Enhanced Soil Emissions of Reactive Nitrogen Gases by Fertilization and Their Impacts on Secondary Air Pollution in Eastern China

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ABSTRACT. Nitrogen fertilizer application is accompanied by intense release of multiple reactive nitrogen (Nr) gases like nitrous acid (HONO), ammonia (NH_3), and nitric oxide (NO) from the soil,

influencing atmospheric chemistry and air pollution. In current emission inventories, post-fertilization soil emissions are poorly characterized due to inaccurate identification of fertilization timing and location. Moreover, pre-existing studies predominantly focus on individual Nr gases, and a comprehensive understanding of simultaneously emitted Nr gases from fertilization and their impacts on air quality is still limited. Here, we developed a novel method to identify the dryland fertilization activity based on satellite and reanalysis datasets. Then, we updated a dynamic soil Nr emissions model (WRF-SoilN-Chem) with lab-derived parameterization and applied it to analyze the time- and space-varying Nr emissions and their effects on air quality. It is estimated that the Nr emissions from a typical fertilization event in the Yangtze River Delta (YRD) region increased ozone (O_3) and nitrate concentrations by 2.5% and 18.2%, respectively. HONO and NH_3 emissions jointly enhanced nitrate via chemical production and gas-particle partitioning. An accurate representation of fertilization and meteorology-emission-chemistry coupled modeling would greatly improve the understanding of the soil Nr emissions and their impacts on regional air pollution.

SYNOPSIS

Our research reveals that the surging emissions of reactive nitrogen after fertilization exacerbate air pollution by contributing to O_3 and $PM_{2.5}$ formation.

1. INTRODUCTION

Agricultural soil emissions are important sources of atmospheric reactive nitrogen (Nr) gases[1–3], including ammonia (NH_3), nitrous acid (HONO) and nitric oxide (NO), which are produced through various microbial and chemical processes. Nitrogen fertilizer input significantly enhances Nr gases emissions by increasing soil nitrogen content for microbial activities[4–6]. Field measurements and lab

experiments[7–9] show that the Nr gases emissions peak 2–4 days later after fertilization with levels 5–100 times higher than in non-fertilization situations. Such intense Nr emissions, especially NH_3, HONO and NO, collectively worsen air quality by contributing to the formation of tropospheric ozone (O_3) and particulate matter (PM) via different chemical pathways[10–16].

A variety of soil Nr emissions models[17–19], including WRF-CMAQ-EPIC[20] (NH_3), WRF-SoilN-Chem[21] (NH_3), Yienger and Levy (YL)[22] (NO), Model of Emissions of Gases and Aerosols from Nature (MEGAN)[23] (NO) and Berkley-Dalhousie Iowa Soil NO Parameterization (BDISNP)[24] (NO) have been developed and applied to investigate the role of soil Nr emissions on air quality. Sha et al.[25] reveal the impacts of soil NO emissions on O_3 in rural California with the BDISNP model. Wang et al.[26] demonstrate that soil HONO emissions aggravate O_3 and nitrate pollution in northern China by WRF-CMAQ coupled with HONO parameterization. Pleim et al.[20] evaluate the NH_3 emitted from fertilized croplands and grasslands in North America with the WRF-CMAQ-EPIC model. These models are effective in the assessment of individual Nr gases species. However, in real agricultural fields, multiple gases are released concurrently after fertilization. A recent study assesses the impact of soil HONO and NO emissions on O_3 and nitrate in a non-fertilization month[27]. So far, there has been little discussion on the simultaneous Nr gases emissions from fertilization and their collective impacts on air quality.

Another challenge in fertilization relevant studies arises from the coarse or inaccurate fertilization spatiotemporal information, which is attributable to the irregularity of farmlands and diverse cultivation practices of smallholder farmers in China[28,29]. As Nr surging emissions appear 2–4 days after fertilization, exact dates and location of fertilization are critical for soil emissions analysis. The pre-existing studies typically obtain fertilization information from crop calendars or surveys of local farmers[30–32]. However, crop calendars with a temporal resolution of 1–2 months are insufficiently

detailed to accurately capture surging soil emissions lasting only 1–2 weeks. Furthermore, the questionnaire method is labor-intensive and time-consuming, particularly in large agricultural regions like the North China Plain (NCP) and the Yangtze River Delta (YRD).

In this work, we aim to clarify the impact of fertilization Nr emissions on regional air quality using a novel approach to identify daily dryland fertilization and an updated emissions model incorporating dynamic soil NO, HONO and NH_3 parameterization. We chose the YRD, which has extensive and fertile croplands, as a case study region to apply this new approach and model for analysis. The results indicate that the intense Nr emissions during a typical fertilization activity noticeably enhance ambient oxidizing capacity and promote nitrate formation. This work advances understanding of the effects of fertilization-induced Nr emissions on air pollution, emphasizing the need for considering agricultural emissions in future air quality strategies.

MATERIALS AND METHODS

2.1 Identification method of fertilization events

In light of the limitation concerning fertilization spatiotemporal information, we propose a novel method for identifying fertilization events in drylands based on multiple datasets, including the crop calendar, land use, satellite soil moisture measurements, and reanalysis data of soil moisture. This methodology is mainly based on two key features in crop cultivation practices.

Firstly, fertilization and irrigation are typically carried out concurrently during the sowing season. In modern cultivation practices, farmers usually drill the fertilizer together with seed to optimize labor efficiency and subsequently irrigate the fertilized soil immediately (within approximately 30 mins)[33,34] to facilitate the fertilizer hydrolysis and maintain soil moisture for seed germination[35]. In regions with higher mechanization level like the NCP or YRD, farmers adopt the Water-Fertilization

Integration (WFI) technique that mixes fertilizer with water and delivers the mixture to soil through a pipe or pressure system to supply nutrients directly to the crop[36]. Such synchrony between the two processes makes it reasonable to identify fertilization events through irrigation occurrence.

Secondly, the disparity of soil moisture dry-wet transition trends between satellite and reanalysis data can serve as an indicator of irrigation activities. Satellite directly detect soil brightness temperature and calculate soil moisture[37,38]. Thus, both natural rainfall and irrigation-induced soil wetting can be detected by the satellite measurements[39]. However, unlike direct measurements by satellite, reanalysis data is generated by combining historical observations with model outputs using data assimilation techniques. This process assimilates precipitation but does not incorporate human activities like irrigation; hence, reanalysis data cannot represent irrigation-induced soil wetting[40,41]. Therefore, a daily comparison between satellite observations and reanalysis data on different areas can identify the timing and location of irrigation. The concurrency of fertilization and irrigation makes it possible to derive the spatiotemporal distribution of fertilization with a spatial resolution of 25 km, following the soil moisture data resolution.

