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Spatial characteristics of change trends of air pollutants in Chinese urban areas during 2016–2020: The impact of air pollution controls and the COVID-19 pandemic

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1. Introduction

Ambient air pollution is a threat to human health and climate. According to the World Health Organization (WHO), 4.2 million deaths occur annually due to air pollution (WHO, 2018), and people in developing countries are exposed to higher levels of air pollution. Fine particles (PM$_{2.5}$) (Chen et al., 2017a; Samoli et al., 2005), PM$_{10}$ (Chen et al., 2017b), nitrogen dioxide (NO$_2$) (Meng et al., 2021), SO$_2$ (Amsalu et al., 2019; Orellano et al., 2021), carbon monoxide (CO) (Dalessio et al., 2021), and ozone (O$_3$) (Wang et al., 2021a) levels have been found to be related to premature mortality, especially to cardiovascular and respiratory mortality.

Air pollution is a threat to public health in China, and several actions and plans have been implemented by Chinese authorities in recent years to mitigate it. This study examined the spatial distribution of changes in urban air pollutants (UAP) in 336 Chinese cities from 2016 to 2020 and their responses to air pollution controls and the COVID-19 pandemic. Based on the harmonic model, decreases in fine particles (PM$_{2.5}$), inhalable particles (PM$_{10}$), nitrogen dioxide (NO$_2$), sulfur dioxide (SO$_2$), and carbon monoxide (CO) levels were found in 90.7%, 91.9%, 94.3%, and 94.3%, and 88.7% of cities, respectively, while an increase in ozone (O$_3$) was found in 87.2% of cities. Notable spatial heterogeneity was observed in the air pollution trends. The greatest improvement in air quality occurred mainly in areas with poor air quality, such as Hebei province and its surrounding cities. However, some areas (i.e., Yunnan and Hainan provinces) with good air quality showed a worsening trend. During the 13th Five-Year Plan period (2016–2020), the remarkable effects of PM$_{2.5}$ and SO$_2$ pollution control plans were confirmed. Additionally, economic growth in 74.2% of the Chinese provinces decoupled from air quality after implementing pollution control measures. In 2020, several Chinese cities were locked down to reduce the spread of COVID-19. Except for SO$_2$, the national air pollution in 2020 improved to a greater extent than that in 2016–2019; In particularly, the contribution of simulated COVID-19 pandemic to NO$_2$ reduction was 66.7%. Overall, air pollution control actions improved urban PM$_{2.5}$, PM$_{10}$, SO$_2$, and CO, whereas NO$_2$ was reduced primarily because of the COVID-19 pandemic.

In recent years, China has suffered from severe air pollution owing to rapid urbanization and industrialization (Yuan et al., 2018; Zhao et al., 2019). In 2017, only 29.3% of Chinese cities met their national air quality standards (Ministry of Ecology and Environment, 2018). In addition, O$_3$ is a hazardous air pollutant with an increasing trend in China (Li et al., 2019; Zhao et al., 2021). To address the air pollution issue, China has implemented a series of actions, such as the “Air Pollution Prevention and Control Action Plan” (2013–2017) and “Blue Sky Defense Battle” (2018–2020). In addition, the State Council of China issued the “13th Five Year Plan (2016–2020) for the protection of the ecological environment” (13th FYP) to tackle the problem of air pollution.

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Several studies assessed the effects of action plans on air pollution between 2013 and 2017. These studies also showed reduced exposure to air pollution and mortality (He et al., 2021; Huang et al., 2018; Zhang et al., 2019). In addition, the COVID-19 pandemic introduced global changes in human activities in 2020, resulting in a decline in air pollution in many cities (Chauhan and Singh, 2020; Filonchyk et al., 2020a; Zeng and Wang, 2022). Currently, an accurate assessment of the pollution in many cities (Chauhan and Singh, 2020; Filonchyk et al., 2019). In addition, the COVID-19 pandemic introduced global changes in human activities in 2020, resulting in a decline in air pollution in many cities (Chauhan and Singh, 2020; Filonchyk et al., 2019). In this study, the harmonic model, this study aimed to (i) identify the overall change trends and the COVID-19 pandemic. Therefore, this study focused on exploring the driving factors of air pollution changes during the 13th FYP period and comprehensively considered the impact of air pollution reduction plans and the COVID-19 pandemic. As urban areas are the main contributors and are also highly sensitive areas to air pollution, based on ground observation data that are mainly distributed in urban areas combined with the harmonic model, this study aimed to (i) identify the overall change trends and the varying trends in PM$_{2.5}$, PM$_{10}$, NO$_x$, SO$_2$, CO, and O$_3$ levels in urban areas under different economic developments in mainland China from 2016 to 2020; (ii) investigate the spatial heterogeneity of urban air pollutants (UAP) and their response to air pollution control plans; and (iii) evaluate the effect of the COVID-19 pandemic on air quality from a national perspective.

