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Non-stop industries were the main source of air pollution during the 2020 coronavirus lockdown in the North China Plain

Zhen Li¹ · Shaocai Yu¹ · Mengying Li¹ · Xue Chen¹ · Yibo Zhang¹ · Jiali Li¹ · Yapping Jiang¹ · Weiping Liu¹ · Pengfei Li² · Eric Lichtfouse³,⁴

Abstract
Despite large decreases of emissions of air pollution during the coronavirus disease 2019 (COVID-19) lockdown in 2020, an unexpected regional severe haze has still occurred over the North China Plain. To clarify the origin of this pollution, we studied air concentrations of fine particulate matter (PM$_{2.5}$), NO$_2$, O$_3$, PM$_{10}$, SO$_2$, and CO in Beijing, Hengshui and Baoding during the lockdown period from January 24 to 29, 2020. Variations of PM$_{2.5}$ composition in inorganic ions, elemental carbon and organic matter were also investigated. The HYSPLIT model was used to calculate backward trajectories and concentration weighted trajectories. Results of the cluster trajectory analysis and model simulations show that the severe haze was caused mainly by the emissions of northeastern non-stopping industries located in Inner Mongolia, Liaoning, Hebei, and Tianjin. In Beijing, Hengshui and Baoding, the mixing layer heights were about 30% lower and the maximum relative humidity was 83% higher than the annual averages, and the average wind speeds were lower than 1.5 m s$^{-1}$. The concentration of NO$_3^-$, SO$_4^{2-}$, NH$_4^+$, organics and K$^+$ were the main components of PM$_{2.5}$ in Beijing and Hengshui, while organics, K$^+$, NO$_3^-$, SO$_4^{2-}$, and NH$_4^+$ were the main components of PM$_{2.5}$ in Baoding. Contrary to previous reports suggesting a southerly transport of air pollution, we found that northeast transport caused the haze formation.

Keywords  Spring festival · COVID-19 · North China Plain · Haze

Introduction
Haze is a weather phenomenon in which dust, smog, and other dry particles affect the atmospheric visibility (Cai et al. 2017; Chang et al. 2009). The USA and European countries began to experience haze in the 1950s, but after decades of governance, air quality has been greatly improved. Unlike the haze in London in 1952 caused by coal burning and Los Angeles smog caused by vehicle emissions in the 1950s, haze in China is the consequence of diverse and high primary emissions, and efficient secondary generations (An et al. 2019). Although the Chinese government has made many years of efforts to control haze by transferring heavy industries from core cities to neighboring cities recently, the North China Plain has a high incidence of heavy haze in winter, and air pollutions there are more serious during the Spring Festival (Xiao et al. 2015; An et al. 2019). The Spring Festival is the most important and lively traditional festival in China, marking the beginning of the Lunar New Year by setting off fireworks and firecrackers. Previous studies have shown that residential heating, industrial production,
and vehicle emissions caused severe haze pollution in North China Plain in winter (Li et al. 2014; Zhang et al. 2010; Kong et al. 2015).

Meanwhile, the topography and emission characteristics in the North China Plain (Zhang et al. 2019) are conducive to the regional transport and accumulation of pollutants because its north and west sides are Yanshan and Taihang Mountains, respectively. For example, in the winter of 2013, a nationwide haze broke out in China, especially in the North China Plain. Due to the fact that North China Plain is the largest industrial urban agglomeration in China, North China Plain bear the brunt of nationwide haze in 2013. The peak mass concentration of PM$_{2.5}$ (aerodynamic equivalent diameter less than or equal to 2.5 μm) in Beijing in January 2013 reached 855 μg m$^{-3}$ (Jin et al. 2020; Sun et al. 2017; Yang et al. 2015; Zheng et al. 2015). In addition to the impacts from local emissions, the average contribution rates of regional transport to Beijing’s PM$_{2.5}$ concentrations were found to exceed 40% during the whole year of 2013 (Ge et al. 2018). Several recent studies have shown that areas outside the North China Plain could bring pollutants into cities in the North China Plain through regional transports (Pu et al. 2020; Zhu et al. 2021).

Meteorological factors are another key factor causing air pollution in North China Plain (NCP) in winter. Stable weather conditions are not conducive to the diffusion and sedimentation of pollutants. When haze frequently occurs in winter, the North China Plain is mostly stable weather, which is generally characterized by low wind speed, low mixed layer height (MLH), and high relative humidity (Bey- rich 1997; Jin et al. 2020; Sun et al. 2016; Yu et al. 2018; Kanawade et al. 2020; Zhu et al. 2018). The mixed layer is regarded as the link between the near surface and the free atmosphere and a low mixing layer height is not conducive to the diffusion of pollutants (Seidel et al. 2010).

Due to the outbreak of the COVID-19 pandemic, the Spring Festival in 2020 was very special with imposing extreme measures of nationwide lockdown by the Chinese government. The national emergency responses include extension of the Spring Festival holiday, suspension of all public transportation, closes of schools and entertainment venues, prohibition of public gatherings, enhancement of social distancing, and stay-at-home orders and closes of non-essential businesses (Wang et al. 2020a, b; Wang et al. 2021). Some recent scientific studies have shown that the improvement of air quality was one of the social benefits (Dutheil et al. 2020; He et al. 2020), while other studies have shown that limiting human activities by lockdown were not the fundamental way to alleviate air pollution (Wang et al. 2020a, b; Chang et al. 2020). It is worth studying why the unexpected haze problem still existed over the North China Plain when the air pollutant emissions were significantly reduced during the COVID-19 pandemic. The reduction of emissions from traffic and industrial sources during the COVID-19 lockdown in China provided a unique nature experiment for evaluating the efficiency of air pollution control measures (Le et al. 2020).

The overall goal of this study is to investigate the regional severe haze over the North China Plain during the COVID-19 lockdown period from January 24 to January 29, 2020, by conducting a detailed analysis of the characteristics of haze at three representative cities in the North China Plain and digging the origins of the haze formation. The study explored the impact of regional transport on the formation of haze by identifying the pollution source areas of the target cities, and assessed the role of meteorological factors and ground emissions in the special atmospheric pollution during the lockdown.