This identification method is suitable for most dryland crops but has limitations on wetland crops such as rice, taro and other semi-aquatic crops, where the reliability of the satellite retrievals is far lower than those in drylands[42]. Though this method is applicable to basal fertilization during the planting season, it may not be feasible during the growing season when irrigation and fertilization can be conducted separately. As dryland constitutes the majority of China's cropland (i.e., 75.45%)[43], and the basal fertilization accounts for 63% of the total fertilizer consumption for dryland crops[44], this method is thus applicable under most cultivation situations in China.

2.2 Implementation of soil HONO and NO emissions in WRF-SoilN-Chem

WRF-SoilN-Chem is a dynamic soil Nr emissions model coupled with meteorology–chemistry transport model WRF-Chem, capable of numerically describing agriculture Nr emissions rate interactively with time- and spatial-varying meteorological and soil conditions. In the original version WRF-SoilN-Chem, dynamic fertilization NH_3 emission flux is obtained by multiplying static basic emissions (Basic E_{NH3}) with dynamic emission correction factors (CF), as shown in Eq (1) and Eq (2). The descriptions of the model parameterization and evaluation are detailed in Ren et al.[21]

$$Flux_{NH3-fertilizer} = Basic\ E(NH_3)_{fertilizer} \times CF_{wind} \times CF_{soil_T} \times CF_{soil_m} \times CF_{rain} \quad (1)$$

$$EF_{static_{fertilizer}} = EF_{0i} \times CF_{pH} \times CF_{method} \times CF_{rate} \times CF_{time} \quad (2)$$

Where $Flux_{NH3-fertilizer}$ (mol km^{-2} h^{-1}) is the dynamic NH_3 emission flux from fertilization. $Basic\ E(NH_3)_{fertilizer}$ (mol km^{-2} h^{-1}) is the basic NH_3 emission flux from fertilization, which is calculated from nitrogen fertilizer rate multiply by static emission factor ($EF_{static_{fertilizer}}$) at a specific condition. CF_{wind}, CF_{soilT}, CF_{soilm} and CF_{rain} are the dynamic meteorological correction factors (CF) for wind speed, soil temperature, soil moisture content and rainfall, respectively. EF0i is the reference emission factor for different nitrogen fertilizers. CF_{pH} is the correction factor for different soil acidity; CF_{method} is for fertilization application method, including basal dressing and top dressing; CF_{rate} is for different application rates, and CF_{time} is for time since fertilizer application, which is used to reflect the variation characters of NH_3 emission after fertilization. The detailed parameterizations of these CF are described in the Table S1.

In this work, we updated the model by incorporating additional soil HONO and NO emissions based on Oswald et al.,[45] Zhang et al.[46] and Wang et al.[26] Given the substantial differences in Nr gases emissions before and after fertilization, we classified agricultural soil emissions into two states:

the post-fertilization state and the background state. Since Nr emissions usually peak 2–4 days after fertilization and return to non-fertilization level within 10 days, the post-fertilization state is defined as "within 10 days after fertilizer application". The background state is defined as "before fertilization" or "10 days later after fertilization" time, when Nr emissions are at a low level.

For HONO emission in the background state, Oswald et al.[45] estimated HONO optimum emission ($F_{N,\ opt}$) from different soil samples like forest, pasture, grassland, wheat and cotton fields. The optimum soil HONO flux is scaled by the function of soil temperature (T) and soil water content (SWC), based on laboratory experiments, the parameterization is Eq (3):

$$F_{SHONO} = F_{N,opt} \times h(T) \times g(SWC) \quad (3)$$

where $F_{N,opt}$ is the optimum flux of HONO in terms of N mass (ng N m^{-2} s^{-1}). *h(T)* is a dimensionless function of HONO emissions with respect to soil T, and *g(SWC)* is the function of HONO emissions on *SWC* (% water-holding capacity (WHC)).

For HONO emissions in the post-fertilization state, we utilized the fertilization parameterization of Wang et al.[26] to represent the enhanced HONO emissions after fertilization within 10 days. The post-fertilization HONO emissions (F_{SHONO}) are predicted as a function of soil temperature and SWC in Eq (4). The detailed equation method and parameters can be found in Wang et al.:[26]

$$F_{SHONO} = F_{emis(HONO)}(Dry\ peak) + F_{emis(HONO)}(wet\ peak) = [F_{N,max_{Dry}} \cdot exp\left(-\frac{\left(SWC-SWC_{C_{Dry}}\right)^2}{{w_{Dry}}^2}\right) + F_{N,max_{Wet}} \cdot exp\left(-\frac{\left(SWC-SWC_{C_{Wet}}\right)^2}{{w_{Wet}}^2}\right)] \times \exp\left[\left(-\frac{43900}{R}\right) \cdot \left(\frac{1}{T}-\frac{1}{298}\right)\right] \quad (4)$$

$F_{emis}(Dry\ peak)$ is the fitted Gaussian function in the low SWC range (0-60% WHC) and $F_{emis}(wet\ peak)$ is in the high SWC range (60-100% WHC), $F_{N,max}$ is the maximum HONO flux at the optimum SWC ($SWC_{Cwet/dry}$) at the reference temperature 298 K, *w* characterizes the width of

the curves. The *dry* and *wet* subscripts represent the parameters on dry peak and wet peak, respectively. R is the gas constant (8.314 J mol^{-1} K^{-1}).

Given that the default WRF-Chem model includes only one pathway for HONO formation via hydroxyl radical (OH) and NO, we integrated additional sources, including vehicle emissions and secondary pathways (Table S2, S3), to enhance the accuracy of HONO concentration estimation and assess the contributions of post-fertilization to HONO.

