The Impact of the Numbers of Monitoring Stations on the National and Regional Air Quality Assessment in China During 2013–18

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ABSTRACT

China national air quality monitoring network has become the core data source for air quality assessment and management in China. However, during network construction, the significant change in numbers of monitoring sites with time is easily ignored, which brings uncertainty to air quality assessments. This study aims to analyze the impact of change in numbers of stations on national and regional air quality assessments in China during 2013–18. The results indicate that the change in numbers of stations has different impacts on fine particulate matter (PM$_{2.5}$) and ozone concentration assessments. The increasing number of sites makes the estimated national and regional PM$_{2.5}$ concentration slightly lower by 0.6−2.2 μg m$^{-3}$ and 1.4−6.0 μg m$^{-3}$ respectively from 2013 to 2018. The main reason is that over time, the monitoring network expands from the urban centers to the suburban areas with low population densities and pollutant emissions. For ozone, the increasing number of stations affects the long-term trends of the estimated concentration, especially the national trends, which changed from a slight upward trend to a downward trend in 2014−15. Besides, the impact of the increasing number of sites on ozone assessment exhibits a seasonal difference at the 0.05 significance level in that the added sites make the estimated concentration higher in winter and lower in summer. These results suggest that the change in numbers of monitoring sites is an important uncertainty factor in national and regional air quality assessments, that needs to be considered in long-term concentration assessment, trend analysis, and trend driving force analysis.

Key words: monitoring network, newly added sites, PM$_{2.5}$, ozone

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Article Highlights:
• The impact of the numbers of monitoring stations on air quality assessment in China during 2013–18 is investigated.
• The newly added sites make the estimated national and most regional PM$_{2.5}$ concentrations slightly lower.
• The newly added sites affect the estimated long-term trends of ozone concentrations.
• The impact of the increasing number of sites on ozone assessment exhibits a seasonal difference.

1. Introduction

China is facing severe compound air pollution with fine particulate matter (PM$_{2.5}$) and ozone (O$_3$) as the key pollutants (Shao et al., 2006; Song et al., 2017; Wang et al., 2017a). High O$_3$ concentrations enhance the oxidation of the atmosphere, which accelerates the transformation of sulfur dioxide (SO$_2$), nitrogen oxide (NOx) and volatile organic compounds (VOCs) into sulfates, nitrates and particulate organic matter, and the generated fine particles further catalyze heterogeneous reactions (Ravishankara, 1997). High O$_3$ concentrations and PM$_{2.5}$ pollution combine to form compound air pollution, which seriously affects air quality in China (Li et al., 2017a; Ning et al., 2018; Wang et al., 2019a; Zhang et al., 2019). To deal with compound air pollution, we first need a complete monitoring network to obtain long-term and high-density observations of PM$_{2.5}$, O$_3$ and their common precursors to research air pollution. The air quality monitoring network can provide an important data basis for comprehending
the complex relationship between PM$_{2.5}$ and O$_3$ and achieving collaborative control and precise prevention and control.

The air quality monitoring network in China has been continuously developing in the past two decades. It started in the 1970s and focused on urban air quality monitoring. At the beginning of the 1990s, it became an air quality monitoring network composed of 103 urban air quality stations monitoring SO$_2$, NO$_x$ and total suspended particulate matter (TSP) concentrations (Zhong et al., 2007). Since 2012, the number of monitored species has increased from 3 to 6: SO$_2$, nitrogen dioxide (NO$_2$), coarse particulate matter (PM$_{10}$), PM$_{2.5}$, carbon monoxide (CO), and O$_3$ (Wang et al., 2012).

As of 2017, the air quality monitoring network covers more than 5000 sites at the national, provincial, city, and county levels (Liang, 2018), and it can be divided into seven categories in terms of monitoring functions. Among them, the national air quality observation network has 1436 monitoring sites in 338 cities and shows a relatively basic monitoring function.

Using an air quality monitoring network to evaluate pollutant concentrations is of great significance for national air quality management and air pollution research. Since 2013, there have been many assessments on air quality in China using the monitoring network that show the concentrations of SO$_2$, PM$_{2.5}$, CO, PM$_{10}$, and NO$_2$ have decreased but the concentration of O$_3$ has increased (Guo et al., 2019; Li et al., 2019a; Fan et al., 2020; Liu et al., 2020). These assessments address the effectiveness of national pollution control policies (Yang et al., 2019; Yuan and Yang, 2019) and point out the key points of the current pollution control in China, which provides an important scientific basis for the strategic transformation of China’s pollution control policy from original emission control to air quality management (Jiang et al., 2020; Lu et al., 2020a). In the field of air pollution research, the monitoring network concentration assessment is an important basic task. The pollutant concentration assessment using monitoring networks is essential for developing accurate pollution prevention and control measures (Zhao et al., 2018), improving the performance of numerical model simulations and predictions (Tang et al., 2011; Lei et al., 2019), researching atmospheric chemical mechanisms (Lu et al., 2019a), and studying environmental health (Liu et al., 2019). However, the monitoring network used for concentration assessments may lead to inaccurate concentration assessments due to the use of different monitoring equipment (Gego et al., 2005) and the layout of monitoring stations (Adams and Kanaroglou, 2016). According to the abovementioned development of China national monitoring network, the number, coverage and representativeness of monitoring stations have changed significantly since the “12th five-year plan” (Dong et al., 2020). What impact will these changes in the observation network have on air quality concentration assessment in China? This study uses surface observations from 2013 to 2018 to explore this problem, focusing on the increase in the number of stations to elucidate the impact of changes in air quality monitoring network on air quality assessment in China. We expect to obtain an improved understanding of the results of air quality concentration assessments and offer a scientific basis for the formulation of national pollution control policies.