**Experimental**

**Observational data**

The study period was from January 24 to January 29, 2020, of which January 24 is the New Year’s Eve of 2020 and the first day when China adopted lockdown measures. Figure 1a shows the North China Plain area (Zhang et al. 2019), which spans 5 provinces (Hebei, Shandong, Henan, Anhui, and Jiangsu) and two municipality cities (Beijing, Tianjin), and is one of the three plains and the most populous plain in China. The three receptor cities selected in this study are Beijing, Hengshui and Baoding, all located in the northern part of the North China Plain. One of the reasons for this selection is that the observational data in the three cities were available. Figure S1 shows that the North China Plain (NCP) area borders Yanshan Mountain to the north, Taihang Mountain to the west, and Bohai Sea to the east. Hourly PM$_{2.5}$, NO$_2$, O$_3$, PM$_{10}$, SO$_2$, and CO concentrations in three representative cities (there are 12, 4 and 6 monitoring stations in Beijing, Hengshui and Baoding cities, respectively) were obtained from China National Environmental Monitoring Center (CNEMC, http://www.cnemc.cn). The hourly chemical component concentrations of PM$_{2.5}$ at three representative cities were obtained from the National Air Pollution Prevention and Control Joint Center (https://view.inews.qq.com/media/6474488). The chemical components include water-soluble inorganic ions (WSIs) (Cl$^-$, NO$_3^-$, SO$_4^{2-}$, NH$_4^+$, Na$^+$, K$^+$, Mg$^{2+}$ and Ca$^{2+}$), elemental carbon (EC) and organic matter (OM). According to the hourly concentrations of PM$_{2.5}$, the pollution levels were divided into light haze (75 $\leq$ PM$_{2.5}$ $<$ 115 μg m$^{-3}$), moderate haze (115 $\leq$ PM$_{2.5}$ $<$ 150 μg m$^{-3}$), heavy haze (150 $\leq$ PM$_{2.5}$ $<$ 250 μg m$^{-3}$), and severe haze (PM$_{2.5} \geq$ 250 μg m$^{-3}$). The observed meteorological data were derived from the National Meteorological Data Center website (http://www.nmic.cn/site/index.html).
The meteorological data with a high spatial resolution of $0.5^\circ \times 0.5^\circ$ required for the Hybrid single-particle Lagrangian integral trajectory (HYSPLIT) model were obtained from the NOAA website (ftp://arlftp.arlhq.noaa.gov/pub/archives/gdas0p5).

The **HYSPLIT model**

We used the HYSPLIT model to calculate backward trajectories and concentration weighted trajectories (CWT). The backward trajectory provides a transport path for air pollutants, which is used to determine whether the high-concentration pollutants in one location originate from the pollution transport of another location (Draxler and Hess 1998). This study calculated the 48 h backward trajectory of PM$_{2.5}$ at a height of 100 m with the starting calculation time points for each day during the study period on 00:00, 03:00, 06:00, 09:00, 12:00, 15:00, 18:00 and 21:00 (UTC time). One back trajectory for one monitoring station were calculated each time. Trajectory cluster analysis based on the trajectory space similarity is used to group a large number of trajectories. In this work, we used the angular distance to do cluster analysis which is based on the trajectory space similarity to group a large number of trajectories. This is mainly because the trajectories can be applied to locate the directions and sources from which the air masses reaching

![Fig. 1](image-url)
the receptor site had transported (Yan et al. 2015). The CWT method assigns a weighted concentration to each grid cell, which is obtained by averaging the sample concentrations associated with the trajectory passing through the grid cell (Reizer and Orza 2018). This method calculates the $CWT_{ij}$ value in each grid cell $(i, j)$ as follows (Hsu et al. 2003; Yan et al. 2015):

$$CWT_{ij} = \frac{1}{\sum_{l=1}^{M} \tau_{ijl}} \sum_{l=1}^{M} C_l \tau_{ijl}$$

where $l$ and $M$ represent the index and the total number of trajectories in the grid cell, respectively. $C_l$ is the concentration detected at the receptor position on the arrival of trajectory $l$. $\tau_{ijl}$ is the time that the trajectory $l$ spent in the grid cell $(i, j)$. A high $CWT_{ij}$ value indicates that the air mass with attributions of high potential pollution has a significant contribution to the receptor site (Yan et al. 2015). In order to find the specific location of the pollution sources, the common pollution source area calculation method of Chen et al. (2020) was adopted to determine the common source area of PM$_{2.5}$ in the three cities during the study period.

**Calculation of meteorological parameters**

This study used mixed layer height (MLH), 10 m wind speed (WS), relative humidity (RH) and static weather index to represent the meteorological conditions in the three recipient cities. The MLH values can be calculated using an empirical method as follows (Nozaki 1973; Beyrich 1997):

$$MLH = \frac{121}{6}(6 - P)(T - T_d) + \frac{0.169P(U_z + 0.257)}{12\ln(Z/z_0)}$$

$$f = 2\Omega \sin \varphi$$

where $P$ is Pasquill's stability coefficient, $T$ is the surface temperature, $T_d$ is the dew point temperature, $U_z$ is the average wind speed (m/s) at the height of $Z$ ($Z = 10$ m), $f$ is the Coriolis parameter (s$^{-1}$), $z_0$ is the surface roughness (0.03–0.2 m for rural area, 0.8–2 m for urban area, and 1.4 m is used in this study), $\Omega$ is the Earth's angular velocity (7.29 $\times$ 10$^{-5}$ rad s$^{-1}$), and $\varphi$ is the latitude value. Pasquill divides atmospheric stability into six levels from A to F (Pasquill 1961), and the six levels from A to F are replaced by 1–6 numbers in this study.

By correlating the air pollutant levels with a combination of meteorological factors in a specific region or city, Zhang et al. (2017) developed the stable weather index (SWI) which included the advection, vertical diffusion and humidity and other meteorological factors (Gong et al. 2021; Liu et al. 2021). A higher SWI value means a weaker diffusion of air pollutants. The detailed information about the calculation method of SWI can be found in Zhang et al. (2017). In this study, the SWT values were calculated and used to assess the atmospheric stable states of the meteorological conditions.

**Modelling meteorological and chemical parameters**

The offline weather research and forecasting (WRFv3.9.1)-Community Multiscale Air Quality Modeling (CMAQv5.3.2) model was used to simulate spatiotemporal variations in meteorological and chemical parameters. Model configurations are listed in Table S1. In this study, the Carbon Bond 6 (CB6r3) scheme and AERO7 module were responsible for gas and aerosol chemistry simulations, respectively. The CMAQ version 5.3.2 (released in 2020) is the latest version and its major science advances in detail can be found in Murphy et al. (2021), Appel et al. (2021) and US EPA (2020). A brief summary relevant to the present study is presented here. Major science advances in the new aerosol module (AERO7) includes significant updates to secondary organic aerosol (SOA) chemistry, updates for chlorine, bromine, iodine chemistry, and halogen chemistries, and the addition of dimethyl sulfide (DMS) chemistry in the CB6r3 chemical mechanism (Appel et al. 2021). The CB6r3 is the most recent version of CB implemented in CMAQ53 with 4 heterogeneous reactions, 38 gas-phase reactions, and 44 gas-phase reactions and 10 heterogeneous reactions for iodine species, and 5 aqueous-phase reactions for bromine species (Appel et al. 2021).

