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Air quality characteristics in Wuhan (China) during the 2020 COVID-19 pandemic

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1. Introduction

Seasonality significantly influences air quality in many cities across the world. China typically experiences worse air quality in winter and spring (Huang et al., 2020a) due to meteorological conditions (D et al., 2015; M et al., 2017), which is controlled by related pollution emission control schemes (LIU et al., 2019). Moreover, China’s air quality has continuously improved since 2013 (Yang et al., 2020a) due to the implementation of the Action Plan for Air Pollution Prevention and Control (CHAI et al., 2020). This plan aims to improve air quality by strengthening the control of industrial, non-point source (dust, catering, etc.), and mobile pollution emissions (motor vehicles and ships). According to online monitoring data, Beijing’s air quality has significantly improved in recent years, with good and excellent air quality rates in spring and winter increasing from 46.3% in 2013 to 47.1% in 2015 and 70.8% in 2019. Similar results have also been reported in Guangzhou and Shanghai (Ministry of Ecology and Environment of China, 2019). Owing to the control of regional air pollution sources in recent years, excellent and good air quality rates in Wuhan have also improved significantly from 28.1% in 2013 to 72.5% in 2019.

The global outbreak of the novel coronavirus (COVID-19) in 2020 led to the significant improvement of air quality in several countries and regions in response to enforced national lockdowns. For example, concentrations of particulate matter with a diameter <10 μm (PM10) and <2.5 μm (PM2.5) during the COVID-19 lockdown in India decreased by approximately 50% compared to pre-lockdown levels (M et al., 2020). Moreover, nitric oxide (NO) and nitrogen dioxide (NO2) concentrations in urban roads of São Paulo in Brazil decreased by 77.3% and 54.3%, respectively (N and U, 2020), and the concentrations of typical pollutants in Milan in Italy also decreased significantly (Collivignarelli et al., 2020). Several studies have also identified close relationships between typical pollutants (such as PM10, PM2.5, sulfur dioxide (SO2), and NO2)

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and the COVID-19 outbreak (Fattorini, D. et al., 2020; Bashir et al., 2020). Prolonged exposure to contaminated environments may also facilitate the spread of the virus. Hence, controlling environmental pollutant emissions is necessary for protecting both the environment and human health (Bilal et al., 2020).

Wuhan is a megacity in central China and the capital city of Hubei Province (located at 113°69′–115°07′ E and 29°98′–31°36′ N). The city had a resident population of 11.21 million in 2019 and was the first provincial capital city to lockdown in 2020 due to COVID-19. The air quality in Wuhan shows notable seasonal variability, with worse air quality in winter and spring (Wang et al., 2018a). This study aimed to determine the dominant factors influencing air quality characteristics during the COVID-19 lockdown in Wuhan using online atmospheric monitoring data and information on road traffic control and enterprise control schemes. Our study also provides a basis for the development of effective regional air quality prevention and control measures in Wuhan.

2. Methods

2.1. Data sources

Atmospheric environmental monitoring data were obtained from the Hubei Provincial Environmental Monitoring Center Station (HBEMS), which is affiliated with the Department of Ecology Environment of Hubei Province and is under the technical guidance of the China National Environmental Monitoring Center. Real-time data from monitoring stations are recorded and audited on the HBEMS online monitoring platform (http://59.172.208.250:8082/), and the monitoring equipment meets the quality control requirements to ensure data credibility (Ministry of Environmental Protection of China, 2018a; Ministry of Environmental Protection of China, 2018b; China National Environmental Monitoring Centre, 2020).

We collected the daily average concentrations of six basic air pollutants from January to May during 2017–2020, namely SO₂, NO₂, carbon monoxide (CO), ozone (O₃), PM₁₀, and PM₂.₅. The data were obtained from nine national control points in Wuhan: Donghu Liyuan, Hanyang Yuehu, Hankou Huaqiao, Wuchang Ziyang, Qingshan Ganghui, Zhuankou Xinqu, Hankou Jiangtan, Donghu Gaoxin, and Wujiaoshan. The daily average VOC concentrations were also collected for 2019 and 2020. Meteorological data from January to April 2017–2020 in Wuhan were obtained from the Wuhan Meteorological Station, including data on wind direction, wind speed, temperature, and relative humidity. Table 1 shows some of the monitoring equipment and models used in the monitoring stations of this study. The spatial distribution of the nine national control points and the location of the weather station are shown in Fig. 1.

