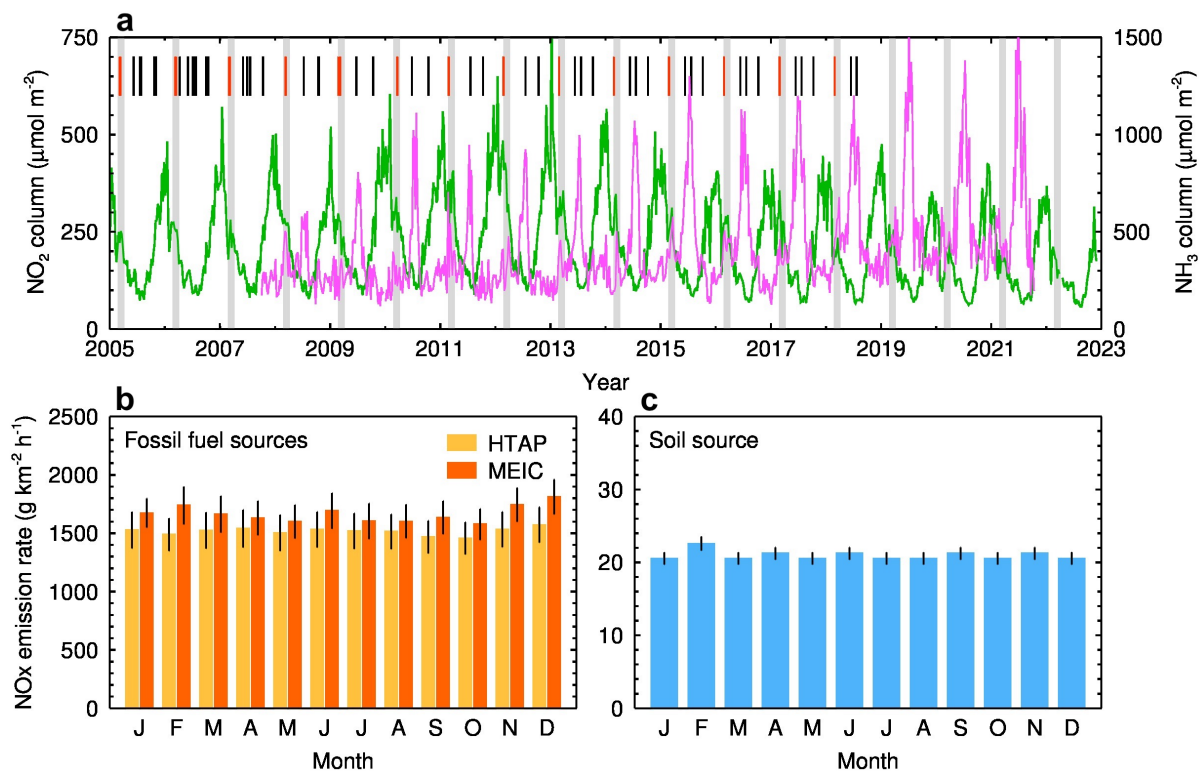


Figure 2. NO₂ column pulses in March, NH₃ column variation, and NO_x emissions from fossil fuel and soil sources over the NCP. (a) Long-term variation of seven-day mean tropospheric NO₂ column observed by OMI during 2005-2022 (green) and NH₃ column retrieved from IASI during 2007-2022 (pink). Intersections of the gray bars and the green lines denote a sub-peak of NO₂ column occurred in each March, and the short bars represent the timing record for agricultural fertilization at Fengqiu station in the NCP, of which the red ones indicate the fertilization period in early spring. (b) Monthly mean NO_x emission rates with $\pm 1\sigma$ standard deviation (SD) in two sets of anthropogenic emission inventories, the HTAP v3 (2005-2018, orange) and MEIC v1.3 (2008-2017, red). (c) Same as (b), but for NO_x emission rates from soils in the HTAP v3 inventory (2005-2018).

As for soil NO_x emissions, they are absent in the regional emission inventory, while in the global emission inventory, soil NO_x emissions fluctuate slightly on a monthly scale, far less than those from fossil fuel combustion, constituting less than 2% of the total (Figure 2c).

Similar to NO_x emissions from fossil fuel combustion, there are no evident sub-peaks of soil NO_x emission to keep pace with the atmospheric NO_2 column. Soil NO_x emissions are even at a lower level in March, significantly less than emissions in adjacent months. Therefore, the known emission inventory fails to explain the occurrence of these sub-peaks. On the other hand, we compute a pollution accumulation index (PAI, Text S1), the product of boundary layer height and wind speed, to semi-quantitatively assess the influence of atmospheric dispersion conditions on NO_2 column. NO_2 column seems to be somewhat dependent on the PAI, yet the noticeable discrepancies between the timing of the sub-peaks and PAI are insufficient to account for the occurrence of each sub-peak (Figure S3). Additionally, it is seen that the daily soil temperature was consistently higher than 0°C during March 2020 and the ten days before (Figure S4). Therefore, the sub-peak of NO_2 column is not expected to be originated from soil thaw either.

3.2 Linkage between NO_2 column pulses and agricultural fertilization

What causes these regular NO_2 sub-peaks occurred over the NCP during the past two decades? Measurements on ammonia (NH_3) column also present similar pulses to those of NO_2 column during the same period, in spite of some differences in the long-term trend (Figure 2a). Although the main peaks of NH_3 column occur in June, the sub-peaks of NH_3 column in March may provide favorable evidence that these NO_2 column sub-peaks are connected to agricultural activities because atmospheric NH_3 is largely originated from fertilizer application in agriculture (Crippa et al., 2023; Huang et al., 2012; Li et al., 2017a). We note that the two time series don't have similar seasonal dynamics, which may raise some questions about whether the March NO_2 peaks represent fertilizer pulses. This discrepancy could be due to several factors: (1) contributions from fossil fuel-derived NO_x , (2) elevated background soil NO_x emissions during the "spring thaw" period, (3) differences in fertilizer type, as fertilizers vary

in their potential for ammonia volatilization, and (4) variation in application methods, e.g., banding or deep soil placement vs broadcasting, especially when the latter is done without incorporation.

Another key evidence is that the occurrence of each sub-peak of the NO₂ column is highly consistent with the record of agricultural fertilization at the Fengqiu cropland ecological station in the NCP during the past decades (Figures 1b, 2a, and S1). The wheat-maize double-cropping system is predominate in the NCP, where the agricultural activities are strongly dependent on the lunar calendar. For winter wheat, the planting date ranges from early to mid-October (after maize harvest). Fertilization is generally divided into three stages: 1) Pre-planting during late September – early October; 2) Jointing stage during mid-March – early April; 3) Grain filling during late April for high-yield fields. The planting date of summer maize ranges from early to mid-June (after wheat harvest), and the stages of fertilization include: 1) At planting during early June; 2) V6-V8 stage during early July; 3) Tasseling stage during late July for high-yield fields. The agricultural fertilization is closely associated with three solar terms, i.e., Waking of Insects in March (the 3rd solar term), Grain in Beard in June (the 9th solar term) and Cold Dew in October (the 17th solar term). During these pulses of the NO₂ column, we found that the pulse in March is more pronounced than those in June and October, because March is the season for a large-scale cultivating in the NCP, accompanied by more land preparation and fertilization. Therefore, we hypothesize that the NO₂ column pulse in March is possibly caused by fertilized croplands that accelerate NO_x emissions from agricultural soils. Field campaigns have measured a high NO emission rate of 266.3 g km⁻² h⁻¹ in croplands after fertilization and irrigation in autumn in eastern China (Tang et al., 2020; Tian et al., 2020) and also other regions (Hickman et al., 2017; Huang et al., 2018; Huber et al., 2020), suggesting that agricultural fertilization is likely a significant source of atmospheric NO_x in major agricultural countries like China.

3.3 Soil NO_x emission mechanism

To examine the role of soil NO_x emissions from agricultural fertilization in the pulses of atmospheric NO₂ column, we introduce a flexible soil NO_x emission module and NH₃ emission into the WRF-Chem model, and perform two simulation experiments that include and exclude soil emissions, respectively (Table 1). Noticeably, there is an acceleration in the release of soil NO_x, and daily mean emission rate increases from 155.6 g km⁻² h⁻¹ to 438.3 g km⁻² h⁻¹ during the simulation period (Figure 3a). In particular, the NO_x emission rate during the post-fertilization phase is significantly higher than those during other phases, consistent with the accelerated soil NO_x release observed in agricultural areas in California after fertilization (Oikawa et al., 2015). On average, the simulated NO_x emission rate in March is 312.9 g km⁻² h⁻¹, between the measured 113.6 g km⁻² h⁻¹ in November in eastern China (Tang et al., 2020) and 988.2 g km⁻² h⁻¹ in September in California (Oikawa et al., 2015), suggesting the rationality of the soil NO_x emission mechanism in the model. We should also note that the BDSNP scheme would be heavily biased towards having a fertilizer-induced emission peak only at planting/green-up.

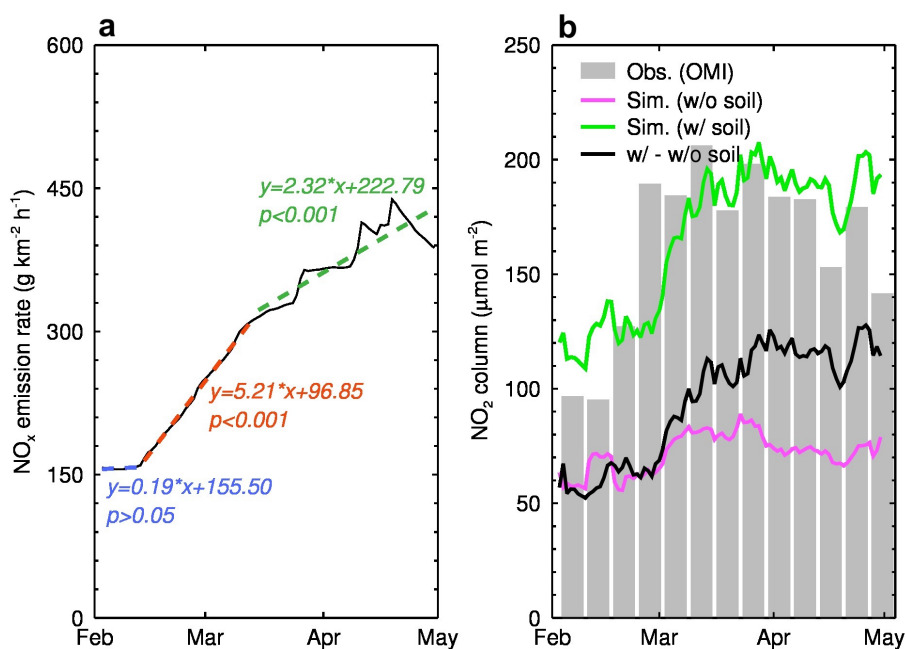


Figure 3. NO_x emissions from agricultural fertilization and resultant NO_2 column during February-April in 2020 over the NCP. (a) Calculated NO_x emission rate from croplands with N-fertilizer application in the model. The black curve represents daily variation in NO_x emission rate around the fertilization, and the blue, red, and green dash lines correspond to the trends of NO_x emission rates in croplands during the pre-fertilization, fertilization and post-fertilization periods, respectively. (b) Observed and simulated NO_2 column. The gray histogram represents NO_2 column observed by satellite (OMI). The green and pink lines represent simulated NO_2 column with and without soil NO_x emissions, and the black line shows the difference between them. The model well replicates the rapid increase in observed NO_2 column by considering soil NO_x emissions from agricultural fertilization.