## 2. Data and Methods

### 2.1. Observation data

We used data from the China National Environmental Monitoring Centre (CNEMC), including hourly observations of PM$_{2.5}$, O$_3$, and NO$_2$ from 1436 air quality monitoring stations. The occasional outliers of observations were filtered out by a fully automated outlier detection method (Wu et al., 2018). To ensure the time continuity of data, we selected stations with less than 20% missing data for PM$_{2.5}$ and O$_3$ observations each year. The network in 2013 is taken as the standard observation network (SON) representing the unchanging situation of sites, while the actual network in each year is regarded as the dynamic observation network (DON) representing the changing situations of sites. We evaluate the impact of the change in numbers of sites on PM$_{2.5}$ and O$_3$ concentration assessments by comparing the estimated pollution concentrations in the two different observation networks. Since some sites included in 2013 were absent in the following years, we removed these missing sites from the SON and DON to ensure that each site in the SON existed in the DON.

Air pollution in China presents regional characteristics (Chan and Yao, 2008; Tao et al., 2014; Rohde and Muller, 2015). Therefore, the research areas of this study include all of China and its six important city clusters, namely, the Sichuan and Chongqing areas (SC), Hubei and Hunan areas (HH), Fenwei Plain (FWP), Yangtze River Delta (YRD), Pearl River Delta (PRD) and Beijing-Tianjin-Hebei region (BTH).

The number of sites in the two observation networks after the abovementioned data preprocessing is shown in Table 1. From 2013 to 2015, the national number of sites experienced rapid growth increasing by 3.3 times from the original 307 sites to 1322 sites. The numbers of stations in six key city clusters also have notable increases. The numbers of stations in four regions (all but the BTH and PRD) increased significantly. From the spatial distribution of sites in 2013 and 2015 across the country (Fig. 1a) and 6 key regions (Fig. 2), it can be seen that the coverage of sites was relatively small and the distribution of sites was concentrated in 2013, while the coverage of sites was significantly larger and the sites were more evenly distributed in 2015.

### 2.2. The population and emissions around the sites

Studies have shown that there is a positive correlation between population density and pollutant concentration (Wang et al., 2019b). Moreover, NOx and VOC emissions can reflect the urbanization, industrialization, and local pollutant emission levels of a region (Degraeuwe et al., 2017; Luo et al., 2017). The increase in the number of air quality monitoring sites in China is accompanied by changes in the monitoring area, population, and pollutant emissions within
3. Results and Discussion

The number of sites in the DON increased significantly from 2013 to 2018. Most newly added stations were located in regions with relatively sparse populations and low emissions except the BTH. We discuss the impact of these newly added stations on assessments of PM$_{2.5}$ and O$_3$ concentrations in China in the following sections.

### 3.1. The impact of observation network changes on PM$_{2.5}$ concentration assessment

#### 3.1.1. National average PM$_{2.5}$ concentration

To assess the impact of the growth of the observation network on PM$_{2.5}$ concentration assessment, we first estimated the national annual average PM$_{2.5}$ concentrations in the SON with 307 sites and the DON with the temporally variable number of sites. Figure 1b shows the variation in the national annual average PM$_{2.5}$ concentration in the SON and DON from 2013 to 2018 and the differences (DON minus SON) between the two observation networks. The national annual average PM$_{2.5}$ concentration in China shows an obvious downward trend during 2013–18 in the SON and DON, which is consistent with previous studies (Zhai et al., 2019; Shao et al., 2021). The downward trends in the SON and DON are nearly the same, but the estimated PM$_{2.5}$ concentrations are different. The estimated national annual average PM$_{2.5}$ concentration in the DON is slightly lower than that in the SON by 0.6–2.2 μg m$^{-3}$. The largest concentration difference between the two observation networks is in 2015. The reason for these differences may be that the newly added sites in the DON are located in regions with low population densities and emissions in comparison with sites in SON, since the PM$_{2.5}$ pollution is higher in urban centers covered by the SON (Wang et al., 2017b; Gao et al., 2020), consequently lowering the estimated national annual average PM$_{2.5}$ concentrations in the DON. This result shows that although more than 1000 sites were added to China national air quality observation network during 2013–18, the impact of these newly added sites on the national annual average PM$_{2.5}$ concentration assessment is relatively slight, which also indicates the strong regional characteristics of PM$_{2.5}$ pollution in China.