The model domain covering China and a portion of East Asia with a horizontal resolution of 12 km $\times$ 12 km and 345 $\times$ 395 grid cells is shown in Fig. S5. The WRFv3.9.1 (Skamarock et al. 2008) model was used to provide meteorological fields for chemical simulations. The configurations and components for the WRF-CMAQ model used in this study are the same as those in Yu et al. (2014) and are summarized in Table S1. The anthropogenic emissions for China were generated by the Emission Inventory of Air Benefit and Cost and Attainment Assessment System (EI-ABaCAS) developed by Tsinghua University (Zheng et al. 2018). The EI-ABaCAS includes 16 anthropogenic sectors: fertilizer application, livestock, domestic bio-fuel, combustion, fossil fuel, solvent and other use, industry combustion, open burning, power plant, cement steel, other industry process, industry solvent use, road and non-road transport. The natural sources for biogenic emissions were calculated inline using the Biogenic Emission Inventory System version 3.14 (BEISv3.14).
Results and discussion

Air pollutants and meteorological conditions during the severe haze period

Figure 1 shows the time series of the concentrations of PM$_{10}$, O$_3$, NO$_2$, SO$_2$, CO, PM$_{2.5}$, and chemical components of PM$_{2.5}$ in the three cities during the study period. The results show that the changing trends of air pollutants in the three cities were obviously different, indicating different PM$_{2.5}$ formation mechanisms. The highest levels were 692 μg m$^{-3}$ for PM$_{10}$ in Baoding, 125 μg m$^{-3}$ for O$_3$ in Beijing, 90 μg m$^{-3}$ for NO$_2$ in Baoding, 81 μg m$^{-3}$ for SO$_2$ in Baoding, and 2.7 mg m$^{-3}$ for CO in Baoding. The maximum PM$_{2.5}$ concentrations in Beijing, Hengshui and Baoding were 266 μg m$^{-3}$ at 13:00 January 28 in Beijing, 286 μg m$^{-3}$ at 11:00, January 26 in Hengshui, and 641 μg m$^{-3}$ at 3:00, January 25 in Baoding. Baoding experienced a one-day severe pollution process from January 24 to 25, during which PM$_{10}$, NO$_2$ and SO$_2$ all reached their peak values. The K$^+$, Mg$^{2+}$, Cl$^−$, and Ca$^{2+}$ ions from fireworks as indicated in Fig. 1 increased sharply after January 24, and reached their peaks on January 25, being consistent with the fact that the air pollution emissions from fireworks and firecrackers flooded into the air on the New Year’s Eve. This indicates that the fireworks and firecrackers still affected the formation of haze in the North China Plain during the lockdown period. In addition to being derived from fireworks and firecrackers, K$^+$ was also a tracer of biomass combustion (Cheng et al. 2013). The increases of K$^+$ also show that the suburban biomass burning contributed significantly to the K$^+$ in the three cities.

In Beijing, after experiencing two peaks and valleys from January 25 to January 26, PM$_{2.5}$ rebounded after 12 noon on January 26 and then remained at high concentrations above 147 μg m$^{-3}$. This shows that after January 26, when local emissions were almost unchanged, regional transport made a certain contribution to Beijing’s PM$_{2.5}$. The main component of PM$_{2.5}$ in Beijing was water soluble inorganic ions (WSIIs) with an average proportion of 53.6% for PM$_{2.5}$. The average concentration of NO$_3^−$ in Beijing was 32.0 μg m$^{-3}$, accounting for 12.1–29.7% of PM$_{2.5}$, with an average proportion of 20.9%, while SO$_4^{2−}$ in Beijing ranged from 10 μg m$^{-3}$ to 27.8 μg m$^{-3}$, accounting for 12.2% of PM$_{2.5}$ on average. The average relative humidity in Beijing was 60.7 ± 16.4% (Fig. S2) during the study period, which promoted the formation of a large amount of SO$_4^{2−}$ from the heterogeneous chemical reactions and liquid phase oxidations of SO$_2$. The concentrations of NH$_4^+$ in Beijing ranged from 8.9 to 21.9 μg m$^{-3}$, with an average proportion of 10.4% for PM$_{2.5}$. Since emissions from motor vehicles and industries have declined to a certain extent under the COVID-19 lockdown (Le et al. 2020), residential heating and non-stop industrial productions in Beijing emitted a certain amount of NO$_3^−$, SO$_4^{2−}$ and NH$_4^+$ precursors.

In Hengshui, there were four peak values of PM$_{2.5}$ concentrations in the early stage with decreasing trends in the later stage. The average percentage of WSIIs in PM$_{2.5}$ in Hengshui was 71.6%. Similar to Beijing, the main components of PM$_{2.5}$ in Hengshui were NO$_3^−$, SO$_4^{2−}$ and NH$_4^+$, with average mass concentrations of 36.1 ± 20.4 μg m$^{-3}$, 22.3 ± 13 μg m$^{-3}$, and 16.9 ± 7.6 μg m$^{-3}$, respectively. The proportion of WSIIs in PM$_{2.5}$ changed over time. For example, on January 25, the main component of PM$_{2.5}$ was SO$_4^{2−}$ and on January 26, the main component of PM$_{2.5}$ turned to be NO$_3^−$, which peaked at night on January 26. Considering that the local emissions were relatively unchanged, this indicates that the contributions of regional transports to PM$_{2.5}$ in Hengshui were considerable, and the formation mechanism of PM$_{2.5}$ in Hengshui has changed. Figure S2 shows that the changes of relative humidity in Hengshui on January 25 were consistent with those of PM$_{2.5}$. It should be noted that there were diurnal differences in the concentrations of NO$_3^−$ in Hengshui with the average concentrations of 27.2 μg m$^{-3}$ (daytime) and 40.7 μg m$^{-3}$ (night). The higher night NO$_3^−$ concentrations in Hengshui might be affected by external sources and stable weather conditions.

In Baoding, unlike Beijing and Hengshui, organic matter dominated PM$_{2.5}$, accounting for 25 ± 6.4% on average. The average concentrations of chemical components of PM$_{2.5}$ in Baoding in the descending order were 60.3 ± 19 μg m$^{-3}$ for organic matter (OM), 35.2 ± 20.1 μg m$^{-3}$ for K$^+$, 33.9 ± 20.4 μg m$^{-3}$ for SO$_4^{2−}$, 26.6 ± 7.0 μg m$^{-3}$ for NO$_3^−$, 25.5 ± 6.8 μg m$^{-3}$ for NH$_4^+$, 16.6 ± 11.7 μg m$^{-3}$ for Cl$^−$, 10.7 ± 3.3 μg m$^{-3}$ for elemental carbon (EC), 8.3 ± 3.4 μg m$^{-3}$ for Ca$^{2+}$, 3.9 ± 1.4 μg m$^{-3}$ for Mg$^{2+}$, and 2.9 ± 0.5 μg m$^{-3}$ for Na$^+$. PM$_{2.5}$ in Baoding peaked on January 25 with sharp drops in the next 24 h by 380 μg m$^{-3}$ because of the diffusion caused by the increases of the mixed layer height (MLH, Fig. S2). The concentration of organic matter (OM) in Baoding was extremely high, indicating that organic gas precursors played a key role in the formation of PM$_{2.5}$ in Baoding. Note that the OM concentrations were still high after Jan 25 in Baoding because of the effects of the regional transports. The very high concentrations of K$^+$ and Cl$^−$ in Baoding’s PM$_{2.5}$ indicate substantial contributions from the emissions of fireworks and firecrackers. This is consistent with the findings of Tang et al (2016), who found that the K$^+$, Cl$^−$, SO$_4^{2−}$, NH$_4^+$, and NO$_3^−$ concentrations in Tianjin’s PM$_{2.5}$ during the firework were 65.8, 42.9, 38.9, 8.7, and 1.9 μg m$^{-3}$, respectively, and that the K$^+$ and Cl$^−$ concentrations during the peak period were 23.5 and 15.5 times higher than those during Chinese Lunar New
Year, respectively. Since fireworks contained charcoal and organic materials used as adhesives, they were responsible for elevated OC and EC concentrations during the firework and firecracker events (Tsai et al. 2012; Kong et al. 2015). Since potassium compounds, e.g., KNO₃, KClO₃, KClO₄, K₂CrO₇, and K₂Cr₂O₇, in black powder are used as the main oxidizers, the corresponding chemical reactions, e.g., 2KClO₃ = 2KCl + 3O₂ and KClO₄ = KCl + 2O₂, can lead to high concentrations of K⁺ and Cl⁻ during the firework and firecracker events (Azhagurajan et al. 2011; Tang et al. 2016).