We evaluate the model performance against satellite-derived NO_2 column. Consequently, the modified model perfectly replicates the sudden increase in NO_2 column linked to agricultural fertilization, while the conventional WRF-Chem model fails to capture the observed NO_2 column pulse in March due to lack of the adopted soil NO_x emission mechanism (Figure 3b). For example, when soil NO_x emission caused by agricultural fertilization is considered, the simulated NO_2 column rapidly increases to the peak in March, matching well with the satellite observation. However, without the contribution of agricultural fertilization, NO_2 column seems to exhibit a weak upward trend, but not significant. Comparing these two scenarios, a substantial NO_x emission from N-fertilizer input in croplands leads to an increase in NO_2 column by 1 to 1.5 times.

We also validate the modified model performance on temporal variations of routine surface pollutant measurements (NO_2 , O_3 and $\text{PM}_{2.5}$) associated with NO_x emissions at the CNEMC sites throughout the simulation period (Figure 4). Although there are some discrepancies between the simulations and observations, e.g., overestimates occur in mid-

February for NO₂ and PM_{2.5} levels, the model generally reproduces hourly variations in each pollutant reasonably. The indices of agreement (IOAs, Text S2) between the simulated and observed near-surface concentrations of NO₂, O₃, and PM_{2.5} are 0.90, 0.91, and 0.88, respectively, and the normalized mean biases (NMBs) for these pollutants are within 10%.

We still cannot ignore the discrepancies between the model results and observations. These biases may largely originate from the soil NO_x emission mechanism. The fertilization dates in the BDSNP mechanism are determined by the beginning and end of the growing season that is derived from the MODIS Land Cover Dynamics product (MCD12Q2) averaged over the years from 2001 to 2004 (Hudman et al., 2012). This may be quite different from practices in 2020, the year we simulated in this study. We use the default assumption in the mechanism that 75% of fertilizer is added at the green-up day with the remaining 25% applied constantly throughout the rest of the season (Hudman et al., 2012). We should note that the 75/25 treatment is an estimate for local farming practice in Mexico in 1994 (Matson et al., 1998), which probably introduces biases in other regions. Typical practice would place more fertilizer at topdressing and less at planting. For example, a 20/80 split was used by GGCM, the largest global crop modeling exercise (Jägermeyr et al., 2021), and a 40/60 split was used for maize production in Northeast China (Zheng et al., 2023). It's also worth noting that the BDSNP mechanism treats topdressing as a series of applications spread out over several weeks or months, rather than as a single event, which could further influence the modeling results. Despite the uncertainties, all of these significant improvements of the modified model we used suggest that soil NO_x emission from agricultural fertilization would exert a crucial influence on regional air quality.

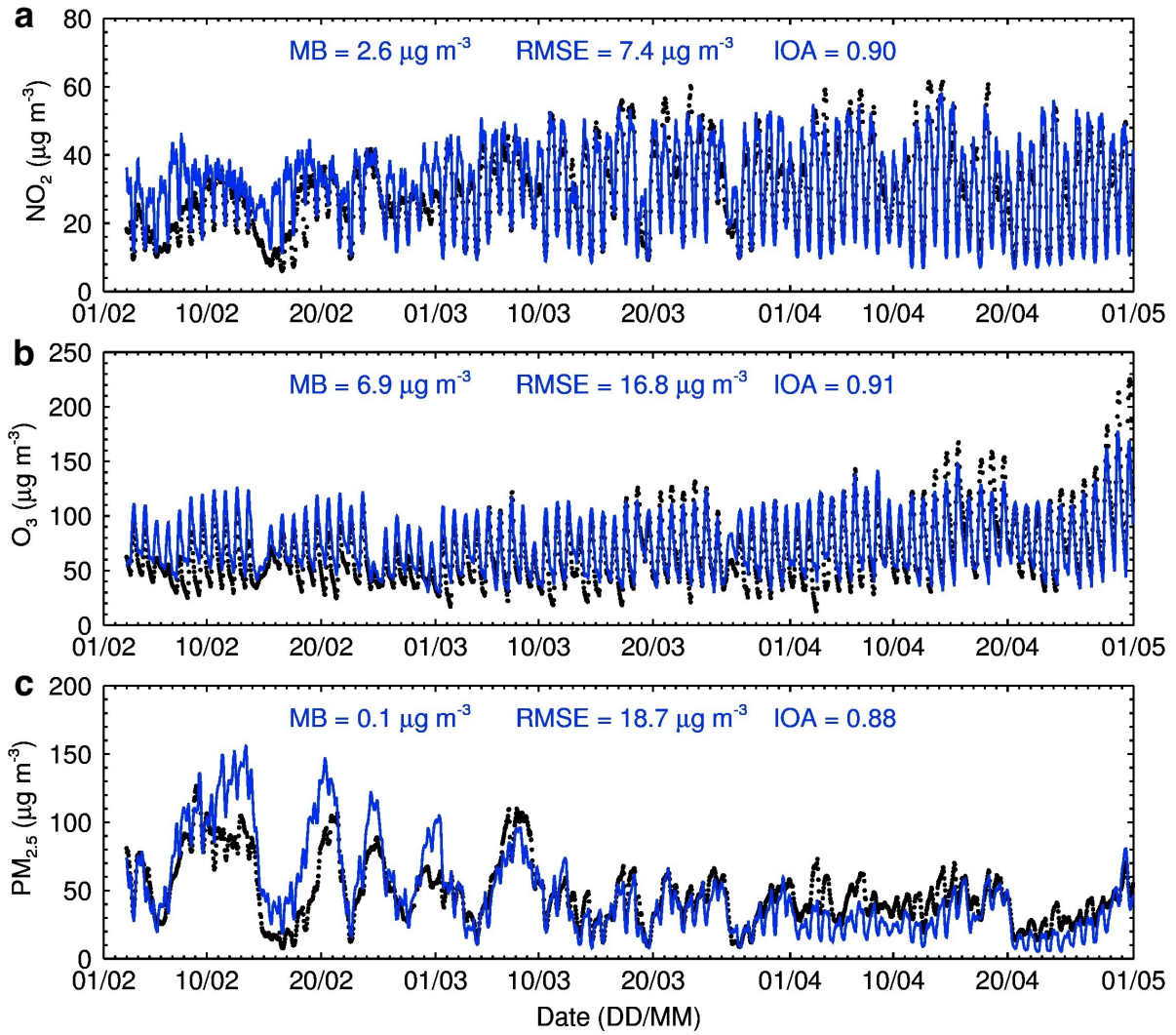


Figure 4. Simulated vs. measured surface pollutants averaged over the monitoring sites of the NCP (Figure S1) during February-April in 2020. (a to c) Temporal variations in surface NO_2 (a), O_3 (b), and $\text{PM}_{2.5}$ (c) mass concentrations. The blue curves denote the model calculation and the black dots denote *in-situ* measurements. Model biases are shown in the central upper position of each figure.

Furthermore, we examine the ability of the model to simulate the ground-level NO_2 mass concentration and NH_3 volume concentration when soil NO_x rapidly releases after fertilization. The reason is that influences of soil emissions on atmospheric NO_2 and NH_3 concentrations are confined in the near-surface layers below 1 km, and the influences diminish rapidly as

altitude increases (Figure 5). This indicates that impact of the soil emissions is primarily concentrated near the ground surface. With soil emissions included or not in the model, we compare the simulated NO_2 and NH_3 concentrations with near-surface observations (Figures 6a and 6b). When there are no soil NO_x emissions from agricultural fertilization, the simulated NO_2 concentration is significantly lower than the observed by $9.4 \mu\text{g m}^{-3}$ during February through April, 2020. While considering these emissions, the mean bias (MB, Text S2) between the simulation and the observation decreases to $2.6 \mu\text{g m}^{-3}$, and the IOA also increases from 0.48 to 0.78. Similarly, the simulated NH_3 concentration is in good agreement with the observed when the soil NH_3 emission related to agricultural fertilization is involved, e.g., the MB decreases from -12.0 ppb to -4.4 ppb, and the IOA increases from 0.56 to 0.64 (Figures 6c and 6d). It is important to acknowledge the limitation posed by the absence of direct comparisons with flux measurements of NO_x emissions from soils, due to the unavailability of such data. The simulated NO_x emission flux from the BDSNP scheme cannot be well examined, which may introduce uncertainties to the predicted emission rates and mixing ratios in the atmosphere.