#### 3.1.2. Average PM$_{2.5}$ concentrations in the six city clusters

To further analyze the impact of observation network...
changes on PM$_{2.5}$ concentration assessments in China’s different city clusters, we estimated the annual average PM$_{2.5}$ concentrations in the SON and DON in the six city clusters. Figures 1c–h show the annual average PM$_{2.5}$ concentrations in the SC, HH, FWP, YRD, BTH, and PRD in the SON and DON from 2013 to 2018 and the differences (DON minus SON) between the two networks. Similar to the country as a whole, in the six city clusters, the downward trends of the
The annual average PM$_{2.5}$ in SON and DON are nearly the same, but the estimated PM$_{2.5}$ concentration values are different. In the SC, HH, and FWP, the annual average PM$_{2.5}$ concentrations in the DON are lower than those in the SON by 1.4–6.0 $\mu$g m$^{-3}$ (Figs. 1c–e). However, the annual average PM$_{2.5}$ concentrations in the DON are slightly higher than those in the SON by 1.1–3.7 $\mu$g m$^{-3}$ in BTH (Fig. 1g, not significant at the 0.05 significance level) and 1.1–3.1 $\mu$g m$^{-3}$ in YRD (Fig. 1f). In the PRD, the annual average PM$_{2.5}$ concentration differences between the two networks are so minor (<1 $\mu$g m$^{-3}$) that they can be ignored (Fig. 1h). The possible reason for the different results among city clusters is that there are different changes in observation networks in these regions. In the SC, HH, and FWP, the newly added sites in the DON are mostly located in the regions with relatively low population densities and pollutant emissions (Table 2), causing lower PM$_{2.5}$ concentrations in the DON. However, the newly added sites in the DON in the BTH are mostly located in areas with relatively high population densities and pollutant emissions (Table 2), causing higher PM$_{2.5}$ concentrations in the DON. The number of newly added sites in the PRD is miniscule (Table 1) and does not have an obvious impact on PM$_{2.5}$ concentration assessment. In the YRD, the newly added sites in the DON are also located in the areas with relatively low population densities and pollutant emissions, but the changes in the population and pollution emissions are not as significant as those in the SC (Table 2). The PM$_{2.5}$ pollution in the northern YRD is more serious than that in the other parts of YRD (Ma et al., 2019). Many newly added sites in DON are in the northern YRD so that the estimated PM$_{2.5}$ concentrations in the DON are higher (Fig. 1f). From the above results, we can see that the impacts of observation network changes on PM$_{2.5}$ concentration assessment are not identical between city clusters and have certain regional characteristics.

### 3.2. The impact of observation network changes on O$_3$ concentration assessment

O$_3$ is a critical pollutant for atmospheric compound pollution in China. It is quite important to explore the influence of observation network changes on O$_3$ concentration assessment. The O$_3$ concentration variation in the boundary layer is complex and affected by not only meteorological conditions

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**Table 2.** The averages of total population, nitrogen oxide (NOx) emissions, and volatile organic compound (VOC) emissions in a 3 km radius around each site in the SON (in 2013) and DON (in 2015).

|                   | Year | Observation network | China | SC  | HH  | FWP | YRD | BTH | PRD |
|-------------------|------|---------------------|-------|-----|-----|-----|-----|-----|-----|
| Population ($\times 10^3$) | 2013 | SON                 | 249   | 145 | 243 | 181 | 259 | 236 | 275 |
|                   | 2015 | DON                 | 157   | 118 | 139 | 170 | 204 | 256 | 263 |
| NOx emissions ($\times 10^3$ kg) | 2013 | SON                 | 1899  | 1728 | 1876 | 1174 | 1906 | 2243 | 1316 |
|                   | 2015 | DON                 | 1073  | 745 | 811 | 921 | 1455 | 2261 | 1273 |
| VOC emissions ($\times 10^7$ mol) | 2013 | SON                 | 2603  | 2545 | 2560 | 1558 | 2894 | 2496 | 2912 |
|                   | 2015 | DON                 | 1303  | 1081 | 1073 | 1076 | 2082 | 2527 | 2839 |

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**Fig. 2.** The distribution of the sites in 2013 (blue dots) and the newly added sites in 2015 (red dots) in (a) SC, (b) HH, (c) FWP, (d) YRD, (e) BTH and (f) PRD and the population in 2015.
and regional transport but also precursor concentrations, photochemical reactions, and VOC/NOx sensitivity (Finlayson-Pitts and Pitts, 1997). The impact of observation network changes on O₃ concentration assessments in China is likewise complicated.

3.2.1. National average O₃ concentration

To explore the impact of observation network changes on estimated O₃ concentration, in a manner similar to PM₂.₅, we first estimated the national annual average daily peak O₃ (O₃ Max) concentrations in the SON and DON. Figure 1i shows the variation in the national annual average O₃ Max concentrations in the two networks from 2013 to 2018 and the differences (DON minus SON) between the two networks. The trends of national O₃ Max in the two observation networks are different and opposite during 2014−15. The trend of the estimated national O₃ Max during 2014−15 is downward in the DON, while it is slightly upward in the SON. Furthermore, the estimated national O₃ Max concentrations in the two networks are also notably different. The national O₃ Max concentrations in the DON are 1.4−5.1 μg m⁻³ lower than those in the SON and the standard deviation of the differences is 1.5 μg m⁻³. The largest concentration difference between the two networks is in 2015. Based on the above results, we can conclude that the newly added sites not only led to the opposite trends of estimated O₃ concentration in China in the two networks during 2014−15, but also made the estimated values of O₃ concentration in the DON lower than those in the SON. This provides a new perspective for understanding the declining trend of estimated O₃ concentration during 2014−15 in the DON (Meng et al., 2017). In studies of the trend of O₃ concentration in China, it is necessary to fully consider the impact of observation station changes on the assessment. In addition, the impact of changes in the location of observation stations on the O₃ concentration assessment should also be considered in the layout of observation stations.

The reason why the newly added sites in the DON have the above influences on the assessment of the national average O₃ concentration is relatively complex, and we will explain it in detail in section 3.2.3.