Figure S2 shows the time series of hourly meteorological elements in the three cities. The meteorological conditions in the three cities during the study period can be summarized as low MLH and wind speed (WS) but high stable weather index (SWI) and relative humidity. For example, the MLH mean values in Beijing, Hengshui, and Baoding were 448.2 ± 294.1, 497.3 ± 209.1, and 330.0 ± 232.2 m, respectively. Comparing to the annual average MLH values in Beijing and Shijiazhuang being 594 ± 183 m and 464 ± 183 m, respectively (Zhu et al. 2018), the MLH values in the three cities during the study period can be considered as relatively low. The average WS in Beijing, Hengshui, and Baoding were 1.4 ± 0.8, 0.8 ± 0.5, and 0.8 ± 0.8 m s⁻¹, respectively. The low wind speeds in these three cities limit the diffusion of local sources. On the other hand, the maximum relative humidity in Beijing, Hengshui and Baoding were 83%, 92% and 97%, respectively. The SWI ranges in Beijing, Hengshui and Baoding were 8–19, 11–20 and 9–21, respectively. The high SWIs in the three cities indicate that the meteorological conditions were stable during the study period. In conclusion, the stable meteorological conditions in the three cities exacerbated the formation of the haze.

#### Regional transport impacts based on the HYSPLIT analysis

Table 1 summarizes the average concentrations of PM₂.₅ and its chemical components, and the percentages for each cluster and three cluster trajectories for each receptor city are shown in Fig. S3. The results of all backward trajectories in Fig. S3 indicate that most of the air masses came from Inner Mongolia, Liaoning, Hebei and Tianjin.

For Beijing, Cluster 1, which accounted for 54.1% and carried 51.0% of the pollution trajectories, transported from the neighboring areas to Beijing through a small area in northern Hebei province. Cluster 1 had the highest concentration of PM₂.₅ (177.3 ± 61.2 μg m⁻³), which was the most important transport path for PM₂.₅ in Beijing. Cluster 2 (3) accounted for 37.7% (8.2%) and carried 39.5% (9.5%) of the pollution trajectories. According to the concentrations of chemical components, Cluster 2 carries the most K⁺ and Cl⁻ probably generated by firework because Cluster 2 was not a cross-sea transport as shown in Fig. S3.

For Hengshui, all cluster trajectories were cross-sea transports in the northeast direction with obvious long-distance transport characteristics as indicated in Fig. S3. Cluster 1 with the longest transport path came from western Inner Mongolia and Liaoning Province, accounting for 43.8% and carrying 44.3% of the pollution trajectories (see Table 1). Clusters 2 and 3 passed through western Liaoning, southern Tianjin, and northern Shandong. The PM₂.₅ concentrations (157.6 μg m⁻³) carried by the air mass in Cluster 2 was the highest. The Na⁺ concentrations in Cluster 2 were the highest, further indicating that cross-sea transports had a certain influence on the formation of PM₂.₅ in Hengshui.

For Baoding, except for Cluster 3, the other two Clusters all passed through the western Liaoning and Inner Mongolia. Cluster 3, which accounted for 21.1% and carried PM₂.₅ with an average concentration of 216.7 μg m⁻³, moved around Baoding, indicating a great impact of local pollution sources on PM₂.₅ in Baoding. The percentages of Clusters 1 and 2 were 45.4% and 33.5%, respectively, and Cluster 2 carried the highest concentration of PM₂.₅ (295.9 μg m⁻³).

To further study the impact of regional transports on the formation of haze, Fig. 2 shows the 48-h backward trajectories in three cities, superimposed by the observed PM₂.₅ concentrations over the North China Plain every 12 h within 48 h. As can be seen, the spatial variations of PM₂.₅ in the three cities were consistent with those of the monitoring stations on the transport paths. The pollution incidents in Beijing were a typical full-time severe pollution event. PM₂.₅ concentrations in Beijing were higher at 13:00 on January 28 than those in other four time points. From January 26 to January 28, there was a rising process of PM₂.₅ in the western cities of Liaoning, which were on the pollution transport path. The concentrations of PM₂.₅ in Beijing remained at a high level above 147 μg m⁻³, indicating that PM₂.₅ in western Liaoning was imported into Beijing along the pollution transport path. Figure 2 indicates that nearby cities located in the pollution transport path had a greater impact on the increases of Beijing’s PM₂.₅. Figure 2 also shows that the spatial variations of PM₂.₅ concentrations in Hengshui and Baoding were consistent with those at the monitoring stations on the transport path. For the pollution event from January 24 to January 26, the cities in Liaoning province located on the northeast transport path for Hengshui experienced a substantial increase of PM₂.₅ within 24 h from 11:00 on January 24 to 11:00 on January 25. The PM₂.₅ concentrations in Hengshui began to rise from 23:00 on January 25, and reached a peak on January 26. Baoding’s PM₂.₅ experienced a sharp rise from 15:00 on January 24. At the same time, the concentrations of PM₂.₅ in cities of western Liaoning and Hebei Provinces on the transport path for Baoding also experienced the same rising processes.
Figure 3 shows the concentration weighted trajectories (CWT) for the entire and severe pollution periods and the concentrations of PM$_{2.5}$ and chemical components for different clusters in Beijing, Hengshui, and Baoding during the study period.

| Cluster | Number of trajectories (percent) | PM$_{2.5}$ (μg m$^{-3}$) | Number of polluted trajectories* | PM$_{2.5}$ (μg m$^{-3}$) |
|---------|----------------------------------|---------------------------|---------------------------------|---------------------------|
| Cluster 1 | 309 (54.1) | 177.3 ± 61.2 | 214 (51) | 150.4 ± 41.1 |
| Cluster 2 | 215 (37.7) | 133.9 ± 64.0 | 166 (39.5) | 157.3 ± 53.3 |
| Cluster 3 | 47 (8.2) | 136.2 ± 50.7 | 40 (9.5) | 150.7 ± 39.6 |

For Beijing, Cluster 1 had the highest concentration of PM$_{2.5}$ (177.3 ± 61.2 μg m$^{-3}$), which was the most important transport path for PM$_{2.5}$ in Beijing.

According to the concentrations of chemical components, Cluster 2 carries the most K$^+$ and Cl$^-$ probably generated by firework because Cluster 2 was not a cross-sea transport as shown in Fig. S3. For Hengshui, the PM$_{2.5}$ concentrations (157.6 μg m$^{-3}$) carried by the air mass in Cluster 2 was the highest. The Na$^+$ concentrations in Cluster 2 were the highest, further indicating that cross-sea transports had a certain influence on the formation of PM$_{2.5}$ in Hengshui. For Baoding, Cluster 3 carried PM$_{2.5}$ with an average concentration of 216.7 μg m$^{-3}$, moved around Baoding, indicating a great impact of local pollution sources on PM$_{2.5}$ in Baoding.