2. Materials and methods

2.1. Data sources

Table 1 lists the data sources used in this study. PM$_{2.5}$, PM$_{10}$, NO$_x$, SO$_2$, CO, and O$_3$ levels and AQI data from 2016 to 2020 for Chinese cities were obtained from the China National Environmental Monitoring Center, a platform for providing air pollution data based on in-situ stations located primarily in urban areas. A continuous 8-h moving average of daily maximal O$_3$ concentrations was used to describe O$_3$ levels, whereas a 24-h average concentration was used to represent daily levels of other pollutants. Monthly average air pollutant levels were calculated by averaging daily data, and then monthly data was used to calculate the annual average. <80% of the daily data for that month would be considered as missing values. In addition, AQI can quantitatively describe the air pollution level using the calculation method issued by the Technical Regulation on Ambient Air Quality Index (HJ663–2012).

In recent decades, China’s rapid economic growth and enormous energy consumption have been accompanied by serious air pollution. To determine the contribution of the rising economy and energy consumption to the deterioration of air quality, national GDP and energy consumption data were obtained from the National Bureau of Statistics.

2.2. Principal component analysis (PCA)

Given that SO$_2$, NO$_x$ and VOCs are precursors of PM$_{2.5}$ (Hodan and Barnard, 2004), the reduction of these pollutants could mitigate PM$_{2.5}$ pollution issue. Therefore, to quantify the mitigation effect of air pollution reduction plans on the PM$_{2.5}$ pollution during the 13th FYP period, this study used PCA to reduce the dimensionality of provincial air pollutant (SO$_2$, NO$_x$, and VOCs) emission reduction plans. PCA is a technique for reducing the dimensionality of datasets and improving data interpretability while minimizing information loss (Jolliffe and Cadima, 2016). PCA is defined as an orthogonal linear transformation of the data into a new coordinate system such that the maximum variance of a scalar projection of the data is at the first coordinate (called the first principal component, PC1), the second largest variance is at the second coordinate, and so on (Jolliffe, 2002). PC1 is defined as the direction that maximizes the variance of the projected data. With provincial air pollutant (SO$_2$, NO$_x$, and VOCs) emission reduction plans as input data, the analysis was run in R (4.0.2).

2.3. Harmonic model

There are apparent seasonal variations in air pollution owing to anthropogenic emissions and changes in meteorological conditions (Xiang et al., 2019; Yang et al., 2017). In this study, the harmonic model was used to capture the inter-annual and seasonal variations of air pollutants in China, which was derived from Wu et al. (2012).

\[ y_i = a_i + b_i \sin(\omega t + c_i) + d_i \cos(\omega t - \theta_i) \]  \( (1) \)

\[ ac_i = 12 \times b \]  \( (2) \)

\[ asc_i = 2 \times (12 \times d) \]  \( (3) \)

where \( y \) is the monthly concentration of air pollutant \( i \) in the study city; \( a \) is the basic concentration of air pollutant; \( b \) is the monthly trend of air pollutant concentration; \( t \) is time expressed as consecutive months (1, 2, ..., 60 in this study), \( \omega \) is the frequency (2\( \pi \)/12 radians per month) that applies to a period of one year, and \( c \) corresponds to the change rate associated with the annual maximum monthly air pollutant concentration (Liu et al., 2015; Wu et al., 2012); \( \theta \) is the angular location of the month with maximum air pollutant level, calculated by the difference between the month and December and multiplied by \( \omega \); \( d \) is the angular location of seasonal variation; \( d \) is the amplitude change rate of seasonal variation; and \( ac \) is the annual change of air pollutant level in each city and assigns stands for the change of seasonal cycle during a year, both calculated by accumulating monthly changes.

The harmonic model was run in MATLAB (2019a) to simulate air pollution change trends in different cities and for different air pollutants from 2016 to 2020. In this study, \( \theta \) was taken as a constant, calculated from the mean value of the angle of each year, and \( d \) was optimized in the range of \( \theta_1 \times 2\pi/12 \) to \( \theta_1 + 2\pi/12 \).