2.2. Research period

The COVID-19 lockdown in Wuhan began on January 23, 2020, and included the implementation of strict home quarantining. Table 2 illustrates the chronology of major events during the COVID-19 pandemic in Wuhan.

In this study, January was considered as the pre-COVID-19 control

Table 1
Station monitoring equipment information.

| Equipment Name | Equipment Model | Equipment Manufacturer       |
|----------------|-----------------|-----------------------------|
| SO₂            | 43i             | Thermo Fisher Scientific     |
| NOₓ            | 42i             | Thermo Fisher Scientific     |
| CO             | API300E         | API (America)                |
| O₃             | 49i             | Thermo Fisher Scientific     |
| PM₂.₅          | Metone1020      | METONE (America)             |
| PM₁₀           | TEOM1405DF      | Thermo Fisher Scientific     |
| VOCs Analyzer | TH-300 B        | Wuhan Tianhong               |
| Meteorological Instrument | WS-600           | LUFFT (Germany)             |

Table 2
Timeline of the COVID-19 pandemic in Wuhan.

| Name                     | Date                  | Events                                                                 |
|--------------------------|-----------------------|------------------------------------------------------------------------|
| Before the COVID-19 lockdown | 12/30/2019–1/23/2020 | Occurrence of patients with unexplained pneumonia in Wuhan, followed by a rapid increment in cases. |
| During the COVID-19 lockdown | 1/23/2020–4/8/2020   | Wuhan was officially under lockdown at 10:00 on January 23; the airport and railway station were shut and residents were instructed to reduce travel. On February 11, all residential roads were closed, except for special vehicles. |
| After the COVID-19 lockdown | 4/8/2020             | At 00:00 on April 8, Wuhan officially lifted its lockdown measures, and traffic gradually resumed to facilitate the resumption of work and production enterprises. |

Fig. 1. Map of Wuhan showing the locations of the nine national control points and weather station.
period, February and March as the COVID-19 control period, and April and May as the post-COVID-19 control period.

2.3. Evaluation method

We calculated the air quality index (AQI) (Ministry of Environmental Protection of China, 2012) to evaluate the air quality characteristics in Wuhan from January to April/May during 2017–2020. AQI is an effective method for describing urban air quality and is widely used in many countries (Tan et al., 2021; Heidarinejad et al., 2018; Hu et al., 2020).

The individual air quality index (IAQI) and pollutant concentration limits (Table 3) of individual pollutants (P) are shown in Table 3. The IAQI was calculated using the following formula:

\[ IAQI_P = \frac{IAQI_{P, Hi} - IAQI_{P, Lo}}{BP_{Hi} - BP_{Lo}} (C_P - BP_{Lo}) + IAQI_{P, Lo} \]  

(1)

where \( C_P \) is the mass concentration value of pollutant \( P \), \( BP_{Hi} \) is the high limit of the pollutant concentration, \( BP_{Lo} \) is the low limit of the pollutant concentration, \( IAQI_{P, Hi} \) is the individual AQI corresponding to \( BP_{Hi} \), and \( IAQI_{P, Lo} \) is the individual AQI corresponding to \( BP_{Lo} \) in Table 3. The AQI was then calculated as the maximum value of each IAQI. The AQI was divided into six air quality levels: excellent (green), good (yellow), lightly polluted (orange), moderately polluted (red), heavily polluted (purple), and severely polluted (brown), as shown in Table 4.

For air quality, the primary pollutant is defined as the pollutant with the largest IAQI. Multiple pollutants were considered primary pollutants if they all produced the highest IAQI.

In this study, we compared the air quality of the different control periods from January to May in 2017–2020 to identify the dominant causes of significant air quality improvement in the winter and spring of 2020.