Values of chemical components are: average mass concentrations (μg m$^{-3}$) and the fraction of chemical component (%) in parentheses for PM$_{2.5}$.

For Beijing, Clusters 2 and 3 had a higher concentration of PM$_{2.5}$ than Cluster 1. This is because the transport paths of Cluster 2 and 3 are mainly through Liaoning province, Hebei province, Qinhuangdao, Tangshan, Langfang and Chengde in Hebei province, and Tianjin, which contributed high concentrations of PM$_{2.5}$ to Beijing. For Hengshui, the potential sources of pollution during the severe pollution period were Panjin, Anshan and Yingkou in Liaoning province. It was worth noting that Liaoning province had a large amount of PM$_{2.5}$ input for Hengshui for all the pollution periods. For Baoding, Cangzhou, Langfang, Chaoyang, Jinzhou and Tianjin were potential source areas during the heavy pollution period. Note that some of the air masses from Liaoning were transported across the Bohai Sea, which is located in the east of the North China Plain.

Identification of common source areas for pollution

The concentrations of PM$_{2.5}$ in the receptor cities were combination effects of local emissions and regional transports. Figure 3 shows the concentration weighted trajectories (CWT) for the entire and severe pollution periods and the CWT for the light, medium and heavy pollution periods are shown in Fig. S4. Figure 3 indicates that different cities had different degrees of contributions to PM$_{2.5}$ in the three cities. The regions with the CWT values higher than 250 μg m$^{-3}$ (see Fig. 3) included Chaoyang and Huludao in Liaoning province, Qinhuangdao, Tangshan, Langfang and Chengde in Hebei province, and Tianjin, all of which contributed high concentrations of PM$_{2.5}$ to Beijing. For Hengshui, the potential sources of pollution during the severe pollution period were Panjin, Anshan and Yingkou in Liaoning province. It was worth noting that Liaoning province had a large amount of PM$_{2.5}$ input for Hengshui for all the pollution periods. For Baoding, Cangzhou, Langfang, Chaoyang, Jinzhou and Tianjin were potential source areas during the heavy pollution period. Note that some of the air masses from Liaoning were transported across the Bohai Sea, which is located in the east of the North China Plain.
Figure 3 shows the common source areas of PM$_{2.5}$ in the three cities during the study period. Controlling the common source area is an effective choice for air pollution control (Yu et al. 2018). As can be seen, the PM$_{2.5}$ concentrations of the three recipient cities partly depend on the cities located on pollution transport paths for them. The northeast transport path through Liaoning province had the greatest impact on PM$_{2.5}$ in the three recipient cities. The results show that the northeast transport path dominates the regional transport for the haze formation in NCP.

Regional transport impacts based on the WRF-CMAQ model simulations

To determine if the northeast transport path had the substantial impact on PM$_{2.5}$ in the three receptor cities during the study period, the WRF-CMAQ model was applied to simulate this regional severe haze episode in the retrospective mode. The concentration weighted trajectories (CWT) values for PM$_{2.5}$ were calculated on the basis of the air mass back trajectories and their associated PM$_{2.5}$ concentrations observed at the three receptor sites to locate the source areas with the largest potential contributions to high concentrations. Figure S6 shows the spatial distributions of the CWT values created by combining all results at the three
cities during the study period. Followed Yu et al. (2016), an emission reduction scenario was created on the basis of the CWT values in Fig. S6 by removing all anthropogenic emissions for grid cells with CWT ≥ 75 μg m⁻³, while those will remain uncontrolled for grid cells with CWT < 75 μg m⁻³. Figures S7 and S8 show the comparisons of the spatial distributions of anthropogenic emissions for primary PM₂.₅, NH₄⁺, NO₃⁻, OM, and SO₄²⁻ concentrations in the reduction case decreased at most by −106.73 μg m⁻³, −8.93 μg m⁻³, −25.96 μg m⁻³, −13.75 μg m⁻³ and −5.51 μg m⁻³ in Beijing, respectively, while they were −144.38 μg m⁻³, −12.01 μg m⁻³, −34.15 μg m⁻³, −21.77 μg m⁻³ and −7.79 μg m⁻³ in Hengshui, respectively, and were −148.32 μg m⁻³, −12.21 μg m⁻³, −35.86 μg m⁻³, −23.68 μg m⁻³ and −8.56 μg m⁻³ in Baoding, respectively. The mean PM₂.₅, NH₄⁺, NO₃⁻, OM, and SO₄²⁻ concentrations in the reduction case could decrease by more than 60% in the three receptor cities relative to the base case. The model sensitive simulations results highlight the fact that regional severe haze in North China Plain during the coronavirus lockdown was caused mainly by the anthropogenic emissions through the northeast transport paths.

**Conclusion**

This study investigated the causes of the regional severe haze over the North China Plain during the COVID-19 lockdown period on the basis of chemical observations in the three cities (Beijing, Baoding and Hengshui) and model simulations. The concentrations of NO₃⁻, SO₄²⁻, NH₄⁺, organic matter and K⁺ were found to be the main components of PM₂.₅ in Beijing and Hengshui, while they were OM and K⁺, NO₃⁻, SO₄²⁻, and NH₄⁺ in Baoding. The results indicate that the emissions from firework, firecrackers, residential heating, and non-stop industries under the stable meteorological conditions with low wind speed, low mixing layer height, and high relative humidity caused this severe haze pollution in North China Plain. The results of backward trajectories in the three cities indicate that most of the air masses influencing the cities came from Inner Mongolia, Liaoning, Hebei and Tianjin. For Hengshui, all cluster trajectories are cross-sea transports in the northeast direction with obvious long-distance transport characteristics by passing the western Inner Mongolia and Liaoning Province. For Baoding, the two cluster trajectories also passed through the western...
Liaoning and Inner Mongolia. It was found that the common source areas for the haze formation in the three cities were located in western Liaoning, northern Hebei, and southern Tianjin as the common source cities. The stagnant airflow and uninterrupted emissions from non-stop industries (e.g., power plants and petrochemical facilities) can still cause regional severe haze formation with high humidity to promote aerosol heterogeneous chemistry. The results of the cluster trajectories and model simulations show that different from the previous studies that mainly found a southerly transport path of air pollution for North China Plain, the northeast transport path dominates the regional transport for this haze formation. This indicates that the transport paths can be variable for a city for different haze events in regional scale. This implies that the variabilities of transport paths should be taken into account in the regulation of air pollution in the North China Plain, especially in the case of short-term severe pollutants.

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Fig. S1 Topographic map showing the locations of Beijing, Hengshui, and Baoding. The North China Plain (NCP) area borders Yanshan Mountain to the north, Taihang Mountain to the west, and Bohai Sea to the east.
Fig. S2  

**a** Hourly wind speed (WS) and relative humidity (RH);  
**b** Hourly mixed layer height (MLH) and stable weather index (SWI) in Beijing (BJ), Hengshui (HS), and Baoding (BD) cities during the study period (January 24 to January 29, 2020). The average RH in Beijing was 60.7±16.4% during the study period, which promoted the formation of a large amount of SO$_4^{2-}$ from the heterogeneous chemical reactions and liquid phase oxidations of SO$_2$.  