2.4. Quantifying the effects of COVID-19 lockdown on the trend of air pollutants

To quantify COVID-19’s contribution to air pollution during the 13th FYP, it is necessary to investigate how the pandemic changed human activities and influenced air pollution. First, for the period 2016–2019, the harmonic model was conducted to evaluate the change

Table 1

| Data               | Year      | Time resolution | Spatial resolution | Data Source                                       |
|--------------------|-----------|-----------------|--------------------|--------------------------------------------------|
| Air pollutants     | 2016–2020 | 1 day           | City               | China National Environmental Monitoring Center Platform (https://www.cnemc.cn) |
| PM$_{2.5}$, PM$_{10}$, NO$_x$, SO$_2$, CO, O$_3$, AQI | 2016–2020 | 1 year          | National, Province | National Bureau of statistics (http://www.stats.gov.cn/) |
| Economic development | 2016–2020 | 1 year          | National, Province | National Bureau of statistics (http://www.stats.gov.cn/) |
| GDP, industry GDP  | 2016–2020 | 1 year          | National, Province | National Bureau of statistics (http://www.stats.gov.cn/) |
| Energy consumption | 2016–2020 | 1 year          | National, Province | National Bureau of statistics (http://www.stats.gov.cn/) |
| Coal, petroleum    | 2016–2020 | 1 year          | National, Province | National Bureau of statistics (http://www.stats.gov.cn/) |

GDP: Gross domestic production.
trend of air pollutants without COVID-effects, assuming that the change trend in 2020 were consistent with the previous four years. The second step involved quantifying the COVID-19 effect on air pollutant concentrations and measuring COVID-19 effect by calculating the relative contribution of the simulated total change and the assumed total change (Eq. 4).

$$rc = \frac{tc - atc}{rc}$$  \hspace{2cm} (4)

where \(rc\) corresponds to the relative contribution of the COVID-19 outbreak to the change in each air pollutant during the 13th FYP; \(rc\) is the total change in air pollutants during the 13th FYP, and \(atc\) is the assumed total change in air pollutant concentrations during the 13th FYP, which was cumulated by the monthly trend simulated by the harmonic model that was applied with data from 2016 to 2020 and 2016 to 2019, respectively.

3. Results and discussion

3.1. The overall change trend of UAP in China during 2016–2020

From 2016 to 2020, the concentrations of PM\(_{2.5}\), PM\(_{10}\), NO\(_2\), SO\(_2\), and CO (i.e., PM\(_{2.5}\), PM\(_{10}\), NO\(_2\), SO\(_2\), and CO in urban areas) all showed a decreasing trend (Table 2), whereas O\(_3\) showed an increasing trend. The density plot of the air pollutants (except O\(_3\)) also gradually narrowed (Fig. S1). Specifically, compared to 2016, PM\(_{2.5}\), PM\(_{10}\), NO\(_2\), SO\(_2\), CO, and AQI levels in 2020 decreased by 22.9%, 22.8%, 12.0%, 50.0%, 24.2%, and 7.8%, respectively; SO\(_2\) decreased the most, followed by PM\(_{10}\), PM\(_{2.5}\), CO, and NO\(_2\). In contrast, O\(_3\) increased by 13.3%.

Furthermore, according to the Environmental Air Quality Standard of China (GB3095–2012), 37.3%, 51.0%, and 88.1% of urban areas in cities met the national level II standard of annual average PM\(_{2.5}\), PM\(_{10}\), and NO\(_2\) in 2016, respectively, whereas 61.2%, 76.7%, and 98.2% of the urban areas satisfied the standards in 2020. From 2016 to 2020, the ratio of cities that meet level I standard of SO\(_2\) increased from 62.1% to 95.5%.

It is noteworthy that the annual consumption of coal and petroleum in mainland China was growing year by year at the same time (Fig. 1). GDP and industrial GDP also grew steadily for the past five years (National Bureau of Statistics, 2020), with a lower increase rate in 2020. China has committed to decoupling the relationship between its economy, energy growth, and air pollution levels. Although the energy structure has been optimized in recent years, coal continues to be the dominant energy source in China (National Bureau of statistics, 2020), accounting for 56.8% of the total energy consumption. From a national perspective, PM\(_{2.5}\), PM\(_{10}\), NO\(_2\), and SO\(_2\) were decoupled from current economic growth and energy consumption, indicating that air pollution controls could offset the additional emissions caused by growing economic activities and energy consumption.

Despite the decoupling of GDP growth and air quality in China on a national scale, rapid economic growth still led to the deterioration of air quality in some areas, such as Yunnan and Fujian provinces (Fig. 1b), which ranked second (64.8%) and fourth (53.9%) in relative growth in GDP in 2020 compared to that in 2016. Similarly, the reduction of air pollutants (except for O\(_3\)) in Tianjin, northeast China (Jilin and Heilongjiang provinces), and Inner Mongolia was accompanied by economic recession. In contrast, high air quality improvements and rapid economic growth (45.0%) were observed in Beijing, the capital of China. From 2010 to 2017, precursors of O\(_3\) non-methane volatile organic compounds (NMVOCs) increased by 11% owing to the growing use of solvents (Zheng et al., 2018). The reductions caused by emission controls on residential and transport activities were far outweighed by the increasing use of solvents (Zheng et al., 2018), which may be the reason for the growth of O\(_3\) levels in VOC-sensitive regions. Therefore, we can summarize this by comparing the AQI of each province in 2020 with that in 2016, under the effects of emissions controls, economic development and air quality decoupled in 74.2% of regions in China.