3.2.2. Average O₃ concentrations in the six city clusters

To explore whether the impacts of observation network changes on different city clusters are similar to those in the country as a whole, we estimated the O₃ Max concentrations of six regions in the SON and DON. Figures 1j−o show the annual average O₃ Max concentrations in the SC, HH, FWP, YRD, BTH, and PRD in the SON and DON from 2013 to 2018 and the differences (DON minus SON) between the two networks. The newly added stations in the DON affect evaluations of the long-term trend and O₃ Max concentration in all regions except the BTH and PRD. The trends of O₃ Max in the DON are upward in 2016 compared with 2015 in the SC (Fig. 1j), slightly upward in 2014 compared with 2013 in the FWP (Fig. 1l), and downward in 2015 compared with 2014 in the YRD (Fig. 1m). These trends are completely opposite to those in the same period in the SON. In the HH, the O₃ Max decrease in 2014 compared with 2013 and increase in 2015 compared with 2014 in the DON are more obvious than those in the same period in the SON (Fig. 1k). Moreover, the differences in the estimated annual average O₃ Max concentrations between the two observation networks are different in the six regions. In the SC and FWP, the O₃ concentrations in the DON are higher than those in the SON and the differences are up to 4.9 μg m⁻³ in 2017 and 8.1 μg m⁻³ in 2014 respectively (Figs. 1j and 1l). However, in the HH and YRD, the O₃ Max concentrations in the DON are lower than those in the SON and the differences are up to 8.2 μg m⁻³ in 2014 and 8.0 μg m⁻³ in 2015 respectively (Figs. 1k and 1m). These results show that, except in the BTH and PRD, the impacts of the newly added stations in the DON on O₃ concentration assessments in the SC, HH, FWP, and YRD are quite significant and distinct in different regions. When considering individual city clusters, we need to account for this impact including site changes.

The reason why the newly added sites in the DON have these different influences on O₃ concentration assessment in the six city clusters is relatively complex, and will also be explained in detail in section 3.2.3.

3.2.3. Seasonal difference in the impact of the newly added sites on O₃ concentration assessment

The above discussion reveals that the newly added sites in the DON affect the evaluations of long-term trends and annual average O₃ Max concentrations in the country as a whole and its six city clusters. Considering the complexity of O₃, we also estimated the differences in the monthly average concentration of NO₂ and O₃ Max between the two observation networks in the country as a whole and its six city clusters. Figure 3 shows the differences (DON minus SON) in the national monthly average concentrations of NO₂ and O₃ Max between the DON and SON from 2013 to 2018. Except in winter, the O₃ concentrations in the DON are lower than those in the SON (Fig. 3a). Furthermore, the NO₂ concentrations at all time periods after 2013 in the DON are significantly lower than those in the SON. Figure 4 shows the differences (DON minus SON) in the monthly average O₃ Max concentrations between the DON and SON in the six city clusters from 2013 to 2018. In the SC, HH, and FWP, the monthly average O₃ Max concentrations in the DON are higher in winter and lower in summer than those in the SON (Figs. 4a−c). In the YRD, the monthly average O₃ Max concentrations in the DON are almost equal to those in the SON in winter but lower than those in the SON in summer (Fig. 4d). In the BTH and PRD, the differences in the monthly average O₃ Max concentrations between the DON and SON are negligible (Figs. 4e and 4f). Figure 5 shows the differences (DON minus SON) in the monthly average NO₂ concentration between the DON and SON in the six city clusters from 2013 to 2018. Similar to the country as a whole, the NO₂ concentrations in the DON in the six city clusters are significantly lower than
those in the SON (Figs. 5a–d) in all regions except the BTH and PRD. As a whole, the seasonal difference of the impact of the newly added sites in DON on the \(O_3\) concentration assessment in regions is more significant than that in the whole country, especially in the SC.

In the country as a whole and most city clusters, the estimated \(O_3\) concentrations in the DON are higher in winter and lower in summer than those in the SON. We postulate a viable mechanism leading to this seasonal difference. In winter, when the particulate matter pollution is serious, the extinction effect of aerosols weakens the photochemical reaction of \(O_3\) formation (Wang et al., 2019c, 2020). Given that in winter the titration of NO on \(O_3\) greatly suppresses \(O_3\) concentration (Zhang et al., 2014) and \(O_3\) production is in VOC-limited regime (Kang et al., 2021), the lower NOx in DON than in SON (Fig. 3b) leads to higher \(O_3\) concentration in DON than in SON. However, in summer, owing to the intense solar radiation, the photochemical reaction of \(O_3\) formation is greatly strengthened (Geng et al., 2008; Zeng et al., 2018). Due to the fact that \(O_3\) production is in NOx-limited or transition regimes in summer (Wang et al., 2021a) and the NOx in DON is lower than that in SON (Fig. 3b), the concentration of \(O_3\) in DON is lower than that in SON. We also note that the VOC emissions in DON are lower than those in SON (Table 2). But the VOC emissions in MEIC inventory only include anthropogenic VOC emissions excluding the biogenic VOC emission. The estimated anthropogenic VOC emissions in China in 2017 using MEIC inventory were 28.5 Tg (Li et al., 2019b), while the estimated biogenic VOC emissions in China in 2018 using the Model of Emissions of Gases and Aerosols from Nature were 58.89 Tg (Li et al., 2020). The role of biogenic VOC emissions on tropospheric photochemistry cannot be neglected (Xie et al., 2007). We surmise that the VOC emissions in DON will greatly increase and the differences of VOC emissions between the SON and DON will shrink after adding the biogenic VOC emissions (Wang et al., 2021b). Therefore, the NOx emission differences might play a major role in the seasonal change of \(O_3\) concentration difference between the two networks. In addition, the \(O_3\) production is not only proportional to concentrations of VOC, but also to their reactivities with OH radicals. For example, the aromatics accounts for 45% of the total \(O_3\) production, although the concentration of aromatics is only 25% of the total VOC concentrations in Shanghai, China (Geng et al., 2008). After building more VOC monitoring stations in the future, we can better explore the impact of VOC differences on the air quality assessment difference between the SON and DON.