Fig. S2 shows that the changes of RH in Hengshui on January 25 were consistent with those of PM$_{2.5}$. 


Fig S3. 48-h air mass back trajectories and clusters from January 24 to January 29, 2020. The results of all backward trajectories in Fig. S3 indicate that most of the air masses came from Inner Mongolia, Liaoning, Hebei and Tianjin. For Beijing, Cluster 1, which accounted for 54.1% and carried 51.0% of the pollution trajectories, transported from the neighboring areas to Beijing through a small area in northern Hebei province. Cluster 1 had the highest concentration of PM$_{2.5}$ ($177.3 \pm 61.2 \mu g \ m^{-3}$), which was the most important transport path for PM$_{2.5}$ in Beijing. Cluster 2 (3) accounted for 37.7% (8.2%) and carried 39.5% (9.5%) of the pollution trajectories in Beijing. For Hengshui, all cluster trajectories are cross-sea transports in the northeast direction with obvious long-distance transport characteristics as indicated in Fig. S3.
Fig S4. Concentration weighted trajectory (CWT) maps of PM$_{2.5}$ for the slight haze (75 ≤ PM$_{2.5}$ < 115 µg m$^{-3}$), moderate haze (115 ≤ PM$_{2.5}$ < 150 µg m$^{-3}$), and heavy haze (150 ≤ PM$_{2.5}$ < 250 µg m$^{-3}$) during the study period (24th January–29th January, 2020). Different cities had different degrees of contributions to PM$_{2.5}$ in the three cities. The regions with the CWT values higher than 250 µg m$^{-3}$ (the severe haze column in Fig. 3) included Chaoyang and Huludao in Liaoning province, Qinhuangdao, Tangshan, Langfang and Chengde in Hebei province, and Tianjin, all of which contributed high concentrations of PM$_{2.5}$ to Beijing.
Fig S5. a Map of China and the location of North China Plain.
b The model domain covering China and a portion of East Asia with a horizontal resolution of 12 km × 12 km and 345×395 grid cells.
Fig S6. **a** The locations of the three receptor cities and other cities. **b** The spatial distributions of the concentration weighted trajectory (CWT) values created by combining all results at the three cities for PM$_{2.5}$ $>$ 75 µg m$^{-3}$ during the study period.
Fig S7. The comparisons of the spatial distributions of anthropogenic emissions for primary PM$_{2.5}$, NH$_3$, NO$_X$ between the base and reduction cases. The unit is monthly emission tons per grid cell.
Fig S8. The comparisons of the spatial distributions of anthropogenic emissions for SO$_2$, volatile organic compounds (VOC), and POC between the base and reduction cases. The unit is monthly emission tons per grid cell.
Fig S9. The daily mean comparisons of the simulation results for the spatial distributions of PM$_{2.5}$ concentrations for the base and reduction cases and their differences for each day from January 24 to 29, 2020.
Fig S10. The maps of the PM$_{2.5}$ reduction amounts between the base and reduction cases overlaid with the wind fields for different times. Fig. S10 clearly shows that the prevailing northeasterly winds during study period brought air pollution continuously from the northeast areas to the NCP regions, causing the severe haze formation there.
Fig S11. Comparisons of the time series of PM$_{2.5}$ and its chemical components concentrations in the three receptor cities for the base and reduction cases. Fig. S11 shows that the peak values for all these pollutants in the reduction cases in the three receptor cities were effectively decreased by more than 60%.
Table S1. The configurations and components for the WRF-CMAQ model. The meteorological fields used in this study are provided by WRF model and the version is 3.9.1 (Stamarock and Klemp, 2008). Configurations of WRF used in this study are the same as those in Yu et al. (2014) and are briefly listed here.

| WRF | Version | 3.9.1 |
|-----|---------|-------|
|     | Short and long wave radiation scheme | Rapid Radiative Transfer Model for General (RRTMG) |
|     | Cloud microphysics | Two-moment scheme |
|     | Planetary boundary layer scheme | Asymmetrical Convective Model version 2 (ACM2) |
|     | Cumulus parameterization | Kain–Fritsch scheme |
|     | Land–surface model | Pleim-Xiu LSM |

| CMAQ | Simulation period | 20 Jan to 30 Jan 2020 |
|------|------------------|-----------------------|
|      | Domain           | Chinese mainland |
|      | Horizontal resolution | 12 km×12 km |
|      | Version          | 5.3.2 (Murphy et al., 2021) |
|      | Chemical mechanism | Carbon Bond version 6 (CB6r3) (Yarwood et al., 2015) |
|      | Aerosol module   | AERO7 (Pye et al., 2013) |
|      | Anthropogenic Emission inventory | Air Benefit and Cost and Attainment Assessment System (EI-ABaCAS) (Zheng et al., 2019) |
|      | Biogenic Emission inventory | Biogenic Emissions Landuse Database, Version 3 (BELD3) |
Table S2. The comparisons of daily means and ranges of PM$_{2.5}$, NH$_4^+$, NO$_3^-$, OM, and SO$_4^{2-}$ concentrations (µg m$^{-3}$) for the base and reduction cases and their differences in Beijing.