For NO, the background and post-fertilization soil NO emission schemes are from Wang et al.[27] and Wang et al.[26] respectively. In these schemes, the NO flux is calculated based on the maximum NO flux under optimum condition ($F_{N,max}$(NO)), and further adjusted by soil water content and soil temperature, similar to the approach used for HONO. The NO parameterization equation (Eq (5)) follows the same form as the HONO equation. The difference lies in the value of parameters like $F_{N,max_{Dry}}$, $F_{N,max_{Wet}}$, $SWC_{C_{Wet}}$, $SWC_{C_{Dry}}$, w_{Wet}, and w_{Dry}, which are derived from the curve fitting of HONO and NO flux data. The specific values of these parameters are provided in Table S4.

$$F_{NO} = F_{emis(NO)}(Dry\ peak) + F_{emis(NO)}(wet\ peak) = [F_{N,max_{Dry}} \cdot exp\left(-\frac{\left(SWC-SWC_{C_{Dry}}\right)^2}{{w_{Dry}}^2}\right) + F_{N,max_{Wet}} \cdot exp\left(-\frac{\left(SWC-SWC_{C_{Wet}}\right)^2}{{w_{Wet}}^2}\right)] \times \exp\left[\left(-\frac{43900}{R}\right)\cdot\left(\frac{1}{T}-\frac{1}{298}\right)\right] \quad (5)$$

Since the measured HONO and NO emissions may include contributions from each other due to the oxidation of NO to HONO, as well as the rapid photolysis of HONO to NO, Wang et al.[26] placed the entire sampling setup inside dark cabinet and dried the soil with purified air throughout the experiment to minimize the transformation between HONO and NO and obtain more accurate measurements. Given the diagnosed model includes individual chemical reaction processes, such

171 explicit HONO and NO soil emissions in the atmospheric chemistry model are essential for
172 providing detailed diagnosis and quantification of the impacts on air quality.

173 **2.3 Model configuration and simulation design**

174 To evaluate the model performance of the updated WRF-SoilN-Chem and the air quality response
175 to soil Nr emissions, we designed six parallel experiments: Base, HONO, NH_3, NO, NH_3+HONO,
176 NH_3+HONO+NO. The Base experiment applied the anthropogenic emission inventory for 2020
177 from the Multi-resolution Emission Inventory for China (MEIC)[47,48] but excluded the fertilization
178 and soil emissions of all Nr gases. Other sensitivity experiments were conducted by introducing
179 additional soil Nr emissions into Base experiment, as detailed in Table 1. We adopt fertilizer
180 application of 100 kg N ha^{-1} during the identified basal fertilization stage, following the fertilization
181 rate statistics in the YRD provided by Zhang et al.[49] To investigate fertilization Nr emission and its
182 impact on air quality during the whole time period of the identified fertilization event (see Results
183 3.2), simulations are conducted from 20 Sep to 10 Oct 2021, with first 10 days as a spin-up time.
184 The model domain covers eastern China (18° N–50° N, 95° E–131° E) with a 20 km grid resolution.
185 Other key settings are detailed in Text S1 and Table S5. The simulations of key surface
186 meteorological parameters and pollutants are validated by available observations in Figure S1–S3.
187 The normalized mean biases of temperatures, RH and wind speed in Nanjing are −3.6%, 1.7%, and
188 35.9%, respectively. The simulation well captures the spatial pattern and pollution level of $PM_{2.5}$,
189 CO, O_3, NO_2, nitrate and ammonium concentrations measured by surface stations in the YRD
190 (Figure S2–S3).

191 As mentioned in Section 2.1, reanalysis data could not accurately simulate irrigation-induced soil
192 wetting. Since soil moisture directly affects soil emissions simulation, we used satellite-detected

moisture to drive the simulation. Based on the method from Jared et al.,[50] we assimilated the satellite soil moisture retrievals at 00 UTC in all experiments.

Table 1. Simulation experiment design considering different sources of soil Nr emissions

Experiments	Additional fertilization emissions considered		
	Soil HONO	Soil NH_3	Soil NO
Base	×	×	×
HONO	√	×	×
NH_3	×	√	×
NO	×	×	√
NH_3+HONO	√	√	×
NH_3+HONO+NO	√	√	√

2.4 Identification of potential air mass source regions

The Lagrangian particle dispersion modeling (LPDM) was conducted using the Hybrid Single-Particle Lagrangian Integrated Trajectory model (HYSPLIT) driven by Global Data Assimilation System (GDAS) to identify the potential sources of air masses during the studied pollution case[51]. The LPDM was run in the 24-hour backward mode, with 3,000 particles released every 2 hours over the Nanjing station. Particle residence times below 100 m were used to identify the "footprint" of air masses arriving at the Nanjing station. The spatiotemporal distributions of these particles were used to identify potential source regions and their relative contributions to air masses at Nanjing station.

2.5 Datasets

2.5.1 In situ measurements

Hourly concentrations of NH_3, HONO, O_3, nitrogen dioxide (NO_2), carbon monoxide (CO), sulfur dioxide (SO_2), and $PM_{2.5}$ were measured at the Station for Observing Regional Processes of the Earth System (SORPES, 32°07'14' N, 118°57'10' E; ~40 m a.s.l.) in Nanjing, Jiangsu province. NH_3 concentrations were measured with a Picarro G2103 gas analyzer (Picarro Inc., USA) that

employs cavity ring-down spectroscopy (CRDS) technique[52]. HONO concentrations were measured using a commercial long-path absorption photometer instrument (LOPAP)[53]. NO_x, O_3, and CO were measured by Thermo Fisher analyzers (TEI 42i, 49i, and 48i, USA, respectively). The $PM_{2.5}$ concentrations were detected using a mass analyzer (Thermo SHARP-5030). Furthermore, aerosol components, including sulfate (SO_4^{2-}), nitrate (NO_3^-) and ammonium (NH_4^+) were measured by a Monitor for Aerosols and Gases in ambient Air (MARGA).

In addition to the data from SORPES, hourly $PM_{2.5}$, SO_2, NO_2, O_3, and CO data from other YRD cities were obtained from China's Air Quality Monitoring Network[54]. The meteorological observations, including wind, temperature, precipitation and relative humidity, were obtained from the National Climatic Data Center's (NCDC) Integrated Surface Database (ISD).

2.5.2 Soil moisture datasets

Soil moisture observations from the satellite were obtained using the Advanced Microwave Scanning Radiometer–Earth Observing System and version 2 (AMSR–E/AMSR2) instruments. Previous assessments have shown that the AMSR instruments are highly sensitive to changes in soil moisture, whether they're caused by rainfall or irrigation[39]. This sensitivity is reflected in the strong correlation between satellite and aircraft soil moisture measurements, both spatially (average R = 0.92 and RMSD = 0.049 $m^3\ m^{-3}$) and over time (R = 0.94 and RMSD = 0.04 $m^3\ m^{-3}$)[55]. The data provided by this product covers a 25×25 km^2 area and is updated daily, making it a valuable resource for monitoring soil conditions.