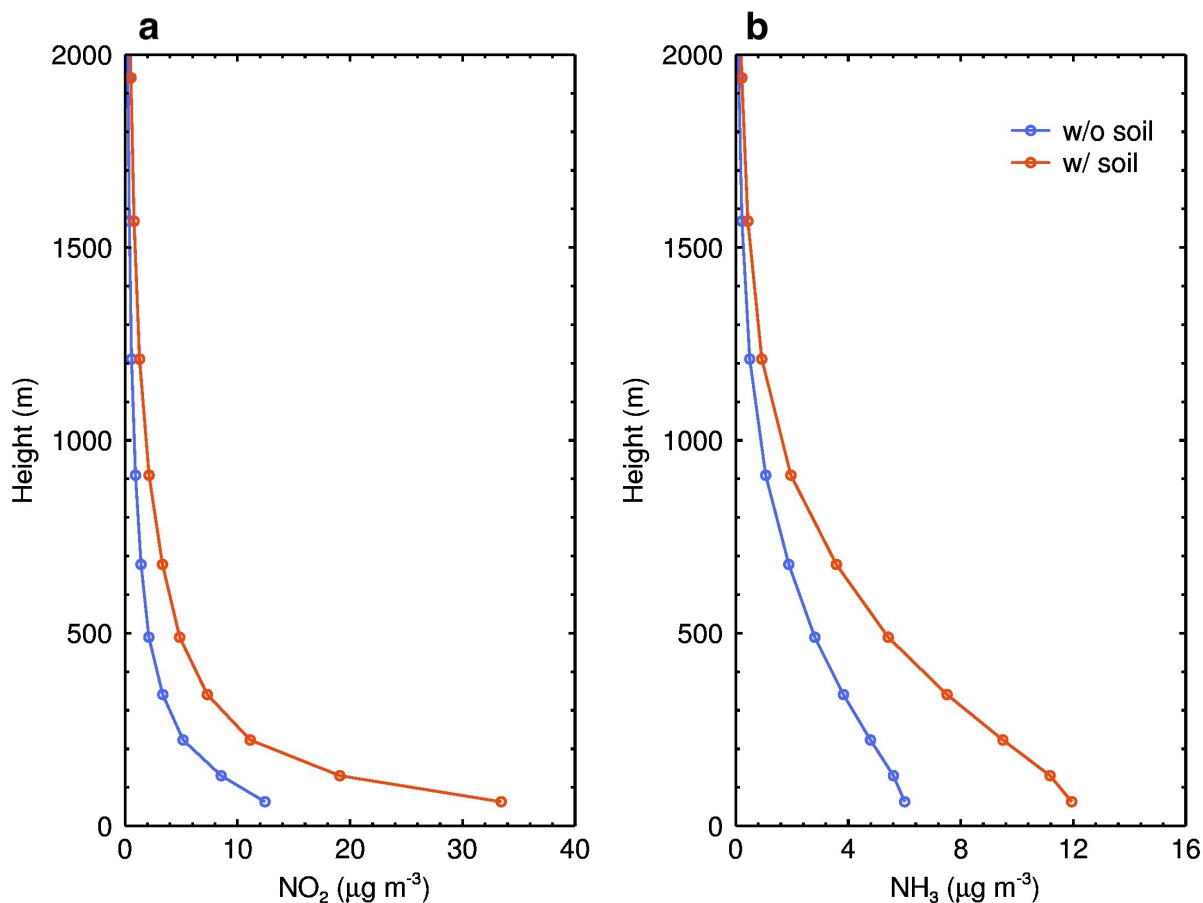


Figure 5. Vertical profiles for impacts of soil emissions on gas pollutants during March 2020 over the NCP. (a) Difference in NO_2 concentration with and without the influence of soil NO_x emission from agricultural fertilization at various heights in the near-surface layers. (b) Similar to (a), but for NH_3 concentration.

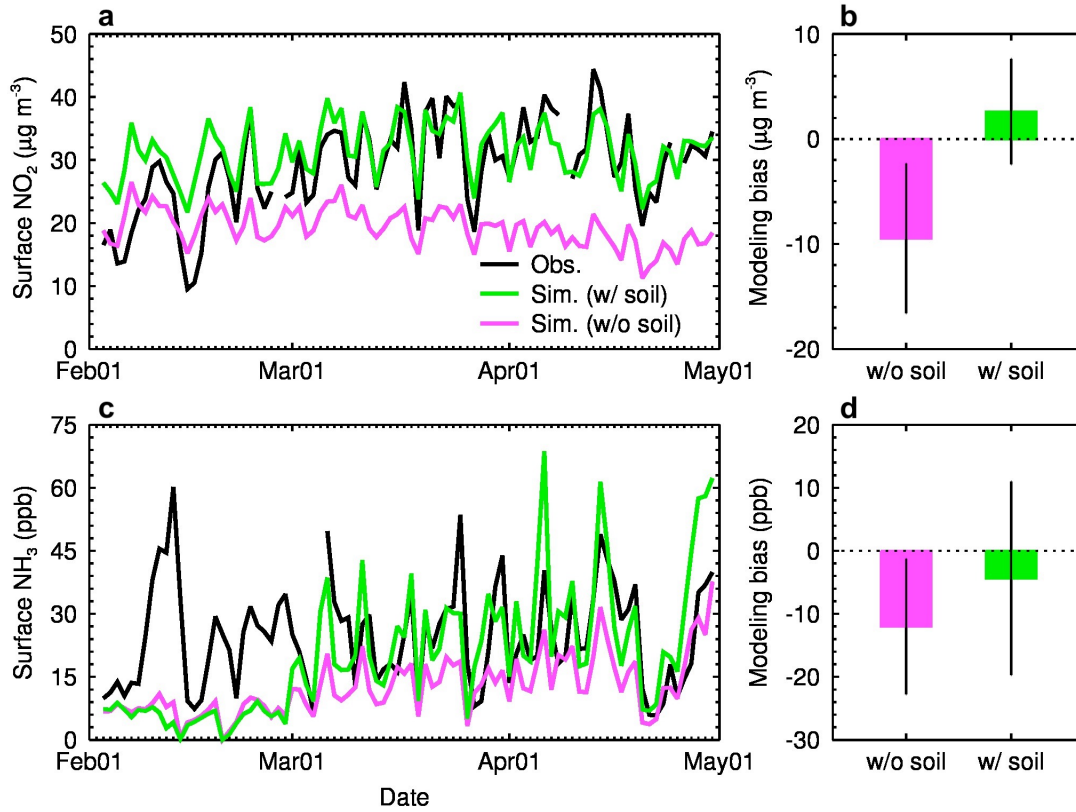


Figure 6. Contribution of soil emissions from agricultural fertilization on surface NO_2 and NH_3 during February through April 2020 over the NCP. (a-b) Change in surface NO_2 concentration with (green) and without (pink) soil NO_x emission from agricultural fertilization, the black line in (a) is for observed surface NO_2 concentration. (c-d) Same as (a-b), but for NH_3 . The error bar in (b and d) denotes $\pm 1\sigma$. NO_2 observations are averaged over the 141 monitoring stations in the study area from the CNEMC network. NH_3 observations are from the rural Xianghe station (Figure 1). According to *in-situ* measurements on NO_2 and NH_3 , the units for NO_2 and NH_3 concentrations are $\mu\text{g m}^{-3}$ and ppb, respectively.

3.4 Significance of soil NO_x emissions from agricultural fertilization for air quality

We perform a model experiment that excludes the soil sources of NO_x and NH_3 in the study domain to examine the impacts of soil emissions on regional air quality. The model results are compared to the benchmark scenario with soil sources involved to examine these impacts. Agricultural fertilization directly leads to substantial increases in atmospheric NO_x and NH_3

concentrations. According to the spatial correlation between land use and NO₂ concentration, NO₂ concentrations increase by more than 15 $\mu\text{g m}^{-3}$ over agricultural areas, with the maximal increments occurring in the densely cultivated southern region of the NCP, exceeding 40 $\mu\text{g m}^{-3}$ (Figures 1b and 7a). While in urban areas, the increase in NO₂ concentration is mostly below 10 $\mu\text{g m}^{-3}$, significantly lower than those in agricultural areas. This indicates that the influence of local emissions originated from agricultural fertilization on air quality primarily concentrate in agricultural areas. Nonetheless, the influence extends to surrounding areas through atmospheric transport, leading to an inhomogeneous increase of NO₂ concentrations across the NCP. Spatial distribution of the increased NH₃ concentration is highly similar to that of the increased NO₂ concentration, but some differences exist in the southeast of the NCP. It should be noted that the NH₃ emission in the model is from Huang et al. (2012), a separate monthly emission inventory. The emission rates of NH₃ in the southeast of the NCP is lower than that of NO from the BDSNP scheme. This indicates nonnegligible discrepancies in the derived emissions between these two approaches, which deserves more in-depth studies.

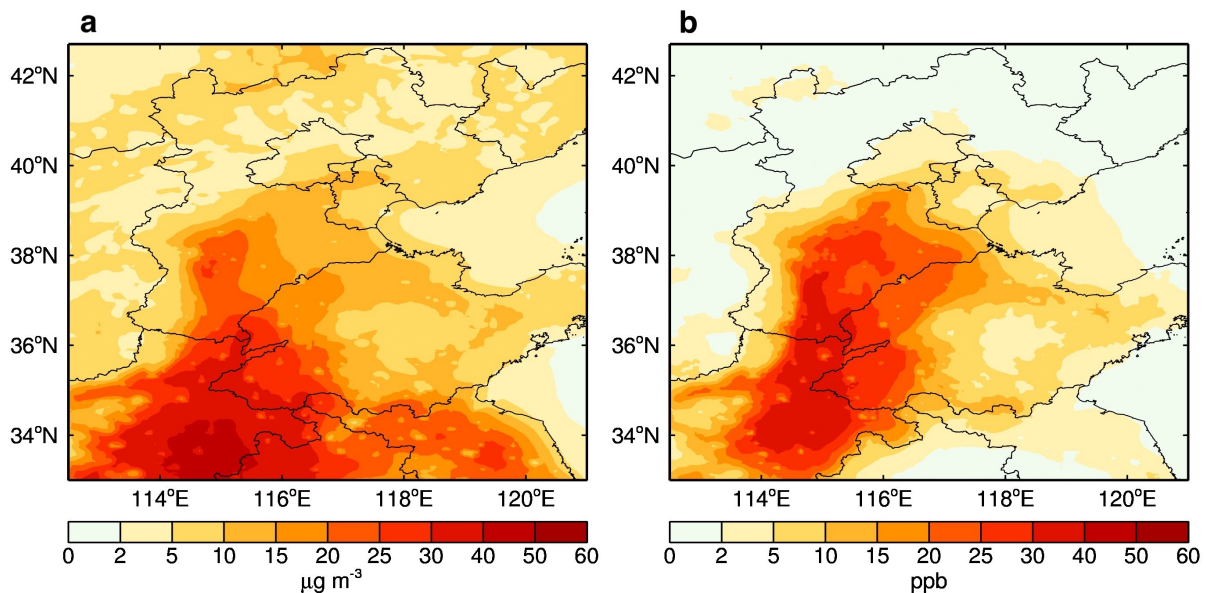


Figure 7. Direct impacts of soil emissions from agricultural fertilization on (a) surface NO₂ and (b) NH₃ during March 2020 over the NCP. (a and b) Spatial distributions of changes in surface NO₂ and NH₃ concentrations due to fertilization-related soil emissions. According to *in-situ*

measurements on NO₂ and NH₃, the units for NO₂ and NH₃ concentrations are $\mu\text{g m}^{-3}$ and ppb, respectively.