### 3.2. Spatial heterogeneity of annual change of UAP and its response to the air pollution reduction plans

A harmonic model was applied to individual cities to simulate the annual evolution of each air pollutant. As depicted in Fig. 2, some simulated air pollutant trends showed a relatively high degree of change.
during the 13th FYP; the average fitted parameters are listed in Table 3. The results showed that the model could effectively explore the seasonal and interannual variation tendency of air pollutants, with the coefficient of determination ($R^2$) ranging from 0.58 to 0.69. The negative $b$ (monthly change) of PM$_{2.5}$, PM$_{10}$, NO$_2$, SO$_2$, and CO indicated a descending trend in those air pollutants levels, of which the simulated annual reductions ($12 \times b$) were 2.3, 3.7, 0.7, 2.5 $\mu$g m$^{-3}$ yr$^{-1}$ and $-0.056$ mg m$^{-3}$ yr$^{-1}$ respectively. The positive $b$ value of O$_3$ indicated an increasing trend during the study period, with an annual increase of 2.6 $\mu$g m$^{-3}$ yr$^{-1}$. In addition, there were decreases in the national annual seasonal cycle ($12 \times 2d$) of 1.4, 3.6, 3.6 $\mu$g m$^{-3}$ yr$^{-1}$, and 0.026 mg m$^{-3}$ yr$^{-1}$ for PM$_{2.5}$, PM$_{10}$, SO$_2$ and CO, respectively. In contrast, the seasonal cycle of NO$_2$ and O$_3$ increased by 0.7 and 4.8 $\mu$g m$^{-3}$ yr$^{-1}$. Accordingly, the decreasing seasonal variation may be due to falling air pollutants in winter, and the increasing seasonal variation for O$_3$ may be due to rising levels in summer (Tian et al., 2020).

With province as the statistical unit, the largest simulated annual reduction in PM$_{2.5}$ (7.2 $\mu$g m$^{-3}$ yr$^{-1}$) and NO$_2$ (3.3 $\mu$g m$^{-3}$ yr$^{-1}$) were found in Beijing (Fig. 3), followed by Hebei province. In addition, the annual PM$_{10}$ reduction in Hebei province was also the largest (8.8 $\mu$g m$^{-3}$ yr$^{-1}$). It should be noted that the Beijing-Tianjin-Hebei region (BTH) has been the most polluted region in China for years. Since 2013, industry-related emissions, including those from coal-fired power plants and steel, iron, and glass industries, have been regulated (Zhang et al., 2020).

![Fig. 2. Trends of monthly air pollution change in urban areas of different cities. The black circles depict the observed monthly value; the red lines denote the simulated monthly value by the harmonic model. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article).](image)

![Fig. 3. Simulated annual change of urban air pollutants (UAP) during 2016–2020 in each province. Units for PM$_{2.5}$, PM$_{10}$, NO$_2$, SO$_2$, O$_3$: $\mu$g m$^{-3}$ yr$^{-1}$; unit for CO: mg m$^{-3}$ yr$^{-1}$.](image)

**Table 3**
The average fitted parameters of the Harmonic model for all cities.

| Pollution type | $a$  | $b$  | $c$  | $d$  | $e$  | $R^2$ | RMSE |
|---------------|------|------|------|------|------|-------|------|
| PM$_{2.5}$    | 43.50| 0.19 | 21.22| 0.49 | -0.06| 0.38  | 0.69 | 9.22 |
| PM$_{10}$     | 75.47| 0.31 | 29.16| 0.50 | -0.15| 0.38  | 0.63 | 13.98|
| NO$_2$        | 28.56| 0.06 | 7.20 | 0.11 | 0.03 | 0.13  | 0.58 | 4.93 |
| SO$_2$        | 20.67| -0.21| 10.34| 0.37 | 0.15 | 0.37  | 0.64 | 3.97 |
| CO            | 0.97 | -0.0046| 0.25 | 0.47 | -0.0011| 0.27 | 0.60 | 0.15 |
| O$_3$         | 80.33| 0.22 | 24.68| -0.50| 0.20 | -0.60 | 0.68 | 13.36|
It is important to note that air quality in Yunnan province slightly deteriorated, the only province where the average PM$_{2.5}$ and PM$_{10}$ concentrations were increasing (0.4 and 0.6 μg m$^{-3}$ yr$^{-1}$, respectively).