The impact of the newly added sites in the DON on \(O_3\) concentration assessment caused by the above mechanism is distinct in different regions. In the BTH and PRD, the changes in the observation network are so small that we can ignore this impact (Figs. 4e and 4f). In the YRD, the differences in population density and emissions between the SON and DON are not as significant as those in the SC (Table 2). Therefore, the above seasonal differences are not obvious in the YRD (Fig. 4d). In HH, although the changes in newly added sites are similar to those in the SC, this region is a major pollutant channel for the north-south transmissions in China (Lu et al., 2019b), where the pollution is greatly affected by external area transmissions. The influence of newly added sites in the DON on the \(O_3\) concentration assessment in the HH is weakened, so the seasonal differences are not as significant as those in the SC (Fig. 4b). Such seasonal differences in these regions are ultimately reflected in the differences in the annual average \(O_3\) concentration between the two observation networks (section 3.2.1 and section 3.2.2).

Based on the above results, we know that the impacts of the newly added sites on \(O_3\) concentrations and trend assessments are significant and show seasonal and regional differences. When evaluating \(O_3\) concentrations, we need to fully consider this seasonal and regional impact.
Conclusions

China national air quality monitoring network experienced a rapid development with the construction of a substantial number of new sites during 2013−18. This study analyzed the impact of the newly added sites on the long-term trends and concentrations assessments of PM$_{2.5}$ and O$_3$ by taking the network in 2013 as the SON and the actual network in each year as the DON. The results are as follows: (1) An increasing number of sites has had almost no influence on the long-term trends of estimated national and regional PM$_{2.5}$ concentrations in China but has slightly affected the

![Fig. 4. The differences in the regional monthly average O$_3$ Max concentrations between the DON and SON from 2013 to 2018 in the (a) SC, (b) HH, (c) FWP, (d) YRD, (e) BTH and (f) PRD. Red indicates higher concentration for the DON, blue indicates lower concentration for the DON, and hatching indicates that the difference is significant at the 0.05 significance level (T-test).]
estimated PM$_{2.5}$ concentrations. The newly added sites lead to the estimated national and most regional (in the SC, HH, and FWP) PM$_{2.5}$ concentrations being lower because the monitoring network expanded from the urban centers to the suburban areas with relatively low population densities and pollutant emissions, where PM$_{2.5}$ pollution is less. (2) The newly added sites significantly affect the assessments of O$_3$ the long-term trends and concentrations in China. The long-term trends of O$_3_{\text{Max}}$ in the DON and SON are opposite in some years, which occurs in the country as a whole (2014–15), the SC (2015–16), FWP (2013–14), and YRD (2014–15). Moreover, the estimated annual average O$_3_{\text{Max}}$ concentrations in the DON are lower in the country as a whole, the HH, and YRD, but higher in the SC and FWP than those in the SON. (3) The impact of the increasing number of sites on O$_3$ assessment exhibits seasonal differences. The estimated monthly national and most regional (in the SC, HH, and FWP) O$_3_{\text{Max}}$ concentrations in the DON

**Fig. 5.** Same as Fig. 4 but for the NO$_2$ concentrations.
are higher in winter and lower in summer than those in the SON. The main reason for this seasonal difference is the difference of ozone sensitivity to precursors in winter and summer.

Of course, there are some uncertainties in this study. To ensure the time continuity of observation data, the setting of the missing data rate determines which observation stations to use in this study. We attempted to use different missing data rates to select stations and found that 20% is the most appropriate for ensuring that the station data are fully utilized, which is consistent with the previous missing measurement rate (Gao et al., 2020). For assessing the national concentrations by averaging the observations of all the sites, the uneven distribution of stations and short lifetime for PM$_{2.5}$ and O$_3$ will unavoidably introduce bias and different species will have different representative errors (Piersanti et al., 2015; Li et al., 2019c). Limited by the actual distribution of existing monitoring stations, we chose this method of averaging all sites as in the previous national assessment (Lu et al., 2020b).

Moreover, in addition to the emission differences, the reason for differences of estimated PM$_{2.5}$ and O$_3$ concentrations between the SON and DON may also include pollution control policies (Zhang et al., 2020), meteorological factors (Tong et al., 2017; Ariya et al., 2018; Bilal et al., 2019) and regional transmission especially for regional air quality assessment (Huang et al., 2018). Additional research is needed to explore these possible reasons. Based on the conclusions of this study, we know that the numbers of monitoring stations affect the estimated PM$_{2.5}$ concentrations, O$_3$ long-term trends, and the estimated O$_3$ concentrations in China. When we assess the air quality, especially for O$_3$ assessments, we need to fully consider the impact of the changes in numbers of monitoring stations on the assessment. In addition, in the process of developing the observation network, the station layout needs to take the influence of the newly added sites on the assessment into account. We need to minimize such influence as much as possible so that the assessment results are more objective and accurate.