A substantial amount of reactive nitrogen from agricultural fertilization suddenly enters the atmosphere, and further affects air quality via photochemical reactions and aerosol chemical transformations profoundly (Seinfeld and Pandis, 2006; Wu et al., 2020). Our results reveal that the NO_x emission induced by N-fertilization significantly suppresses the early-spring O₃ production in the NCP, which varies remarkably with the land use, approximately twice as strong in agricultural areas as in urban areas. For instance, in agricultural areas, the emission in croplands reduces nocturnal and diurnal O₃ by $30.1 \pm 6.5 \mu\text{g m}^{-3}$ ($37.5 \pm 8.1\%$) and $15.0 \pm 3.7 \mu\text{g m}^{-3}$ ($18.7 \pm 4.6\%$), respectively, while in urban areas, the corresponding O₃ reductions are $15.6 \pm 4.7 \mu\text{g m}^{-3}$ ($15.6 \pm 4.7\%$) and $9.7 \pm 3.2 \mu\text{g m}^{-3}$ ($10.6 \pm 3.4\%$), respectively (Figure 8). Based on the diurnal cycle of the change in O₃ concentrations ($\Delta[\text{O}_3]$), we also find that the nighttime O₃ reduction is much higher than the daytime reduction (Figure 9). The $\Delta[\text{O}_3]$ caused by agricultural fertilization is linearly and negatively correlated with the change in NO₂ concentration ($\Delta[\text{NO}_2]$) (Figures 10a-d), and the negative correlation is more pronounced at night ($r < -0.99$ and $p < 0.001$ for both the agricultural and the urban areas, Figures 10a and b). This suggests that the O₃ concentration strongly depends on the change in NO_x levels in the NCP during early spring. Continuous agricultural NO_x (mainly NO) emissions inhibit the O₃ formation, which indicates that the NO_x abundance is excess for O₃ formation over the NCP. Huang et al., (2023) similarly reported that variations in O₃ concentrations were inversely related to the changes in soil NO_x emissions in the NCP. Lu et al., (2021) also reported a NO_x-saturated O₃ formation regime in the NCP. On the other hand, a negative correlation between $\Delta[\text{NO}_2]$ and the change in daytime OH radical ($\Delta[\text{OH}]_{\text{day}}$) suggests that the $\Delta[\text{NO}_2]$ also moderately regulates $\Delta[\text{OH}]_{\text{day}}$ ($r = -0.50$ for agricultural areas and $r = -0.43$ for urban areas, p

< 0.001, Figures 10e and f) through decreasing O₃ levels and reactions of NO₂ with OH radical. Both OH radical and O₃ are critical oxidants in the atmosphere, and the decrease by the excessive NO_x emission from agricultural fertilization weakens atmospheric oxidizing capacity (AOC) (Feng et al., 2021). The decreased AOC can further slow down the oxidation processes in homogeneous and heterogeneous reactions, unfavorable for the formation of secondary aerosols. We note that soil nitrous acid (HONO) emission, which are not included in these modeling experiments, can also perturb atmospheric chemistry and the AOC (Feng et al., 2022; Tan et al., 2023) via providing NO and OH through photolysis. The emission rate of HONO from soil is much less than that of NO_x in the NCP (Tan et al., 2023), which increases daytime O₃ and OH concentrations slightly during summer (Feng et al., 2022; Tan et al., 2023). However, the influence in springtime still remains to be elucidated.

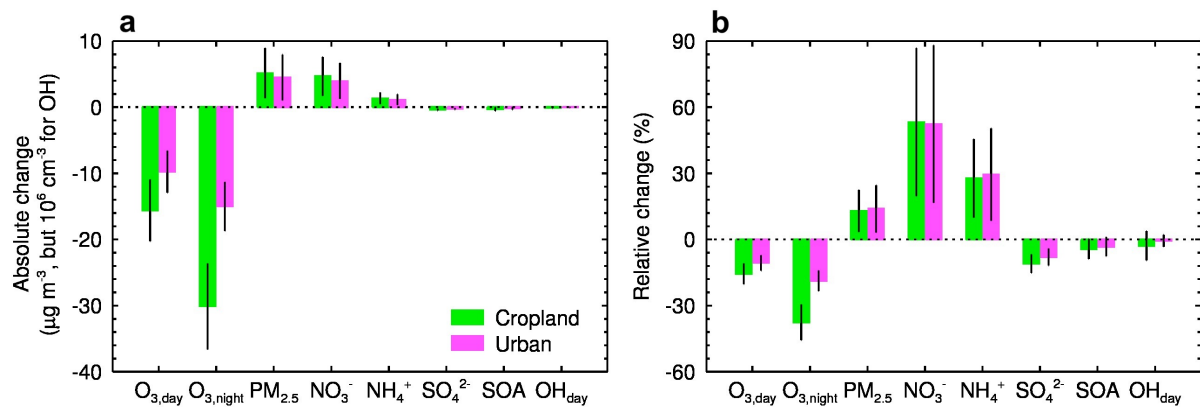


Figure 8. Complex impacts of agricultural fertilization on O₃, PM_{2.5}, and OH during March 2020 over the NCP. (a) Changes in mass concentrations of O₃, PM_{2.5}, aerosol constituents, i.e., nitrate, ammonium, sulfate and secondary organics, and OH radical due to soil NO_x emission from agricultural fertilization in agricultural (green) and urban (pink) areas. The error bar denotes ±1σ. (b) Same as (a), but for percentage changes.

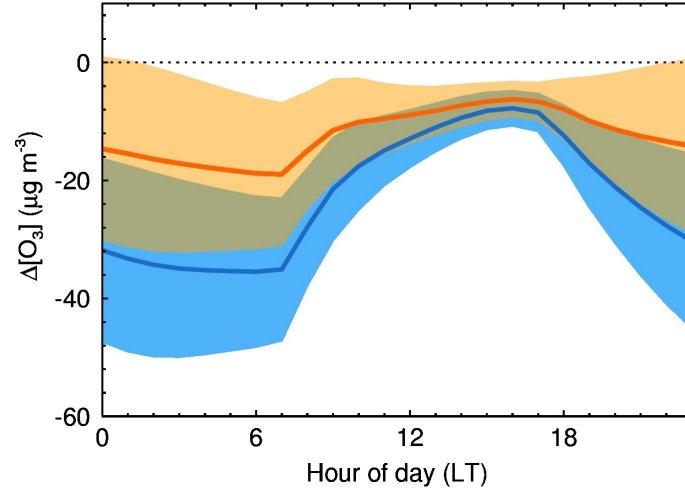


Figure 9. Secondary impact of soil NO_x emissions from agricultural fertilization on surface O₃ during March 2020 over the NCP. Diurnal cycles of changes in surface O₃ concentrations due to fertilization-related soil emissions over croplands and urban areas in the NCP. The blue and orange shadings show $\pm 1\sigma$ of the data.

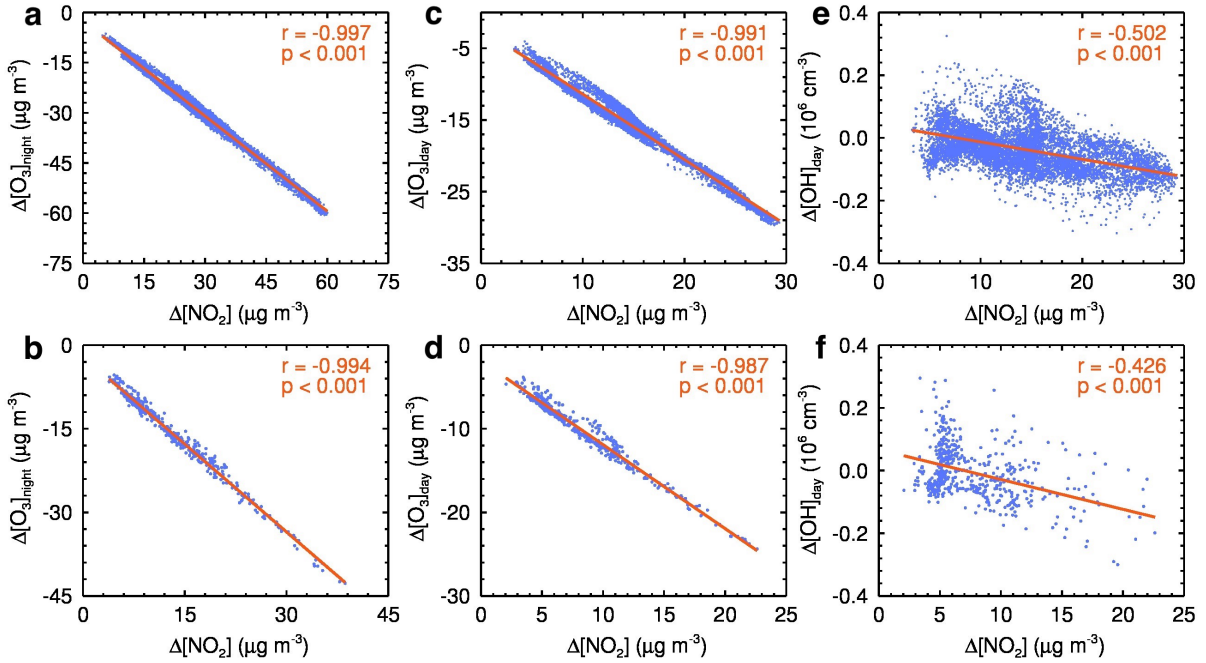


Figure 10. Changes in surface NO₂ and related photochemical products. (a to f) Correlation between $\Delta[\text{NO}_2]$ and $\Delta[\text{O}_3]$ or $\Delta[\text{OH}]$ during March 2020 over the NCP. (a to d) Change in O₃ concentration is strongly dependent on change in NO₂ concentration due to agricultural fertilization in both agricultural (a and c) and urban (b and d) areas, and the dependence is more

pronounced at night, i.e., correlation coefficient $r = -0.997$ ($r = -0.994$) at night and $r = -0.991$ ($r = -0.987$) at daytime in agricultural (urban) areas. (e and f) Change in daytime OH radical is also significantly influenced by change in NO_2 concentration in both agricultural (e) and urban (f) areas.

Interestingly, these findings regarding the impacts of soil NO_x emission on O_3 formation in spring are different from previous studies revealing that agricultural NO_x emissions enhance the O_3 formation in summer over the NCP (Tan et al., 2023; Wang et al., 2022a) and northeast China (Shen et al., 2023) and in the Imperial Valley, California (Oikawa et al., 2015). Similar scenarios are also reported during the growing season of crops in sub-Saharan Africa (Hickman et al., 2017; Huang et al., 2018). This is largely attributed to the sensitivity of O_3 to its precursors. During early spring, a large amount of agricultural NO_x (mainly NO) emission causes a NO titration effect during daytime, decreasing O_3 concentrations, when the O_3 chemistry is under the VOC-sensitive (NO_x -saturated) or the transitional regimes (Figure S6) (Sillman, 1995). In contrast, the O_3 formation chemistry in summer shifts from VOCs-sensitive to NO_x -sensitive (Sha et al., 2021; Wang et al., 2022a). In this scenario, the O_3 production is primarily controlled by NO_x emissions, meaning that the O_3 concentration increases with rising NO_x levels. This seasonal difference in O_3 sensitivity to its precursors highlights a seasonally dependent response of O_3 production to agricultural fertilization.