Interestingly, the phenomenon of “as one falls, another rise” occurred. For example, the annual decrease of SO$_2$ (10.6 μg m$^{-3}$ yr$^{-1}$) and CO (0.15 mg m$^{-3}$ yr$^{-1}$) in Shanxi province ranked first in China, but the increase of NO$_2$ (0.5 μg m$^{-3}$ yr$^{-1}$) and O$_3$ (5.7 μg m$^{-3}$ yr$^{-1}$) both ranked second in China. Coal combustion is the primary source of SO$_2$ (Ai et al., 2021), and the Shanxi province has been the area with the most severe SO$_2$ pollution in recent years, while the average SO$_2$ concentration in Shanxi met the national level I standard in 2020; In other words, the desulfurization policy effectively reduced SO$_2$ emissions in the Shanxi province. A similar phenomenon occurred in Ningxia region, with the decrease of SO$_2$ (second, 5.9 μg m$^{-3}$ yr$^{-1}$) accompanied with the increase of NO$_2$ (first, 0.6 μg m$^{-3}$ yr$^{-1}$). The rapid growth of motor vehicles may be responsible for the high levels of O$_3$ and NO$_2$ in the Shanxi province (Yan and Peng, 2016). In addition, O$_3$ showed an increasing trend across all the provinces.

Fig. 4. Spatial distribution of annual variation in PM$_{2.5}$, PM$_{10}$, NO$_2$, SO$_2$, CO, and O$_3$. Units for PM$_{2.5}$, PM$_{10}$, NO$_2$, SO$_2$, O$_3$: μg m$^{-3}$ yr$^{-1}$; unit for CO: mg m$^{-3}$ yr$^{-1}$.
From the perspective of prefecture-level cities, decreases in PM$_{2.5}$, PM$_{10}$, NO$_2$, SO$_2$, and CO levels were found in 90.7%, 91.9%, 75.2%, 94.3%, and 88.7% of cities, respectively, whereas an increase in O$_3$ was found in 87.2% of cities (Fig. 4). The highest increase and decrease in PM$_{10}$ were found in the cities of Xinjiang, namely Hotan and Kashgar. There are large desert areas in the region where PM, especially PM$_{10}$, is vulnerable to dust-stormy events. The cities with high PM$_{2.5}$, and NO$_2$ reduction were concentrated in the BTH and its surrounding area, among which Baoding had the greatest reduction in PM$_{2.5}$ up to 10.2 μg m$^{-3}$ yr$^{-1}$, and Xingtai had the greatest reduction in NO$_2$, by 4.4 μg m$^{-3}$ yr$^{-1}$. In contrast, the increases in PM$_{2.5}$ were mainly found in southwest China, especially in Yunnan province, where 11 out of 16 cities had a growth in PM$_{2.5}$. Moreover, Shanxi and its surroundings had a great reduction in SO$_2$ and CO, and eight of the top ten cities with the highest decreases in SO$_2$ were all in Shanxi province, ranging from 9.6 to 16.2 μg m$^{-3}$ yr$^{-1}$. In contrast to other pollutants, increasing O$_3$ was found in most cities, and the high values were mainly concentrated in the cities of Tibet and Shanxi province. Notably, Ordos (Inner Mongolia), Nuijiang and Pu’er (Yunnan province) experienced increases in all air pollutants (except SO$_2$) from 2016 to 2020.

During the 13th FYP, China promulgated a series of actions to address air pollution issues, especially in key areas such as the BTH and its surrounding areas (“2 + 26” cities), the Fenwei Plain and the Yangtze River Delta (Fig. 5a). These actions can be summarized as follows: (i) optimization of the industrial structure, including the coal-fired power plants that had already met the ultra-low emission transformation of 890 million KW by the end of 2019. The "2+26" cities and Fenwei Plain identified >70 thousand enterprises as ‘scattered pollution’, and special investigations were conducted, and enterprises that failed to meet the standard were cleaned up and rectified. In addition, small and inefficient boilers were eliminated in the key area, and energy-saving and ultra-low emissions were transformed into larger boilers. (ii) Promotion of clean and low-carbon energy in residential areas. >25 million households that use coal bulk had been renovated. Instead, the use of natural gas and electricity was encouraged. (iii) Strengthening of vehicle emission control. The VI vehicle emission standards for light vehicles in China were applied nationwide, and clean energy vehicles were actively promoted during the 13th FYP (The State Council Information Office of China, 2020).

In the 13th FYP, China outlined its goals, including reducing nationwide emissions of SO$_2$, NO$_x$, and VOCs (volatile organic compounds) by 15%, 15%, and 10%, respectively. Furthermore, each province set the planned emission reduction ratios for the 13th FYP (Fig. 5b). Among the regions, BTH and its surrounding areas set the most stringent targets for reduction ratios of SO$_2$, NO$_x$, and VOCs emissions, as the air pollution in this region has been the most serious in recent years. In particular, for Beijing, the capital of China, the targets for reducing SO$_2$ and VOCs ranked first. In addition, there are 16 provinces that set targets for reducing VOCs emissions, which are the major precursors of O$_3$.