### Table 3

| Pollutant | Concentration limit |
|-----------|---------------------|
| SO\(_2\)  | 24-h average/µg/m\(^3\) 1-h average/µg/m\(^3\) |
| NO\(_2\)  | 24-h average/µg/m\(^3\) 1-h average/µg/m\(^3\) |
| NO\(_x\)  | 24-h average/µg/m\(^3\) 1-h average/µg/m\(^3\) |
| PM\(_{2.5}\) | 24-h average/µg/m\(^3\) 1-h average/µg/m\(^3\) |
| CO       | 24-h average/µg/m\(^3\) 1-h average/µg/m\(^3\) |
| O\(_3\)   | 8-h average/µg/m\(^3\) 24-h average/µg/m\(^3\) |
| O\(_3\)   | 24-h average/µg/m\(^3\) 24-h average/µg/m\(^3\) |

| Pollutant | Concentration limit |
|-----------|---------------------|
| SO\(_2\)  | 24-h average/µg/m\(^3\) 1-h average/µg/m\(^3\) |
| NO\(_2\)  | 24-h average/µg/m\(^3\) 1-h average/µg/m\(^3\) |
| NO\(_x\)  | 24-h average/µg/m\(^3\) 1-h average/µg/m\(^3\) |
| PM\(_{2.5}\) | 24-h average/µg/m\(^3\) 24-h average/µg/m\(^3\) |
| CO       | 24-h average/µg/m\(^3\) 1-h average/µg/m\(^3\) |
| O\(_3\)   | 8-h average/µg/m\(^3\) 24-h average/µg/m\(^3\) |
| O\(_3\)   | 24-h average/µg/m\(^3\) 24-h average/µg/m\(^3\) |

### Table 4

| Air Quality Index | Air Quality Index Level | Air Quality Index Categories and Colors |
|------------------|-------------------------|----------------------------------------|
| 0–50             | Level 1                 | Excellent                               |
| 51–100           | Level 2                 | Good                                    |
| 101–150          | Level 3                 | Lightly polluted                        |
| 151–200          | Level 4                 | Moderately polluted                     |
| 201–300          | Level 5                 | Heavily polluted                        |
| ≥300             | Level 6                 | Severely polluted                       |

Fig. 2. Changes in air quality and the percentage of days with an AQI of excellent and good in Wuhan from January to April 2017–2020.

### 3. Results and discussion

#### 3.1. Analysis of air quality variability

#### 3.1.1. January to April air quality variability during 2017–2020

Figs. 2 and 3 show the calculated AQI of Wuhan from January to April 2017–2020. The number of days with excellent air quality from January to April 2020 was approximately 2–3 times that during the same period in 2017–2019. Moderately and heavily polluted days were observed in 2017–2019 but not in 2020, and lightly polluted days were observed in March. As a result, excellent and good air quality rates reached 90.1% in January–April 2020, which was significantly higher than the rates during the same period in 2017, 2018, and 2019 (71.9%, 70%, and 72.5%, respectively). In particular, excellent and good air quality rates reached 100% in March 2020. As shown in Fig. 3, the AQI of Wuhan in January 2020 was significantly lower than that in January 2017–2019. Although a small number of lightly polluted days also occurred in April 2020, the maximum AQI of the lightly polluted days was similar to the AQI of good days. The cause for this significant improvement in air quality is likely related to the city-wide COVID-19 lockdown from January 23, which included strict traffic controls (preventing people entering and leaving the city), the closure of non-essential enterprises, and strict citizen home quarantine measures.

Table 5 and Fig. 4 show the monthly changes in primary pollutants in
Due to the significant increase in excellent air quality days in January–April 2020, the number of days with $\text{NO}_2$, $\text{PM}_{10}$, and $\text{PM}_{2.5}$ as the primary pollutants in 2020 were significantly lower than those in 2017–2019. However, the number of days with $\text{O}_3$ as the primary pollutant increased by 80% from 15 days in 2019 to 27 days in 2020. In particular, most days with $\text{O}_3$ as the primary pollutant in 2020 occurred in April, which is likely due to the decrease in atmospheric $\text{NO}_x$ or $\text{NO}_2$ concentrations during this period (Yu et al., 2020).