We also quantify the impact of agricultural fertilization on $\text{PM}_{2.5}$ concentrations. The NCP is characterized by an excess of NH_3 , in which nitrate formation is highly sensitive to NO_2 concentration and AOC due to NO_2 oxidation to NO_3^- via gas-phase and heterogeneous reactions (Feng et al., 2018; Fu et al., 2020; Liu et al., 2019; Wen et al., 2018). As atmospheric NO_2 and NH_3 concentrations rapidly increase due to emissions from fertilized croplands, nitrate aerosol in agricultural (urban) areas rises by 4.7 (4.0) $\mu\text{g m}^{-3}$, corresponding to the

increased percentage of 53.2% (52.3%), while ammonium aerosol rises by 1.3 (1.1) $\mu\text{g m}^{-3}$ in agricultural (urban) areas, with an increased percentage of 27.7% (29.4%) (Figure 8). However, sulfate aerosol shows a slight decrease both in agricultural and urban areas (Figure 8a). The reason is that an extra NO_x emission from agricultural fertilization enhances nitrate formation but lowers AOC, which hinders sulfate formation. Similar to sulfate aerosol, secondary organic aerosol (SOA) also has a slight reduction (Figure 8a). The formation of SOA greatly depends on the AOC level, so decreased AOC due to NO_x emission from agricultural fertilization does not favor the conversion of organic precursors, such as VOCs and semi-volatile primary organic aerosols, into SOA.

In general, due to the NO_x emission from agricultural fertilization, $\text{PM}_{2.5}$ concentration increases by 5.1 (4.5) $\mu\text{g m}^{-3}$ (Figure 8a), corresponding to a percentage change of 12.9% (13.9%) over agricultural (urban) areas in the NCP (Figure 8b). There is no significant difference in $\text{PM}_{2.5}$ increments between agricultural and urban areas. Nitrate aerosol is primarily responsible for the increased $\text{PM}_{2.5}$, accounting for 92.2% and 88.9% in these two types of regions, respectively. Our results also indicate that changes in $\text{PM}_{2.5}$ and nitrate in urban areas are more sensitive to the change in NO_2 concentration. For instance, the ratios of nitrate change to NO_2 change ($\Delta[\text{NO}_3^-]/\Delta[\text{NO}_2] = 0.20$) and $\text{PM}_{2.5}$ change to NO_2 change ($\Delta[\text{PM}_{2.5}]/\Delta[\text{NO}_2] = 0.24$) in urban areas are both higher than those in agricultural areas ($\Delta[\text{NO}_3^-]/\Delta[\text{NO}_2] = 0.13$ and $\Delta[\text{PM}_{2.5}]/\Delta[\text{NO}_2] = 0.15$, Figures 11a and b), indicating that the conversion of NO_2 to nitrate aerosol is more efficient in urban areas. Consequently, the increased percentages of $\text{PM}_{2.5}$ and ammonium aerosol in urban areas are higher than those in agricultural areas (Figure 8b). Additionally, the ongoing stringent control measures on emission sources significantly reduce anthropogenic emissions in urban areas, thus the impact of agricultural fertilization on urban air quality is becoming more pronounced (Figure S7). Since soil NO_x emission is sensitive to soil temperature, as global warming is ongoing, routine events

like agricultural fertilization will continue to have amplified impacts on air quality with the joint help of atmospheric dispersion/transport and chemical transformation processes (Bennetzen et al., 2016; Ma et al., 2022; Tubiello et al., 2013). These impacts are not confined in agricultural areas alone, but extend to surrounding cities.

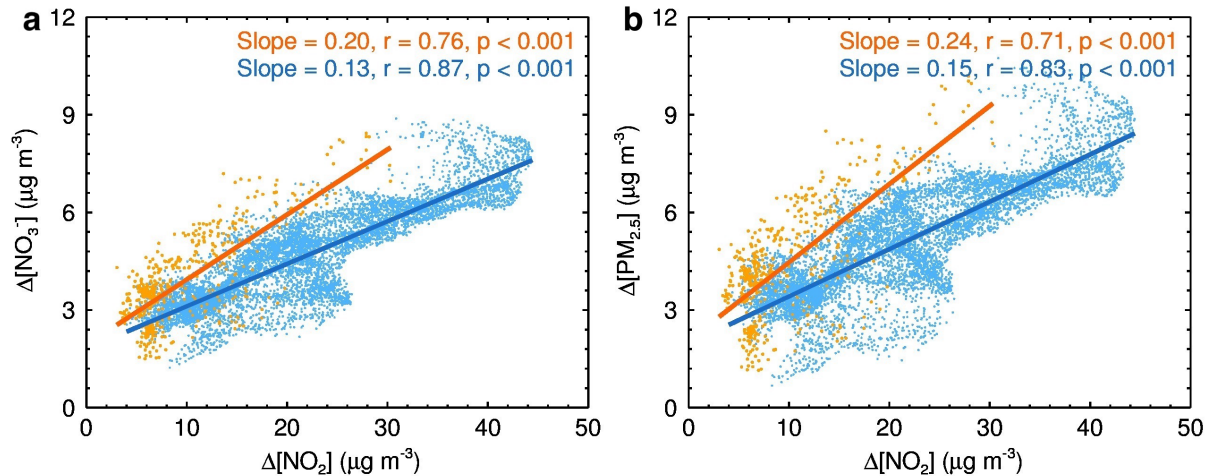


Figure 11. **Simulated changes** in surface NO_2 and related aerosol-chemistry products during March 2020 over the NCP. (a) Comparison of NO_2 conversion to nitrate aerosol (NO_3^-) formation between agricultural and urban areas, i.e., $\Delta[\text{NO}_3^-]/\Delta[\text{NO}_2] = 0.20$ in urban areas and $\Delta[\text{NO}_3^-]/\Delta[\text{NO}_2] = 0.13$ in agricultural areas, indicates that change in nitrate in urban areas are more sensitive to the change in NO_2 concentration and NO_2 conversion to NO_3^- is more efficient. (b) NO_2 conversion to $\text{PM}_{2.5}$ formation is similar to (a), because nitrate aerosol is the most affected among the various aerosol constituents. The blue and orange colors correspond to the agricultural and urban areas, respectively.

4 Conclusions and Implications

Impact of soil NO_x emissions from agricultural fertilization on the atmospheric environment remains unclear worldwide (Guo et al., 2020; Huang et al., 2018; Sha et al., 2021; Shen et al., 2023). In particular, this issue has not yet received enough attention in China, where substantial N-fertilizers are year by year consumed due to extensive agricultural cultivation areas (Sun et

al., 2022; Vitousek et al., 2009; Zhao et al., 2006). Our results indicate that agricultural fertilization is highly responsible for the periodic pulse of the atmospheric NO₂ column in the NCP over the past two decades. A two-decade record of fertilization events at a research station and model results both provide evidence consistent with a cause-to-effect relationship. For example, the fertilization timing is found to match well with the occurrence of satellite-derived NO₂ column pulse in the region. Moreover, the model reasonably captures the regular sub-peak of the NO₂ column in March by introducing an independent module that specifically describes soil NO_x emissions from agricultural fertilization.

These additional NO_x emissions released by croplands directly lead to an elevated level of surface NO_x concentrations. Consequently, the increased atmospheric NO_x concentration significantly inhibits O₃ production in early spring, distinct from the impacts in summer (Sha et al., 2021; Wang et al., 2022a), but enhances nitrate formation. For example, soil emissions linked to agricultural fertilization dramatically reduce nighttime O₃ concentrations by 30.1 µg m⁻³ and 15.0 µg m⁻³ in croplands and urban areas, respectively. During daytime, the decreased O₃ concentration is 15.6 µg m⁻³ and 9.7 µg m⁻³, respectively. In contrast, soil emissions elevate ambient PM_{2.5} concentrations by more than 4.5 µg m⁻³, accounting for 12% of the PM_{2.5} mass over the NCP in March 2020. The opposite effects are challengeable for China to improve air quality, because China is the world's largest consumer of food, of which food production strongly depends on N-fertilizers input. As the emission from fossil fuel combustion has been gradually decreasing, emissions from agricultural fertilization are with increasing implications for air quality. We thus highlight that reducing NO_x emissions from agricultural fertilization is of great importance to air quality improvement. In China, the excessive use of N-fertilizer still remains severe (Sun et al., 2022; Vitousek et al., 2009; Zhao et al., 2006), though a lot of efforts have taken to increase the N-fertilizer efficiency and to reduce N losses from fertilizer (Li et al., 2018; Qiao et al., 2022; YAN et al., 2008). Fortunately, the consumption of N-fertilizer

reached its peak in 2014 in China and has been decreasing since then (Yu et al., 2022). Policymakers should manage to further reduce emissions from N-fertilizers application, for example, improving N-fertilizers efficiency and developing alternative fertilizers friendly to the environment are highly necessary. These measures will greatly minimize the adverse effects of agricultural fertilization on air quality, human health, and the ecological environment. Nevertheless, one should be aware of the limitation in the present case study that there are only three months of simulation as the basis for all of the insights into the soil NO_x emission and its influences on atmospheric chemistry and composition. More studies in terms of soil NO_x emissions, particularly during springtime, are in need to validate and generalize our model results.

We should not ignore the uncertainties regarding the BDSNP scheme. The default 75/25 split in fertilizer application may be not widely suitable for the globe. Using a 20/80 split, as is commonly used in crop modeling (Jägermeyr et al., 2021), or a 40/60 split, as has been reported to be common in Northeast China (Zheng et al., 2023), would be expected to result in differences in the magnitude and timing of emissions compared to the default scheme. Among other impacts, there is no canopy interception of emitted soil NO_x at planting, which would result in substantially larger emissions to the atmosphere under the default 75/25 split than in a 20/80 split. In addition, because BDSNP applies the 25% topdressing application evenly over the growing season following the 75% basal application, it is less likely to produce sizable pulses of emissions. Because fertilizer applications are kept constant and global fertilizer emissions are constrained to 1.8 Tg N, the BDSNP mechanism is unable to reproduce historical trends or capture significant interannual variability in emissions.

The discrepancies between NO₂ and NH₃ column densities suggest substantial differences in the soil emission mechanisms of NO_x and NH₃, especially after fertilization. Future studies

could incorporate a dynamic bidirectional NH₃ scheme alongside the BDSNP scheme to further investigate the nature of fertilizer-induced emission pulses.