The first principal component (PC1) of the planned air pollutant (NO$_x$, SO$_2$, and VOCs) emission reduction ratios accounted for >85%, indicating that PC1 was able to reflect most of the original data. Therefore, PC1 was used to fit the relationship between the provincial air pollution reduction plans and annual PM$_{2.5}$ change. The relationships between provincial planned NO$_x$, SO$_2$, and VOCs reduction ratios and annual NO$_x$, SO$_2$, and O$_3$ changes are illustrated in Fig. 6. Significant relationships were found between air pollution reduction plans and PM$_{2.5}$, NO$_x$, and SO$_2$, while the model of PM$_{2.5}$ had the highest fitness. Based on the slopes of the linear models, the efficiency of the NO$_x$ reduction plans was much lower than that of SO$_2$ and PM$_{2.5}$. In addition, the VOCs control plans failed to reduce O$_3$ effectively.

The results of the present study revealed the effectiveness of action plans, especially for SO$_2$, which peaked in 2006, while 95.5% of cities met the level I standard (<20 μg/m$^3$) in 2020. The reductions in PM$_{2.5}$ in the BTH were the greatest, where the strictest controls were implemented. However, 70% of cities’ urban areas in the BTH region still failed to satisfy the PM$_{2.5}$ level II standard (<35 μg/m$^3$) of GB3095–2012 in 2020. Among the air pollutants with a declining trend, the rate of decrease in NO$_2$ was the lowest. Traffic emissions are the main contributor of NO$_x$ (NO + NO$_2$) (Agudelo-Castaneda et al., 2020; Gao et al., 2021; Jin et al., 2021). Zhang et al. (2019) pointed out certain control actions in the industrial sector that were effective from 2013 to 2017, such as strengthening industrial emission standards, upgrading industrial boilers, and ‘phasing out outdated industrial capacities’. However, industrial combustion sources lacked effective controls on NO$_x$ (Zheng et al., 2018).

Although 16 out of 31 provinces had set the goal of reducing VOCs emissions, an increase in O$_3$ occurred in all provinces and it was considered to be the main daily primary pollutant in China. Addressing O$_3$ pollution is challenging because of the complexity of atmospheric photochemical formation, which is sensitive to NO$_x$, VOCs, and meteorological conditions (Tang et al., 2012). There is a nonlinear relationship between O$_3$, VOCs and NO$_x$, and the formation of O$_3$. Individual regions can be classified into ‘VOCs-sensitivity’ and ‘NO$_x$-sensitivity’ (Sillman, 1999). The O$_3$ concentration could also be affected by PM$_{2.5}$, as the reduced PM$_{2.5}$ suppresses the termination reactions of O$_3$-consuming radicals, thereby increasing photochemical reactions and O$_3$ concentrations (Lu et al., 2020). On the other hand, reduced particle surfaces represent less heterogeneous reactions involving atmospheric oxidants occur, which may lead to termination reactions of some O$_3$-consuming radicals (Lou et al., 2014). On the other hand, the reduced
aerosol radiative effect induces an increase in solar radiation that cause higher photolysis rates and more O$_3$ production (Benas et al., 2013). It is worth mentioning that the mitigation of PM$_{2.5}$ is one of the long-term goals for “beautiful China”. Given the current air pollution issues, it is particularly necessary for individual regions to launch targeted collaborative control of VOCs, NO$_x$, and PM$_{2.5}$.

### 3.3. The effect of the COVID-19 pandemic in 2020 on the air pollutants total change

As of the end of 2020, COVID-19 cases had been identified in 210 countries (WHO, 2020), and the stringency policy played a significant role in diminishing the COVID-19 total cases (Bilgili et al., 2021). Many regions implemented ‘lockdown’ measures to curb the spread of COVID-19 in 2020. Meanwhile, the emissions from human activities were reduced. Based on the in-situ observed data, we found that the levels of all air pollutants decreased in 2020 compared to those in 2019 (Fig. 7). Except for SO$_2$ and CO, the greatest year-to-year decrease rates of each air pollutant were found in 2020, indicating that China’s air quality improved at a faster rate in 2020 than in previous years. In particular, NO$_2$ and O$_3$ remained stable or increased from 2016 to 2019 but decreased significantly in 2020.

Compared to 2019, the highest reduction in PM$_{2.5}$ (~7.1 µg/m$^3$) in 2020 was found in the Hubei province rather than the BTH region (Fig. 8), and a high reduction of NO$_2$ was also found in the region, second only to the capital Beijing. The COVID-19 pandemic in Hubei province was among the most serious in China, and the strictest ‘lockdown’ measures were implemented. With the spread of epidemic control, China’s anthropogenic emissions rebounded in April, and the provinces in China presented nearly synchronous declines and rebounded in the anthropogenic emissions (Zheng et al., 2020), while Hubei and the provinces surrounding Beijing recovered more slowly owing to the extension of lockdown measures (Zheng et al., 2020). Additionally, the absolute decrease in public traffic in Beijing was the greatest, with the passenger volume of public transportation decreasing by 307–498 million passengers per month from February to May 2020 (Gao et al., 2021). Therefore, the greatest NO$_2$ reduction in 2020 was found in Beijing and Hubei province.