|                | PM$_{2.5}$ | NH$_4^+$ | NO$_3^-$ | OM   | SO$_4^{2-}$ |
|----------------|------------|----------|----------|-------|-------------|
|                | Range      | Average  | Range    | Average | Percentage (%) |
| Base           | 13.7~53.3  | 30.7     | 8~26.6   | 14.9   | -51.50%     |
| Reduction      | 1.7~9      | 4.6      | 0.5~3.6  | 1.6    | -66.70%     |
| Reduction-Base | -26.7      | -1.86    | -5.39    | -2.9   | -64.40%     |
|                | -2.4~11.3  | 5.6      | 1.3~6.1  | 2.9    | -48.40%     |
|                | 1.1~2.6    | 2        | 1.1~1.4  | 1.2    | -36.40%     |
| Base           | Range      | Average  | Range    | Average | Percentage (%) |
| 2020/1/24      | 44.9~83.7  | 61.7     | 19.2~46.4| 29.6   | -52.00%     |
| Reduction      | 2.8~6.6    | 4.6      | 1~3.1    | 1.9    | -57.90%     |
| Reduction-Base | -37.97     | -3.41    | -9.5     | -4.3   | -57.90%     |
|                | -7~18.4    | 12.6     | 2.5~8.9  | 5.3    | -47.40%     |
|                | 2.7~4.7    | 9.2      | 2.3~8.1  | 4.8    | -45.60%     |
| Base           | Range      | Average  | Range    | Average | Percentage (%) |
| 2020/1/25      | 25.2~143.5| 71.4     | 11.7~46.2| 23.8   | -66.60%     |
| Reduction      | 1.3~11.7   | 5.2      | 0.4~3.5  | 1.4    | -73.30%     |
| Reduction-Base | -100.73    | -8.85    | -25.96   | 3.8    | -74.00%     |
|                | -25.85     | -10.7    | -12.91   | 3.9    | -60.70%     |
|                | 1.7~7.4    | 9.9      | 1.6~8    | 1.7    | -60.40%     |
| Base           | Range      | Average  | Range    | Average | Percentage (%) |
| 2020/1/26      | 50.9~153.1| 85.3     | 22.6~84.8| 42     | -50.80%     |
| Reduction      | 3.7~13.9   | 7.2      | 1.3~7.4  | 3.4    | -52.70%     |
| Reduction-Base | -71.44     | -6.55    | -18.73   | -10.2  | -49.80%     |
|                | 9.5~42.2   | 20.5     | 3.6~23.5 | 10.3   | -41.70%     |
|                | 7~25.8     | 12.6     | 3.7~17. | 7.3    | -56.00%     |
|                | 3.4~6.7    | 4.7      | 1.5~3    | 2.1    |            |
| Base           | Range      | Average  | Range    | Average | Percentage (%) |
| 2020/1/27      | 52.6~154.8| 99.4     | 21~74.2  | 40     | -59.70%     |
| Reduction      | 3.3~12.9   | 7.6      | 0.7~6.5  | 2.8    | -63.10%     |
| Reduction-Base | -106.73    | -8.93    | -25.46   | -13.7  | -64.00%     |
|                | 7.8~38.4   | 20.8     | 1.5~20.4 | 7.5    | -53.60%     |
|                | 8.3~24.5   | 15.6     | 4.2~14.4 | 7.2    | -54.40%     |
|                | 3.4~8.1    | 5.8      | 1.6~3.3  | 2.7    |            |
| Base           | Range      | Average  | Range    | Average | Percentage (%) |
| 2020/1/28      | 16.8~44.8  | 27.8     | 10.2~18.4| 14.6   | -59.70%     |
| Reduction      | 0.5~2.8    | 1.6      | 0.3~0.8  | 0.7    | -63.10%     |
| Reduction-Base | -106.73    | -4.8     | -13.3    | -8.4   | -64.00%     |
|                | 0.6~7.2    | 2.8      | 0.3~1.7  | 2.4    | -53.60%     |
|                | 2~8.1      | 4.2      | 1.2~4.2  | 2      | -54.40%     |
|                | 1.7~3.8    | 2.9      | 1.4~2.5  | 2      |            |
| Base           | Range      | Average  | Range    | Average | Percentage (%) |
| 2020/1/29      | 16.8~44.8  | 27.8     | 10.2~18.4| 14.6   | -47.60%     |
| Reduction      | 0.5~2.8    | 1.6      | 0.3~0.8  | 0.7    | -55.10%     |
| Reduction-Base | -106.73    | -4.8     | -13.3    | -8.4   | -70.40%     |
|                | 0.6~7.2    | 2.8      | 0.3~1.7  | 2.4    | -43.30%     |
|                | 2~8.1      | 4.2      | 1.2~4.2  | 2      | -30.40%     |
|                | 1.7~3.8    | 2.9      | 1.4~2.5  | 2      |            |
Table S3. The comparisons of daily means and ranges of PM$_{2.5}$, NH$_4^+$, NO$_3^-$, OM, and SO$_4^{2-}$ concentrations (µg m$^{-3}$) for the base and reduction cases and their differences in Hengshui.

|         | PM$_{2.5}$ | NH$_4^+$ | NO$_3^-$ | OM | SO$_4^{2-}$ |
|---------|------------|----------|----------|----|-------------|
|         | Range      | Average  | Range    | Average | Range      | Average  |
| Base    | 17.4–47.7  | 27.4     | 5.1–24.1 | 10.7     | -25.67     | -16.7    |
| 2020/1/24 Reduction | 0.7–2.8    | 1.5      | 0.1–3.5  | 0.3      | -2.42      | -1.3     |
|         | 1.5–8.6    | 4.4      | 0.7–7.2  | 0.7      | -7.76      | -3.7     |
|         | 2.4–11.7   | 5        | 1–1.6    | 2.3      | -4.68      | -2.7     |
|         | 1.4–2.9    | 1.9      |          |          | -1.29      | -0.7     |
|         |           |          |          |          |            |          |
| Reduction-Base | Maximum     | -60.90%  | -82.70%  | -83.90%  | -54.40%    | -35.20%  |
|         | Average    | -60.90%  | -82.70%  | -83.90%  | -54.40%    | -35.20%  |
| Base    | Range      | Average  | Range    | Average  | Range      | Average  |
| 2020/1/25 Reduction | 35.9–60.2  | 46.3     | 18.1–43.6| 30.2     | -25.97     | -16.1    |
|         | 1.4–5.2    | 3.1      | 0.9–3.4  | 1.7      | -2.22      | -1.4     |
|         | 2.6–14.9   | 8.7      | 1.7–9.3  | 4.6      | -7.03      | -4.1     |
|         | 4.2–12     | 8.1      | 2.7–9.2  | 5.4      | -5.98      | -2.7     |
|         | 2.2–4.1    | 3        | 1.2–3.7  | 2.3      | -1.16      | -0.8     |
|         |            |          |          |          |            |          |
| Reduction-Base | Percentage (%) | -60.90%  | -82.70%  | -83.90%  | -54.40%    | -35.20%  |
| Base    | Range      | Average  | Range    | Average  | Range      | Average  |
| 2020/1/26 Reduction | 59.5–148.7 | 103      | 13.6–37.9| 27.1     | -114.27    | -75.9    |
|         | 3.4–11.6   | 6.3      | 0.5–2.2  | 1.3      | -10.26     | -5.1     |
|         | 9.7–33.6   | 17.3     | 1.5–6.9  | 3.2      | -30.07     | -14.1    |
|         | 12.6–27.1  | 19.3     | 3.3–9.2  | 5.5      | -19.98     | -13.8    |
|         | 3–9.3      | 6        | 1.1–3    | 2        | -6.28      | -4.1     |
|         |            |          |          |          |            |          |
| Reduction-Base | Percentage (%) | -73.70%  | -80.00%  | -81.40%  | -71.40%    | -67.20%  |
| Base    | Range      | Average  | Range    | Average  | Range      | Average  |
| 2020/1/27 Reduction | 44.1–128.2 | 69.3     | 12.8–53.9| 26.8     | -74.32     | -42.6    |
|         | 2.5–9.3    | 4.1      | 0.3–2.1  | 1.1      | -7.16      | -3       |
|         | 3.8–26.1   | 10.3     | 1.1–5.7  | 2.4      | -20.43     | -7.9     |
|         | 6.3–23.4   | 10.8     | 2.5–14.2 | 5        | -9.22      | -5.8     |
|         | 2.4–7.3    | 4.9      | 1.1–3.7  | 2        | -4.12      | -2.6     |
|         |            |          |          |          |            |          |
| Reduction-Base | Percentage (%) | -61.40%  | -73.70%  | -76.50%  | -53.50%    | -53.40%  |
| Base    | Range      | Average  | Range    | Average  | Range      | Average  |
| 2020/1/28 Reduction | 101.7–218.1| 162.2    | 58.1–133.7| 96.2     | -144.38    | -66      |
|         | 4.1–18.5   | 11.2     | 2.4–11.7 | 6.8      | -12.01     | -4.5     |
|         | 5.5–55.1   | 29.8     | 3.7–36   | 18.8     | -34.15     | -10.9    |
|         | 14.1–39    | 29.6     | 7.9–28   | 16.6     | -21.72     | -9.9     |
|         | 7.8–11     | 9.8      | 3.4–6.9  | 5.4      | -7.63      | -4.4     |
|         |            |          |          |          |            |          |
| Reduction-Base | Percentage (%) | -40.70%  | -39.70%  | -36.70%  | -37.40%    | -45.10%  |
| Base    | Range      | Average  | Range    | Average  | Range      | Average  |
| 2020/1/29 Reduction | 60.9–178.2 | 101.7    | 19.5–46.1| 27.8     | -136.97    | -74      |
|         | 3.4–13.4   | 6.9      | 0.7–2.9  | 1.4      | -11.06     | -5.6     |
|         | 7.6–37.4   | 18.4     | 1–8.4    | 3.3      | -30.87     | -15.1    |
|         | 8.1–30.7   | 16.8     | 2.6–10.6 | 5.8      | -21.77     | -11      |
|         | 4.7–10.2   | 6.4      | 1.4–2.7  | 2.1      | -7.79      | -4.3     |
|         |            |          |          |          |            |          |
| Reduction-Base | Percentage (%) | -72.70%  | -80.20%  | -82.20%  | -65.40%    | -67.80%  |
Table S4. The comparisons of daily means and ranges of PM$_{2.5}$, NH$_4^+$, NO$_3^-$, OM, and SO$_4^{2-}$ concentrations (µg m$^{-3}$) for the base and reduction cases and their differences in Baoding.