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REFERENCES

Adams, M. D., and P. S. Kanaroglou, 2016: A criticality index for air pollution monitors. Atmospheric Pollution Research, 7, 482–487, https://doi.org/10.1016/j.apr.2015.11.004.

Ariya, P. A., A. Dastoor, Y. Nazarenko, and M. Amyot, 2018: Do snow and ice alter urban air quality? Atmos. Environ., 186, 266–268, https://doi.org/10.1016/j.atmosenv.2018.05.028.

Bilal, M., and Coauthors, 2019: Characteristics of fine particulate matter (PM$_{2.5}$) over urban, suburban, and rural areas of Hong Kong. Atmosphere, 10, 496, https://doi.org/10.3390/atmos10090496.

Chan, C. K., and X. H. Yao, 2008: Air pollution in mega cities in China. Atmos. Environ., 42, 1–42, https://doi.org/10.1016/j.atmosenv.2007.09.003.

Degraeuwe, B., P. Thunis, A. Clappier, M. Weiss, W. Lefebvre, S. Janssen, and S. Vranckx, 2017: Impact of passenger car NOx emissions on urban NO$_2$ pollution - Scenario analysis for 8 European cities. Atmos. Environ., 171, 330–337, https://doi.org/10.1016/j.atmosenv.2017.10.040.

Dong, J. D., X. L. Chen, X. B. Cai, Q. Q. Xu, Y. T. Guan, T. H. Li, S. Y. Liu, and F. Chen, 2020: Analysis of the temporal and spatial variation of atmospheric quality from 2015 to 2019 based on China atmospheric environment monitoring station. Journal of Geo-Information Science, 22, 1983–1995, https://doi.org/10.1016/j.jgs.2020.200212.

Fan, H., C. F. Zhao, and Y. K. Yang, 2020: A comprehensive analysis of the spatio-temporal variation of urban air pollution in China during 2014–2018. Atmos. Environ., 220, 117066, https://doi.org/10.1016/j.atmosenv.2019.117066.

Finlayson-Pitts, B. J., and J. N. Pitts Jr., 1997: Tropospheric air pollution: Ozone, airborne toxics, polycyclic aromatic hydrocarbons, and particles. Science, 276, 1045–1051, https://doi.org/10.1126/science.276.5315.1045.

Gao, L., X. Yue, X. Y. Meng, L. Du, Y. D. Lei, C. G. Tian, and L. Qiu, 2020: Comparison of ozone and PM$_{2.5}$ concentrations over urban, suburban, and background sites in China. Adv. Atmos. Sci., 37, 1297–1309, https://doi.org/10.1007/s00376-020-0054-2.

Gego, E. L., P. S. Porter, J. S. Irwin, C. Hogrefe, and S. T. Rao, 2005: Assessing the comparability of ammonium, nitrate and sulfate concentrations measured by three air quality monitoring networks. Pure Appl. Geophys., 162, 1919–1939, https://doi.org/10.1007/s00024-005-2698-3.

Geng, F. H., X. X. Tie, J. M. Xu, G. Q. Zhou, L. Peng, W. Gao, X. Tang, and C. S. Zhao, 2008: Characterizations of ozone, NO$_x$, and VOCs measured in Shanghai, China. Atmos. Environ., 42, 6873–6883, https://doi.org/10.1016/j.atmosenv.2008.05.045.

Guo, H., X. F. Gu, G. X. Ma, S. Y. Shi, W. N. Wang, X. Zuo, and X. C. Zhang, 2019: Spatial and temporal variations of air quality and six air pollutants in China during 2015–2017. Scientific Reports, 9, 15201, https://doi.org/10.1038/s41598-019-50655-6.

Huang, D., Q. L. Li, X. X. Wang, G. X. Li, L. Q. Sun, B. He, L. Zhang, and C. S. Zhang, 2018: Characteristics and trends of ambient ozone and nitrogen oxides at urban, suburban, and rural sites from 2011 to 2017 in Shenzhen, China. Sustainability, 10, 4530, https://doi.org/10.3390/su10124530.

Jiang, B. W., Y. G. Li, and W. X. Yang, 2020: Evaluation and treatment analysis of air quality including particulate pollutants: A case study of Shandong province, China. International Journal of Environmental Research and Public Health, 17, 9476, https://doi.org/10.3390/ijerph171249476.

Kang, M. J., J. Zhang, H. L. Zhang, and Q. Ying, 2021: On the relevance of observed ozone increase during COVID-19 lockdown to summertime ozone and PM$_{2.5}$ control policies in China. Environmental Science & Technology Letters, 8, 289–294, https://doi.org/10.1021/acs.estlett.1c00036.

Lei, K., and Coauthors, 2019: Improved inversion of monthly ammonia emissions in China based on the Chinese Ammonia Monitoring Network and ensemble Kalman filter. Environmental Science & Technology, 53, 12 529–12 538, https://doi.org/10.1021/acs.est.9b02701.
Li, F., and Coauthors, 2019c: Estimation of representative errors of surface observations of air pollutant concentrations based on high-density observation network over Beijing-Tianjin-Hebei region. Chinese Journal of Atmospheric Sciences, 43, 277–284. https://doi.org/10.3878/j.issn.1006-9095.1804.17267. (in Chinese with English abstract)

Li, G. H., and Coauthors, 2019a: Fast photochemistry in winter-spring surface ozone in eastern China during the non-winter season of 2015: Observations and source attributions. Atmospheric Chemistry and Physics, 17, 2759–2774, https://doi.org/10.5194/acp-17-2759-2017.