The hourly soil moisture reanalysis data used in this study is from ECMWF Reanalysis v5 (ERA5)[56]. ERA5 assimilates precipitation data from the Tropical Rainfall Measuring Mission Microwave Imager (TRMM–MI), and demonstrated outstanding soil moisture dynamic properties in past evaluations[57].

2.5.3 Crop planting datasets

The planting day of crops in YRD region is obtained from the Global Gridded Crop Model Intercomparison (GGCMI) Phase 3 dataset[58]. It provides in each 0.5° land grid cell the planting day and maturity day for 18 different crops, separating rainfed and irrigated systems. To represent the crop planting intensity in different seasons, we calculate the crop planting ratio in each month as follows:

$$Planting\ ratio_{(m)}(\%) = \frac{F_{(m)}}{\sum_{m=1}^{12} F_{(m)}} = \frac{\sum_i \sum_j \sum_c N_{(c,i,j,m)}}{\sum_m \sum_i \sum_j \sum_c N_{(c,i,j,m)}} \quad (6)$$

where *c* represents the crop type, *m* denotes the month in a year, and *i* and *j* are the location indices of the data grid. *N* indicates the planting event, where *N=1* means a planting event occurs, and *N=0* otherwise. $F_{(m)}$ is the frequency of planting events in each month, calculated by summing up the planting events for each crop type, each grid, and each day within the month.

3. RESULTS AND DISCUSSION

3.1 Enhanced HONO and NH_3 concentration during the fertilization period

The YRD region, covering approximately 126,505 km^2 of cultivated land (9.3% of the national cultivated land area), features the most fertile soil in China[59] and is highly suitable conditions for crop farming. Rice, wheat and rapeseed are the main economic crops grown in the YRD region[60]. Rice is the staple crop distributed across most areas of the YRD region, whereas winter wheat and rapeseed are mainly cultivated in northern (Figure 1a) and eastern Jiangsu province (Figure 1b), respectively. According to the crop calendar, early rice and spring corn are sown from March to April, and winter wheat and rapeseed are predominantly planted in October. Therefore, the crop planting seasons in the YRD are mainly concentrated in March, April and October (Figure 1e).

257 Intensive agricultural crop cultivation makes the YRD a hot spot region for agricultural Nr gases
258 emissions[61]. Persistently extensive planting activities accompanied with fertilization lead to
259 substantial locally emitted Nr pollutants, and thus ambient Nr concentrations and crop planting ratio
260 exhibit similar peaking time in the planting season. Multiyear observations (2017–2021) from
261 Nanjing SORPES station showed that the average HONO, the maximum NH_3 concentration and the
262 crop planting ratio peaked in January, March and October (Figure 1c, 1e). The Pearson correlation
263 coefficient between Nr concentrations and planting ratio is 0.58. Besides Nanjing, other rural sites
264 (for example, Fengyang in Anhui province and Fenghua in Zhejiang province) in the YRD exhibited
265 a similar phenomenon of higher Nr concentrations in the sowing season (Figure 1d). As clearly
266 demonstrated in Figure 1d and Figure S4, the substantial enhancement of both ground and satellite
267 observed NH_3 in October (the sowing month) than in September (the warmer month), indicates a
268 more important role of fertilization than warmer temperature in Nr emissions.