Similar to the period of 2016–2019, the largest reductions in SO$_2$ and
The annual change of urban air pollutants (UAP) of each province in 2016–2019 and 2020. 2016–2019: simulated annual change of air pollution during 2016–2019 by the harmonic model; 2020: the annual change of air pollution in 2020 compared to 2019.

CO were also found in Shanxi province. Compared with the simulated annual change in air pollution during 2016–2019, there were 71.0%, 87.1%, 93.5%, 12.9%, 45.2%, and 93.5% provinces in 2020 experienced a greater drop or a smaller increase in PM_{2.5}, PM_{10}, NO_{2}, SO_{2}, CO, and O_3, respectively. Among these air pollutants, the largest difference between 2020 and 2016 was found for O_3 followed by NO_{2}. Furthermore, O_3 in all provinces showed an increasing trend during 2016–2019, while O_3 in 26 out of 31 provinces showed a reduction in 2020. Many studies reported either minimal to no reduction or even an increase in surface O_3 concentrations during the lockdown period (Adam et al., 2021), and the O_3 increase during the lockdown period was attributed to meteorological conditions and weak titration effects (Adam et al., 2021; Su et al., 2021). In addition, Qi et al. (2021) pointed out that the rise in ozone during the lockdown period was the combined effect of a substantial increase at night (58%–91%) and a small reduction in the daytime (1%–17%). In contrast, Pey and Cerro (2022) pointed out that the mitigation of regional O_3 pollution during the lockdown over southwestern Europe was related to the reduction in hemispheric background concentrations. A decline in O_3 concentrations was also observed in the Pearl River Delta (China), where O_3 formation rates are under a NO_x-limited regime; therefore, the reduction in NO_x would diminish O_3 generation rates (Wang et al., 2021b). It is worth noting that the rate of ozone production varies nonlinearly with VOCs and NO_x emissions, and air quality can initially worsen when NO_x emissions are reduced; however, the total amount of ozone produced ultimately increases with increasing NO_x emissions (Lin et al., 1988; Wei et al., 2022). Similarly, Wei et al. (2022) found that O_3 concentrations have increased since 2015 but declined in 2020, which was attributed to the coordinated control of air pollution and ongoing COVID-19 effects. NMVOCs emissions in China were estimated to decrease by 2–26% between January and March 2020 compared to that in the same period in 2019 (Zheng et al., 2020), combined with the reduction in NO_x and changes in meteorological conditions (Ding et al., 2023), which could explain the reason behind the reduction in O_3 in 2020.

In China, the COVID-19 pandemic in Hubei province was the most serious. The cities in Hubei province were under lockdown in February and March 2020. As shown in Fig. S2, the air pollution levels in Hubei province varied considerably by season. Except for O_3, the lowest monthly air pollution level of 2016–2019 average value was found in July, and O_3 reached its highest value in summer. In 2020, the lowest PM_{2.5}, PM_{10}, SO_{2}, CO, and AQI values were found in summer (July or August). In contrast, the lowest monthly levels of NO_2 were found during the lockdown, which is consistent with previous research investigating the effect of the lockdown on NO_2 (Gao et al., 2021; Sulaymon et al., 2021). As vehicle emissions are a major source of NO_2, lockdowns severely restrict travel. During the summer, favorable meteorological conditions had a greater impact on reducing PM_{2.5}, PM_{10}, SO_{2}, and CO levels relative to the lockdown period, demonstrating the important influence of meteorological conditions on seasonal air quality.

In addition, we compared the two models, which were applied to the data for the 2016–2019 and 2016–2020 periods, respectively. The simulated annual average drop in PM_{2.5}, PM_{10}, NO_{2}, and CO levels in 2016–2020 was greater than that in 2016–2019, and the simulated O_3 levels in 2016–2020 increased less than that in 2016–2019. Assuming that the change in air pollution in 2020 was the same as that in the previous four years, the assumed total changes are presented in Table 4. Based on the difference between the total change and the assumed total change, the 2020 COVID-19 pandemic amplified the reduction in the national average PM_{2.5}, PM_{10}, NO_{2}, O_3 and CO by 0.6, 4.2, 2.4, 7.2 μg/m^3 and 0.01 mg/m^3, respectively. Compared with the total reduction during the 13th FYP period, COVID-19 contributed to the total reduction of PM_{2.5}, PM_{10}, NO_{2}, and CO by 5.3%, 22.6%, 66.7%, and 3.9%, respectively. In addition, the continuous growth of O_3 was temporarily suspended. In contrast, SO_{2} did not experience a greater reduction in 2020 compared to that in 2016–2019. On the one hand, SO_{2} failed to show a remarkable decrease in several megacities during the lockdown period (Gao et al., 2021; Filipchyk et al., 2020b) also found that SO_{2} emissions decreased slightly from January to March 2020 compared to those in 2019, as SO_{2} was mainly produced by burning coal (Filipchyk et al., 2020b), domestic heating, and industries such as power plants, steel, and coke that have uninterrupted production processes and need to operate during the lockdown period (Ministry of Ecology and Environment of the People’s Republic of China, 2020). On the other hand, from 2016 to 2019, the average SO_{2} concentration dropped from 20.2 μg/m^3 to 11.3 μg/m^3, which satisfied the level I standard of SO_{2} (20 μg/m^3), and the COVID-19 pandemic that may weaken the effectiveness of SO_{2} control. Overall, PM_{2.5}, PM_{10}, NO_{2}, and CO pollution was reduced to a greater extent in 2020 than in 2016–2019. The air quality improved on a national scale as a result of the COVID-19 outbreak.