Li, K. D., and Coauthors, 2019b: Widespread and persistent ozone pollution in eastern China during the non-winter season of 2015: Observations and source attributions. Atmospheric Chemistry and Physics, 17, 733, 139301, https://doi.org/10.1016/j.atmoscp.2020.139301.

Li, M., and Coauthors, 2019b: Persistent growth of anthropogenic non-methane volatile organic compound (NMVOC) emissions in China from 2001−2016: The roles of land cover change and climate variability. Environmental Science & Technology, 53, 10 676−10 684, https://doi.org/10.1021/acs.est.9b02422.

Li, M., and Coauthors, 2019b: Persistent growth of anthropogenic non-methane volatile organic compound (NMVOC) emissions in China during 1990−2017: Drivers, speciation and ozone formation potential. Atmospheric Chemistry and Physics, 19, 8897–8913, https://doi.org/10.5194/acp-19-8897-2019.

Liang, G. L., 2018: Talking about the site selection of environment quality monitoring sites in the new era. Guangdong Chemical Industry, 45, 160, 177, https://doi.org/10.3969/j.issn.1007-1865.2018.10.072.

Liu, C., and Coauthors, 2019: Ambient particulate air pollution and daily mortality in 652 cities. The New England Journal of Medicine, 381, 705–715, https://doi.org/10.1056/NEJMoa1817364.

Liu, Q. C., X. Y. Li, T. Liu, and X. J. Zhao, 2020: Spatio-temporal correlation analysis of air quality in China: Evidence from provincial capitals data. Sustainability, 12, 2486, https://doi.org/10.3390/su12062486.

Lu, K. D., and Coauthors, 2019a: Fast photochemistry in wintertime haze: Consequences for pollution mitigation strategies. Environmental Science & Technology, 53, 10 676–10 684, https://doi.org/10.1021/acs.est.9b02422.

Lu, M. M., and Coauthors, 2019b: Investigating the transport mechanism of PM$_{2.5}$ pollution during January 2014 in Wuhan, central China. Adv. Atmos. Sci., 36, 1217–1234, https://doi.org/10.1007/s00376-019-8260-5.

Lu, X., and Coauthors, 2020a: Progress of air pollution control in China and its challenges and opportunities in the Ecological Civilization Era. Engineering, 6, 1423–1431, https://doi.org/10.1016/j.eng.2020.03.014.

Lu, X., L. Zhang, X. L. Wang, M. Gao, K. Li, Y. Z. Zhang, X. Yue, and Y. H. Zhang, 2020b: Rapid increases in warm-season surface ozone and resulting health impact in China since 2013. Environmental Science & Technology Letters, 7, 240–247, https://doi.org/10.1021/acs.estlett.0c00171.

Luo, X.-S., Z. Zhao, Y. Chen, X. L. Ge, Y. Huang, C. Suo, X. Sun, and D. Zhang, 2017: Effects of emission control and meteorological parameters on urban air quality showed by the 2014 Youth Olympic Games in China. Fresenius Environmental Bulletin, 26, 4798–4807.

Ma, T., and Coauthors, 2019: Air pollution characteristics and their relationship with emissions and meteorology in the Yangtze River Delta region during 2014–2016. Journal of Environmental Sciences, 83, 8–20, https://doi.org/10.1016/j.jes.2019.02.031.

Meng, X. Y., Z. Y. Gong, C. X. Ye, S. Wang, H. Sun, and X. Zhang, 2017: Characteristics of ozone concentration variation in 74 Cities from 2013 to 2016. Environmental Monitoring in China, 33, 101–108, https://doi.org/10.19316/jnss.2017.10.05.15. (in Chinese with English abstract)

Ning, C. G., S. G. Wang, M. J. Ma, C. J. Ni, Z. W. Shang, J. X. Wang, and J. X. Li, 2018: Characteristics of air pollution in different zones of Sichuan Basin, China. Science of the Total Environment, 612, 975–984, https://doi.org/10.1016/j.scitotenv.2017.08.205.

Piersanti, A., L. Vitali, G. Righini, G. Cremona, and L. Ciancarrelli, 2015: Spatial representativeness of air quality monitoring stations: A grid model based approach. Atmospheric Pollution Research, 6, 953–960, https://doi.org/10.1016/j.apr.2015.04.005.

Ravishankara, A. R., 1997: Heterogeneous and multiphase chemistry in the troposphere. Science, 276, 1058–1065, https://doi.org/10.1126/science.276.5315.1058.

Rohde, R. A., and R. A. Muller, 2015: Air pollution in China: Mapping of concentrations and sources. PLoS One, 10, e0135749, https://doi.org/10.1371/journal.pone.0135749.

Shao, M., X. Y. Tang, Y. H. Zhang, and W. J. Li, 2006: City clusters in China: Air and surface water pollution. Frontiers in Ecology and the Environment, 4, 353–361, https://doi.org/10.1890/1540-9295(2006)004[0353:CCICAA]2.0.CO;2.

Shao, M., and Coauthors, 2021: Quantifying the role of PM$_{2.5}$ dropping in variations of ground-level ozone: Inter-comparison between Beijing and Los Angeles. Science of the Total Environment, 788, 147712, https://doi.org/10.1016/j.scitotenv.2021.147712.