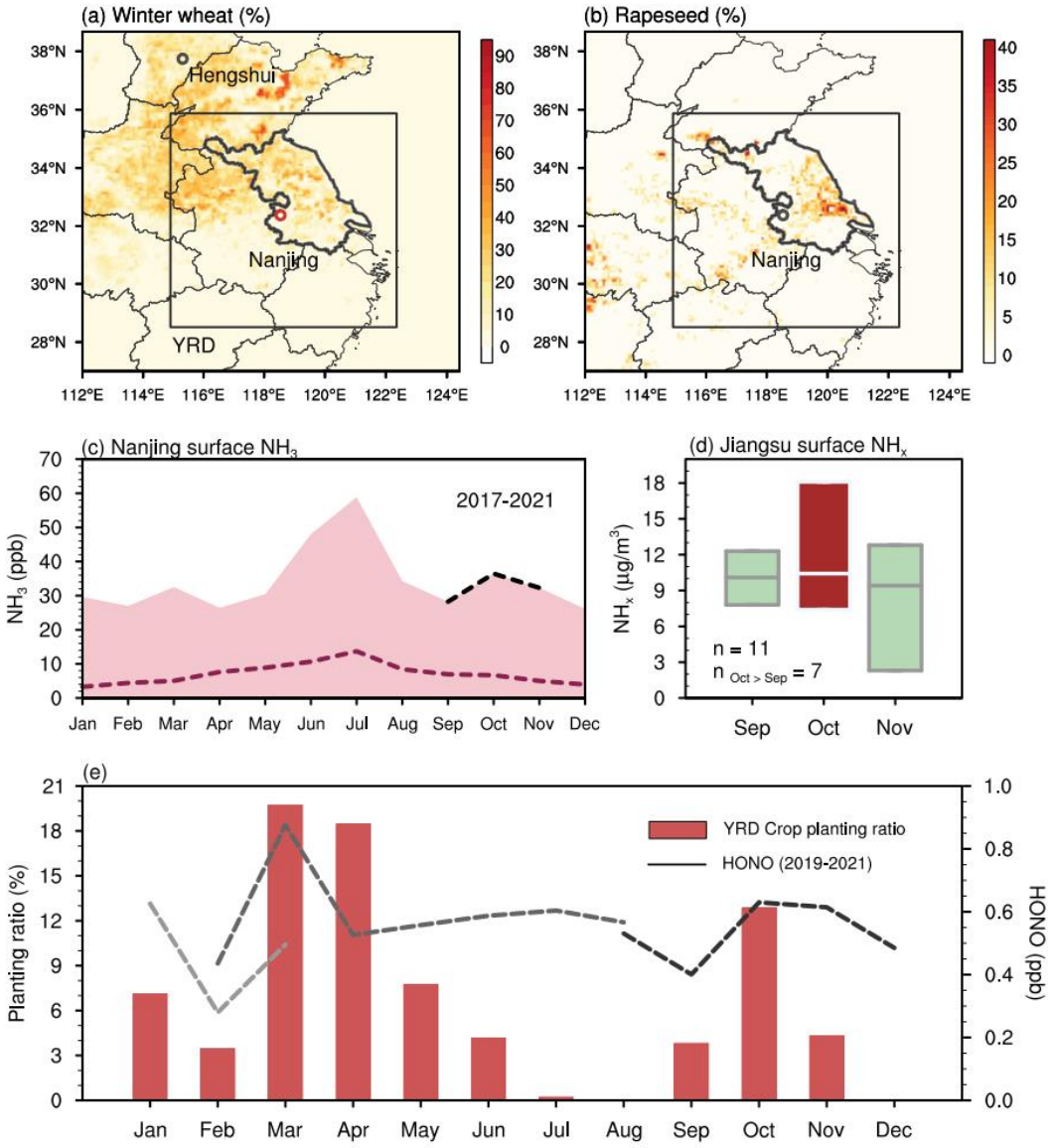

Figure 1. Observational evidence of fertilization effects on Nr concentrations. Percentage of (a) winter wheat and (b) rapeseed coverage[62] at 20 m resolution in the YRD region (black box), and the black thick line outlines the Jiangsu province. (c) Averaged NH_3 concentration in Nanjing SORPES site during 2017–2021, the upper and lower edge of the filled area is 99% and 1%. (d) The mean surface NH_x (NH_3(gas) + NH_4^+(aerosol)) concentration of the YRD region sites from 2011–2015 according to the nationwide nitrogen deposition monitoring network (NNDMN) dataset[63], n is the count of valid value. (e) The YRD seasonal crop planting ratio and HONO concentration in Nanjing.

3.2 Post-fertilization soil Nr emissions and their regional transport

By evaluating the method in Section 2.1 and applying it to identify a typical fertilization event in the YRD, we can investigate the post-fertilization effects on air quality. Previous field measurements of post-fertilization fluxes in the YRD region have supported our identification method with observations of the simultaneous increase in Nr emissions and soil moisture following fertilization and irrigation during the sowing period[64,65]. The high-resolution field measurements conducted by our research group in a crop field in Hengshui city (Hebei province of China) provides another opportunity to verify our identification method. As shown in Fig. 2a, during the corn planting season (April 2012), as farmers applied fertilizer and irrigated the soil, both the NH_3 emission flux and soil moisture simultaneously increased by 4–6 times on April 12th. The soil moisture from satellite observations and reanalysis data showed different dry-wet transitions by the irrigation. Specifically, the soil moisture from the SAC-D (Satélite de Aplicaciones Científicas) satellite effectively captures the dry-wet transition caused by irrigation and rainfall events on April 12th and April 24th, respectively. However, the Global Land Evaporation Amsterdam Model (GLEAM) and ERA5, which only assimilate precipitation, only indicate soil moisture increase due to rainfall on April 24th. In short, the combination of the satellite and reanalysis data is able to distinguish the fertilization event.

Considering the noisy performance of satellite data on waterlogged paddy fields, we examined the bare drylands sown with wheat and rapeseed in October in the YRD region. Figure 2b shows different temporal trends between ERA5 reanalysis and the AMSR satellite datasets in early October. The satellite-observed soil moisture increased continuously from October 1st to 5th, while the ERA5 value remained stable. The Integrated Multi-satellitE Retrievals for Global Precipitation Measurement (IMERG) (Figure S5) shows no precipitation during this period, suggesting that the

soil wetting was dominantly caused by irrigation. As displayed in Figure 2c, the soil wetting was most pronounced in eastern and northern Jiangsu province, overlapping with the wheat and rapeseed cultivation areas, further confirming the occurrence of irrigation and fertilization. Besides this observational evidence, the official guidelines from the Ministry of Agriculture and Rural Affairs recommend early October (Oct 1st–10th) as the optimum time for sowing and fertilizing winter wheat and rapeseed in the "Huaihe River Basin" and "Yangtze-Huaihe region"[66] in 2021. The above analyses confirm the occurrence of fertilizer application from October 1st to 5th, and support the feasibility of the proposed method for determining the timing and location of fertilization.

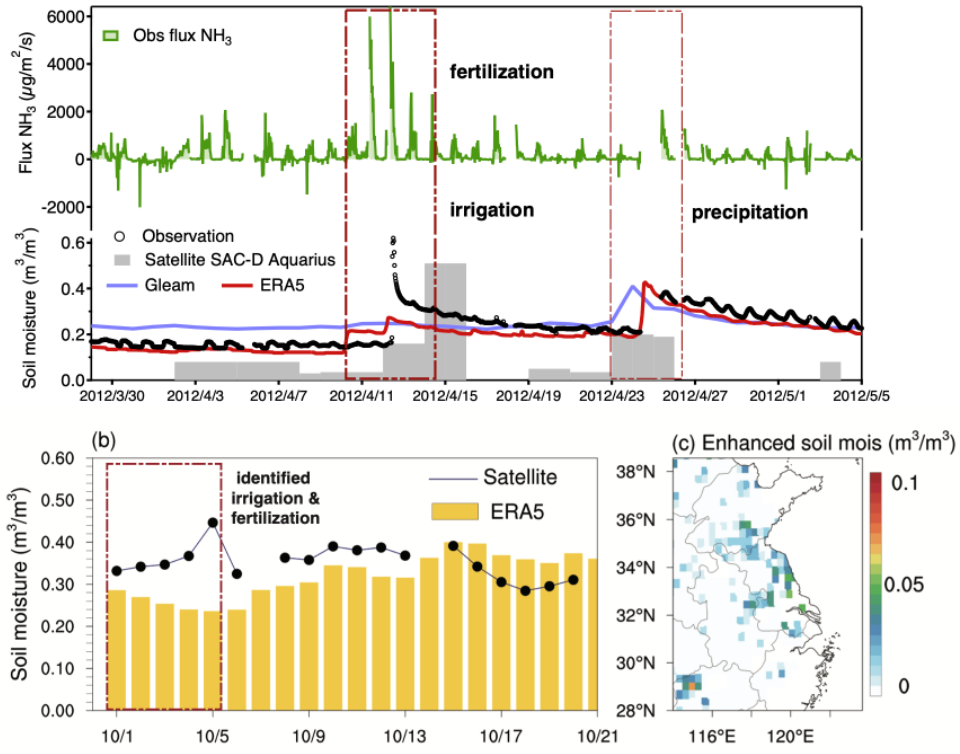

Figure 2. Identification of the timing and location of a fertilization and irrigation event. (a) Time series of the observed ammonia flux and moisture in Hengshui farmland, Hebei province of China. The grey bar refers to the soil moisture from satellite SAC-D Aquarius, and the purple and red lines

313 are the GLEAM and ERA5 reanalysis datasets, respectively. The black circle markers indicate the
314 field measurements. (b) Daily satellite-detected and ERA5 soil moisture on Jiangsu croplands in
315 October 2021, and the bounding red box represents the identified irritation and fertilization dates.
316 (c) The enhanced soil moisture from Oct 1st to Oct 5th 2021.