Data availability

The OMI satellite data are from the NASA Goddard Space Flight Center, Goddard Earth Sciences Data and Information Services Center (GES DISC) (https://disc.gsfc.nasa.gov/datasets/OMI_MINDS_NO2d_1.1/summary) and the IASI satellite observations are from the IASI Portal (https://iasi.aeris-data.fr/nh3_iasi_a_arch). The real-time hourly air pollutant measurements including NO₂, O₃, and PM_{2.5} are released by Ministry of Ecology and Environment, China and can be accessed on the website <https://quotsoft.net/air/>. The MEIC Group and the EDGAR Team for the MEIC and HATP emission inventories which are available at <http://www.meicmodel.org> and https://edgar.jrc.ec.europa.eu/dataset_htap_v3, respectively.

Author contribution

TF and GL conceptualized the ideas, verified the conclusions, and revised the paper. TF conducted research, designed the experiments, carried out the methodology, performed the simulation, processed the data, prepared the data visualization, and prepared the paper, with contributions from all authors. SZ and NB provided the treatment of meteorological data, analyzed the study data, validated the model performance, and reviewed the paper. XL, YP, YS, and RW provided the observation data and emission inventories, and reviewed the paper. XT and LM provided critical reviews in the pre-publication stage.

Competing interests

The authors declare no conflicts of interest relevant to this study.

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References

- Almaraz, M., Bai, E., Wang, C., Trousdell, J., Conley, S., Faloona, I. and Houlton, B. Z.: Agriculture is a major source of NO_x pollution in California, *Sci Adv*, 4(1), eaao3477, doi:10.1126/sciadv.aao3477, 2018.
- Anderson, I. C. and Levine, J. S.: Simultaneous field measurements of biogenic emissions of nitric oxide and nitrous oxide, *J Geophys Res*, 92, 965–976, doi:10.1029/JD092iD01p00965, 1987.
- Bauwens, M., Compernelle, S., Stavrakou, T., Müller, J. F., van Gent, J., Eskes, H., Levelt, P. F., van der A, R., Veefkind, J. P., Vlietinck, J., Yu, H. and Zehner, C.: Impact of Coronavirus Outbreak on NO₂ Pollution Assessed Using TROPOMI and OMI Observations, *Geophys. Res. Lett.*, 47(1), e87978, doi:10.1029/2020GL087978, 2020.
- Bennetzen, E. H., Smith, P. and Porter, J. R.: Decoupling of greenhouse gas emissions from global agricultural production: 1970-2050, *Global Change Biol*, 22(2), 763–781, doi:10.1111/gcb.13120, 2016.
- Bouwman, A. F., Boumans, L. J. M. and Batjes, N. H.: Modeling global annual N₂O and NO emissions from fertilized fields, *Global Biogeochem. Cycles*, 16(4), 1080–28–9, doi:10.1029/2001GB001812, 2002.
- Buchholz, R. R., Emmons, L. K., Tilmes, S. and The CESM Development Team: CESM2.1/CAM-chem Instantaneous Output for Boundary Conditions, UCARNCAR - Atmospheric Chemistry Observations and Modeling Laboratory, doi:10.5065/NMP7-EP60, 2019.

657 Cárdenas, L., Rondón, A., Johansson, C. and Sanhueza, E.: Effects of soil moisture,
658 temperature, and inorganic nitrogen on nitric oxide emissions from acidic tropical savannah
659 soils, *J Geophys Res*, 98(D8), 14783–14790, doi:10.1029/93JD01020, 1993.

660 Chen, F. and Dudhia, J.: Coupling an advanced land surface-hydrology model with the Penn
661 State-NCAR MM5 modeling system. Part II: Preliminary model validation, *Mon. Weather*
662 *Rev.*, 129(4), 569–585, doi:10.1175/1520-0493(2001)129<0569:caalsh>2.0.co;2, 2001.

663 Chou, M.-D. and Suarez, M. J.: A solar radiation parameterization for atmospheric studies,
664 edited by M. J. Suarez. 1999.

665 Chou, M.-D., Suarez, M. J., Liang, X.-Z. and Yan, M. M. H.: A thermal infrared radiation
666 parameterization for atmospheric studies, edited by M. J. Suarez. 2001.

667 Clarisse, L., Franco, B., Van Damme, M., Di Gioacchino, T., Hadji-Lazaro, J., Whitburn, S.,
668 Noppen, L., Hurtmans, D., Clerbaux, C. and Coheur, P.: The IASI NH₃ version 4 product:
669 averaging kernels and improved consistency, *Atmos. Meas. Tech.*, 16(21), 5009–5028,
670 doi:10.5194/amt-16-5009-2023, 2023.

671 Crippa, M., Guizzardi, D., Butler, T., Keating, T., Wu, R., Kaminski, J., Kuenen, J.,
672 Kurokawa, J., Chatani, S., Morikawa, T., Pouliot, G., Racine, J., Moran, M. D., Klimont, Z.,
673 Manseau, P. M., Mashayekhi, R., Henderson, B. H., Smith, S. J., Suchyta, H., Muntean, M.,
674 Solazzo, E., Banja, M., Schaaf, E., Pagani, F., Woo, J.-H., Kim, J., Monforti-Ferrario, F.,
675 Pisoni, E., Zhang, J., Niemi, D., Sassi, M., Ansari, T. and Foley, K.: The HTAP_v3 emission
676 mosaic: merging regional and global monthly emissions (2000–2018) to support air quality
677 modelling and policies, *Earth System Science Data*, 15(6), 2667–2694, doi:10.5194/essd-15-
678 2667-2023, 2023.

679 Davidson, E. A.: Sources of nitric oxide and nitrous oxide following wetting of dry soil, *Soil*
680 *Science Society of America Journal*, 56(1), 95–102,
681 doi:10.2136/sssaj1992.03615995005600010015x, 1992.

682 Davidson, E. A., Potter, C. S., Schlesinger, P. and Klooster, S. A.: Model estimates of
683 regional nitric oxide emissions from soils of the southeastern United States, *Ecological*
684 *Applications*, 8(3), 748–759, doi:10.1890/1051-0761(1998)008[0748:MEORNO]2.0.CO;2,
685 1998.

686 Emmons, L. K., Schwantes, R. H., Orlando, J. J., Tyndall, G., Kinnison, D., Lamarque, J.-F.,
687 Marsh, D., Mills, M. J., Tilmes, S., Bardeen, C., Buchholz, R. R., Conley, A., Gettelman, A.,
688 Garcia, R., Simpson, I., Blake, D. R., Meinardi, S. and Pétron, G.: The Chemistry Mechanism
689 in the Community Earth System Model Version 2 (CESM2), *J. Adv. Model. Earth Syst.*,
690 12(4), e2019MS001882, doi:10.1029/2019MS001882, 2020.

691 Feng, T., Bei, N., Zhao, S., Wu, J., Li, X., Zhang, T., Cao, J., Zhou, W. and Li, G.:
692 Wintertime nitrate formation during haze days in the Guanzhong basin, China: A case study,

693 Environ. Pollut., 243, 1057–1067, doi:10.1016/j.envpol.2018.09.069, 2018.

694 Feng, T., Zhao, S., Hu, B., Bei, N., Zhang, X., Wu, J., Li, X., Liu, L., Wang, R., Tie, X. and
 695 Li, G.: Assessment of Atmospheric Oxidizing Capacity Over the Beijing-Tianjin-Hebei
 696 (BTH) Area, China, *J. Geophys. Res. Atmos.*, 126(7), e2020JD033834,
 697 doi:10.1029/2020JD033834, 2021.

698 Feng, T., Zhao, S., Liu, L., Long, X., Gao, C. and Wu, N.: Nitrous acid emission from soil
 699 bacteria and related environmental effect over the North China Plain, *Chemosphere*, 287,
 700 132034, doi:10.1016/j.chemosphere.2021.132034, 2022.

701 Fu, X., Wang, T., Gao, J., Wang, P., Liu, Y., Wang, S., Zhao, B. and Xue, L.: Persistent
 702 Heavy Winter Nitrate Pollution Driven by Increased Photochemical Oxidants in Northern
 703 China, *Environ. Sci. Technol.*, 54(7), 3881–3889, doi:10.1021/acs.est.9b07248, 2020.

704 Galbally, I. E. and Roy, C. R.: Loss of fixed nitrogen from soils by nitric oxide exhalation,
 705 *Nature*, 275(5), 734–735, doi:10.1038/275734a0, 1978.

706 Gong, C., Wang, Y., Tian, H., Kou-Giesbrecht, S., Vuichard, N. and Zaehle, S.: Uncertainties
 707 in fertilizer-induced emissions of soil nitrogen oxide and the associated impacts on ground-
 708 level ozone and methane, *EGUsphere*, 1–17, doi:10.5194/egusphere-2025-1416, 2025.

709 Guenther, A., Karl, T., Harley, P., Wiedinmyer, C., Palmer, P. I. and Geron, C.: Estimates of
 710 global terrestrial isoprene emissions using MEGAN (Model of Emissions of Gases and
 711 Aerosols from Nature), *Atmos. Chem. Phys.*, 6(11), 3181–3210, doi:10.5194/acp-6-3181-
 712 2006, 2006.

713 Guo, L., Chen, J., Luo, D., Liu, S., Lee, H. J., Motallebi, N., Fong, A., Deng, J., Rasool, Q.
 714 Z., Avise, J. C., Kuwayama, T., Croes, B. E. and FitzGibbon, M.: Assessment of Nitrogen
 715 Oxide Emissions and San Joaquin Valley PM_{2.5} Impacts From Soils in California, *J.*
 716 *Geophys. Res. Atmos.*, 125(24), e2020JD033304, doi:10.1029/2020JD033304, 2020.

717 Hall, S. J., Matson, P. A. and Roth, P. M.: NO_x EMISSIONS FROM SOIL: Implications for
 718 Air Quality Modeling in Agricultural Regions, *Annu. Rev. Energy. Environ.*, 21(1), 311–346,
 719 doi:10.1146/annurev.energy.21.1.311, 1996.

720 Hickman, J. E., Huang, Y., Wu, S., Diru, W., Groffman, P. M., Tully, K. L. and Palm, C. A.:
 721 Nonlinear response of nitric oxide fluxes to fertilizer inputs and the impacts of agricultural
 722 intensification on tropospheric ozone pollution in Kenya, *Global Change Biol*, 23(8), 3193–
 723 3204, doi:10.1111/gcb.13644, 2017.

724 Hong, S. Y. and Lim, J.: The WRF single-moment 6-class microphysics scheme (WSM6),
 725 *Asia-Pac. J. Atmos. Sci.*, 42(2), 129–151, 2006.