Song, C. B., and Coauthors, 2017: Air pollution in China: Status and spatiotemporal variations. Environmental Pollution, 227, 334–347, https://doi.org/10.1016/j.envpol.2017.04.075.

Tang, X., J. Zhu, Z. F. Wang, and A. Gbaguidi, 2011: Improvement of ozone forecast over Beijing based on ensemble Kalman filter with simultaneous adjustment of initial conditions and emissions. Atmospheric Chemistry and Physics, 11, 12 901–12 916, https://doi.org/10.5194/acp-11-12901-2011.

Tao, J., and Coauthors, 2014: PM$_{2.5}$ pollution in a megacity of southwest China: Source apportionment and implication. Atmospheric Chemistry and Physics, 14, 8679–8699, https://doi.org/10.5194/acp-14-8679-2014.

Tong, L., and Coauthors, 2017: Characteristics of surface ozone and nitrogen oxides at urban, suburban and rural sites in Ningbo, China. Atmospheric Research, 187, 57–68, https://doi.org/10.1016/j.atmosres.2016.12.006.

Wang, H., and Coauthors, 2021b: A long-term estimation of biogenic volatile organic compound (BVOC) emission in China from 2001–2016: The roles of land cover change and climate variability. Atmospheric Chemistry and Physics, 21, 4825–4848, https://doi.org/10.5194/acp-21-4825-2021.

Wang, N., X. P. Lyu, X. J. Deng, X. Huang, F. Jiang, and A. J. Ding, 2019a: Aggravating O$_3$ pollution due to NOx emission control in eastern China. Science of the Total Environment, 677, 732–744, https://doi.org/10.1016/j.scitotenv.2019.04.
Wang, S., N. D. Ding, R. B. Wang, S. Y. Xie, and X. Zhang, 2012: Study on the settings of ambient air quality monitoring sites in China. *Environment and Sustainable Development, 37*, 21–25, https://doi.org/10.3969/j.issn.1673-288X.2012.04.005. (in Chinese with English abstract)

Wang, S. J., C. S. Zhou, Z. B. Wang, K. S. Feng, and K. Hubacek, 2017b: The characteristics and drivers of fine particulate matter (PM$_{2.5}$) distribution in China. *Journal of Cleaner Production, 142*, 1800–1809, https://doi.org/10.1016/j.jclepro.2016.11.104.

Wang, T., L. K. Xue, P. Brimblecombe, Y. F. Lam, L. Li, and L. Zhang, 2017a: Ozone pollution in China: A review of concentrations, meteorological influences, chemical precursors, and effects. *Science of the Total Environment, 575*, 1582–1596, https://doi.org/10.1016/j.scitotenv.2016.10.081.

Wang, W. J., X. Li, M. Shao, M. Hu, L. M. Zeng, Y. S. Wu, and T. Y. Tan, 2019c: The impact of aerosols on photolysis frequencies and ozone production in Beijing during the 4-year period 2012–2015. *Atmospheric Chemistry and Physics, 19*, 9413–9429, https://doi.org/10.5194/acp-19-9413-2019.

Yuan, G. H., and W. X. Yang, 2019: Evaluating China’s air pollution control policy with extended AQI indicator system: Example of the Beijing-Tianjin-Hebei Region. *Sustainability, 11*, 939, https://doi.org/10.3390/su11030939.

Zeng, P., and Coauthors, 2018: Causes of ozone pollution in summer in Wuhan, Central China. *Environmental Pollution, 241*, 852–861, https://doi.org/10.1016/j.envpol.2018.05.042.

Zhai, S. X., and Coauthors, 2019: Fine particulate matter (PM$_{2.5}$) trends in China, 2013-2018: Separating contributions from anthropogenic emissions and meteorology. *Atmospheric Chemistry and Physics, 19*, 10 031–10 041, https://doi.org/10.5194/acp-19-10031-2019.

Zhao, S. P., Y. Yu, D. Y. Yin, D. H. Qin, J. J. He, and L. X. Zhang, 2014: Seasonal and diurnal variations of atmospheric peroxyacetyl nitrate, peroxypropionyl nitrate, and carbon tetrachloride in Beijing. *Journal of Environmental Sciences, 26*, 65–74, https://doi.org/10.1016/S1001-0742(13)63824-4.

Zheng, P., and Coauthors, 2019: Drivers of improved PM$_{2.5}$ air quality in China from 2013 to 2017. *Proceedings of the National Academy of Sciences of the United States of America, 116*, 24 463–24 469, https://doi.org/10.1073/pnas.1907951116.

Zhang, Z. H., G. X. Zhang, S. F. Song, and B. Su, 2020: Spatial heterogeneity influences of environmental control and informal regulation on air pollutant emissions in China. *International Journal of Environmental Research and Public Health, 17*, 4857, https://doi.org/10.3390/ijerph17134857.

Zhai, S. X., and Coauthors, 2018: Trends in China’s anthropogenic emissions since 2010 as the consequence of clean air actions. *Atmospheric Chemistry and Physics, 18*, 4 095–4 111, https://doi.org/10.5194/acp-18-4095-2018.

Zhong, L.-J., J. Y. Zheng, G. Q. Lei, J. Chen, and W. W. Che, 2007: Analysis of current status and trends of air quality monitoring in the Beijing-Tianjin-Hebei Region. *Adv. Atmos. Sci., 24*, 1313–1324, https://doi.org/10.1007/s00376-006-11315-2.