317 An air pollution episode occurring 2 days after the extensive fertilization showed a plausible
318 connection with fertilization-related nitrogen emissions. As shown in Fig. 3a, on the morning of Oct
319 7th, high levels of HONO, NH_3, sulfate, nitrate and ammonium (SNA) with the prevailing northerly
320 wind were recorded in Nanjing SORPES station. The pollution episode lasted for 1 day with
321 maximum NH_3 reaching up to 30 ppb, which was 5 times higher than clean state. The satellite
322 retrieval also demonstrated the NH_3 hotspot in Jiangsu province on Oct 7th (Figure S6). As
323 agricultural activity is the major source of NH_3, the sudden rise in pollutant concentrations
324 accompanied by changing wind direction indicated that both agricultural emissions and regional
325 transport are probable causes of this pollution event.

326 The Goddard Earth Observing System (GEOS) model with assimilation showed that the nitrate
327 and sulfate pollution was partly transported from North and South Korea to Nanjing along with the
328 northerly wind (Figure S7). The base experiment without post-fertilization was able to reproduce
329 the rise of nitrate concentration caused by such regional transport, but still obviously underestimated
330 it (Figure S8), indicating the missing of potential source. Source attribution based on LPDM showed
331 that the air mass on the polluted (Oct 7th) day originated from the ocean and passed through the
332 extensively fertilized rapeseed plots (Figure 3b). Along the transport pathway, the upstream soil-
333 emitted Nr gases were carried to Nanjing, contributing to the observed increase in fine particle
334 pollution, particularly ammonia and associated secondary aerosols.

The lifetime of ammonia is approximately 12 hours[67] and our calculations indicate that such lifetime is sufficient long for NH_3 emitted from the farthest Jiangsu fields to be transported to Nanjing. In comparison, noontime HONO is short-lived (10–20 mins) and originates from various sources, including traffic emissions and heterogeneous reactions. Due to its rapid photolysis, HONO emitted during the daytime from eastern Jiangsu is unlikely to reach Nanjing SORPES via long-range transport. Only emissions from nearby farmlands can contribute to HONO levels observed at SORPES station. However, the HONO emitted from distant farmlands can contribute to the formation of longer-lived pollutants such as O_3 or SNA, which exert broader and prolonged effects[68].

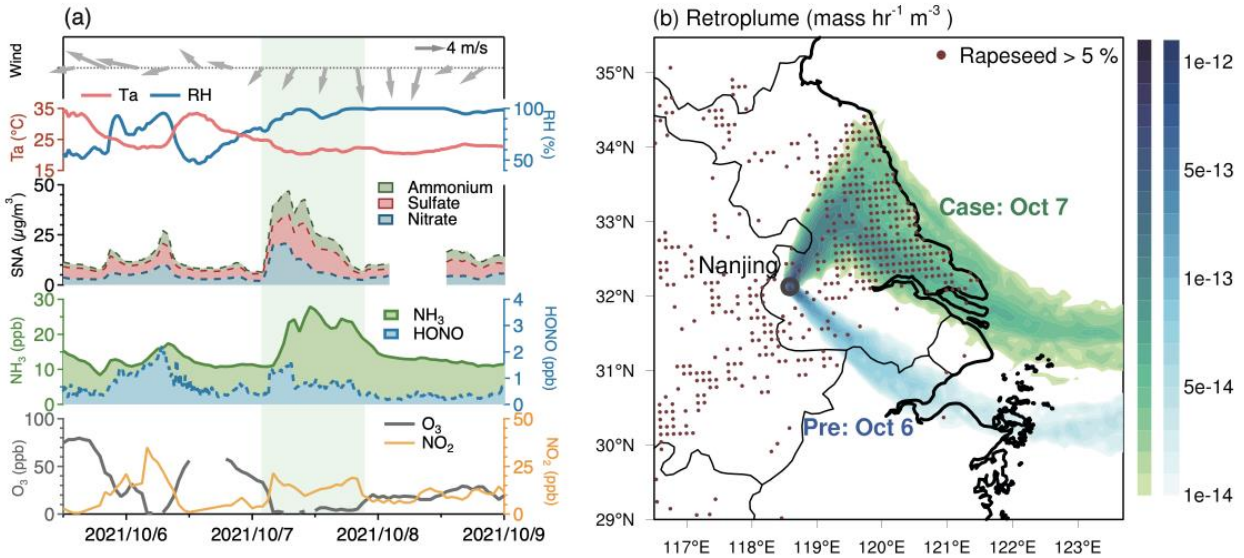

Figure 3. A typical air pollution episode and the potential impact from agricultural sources. (a) Time series of meteorological parameters (air temperature (Ta), relative humidity (RH), wind), the concentrations of NH_3, HONO, O_3, NO_2, and ammonium, sulfate and nitrate at the SORPES station in Nanjing during October 5th–9th 2021. The pollution period is marked by the green box. (b) 24-

hour backward retroplume of Nanjing on clean day Oct 6th (blue) and polluted day Oct 7th (green). The brown dots represent the agricultural fields with more than 5% rapeseed planting.