|          | PM$_{2.5}$ | NH$_4^+$ | NO$_3^-$ | OM     | SO$_4^{2-}$ |
|----------|-----------|----------|----------|--------|------------|
|          | Range     | Average  | Range    | Average | Percentage (%) |
| Base     | 35~66.3   | 51.0     | 1.8~3.8  | 6.7     | 3.5        |
|          | 3.7~11.5  | 10.1     | 1~7.1    | 2.8~13  | 1.2~4.2    |
|          | 5.1~16.3  | 2.5      | 20.9     | 6.9~23.8| 2.4~10.3   |
|          | 1.9~5.2   |          |          |         |            |
| Reduction| Range     | Average  | Range    | Average | Percentage (%) |
| 2020/1/24| 14.1~47.6 | 28.9     | 0.4~2.3  | 1       | -31.53%    |
|          | 4~7.1     | 6.7      | 0.5~1.6  | 2.3     | 2.29%      |
|          | 2.8~13.6  | 1.35     | 0.4~1.6  | 6.7     | -7.01%     |
|          | 1.2~4.2   |          |          |         |            |
| Reduction-Base | Maximum | Average | Percentage (%) |
| 2020/1/24 | -31.53%   | -22.1%   | -61.20%  | -35.20% | -24.20%    |
| Base     | Range     | Average  | Range    | Average | Percentage (%) |
| 2020/1/25| 58~155.4  | 78.8     | 3.6~9.2  | 5.7     | -23.23%    |
|          | 9~23.4    | 7.0      | 4.2~6.4  | 9.2     | -4.7%      |
|          | 4.3~13.7  |          |          |         |            |
| Reduction | Range     | Average  | Range    | Average | Percentage (%) |
| 2020/1/25| 39~123.6  | 32.2     | 2.2~6.4  | 4.6     | -2.88%     |
|          | 5.9~16.3  | 2.6      | 3.6~7.1  | 3.5     | -9.06%     |
|          | 2.4~10.3  |          |          |         |            |
| Reduction-Base | Maximum | Average | Percentage (%) |
| 2020/1/25 | -23.23%   | -17.7%   | -50.30%  | -25.50% | -15.20%    |
| Base     | Range     | Average  | Range    | Average | Percentage (%) |
| 2020/1/26| 55.2~221.9| 119.7    | 3.8~18.7 | 8.9     | -25.2%     |
|          | 9.3~55.6  | 19.1     | 24.9     | 7.0     | -1.7%      |
|          | 3~11.5    |          |          |         |            |
| Reduction | Range     | Average  | Range    | Average | Percentage (%) |
| 2020/1/26| 29.7~119.8| 67.7     | 1.7~9.5  | 4.6     | -12.4%     |
|          | 5.9~24.5  | 6.3      | 12.5     | 3.7     | 0.4%       |
|          | 1.7~7.0   |          |          |         |            |
| Reduction-Base | Maximum | Average | Percentage (%) |
| 2020/1/26 | -102.19%  | -87.8%   | -48.70%  | -38.20% | -39.00%    |
| Base     | Range     | Average  | Range    | Average | Percentage (%) |
| 2020/1/27| 147.5~294.7| 232.1    | 9.8~22.9 | 61.6    | -87.8%     |
|          | 17.4~66   | 16.1     | 34.5     | 40.0    | -0.8%      |
|          | 7~21.9    |          |          |         |            |
| Reduction | Range     | Average  | Range    | Average | Percentage (%) |
| 2020/1/27| 89~207.5  | 144.3    | 6.7~15   | 9.3     | 123.05%    |
|          | 10~42.5   | 44.5     | 24.9     | 5.9     | -6.8%      |
|          | 3~16.3    |          |          |         |            |
| Reduction-Base | Maximum | Average | Percentage (%) |
| 2020/1/27 | -123.05%  | -108.6%  | -42.00%  | -44.10% | -35.60%    |
| Base     | Range     | Average  | Range    | Average | Percentage (%) |
| 2020/1/28| 165.3~279.4| 220.9    | 11.3~22.4| 16.5    | -148.32%   |
|          | 24.9~68.4 | 46.8     | 38.6     | 11.8    | -7.2%      |
|          | 32.1~49.8 |          |          |         |            |
| Reduction | Range     | Average  | Range    | Average | Percentage (%) |
| 2020/1/28| 74.8~199 | 112.3    | 3.8~15.6 | 8.6     | -108.6%    |
|          | 6.2~45.5  | 24.9     | 20.9     | 5.9     | -21.8%     |
|          | 3.4~10.1  |          |          |         |            |
| Reduction-Base | Maximum | Average | Percentage (%) |
| 2020/1/28 | -148.32%  | -108.6%  | -47.60%  | -46.00% | -49.60%    |
| Base     | Range     | Average  | Range    | Average | Percentage (%) |
| 2020/1/29| 57.6~118.5| 85.4     | 3.3~10   | 6.0     | -75.41%    |
|          | 6.1~27.7  | 15.1     | 14.2     | 6.2     | -6.02%     |
|          | 4.3~7.3   |          |          |         |            |
| Reduction | Range     | Average  | Range    | Average | Percentage (%) |
| 2020/1/29| 30.5~73.7 | 48.5     | 1.6~7.1  | 3.1     | -36.9%     |
|          | 3.3~21.3  | 4.0      | 7.6~5.3  | 7.5     | -2.9%      |
|          | 2.2~5.5   |          |          |         |            |
| Reduction-Base | Maximum | Average | Percentage (%) |
| 2020/1/29 | -75.41%   | -36.9%   | -50.50%  | -37.10% | -35.20%    |