726 Huang, L., Fang, J., Liao, J., Yarwood, G., Chen, H., Wang, Y. and Li, L.: Insights into soil

727 NO emissions and the contribution to surface ozone formation in China, *Atmos. Chem.*
728 *Phys.*, 23(23), 14919–14932, doi:10.5194/acp-23-14919-2023, 2023.

729 Huang, X., Song, Y., Li, M., Li, J., Huo, Q., Cai, X., Zhu, T., Hu, M. and Zhang, H.: A high-
730 resolution ammonia emission inventory in China, *Global Biogeochem. Cycles*, 26(1),
731 GB1030, doi:10.1029/2011GB004161, 2012.

732 Huang, Y., Hickman, J. E. and Wu, S.: Impacts of enhanced fertilizer applications on
733 tropospheric ozone and crop damage over sub-Saharan Africa, *Atmos. Environ.*, 180, 117–
734 125, doi:10.1016/j.atmosenv.2018.02.040, 2018.

735 Huber, D. E., Steiner, A. L. and Kort, E. A.: Daily Cropland Soil NO_x Emissions Identified
736 by TROPOMI and SMAP, *Geophys. Res. Lett.*, 47(2), e89949, doi:10.1029/2020GL089949,
737 2020.

738 Hudman, R. C., Moore, N. E., Mebust, A. K., Martin, R. V., Russell, A. R., Valin, L. C. and
739 Cohen, R. C.: Steps towards a mechanistic model of global soil nitric oxide emissions:
740 implementation and space based-constraints, *Atmos. Chem. Phys.*, 12(1), 7779–7795,
741 doi:10.5194/acp-12-7779-2012, 2012.

742 IPCC: Short-lived Climate Forcers, in *Climate Change 2021 – The Physical Science Basis:*
743 *Working Group I Contribution to the Sixth Assessment Report of the Intergovernmental*
744 *Panel on Climate Change*, pp. 817–922, Cambridge University Press, Cambridge. 2023.

745 Janjić, Z. I.: Nonsingular implementation of the Mellor–Yamada level 2.5 scheme in the
746 NCEP Meso model. 2002.

747 Janssens-Maenhout, G., Crippa, M., Guizzardi, D., Dentener, F., Muntean, M., Pouliot, G.,
748 Keating, T., Zhang, Q., Kurokawa, J., Wankmüller, R., Denier van der Gon, H., Kuenen, J. J.
749 P., Klimont, Z., Frost, G., Darras, S., Koffi, B. and Li, M.: HTAP-v2.2: A mosaic of regional
750 and global emission grid maps for 2008 and 2010 to study hemispheric transport of air
751 pollution, *Atmos. Chem. Phys.*, 15(19), 11411–11432, doi:10.5194/acp-15-11411-2015,
752 2015.

753 Jägermeyr, J., Müller, C., Ruane, A. C., Elliott, J., Balkovic, J., Castillo, O., Faye, B., Foster,
754 I., Folberth, C., Franke, J. A., Fuchs, K., Guarin, J. R., Heinke, J., Hoogenboom, G., Iizumi,
755 T., Jain, A. K., Kelly, D., Khabarov, N., Lange, S., Lin, T.-S., Liu, W., Mialyk, O., Minoli,
756 S., Moyer, E. J., Okada, M., Phillips, M., Porter, C., Rabin, S. S., Scheer, C., Schneider, J.
757 M., Schyns, J. F., Skalsky, R., Smerald, A., Stella, T., Stephens, H., Webber, H., Zabel, F.
758 and Rosenzweig, C.: Climate impacts on global agriculture emerge earlier in new generation
759 of climate and crop models, *Nature Food*, 2(11), 873–885, 2021.

760 Kurokawa, J. and Ohara, T.: Long-term historical trends in air pollutant emissions in Asia:
761 Regional Emission inventory in ASia (REAS) version 3, *Atmos. Chem. Phys.*, 20(21),
762 12761–12793, doi:10.5194/acp-20-12761-2020, 2020.

763 Lamsal, L. N., Krotkov, N. A., Vasilkov, A., Marchenko, S., Qin, W., Yang, E.-S., Fasnacht,
 764 Z., Joiner, J., Choi, S., Haffner, D., Swartz, W. H., Fisher, B. and Bucsela, E.: Ozone
 765 Monitoring Instrument (OMI) Aura nitrogen dioxide standard product version 4.0 with
 766 improved surface and cloud treatments, *Atmos. Meas. Tech.*, 14(1), 455–479,
 767 doi:10.5194/amt-14-455-2021, 2021.

768 Laville, P., Lehuger, S., Loubet, B., Chaumartin, F. and Cellier, P.: Effect of management,
 769 climate and soil conditions on N₂O and NO emissions from an arable crop rotation using
 770 high temporal resolution measurements, *Agr Forest Meteorol*, 151(2), 228–240,
 771 doi:10.1016/j.agrformet.2010.10.008, 2011.

772 Li, G., Bei, N., Tie, X. and Molina, L. T.: Aerosol effects on the photochemistry in Mexico
 773 City during MCMA-2006/MILAGRO campaign, *Atmos. Chem. Phys.*, 11(11), 5169–5182,
 774 doi:10.5194/acp-11-5169-2011, 2011a.

775 Li, G., Lei, W., Bei, N. and Molina, L. T.: Contribution of garbage burning to chloride and
 776 PM_{2.5} in Mexico City, *Atmos. Chem. Phys.*, 12(18), 8751–8761, doi:10.5194/acp-12-8751-
 777 2012, 2012.

778 Li, G., Lei, W., Zavala, M., Volkamer, R., Dusanter, S., Stevens, P. and Molina, L. T.:
 779 Impacts of HONO sources on the photochemistry in Mexico City during the MCMA-
 780 2006/MILAGO Campaign, *Atmos. Chem. Phys.*, 10(14), 6551–6567, doi:10.5194/acp-10-
 781 6551-2010, 2010.

782 Li, G., Zavala, M., Lei, W., Tsimpidi, A. P., Karydis, V. A., Pandis, S. N., Canagaratna, M.
 783 R. and Molina, L. T.: Simulations of organic aerosol concentrations in Mexico City using the
 784 WRF-CHEM model during the MCMA-2006/MILAGRO campaign, *Atmos. Chem. Phys.*,
 785 11(8), 3789–3809, doi:10.5194/acp-11-3789-2011, 2011b.

786 Li, M., Liu, H., Geng, G., Hong, C., Liu, F., Song, Y., Tong, D., Zheng, B., Cui, H., Man, H.,
 787 Zhang, Q. and He, K.: Anthropogenic emission inventories in China: A review, *National*
 788 *Science Review*, 4(6), 834–866, doi:10.1093/nsr/nwx150, 2017a.

789 Li, M., Zhang, Q., Kurokawa, J.-I., Woo, J.-H., He, K., Lu, Z., Ohara, T., Song, Y., Streets,
 790 D. G., Carmichael, G. R., Cheng, Y., Hong, C., Huo, H., Jiang, X., Kang, S., Liu, F., Su, H.
 791 and Zheng, B.: MIX: a mosaic Asian anthropogenic emission inventory under the
 792 international collaboration framework of the MICS-Asia and HTAP, *Atmos. Chem. Phys.*,
 793 17(2), 935–963, doi:10.5194/acp-17-935-2017, 2017b.

794 Li, T., Zhang, W., Yin, J., Chadwick, D., Norse, D., Lu, Y., Liu, X., Chen, X., Zhang, F.,
 795 Powlson, D. and Dou, Z.: Enhanced-efficiency fertilizers are not a panacea for resolving the
 796 nitrogen problem, *Global Change Biol*, 24(2), e511–e521, doi:10.1111/gcb.13918, 2018.

797 Liu, C., Zheng, X., Zhou, Z., Han, S., Wang, Y., Wang, K., Liang, W., Li, M., Chen, D. and
 798 Yang, Z.: Nitrous oxide and nitric oxide emissions from an irrigated cotton field in Northern

799 China, *Plant Soil*, 332(1), 123–134, doi:10.1007/s11104-009-0278-5, 2010.

800 Liu, L., Wu, J., Liu, S., Li, X., Zhou, J., Feng, T., Qian, Y., Cao, J., Tie, X. and Li, G.:
 801 Effects of organic coating on the nitrate formation by suppressing the N₂O₅ heterogeneous
 802 hydrolysis: a case study during wintertime in Beijing–Tianjin–Hebei (BTH), *Atmos. Chem.*
 803 *Phys.*, 19(12), 8189–8207, doi:10.5194/acp-19-8189-2019, 2019.

804 Liu, X. J., Mosier, A. R., Halvorson, A. D. and Zhang, F. S.: Tillage and Nitrogen
 805 Application Effects on Nitrous and Nitric Oxide Emissions from Irrigated Corn Fields, *Plant*
 806 *Soil*, 276(1), 235–249, doi:10.1007/s11104-005-4894-4, 2005.

807 Liu, X., Ju, X., Zhang, F., Pan, J. and Christie, P.: Nitrogen dynamics and budgets in a winter
 808 wheat–maize cropping system in the North China Plain, *Field Crops Research*, 83(2), 111–
 809 124, doi:10.1016/S0378-4290(03)00068-6, 2003.

810 Liu, X., Zhang, Y., Han, W., Tang, A., Shen, J., Cui, Z., Vitousek, P., Erismann, J. W.,
 811 Goulding, K., Christie, P., Fangmeier, A. and Zhang, F.: Enhanced nitrogen deposition over
 812 China, *Nature*, 494(7438), 459–462, doi:10.1038/nature11917, 2013.

813 Lu, X., Ye, X., Zhou, M., Zhao, Y., Weng, H., Kong, H., Li, K., Gao, M., Zheng, B., Lin, J.,
 814 Zhou, F., Zhang, Q., Wu, D., Zhang, L. and Zhang, Y.: The underappreciated role of
 815 agricultural soil nitrogen oxide emissions in ozone pollution regulation in North China, *Nat.*
 816 *Commun.*, 12(1), 5021–9, doi:10.1038/s41467-021-25147-9, 2021.

817 Ma, R., Yu, K., Xiao, S., Liu, S., Ciais, P. and Zou, J.: Data-driven estimates of fertilizer-
 818 induced soil NH₃, NO and N₂O emissions from croplands in China and their climate change
 819 impacts, *Global Change Biol.*, 28(3), 1008–1022, doi:10.1111/gcb.15975, 2022.

820 Matson, P. A., Naylor, R. and Ortiz-Monasterio, I.: Integration of Environmental,
 821 Agronomic, and Economic Aspects of Fertilizer Management, *Science*, 280(5360), 112–115,
 822 doi:10.1126/science.280.5360.112, 1998.