3.3 Quantifying fertilized soil Nr emissions and their impacts on air pollution

To investigate the impacts of fertilization emissions on air quality, we performed diagnosed simulations with an air quality model incorporating with dynamic soil Nr emissions (WRF-SoilN-Chem, see Methods). In the YRD region, the simulated emissions of NH_3 and HONO (Figure 4a, 4b) and NO (Figure S9) during the post-fertilization period (October 1st–10th) were 28.5, 1.09 and 0.19 mol km^{-2} hr^{-1}, respectively, in good agreement with the Nr flux measured by Zhao et al.[69], Xia et al.[64] and Wang et al[27] in the YRD (Table S6). The spatial patterns of these Nr emissions show notable heterogeneity across different species. For HONO and NO, emissions were concentrated in eastern and northern Jiangsu, where soil moisture was elevated (~30% WHC) after irrigation (Figure S10). For NH_3, high emissions are not only concentrated in eastern and northern Jiangsu but also in the southern Jiangsu. This spatial heterogeneity can be attributed to their distinct sensitivities to soil moisture and soil temperature. According to the Nr emission parameterizations, the NH_3 emission peaks at 50% WHC[70], the HONO peaks at 25% and 90% WHC, and the NO at 10% WHC (Figure 4c). Thus, the irrigated soil moisture conditions (~30% WHC) are more favorable for peak HONO emissions to occur. Under the same soil temperature, the NH_3 emission factor is larger than HONO (Figure 4d)[71,72]. During the case period, the soil temperature in the southern Jiangsu was 7–9 °C higher than that in the northern part. Such high temperature led to a substantial increase in NH_3 emissions in the southern region, resulting in regional heterogeneity in Nr emissions.

The fertilization Nr emissions significantly improved the simulation of relevant pollutants in Nanjing SORPES site during the pollution period (Figure 4e–j, Figure S11). Agricultural NH_3 emissions from upstream regions improved the simulation of NH_3 in Nanjing, both in magnitude

and temporal variation (Figure 4e). As an alkaline gas, NH_3 can react with ambient sulfuric acid (H_2SO_4) and nitric acid (HNO_3) to form $(NH_4)_2SO_4$, NH_4HSO_4 and NH_4NO_3 aerosols; hence, the enhanced NH_3 also mitigates the underestimation of NH_4^+, NO_3^-, and $PM_{2.5}$ (Figure S12). The base experiment revealed a slight underestimation of HONO compared to NH_3 (Figure 4f), indicating that local non-agricultural sources dominate HONO production at SORPES (Figure S13a). Despite this, additional soil HONO emissions contributed to elevated the nitrate and $PM_{2.5}$ concentrations (Figure S12). In contrast, soil NO emissions were negligible relative to HONO and NH_3, resulting in minimal effect on air quality in this case (Figure S14).

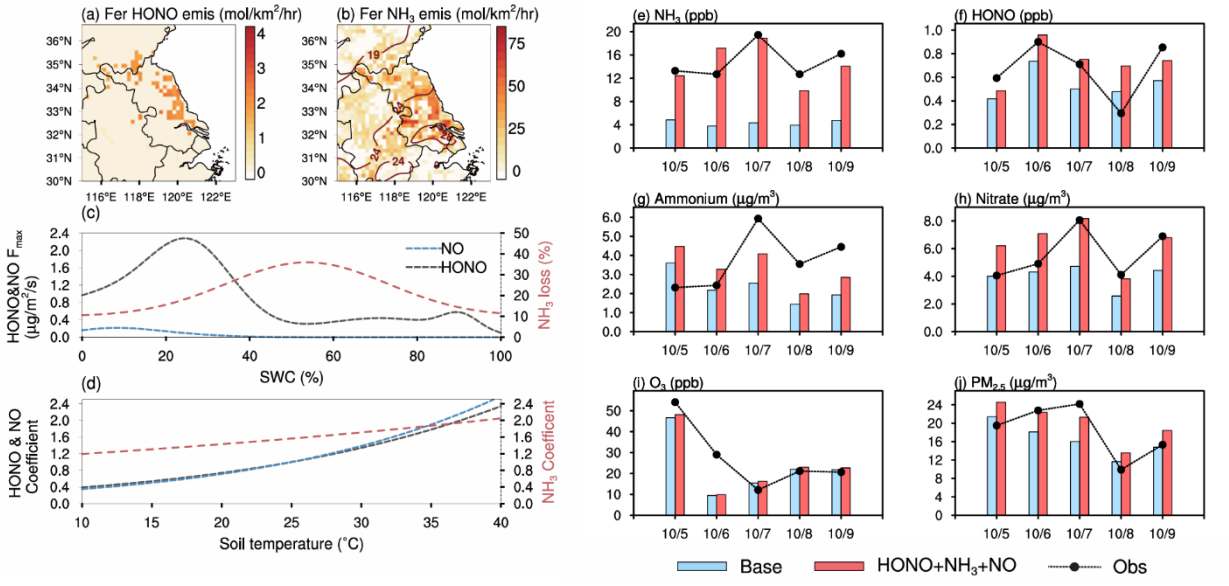

Figure 4. Improved model performance on air quality by including fertilization Nr emissions. Spatial distribution of simulated (a) HONO and (b) NH_3 fertilization emissions and soil temperature (isolines in unit of ℃) during the case period (October 1st–10th). The sensitivity of Nr emissions to (c) soil water content (SWC) and (d) soil temperature. (e–j) Daily observed (black line) and simulated (blue and red bars) NH_3, HONO, ammonium, nitrate, O_3 and $PM_{2.5}$ concentration in Nanjing during the pollution episode.

To quantitatively understand the enhanced secondary pollution by soil Nr emissions, we diagnosed the chemical processes in the YRD region. As shown in Figure 5, with fertilization emissions, the average simulated HONO and NH_3 mixing ratio in the YRD region increased by 0.5 ppb (118.5%) and 5.9 ppb (39.7%), respectively. The photolysis of HONO is an important source of atmospheric OH that regulates the atmospheric oxidizing capacity and the production of secondary pollutants such as O_3. Along with the enhanced HONO, the OH and O_3 also increased by 11.0% (Figure S13b) and 2.5% (Figure 5e), respectively. Diagnostic analysis showed that such enhanced atmospheric oxidizing capacity facilitates the formation of HNO_3 by accelerating the chemical conversion of NO_x to HNO_3 (Figure 5c). Moreover, the soil-emitted NH_3 facilitates the HNO_3 gas-particulate conversion by neutralizing HNO_3 gas, thus facilitating particulate nitrate formation (Figure 5d). The increased emissions of HONO and NH_3 each play a dominant role in the two major steps required for NO_3^- production, collectively contributing to the overall production of nitrate (Figure 5g). Thus, sensitivity experiments showed that the joint effects of HONO and NH_3 emissions on nitrate formation are larger than the sum of the individual impacts (Figure S12). Such nonlinear response of the nitrate formation is mainly caused by thermodynamic equilibrium as detailed in Text S2. This single fertilization event resulted in a 18.2% increase in nitrate concentrations in YRD, demonstrating the significance of crop fertilization in Nr emissions and regional air pollution.