During 2016–2020, the concentrations of PM_{2.5}, PM_{10}, SO_{2}, and CO in China were effectively reduced, although most regions experienced economic growth, highlighting the success of the action plans. Anthropogenic emissions were reduced as a result of the outbreak of COVID-19, with NO_x illustrating the largest reductions among the air pollutants (Zheng et al., 2020), which could explain the COVID-19 pandemic contributed the most to NO_x reduction during the 13th FYP. The total increase in O_3 during the 13th FYP also declined owing to the pandemic. Although some meteorological factors also influence the variation in air quality, such as monthly anomalies and seasonal variations, their effect on the national 5-year interannual variation is relatively small (Zhang et al., 2020).

### Table 4

| PM_{2.5} | PM_{10} | NO_{2} | SO_{2} | CO | O_3 |
|----------|---------|-------|-------|----|-----|
| 10.8     | 14.4    | 1.2   | 15.0  | 0.28| 20.4|
| -11.4    | -18.6   | -3.6  | -12.6 | -0.28| 13.2|
| -0.6     | -4.2    | -2.4  | 2.4   | -0.01| -7.2|
| 5.3      | 22.6    | 66.7  | -19.0 | 3.9 | 54.5|

Units for PM_{2.5}, PM_{10}, SO_{2}, O_3: μg/m^3, unit for CO: mg/m^3; the negative numbers indicate that the air pollutant reduction decreased in 2020 compared with that in 2016–2019.
especially for NO2, meteorological factors have little influence on the variation (Gao et al., 2021). Thus, meteorological factors were not considered to quantify the interannual changes in this study. However, for secondary pollutants, such as O3, the simulated effect of the COVID-19 may not be sufficiently accurate. Overall, PM2.5, PM10, SO2, and CO improved owing to the implementation of air pollution controls, while the reduction in NO2 was primarily because of the COVID-19 lockdown. The effectiveness of the 13th FYP NOx control plan would be overestimated if based only on the NO2 concentration in 2020. The long-term effect of NO2 on mortality is as great as that of PM2.5 (Faustini et al., 2014). In 2021, the World Health Organization further tightened the NO2 guidelines (from 40 μg/m3 to 10 μg/m3) and emphasized NO2 health hazards. Therefore, more effective national-level control measures are required to reduce NO2 emissions.

4. Conclusions

Several initiatives to mitigate air pollution were implemented during China’s 13th FYP. Under the implementation of air pollution control, economic growth decoupled from air quality in >70% of China’s regions. Based on the harmonic model, PM2.5, PM10, NO2, SO2, and CO indicated a descending trend, whereas O3 showed an increasing trend during the study period, among which the change rate of SO2 was the highest. In addition, the remarkable effects of PM2.5 and SO2 pollution control plans were confirmed, whereas the effect of NO2 control plan was slight.

In addition, the COVID-19 pandemic in 2020 led to a greater improvement in national air quality, especially for NO2. Over the period 2016–2020, the COVID-19 outbreak reduced NO2 by approximately two-thirds. In summary, PM2.5, PM10, SO2, and CO levels were greatly improved in urban areas primarily as a result of air pollution control plans, whereas NO2 levels mainly decreased following the COVID-19 pandemic. The effectiveness of the 13th FYP NOx control plan would be overestimated if based only on the NO2 concentration in 2020. More effective national-level control measures are required to reduce NO2 emissions.

CRediT authorship contribution statement

Chanchan Gao: Conceptualization, Methodology, Visualization, Writing – original draft, Writing – review & editing. Fengying Zhang: Resources, Formal analysis. Dekun Fang: Writing – review & editing. Qingtao Wang: Visualization. Min Liu: Conceptualization, Methodology, Writing – review & editing, Supervision.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

The authors do not have permission to share data.

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Appendix A. Supplementary data

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