823 Murray, L. T., Jacob, D. J., Logan, J. A., Hudman, R. C. and Koshak, W. J.: Optimized
 824 regional and interannual variability of lightning in a global chemical transport model
 825 constrained by LIS/OTD satellite data, *J. Geophys. Res. Atmos.*, 117(D), D20307,
 826 doi:10.1029/2012JD017934, 2012.

827 Oikawa, P. Y., Ge, C., Wang, J., Eberwein, J. R., Liang, L. L., Allsman, L. A., Grantz, D. A.
 828 and Jenerette, G. D.: Unusually high soil nitrogen oxide emissions influence air quality in a
 829 high-temperature agricultural region, *Nat. Commun.*, 6(1), 8753–10,
 830 doi:10.1038/ncomms9753, 2015.

831 Potter, P., Ramankutty, N., Bennett, E. M. and Donner, S. D.: Characterizing the Spatial
 832 Patterns of Global Fertilizer Application and Manure Production, *Earth Interactions*, 14, 1–
 833 22, doi:10.1175/2009EI288.1, 2010.

- 834 Qiao, J., Wang, J., Zhao, D., Zhou, W., Schwenke, G., Yan, T. and De Li Liu: Optimizing N
835 fertilizer rates sustained rice yields, improved N use efficiency, and decreased N losses via
836 runoff from rice-wheat cropping systems, *Agriculture, Ecosystems & Environment*, 324,
837 107724, doi:10.1016/j.agee.2021.107724, 2022.
- 838 Seinfeld, J. H. and Pandis, S. N.: *Atmospheric Chemistry and Physics - From Air Pollution to*
839 *Climate Change*, 2nd ed., John Wiley & Sons, New Jersey. 2006.
- 840 Sha, T., Ma, X., Zhang, H., Janecek, N., Wang, Y., Wang, Y., Castro García, L., Jenerette,
841 G. D. and Wang, J.: Impacts of Soil NO_x Emission on O₃ Air Quality in Rural California,
842 *Environ. Sci. Technol.*, 55(10), 7113–7122, doi:10.1021/acs.est.0c06834, 2021.
- 843 Shen, Y., Xiao, Z., Wang, Y., Xiao, W., Yao, L. and Zhou, C.: Impacts of Agricultural Soil
844 NO_x Emissions on O₃ Over Mainland China, *J. Geophys. Res. Atmos.*, 128(4),
845 e2022JD037986, doi:https://doi.org/10.1029/2022JD037986, 2023.
- 846 Sillman, S.: The use of NO_y, H₂O₂, and HNO₃ as indicators for ozone-NO_x-hydrocarbon
847 sensitivity in urban locations, *J. Geophys. Res. Atmos.*, 100(D7), 14175–14188,
848 doi:10.1029/94JD02953, 1995.
- 849 Stehfest, E. and Bouwman, L.: N₂O and NO emission from agricultural fields and soils under
850 natural vegetation: summarizing available measurement data and modeling of global annual
851 emissions, *Nutrient Cycling in Agroecosystems*, 74(3), 207–228, doi:10.1007/s10705-006-
852 9000-7, 2006.
- 853 Steinkamp, J. and Lawrence, M. G.: Improvement and evaluation of simulated global
854 biogenic soil NO emissions in an AC-GCM, *Atmos. Chem. Phys.*, 11(1), 6063–6082,
855 doi:10.5194/acp-11-6063-2011, 2011.
- 856 Sun, X., Ritzema, H., Huang, X., Bai, X. and Hellegers, P.: Assessment of farmers' water and
857 fertilizer practices and perceptions in the North China Plain, *Irrigation and Drainage*, 71(4),
858 980–996, doi:https://doi.org/10.1002/ird.2719, 2022.
- 859 Tan, W., Wang, H., Su, J., Sun, R., He, C., Lu, X., Lin, J., Xue, C., Wang, H., Liu, Y., Liu,
860 L., Zhang, L., Wu, D., Mu, Y. and Fan, S.: Soil Emissions of Reactive Nitrogen Accelerate
861 Summertime Surface Ozone Increases in the North China Plain, *Environ. Sci. Technol.*,
862 57(34), 12782–12793, doi:10.1021/acs.est.3c01823, 2023.
- 863 Tang, K., Qin, M., Fang, W., Duan, J., Meng, F., Ye, K., Zhang, H., Xie, P., Liu, J., Liu, W.,
864 Feng, Y., Huang, Y. and Ni, T.: An automated dynamic chamber system for exchange flux
865 measurement of reactive nitrogen oxides (HONO and NO_x) in farmland ecosystems of the
866 Huaihe River Basin, China, *Sci. Total Environ.*, 745(C), 140867,
867 doi:10.1016/j.scitotenv.2020.140867, 2020.
- 868 Tian, D., Zhang, Y., Mu, Y., Liu, J. and He, K.: Effect of N fertilizer types on N₂O and NO

emissions under drip fertigation from an agricultural field in the North China Plain, *Sci. Total Environ.*, 715(C), 136903, doi:10.1016/j.scitotenv.2020.136903, 2020.

Tubiello, F. N., Salvatore, M., Rossi, S., Ferrara, A., Fitton, N. and Smith, P.: The FAOSTAT database of greenhouse gas emissions from agriculture, *Environ. Res. Lett.*, 8(1), 015009, doi:10.1088/1748-9326/8/1/015009, 2013.

Vinken, G. C. M., Boersma, K. F., Maasakkers, J. D., Adon, M. and Martin, R. V.: Worldwide biogenic soil NO_x emissions inferred from OMI NO₂ observations, *Atmos. Chem. Phys.*, 14(1), 10363–10381, doi:10.5194/acp-14-10363-2014, 2014.

Vitousek, P. M., Naylor, R., Crews, T., David, M. B., Drinkwater, L. E., Holland, E., Johnes, P. J., Katzenberger, J., Martinelli, L. A., Matson, P. A., Nziguheba, G., Ojima, D., Palm, C. A., Robertson, G. P., Sanchez, P. A., Townsend, A. R. and Zhang, F. S.: Nutrient Imbalances in Agricultural Development, *Science*, 324, 1519–1520, doi:10.1126/science.1170261, 2009.

Wang, R., Bei, N., Wu, J., Li, X., Liu, S., Yu, J., Jiang, Q., Tie, X. and Li, G.: Cropland nitrogen dioxide emissions and effects on the ozone pollution in the North China plain, *Environ. Pollut.*, 294, 118617, doi:10.1016/j.envpol.2021.118617, 2022a.

Wang, Y., Yao, Z., Zheng, X., Subramaniam, L. and Butterbach-Bahl, K.: A synthesis of nitric oxide emissions across global fertilized croplands from crop-specific emission factors, *Global Change Biol*, 28(14), 4395–4408, doi:10.1111/gcb.16193, 2022b.

Wen, L., Xue, L., Wang, X., Xu, C., Chen, T., Yang, L., Wang, T., Zhang, Q. and Wang, W.: Summertime fine particulate nitrate pollution in the North China Plain: increasing trends, formation mechanisms and implications for control policy, *Atmos. Chem. Phys.*, 18(15), 11261–11275, doi:10.5194/acp-18-11261-2018, 2018.

Wu, J., Bei, N., Hu, B., Liu, S., Wang, Y., Shen, Z., Li, X., Liu, L., Wang, R., Liu, Z., Cao, J., Tie, X., Molina, L. T. and Li, G.: Aerosol–photolysis interaction reduces particulate matter during wintertime haze events, *Proc. Natl. Acad. Sci. U.S.A.*, 117(18), 9755–9761, doi:10.1073/pnas.1916775117, 2020.

YAN, X., JIN, J.-Y., He, P. and LIANG, M.-Z.: Recent Advances on the Technologies to Increase Fertilizer Use Efficiency, *Agricultural Sciences in China*, 7(4), 469–479, doi:10.1016/S1671-2927(08)60091-7, 2008.

Yan, X., Ohara, T. and Akimoto, H.: Statistical modeling of global soil NO_x emissions, *Global Biogeochem. Cycles*, 19(3), GB3019, doi:10.1029/2004GB002276, 2005.

Yienger, J. J. and Levy, H.: Empirical model of global soil-biogenic NO_x emissions, *J. Geophys. Res. Atmos.*, 100(D), 11447–11464, doi:10.1029/95JD00370, 1995.

Yu, Z., Liu, J. and Kattel, G.: Historical nitrogen fertilizer use in China from 1952 to 2018,

903 Earth System Science Data, 14(11), 5179–5194, doi:10.5194/essd-14-5179-2022, 2022.

904 Zhang, Q., Streets, D. G. and Carmichael, G. R.: Asian emissions in 2006 for the NASA
 905 INTEx-B mission, *Atmos. Chem. Phys.*, 9(14), 5131–5153, doi:10.5194/acp-9-5131-2009,
 906 2009.

907 Zhang, R., Wang, G., Guo, S., Zamora, M. L., Ying, Q., Lin, Y., Wang, W., Hu, M. and
 908 Wang, Y.: Formation of urban fine particulate matter, *Chem. Rev.*, 115(10), 3803–3855,
 909 doi:10.1021/acs.chemrev.5b00067, 2015.

910 Zhang, Y., Liu, J., Mu, Y., Pei, S., Lun, X. and Chai, F.: Emissions of nitrous oxide, nitrogen
 911 oxides and ammonia from a maize field in the North China Plain, *Atmos. Environ.*, 45(17),
 912 2956–2961, doi:10.1016/j.atmosenv.2010.10.052, 2011.

913 Zhao, M., Tian, Y., Zhang, M., Yao, Y., Ao, Y., Bin Yin and Zhu, Z.: Nonlinear response of
 914 nitric oxide emissions to a nitrogen application gradient: A case study during the wheat
 915 season in a Chinese rice-wheat rotation system, *Atmos. Environ.*, 102(C), 200–208,
 916 doi:10.1016/j.atmosenv.2014.11.052, 2015.

917 Zhao, R.-F., Chen, X.-P., Zhang, F.-S., Zhang, H., Schroder, J. and Römheld, V.:
 918 Fertilization and Nitrogen Balance in a Wheat–Maize Rotation System in North China,
 919 *Agron. J.*, 98(4), 938–945, doi:10.2134/agronj2005.0157, 2006.

920 Zheng, Y., Ji, J. and Liu, S.: Effect of topdressing time on spring maize yield and nitrogen
 921 utilization in black soil of northeast China, *Sci. Rep.*, 13(1), 11841–10, doi:10.1038/s41598-
 922 023-38724-3, 2023.

923 Zhu, Z., Stewart, B. A. and Fu, X.: Double cropping wheat and corn in a sub-humid region of
 924 China, *Field Crops Research*, 36(3), 175–183, doi:10.1016/0378-4290(94)90109-0, 1994.

925