The responses of nitrate production show great regional disparities depending on its chemical sensitivity to NH_3 or HNO_3 according to the thermodynamic equilibrium principle[14,73]. For the YRD region, where the nitrate formation regime is mainly NH_3-limited[74], increasing NH_3 alone significantly enhanced nitrate formation (Figure S16, S17). Besides, the secondary formation of sulfate can also be influenced by NH_3. For instance, the rate of sulfate formation pathways like

aqueous-phase oxidation of SO_2 by NO_2, O_3, and Mn-catalyzed oxidation on aerosol surfaces, are positively correlated to aerosol pH[75]. In NH_3-limited regime, the aerosol pH notably increased with excess NH_3 due to lower aqueous H^+ concentrations[76], likely resulting in a higher sulfate formation rate during this pollution event.

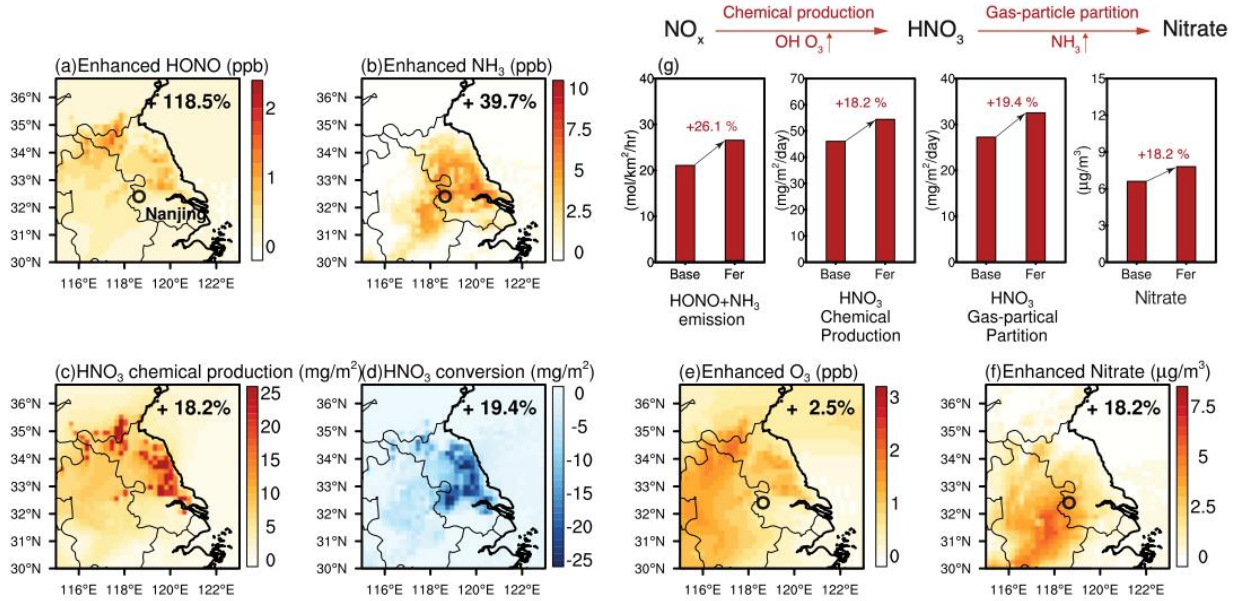

Figure 5. Impacts of soil Nr emissions after fertilization on O_3 and secondary aerosol pollution in the YRD region. Distribution of enhanced (a) HONO, (b) NH_3, (c) HNO_3 chemical production, (d) HNO_3 gas-particle conversion, (e) O_3, and (f) nitrate concentration by fertilization Nr emissions during 1st–10th October 2021. (g) The key processes and factors driving the O_3 and nitrate formation in the map area.

Though fertilization-induced emissions of nitrogen-containing trace gases are dynamically depicted in this work, some limitations and uncertainties need further efforts. In our simulations, it is assumed that all fields within a single grid applied fertilizer on the same day, which might not be that realistic and thus introduce some biases in emission estimation. Furthermore, our simulations

only considered fertilizer emissions from drylands, which means the impact of soil emissions on air quality could be even greater while taking other agricultural land types (e.g., rice paddies) into account. Additionally, the rewetting of soils through rainfall or irrigation can trigger nitrogenous gas emissions (the Birch effect)[77–79], which was not accounted for in our simulations. Despite these limitations, our results clearly demonstrate the significant influence and spatial heterogeneity of agricultural emissions on air quality and the necessity of including them in air quality modelling.

4. IMPLICATIONS

We developed a novel fertilization identification method and a dynamic emission model to access soil Nr emissions and their combined impacts on air pollution in China, with a specific focus on the agriculture-intensive YRD region. Compared to the widely used monthly emission rates based on static statistics, the updated agricultural emission estimation in this work is capable of depicting high-resolution Nr emissions by pinpointing fertilization activities and considering time- and spatial-varying meteorological and soil conditions. When coupling with air quality model, dynamic NH_3 and HONO emissions would more accurately capture the day-to-day variability in the ambient concentration of $PM_{2.5}$ and O_3 precursors. Such dynamic emission and its coupling with air quality model could shed more light on the role of fertilization on regional air quality, particularly during post-fertilization period. Our simulations show that Nr emissions from a typical fertilization event collectively contribute to a 2.5% increase in ozone concentrations and a 18.2% enhancement in nitrate levels over the YRD region. As China's pollutant emissions controls have been relatively stringent for the industrial and transportation sectors in the past decade, the impacts of soil emissions on the nitrogen cycle and atmospheric environment are expected to be increasingly prominent, especially during the fertilization period. Given that there are no specific regulations and standards for controlling emissions of air pollutants from agricultural activities currently, it is of great urgency

to investigate the impacts of agricultural emissions on air pollution and to incorporate them into future air quality improvement strategies. Additionally, there is a need for more comprehensive and accurate data regarding agricultural management, such as fertilizer application rate, timing, types and locations of fertilizer uses, which are essential for developing effective models and making informed and cost-effective decisions. By addressing these research gaps, more targeted strategies to mitigate the adverse effects of agriculture on air quality could be achieved.

ASSOCIATED CONTENT

Supporting Information

Detailed parameterized HONO source mechanisms included in the model; explanation of methods and modeling parameters employed; data quality control and model evaluation with observed air pollutants and meteorological elements; simulated daily average soil water content distribution in Jiangsu; model simulated impacts of soil NH_3, HONO and NO emissions after fertilization in the YRD region.

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Notes

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