### 3.1.2. Pollutant concentration variability from January to April 2017–2020

The air quality level and primary pollutants are linked to pollutant concentrations. The average concentrations of the six typical pollutants in Wuhan from January to April during 2017–2020 are shown in Fig. 5. The concentrations of $\text{SO}_2$, $\text{NO}_2$, $\text{CO}$, $\text{PM}_{10}$, and $\text{PM}_{2.5}$ decreased by

| Primary Pollutants | 2017 | 2018 | 2019 | 2020 |
|-------------------|------|------|------|------|
|                   | days/d | percentage | days/d | percentage | days/d | percentage | days/d | percentage |
| None              | 9      | 7.50%    | 10     | 8.33%    | 11     | 9.17%    | 34     | 28.10%     |
| $\text{NO}_2$     | 15     | 12.50%   | 19     | 15.83%   | 16     | 13.33%   | 2      | 1.65%      |
| $\text{O}_3$      | 4      | 3.33%    | 4      | 3.33%    | 15     | 12.50%   | 27     | 22.31%     |
| $\text{PM}_{2.5}$ | 69     | 57.50%   | 63     | 52.50%   | 66     | 55.00%   | 48     | 39.67%     |
| $\text{PM}_{10}$  | 23     | 19.17%   | 24     | 20.00%   | 12     | 10.00%   | 9      | 7.44%      |
| $\text{PM}_{2.5}$ & $\text{O}_3$ | 0      | 0.00%    | 0      | 0.00%    | 0      | 0.00%    | 1      | 0.83%      |

*PM$_{2.5}$ and O$_3$ are sometimes listed as the primary pollutants in 2020.
17–43% since 2017 due to the effective control of air pollution emissions in recent years (Wang, 2020a). However, the average January–April concentrations of SO$_2$, NO$_2$, CO, PM$_{10}$, and PM$_{2.5}$ decreased by 6.95%, 38.23%, 18.24%, 30.25%, and 32.92%, respectively, from 2019 to 2020. NO$_2$ showed the largest decrease, followed by PM$_{10}$, PM$_{2.5}$, CO, and SO$_2$, which is consistent with previous reports (Bauwens et al., 2020).

SO$_2$ concentrations in urban areas are strongly related to emissions from metallurgical and coal-fired enterprises (Sun, 2015), while NO$_2$ concentrations are closely related to the number of motor vehicles (Samoli et al., 2019; Lu et al., 2020). PM$_{10}$ and PM$_{2.5}$ concentrations are related to industrial emissions, motor vehicle exhaust emissions, and road dust (Wang et al., 2018b; Zhao, 2019; M et al., 2020), and are also affected by SO$_2$ and NO$_2$ (Mi et al., 2019; Deng et al., 2018). Atmospheric CO concentrations are also closely related to motor vehicle emissions (Sun et al., 2017). The observed changes in NO$_2$, CO, PM$_{10}$, and PM$_{2.5}$ concentrations in this study were closely related to the restriction of motor vehicles and the suspension of non-essential enterprises during the 2020 COVID-19 lockdown (S and B, 2020; Jiang, 2017).

In contrast to the decrease in most pollutants in this study, the January–April O$_3$ concentrations increased by 22.26% from 2019 to 2020. The cause for this increase is likely related to meteorological conditions as well as the changes in NO$_2$, CO, and VOC concentrations (Yang et al., 2016; Yang et al., 2020b). The complex nonlinear relationship between O$_3$ and its precursors is a significant issue in the control of O$_3$ pollution (Ying and Ke, 2020). Based on the relationship between O$_3$ and its precursors, O$_3$ generation is typically classified as either VOC-sensitive, NOx-sensitive, or mixed-sensitive (Wang et al., 2019a). O$_3$ in Wuhan was found to be highly sensitive to VOCs (Hui L R et al., 2018). Moreover, a reduction in NOx can also increase in O$_3$ concentrations (Hossain et al., 2021).

According to the basic composition of VOCs and their classification (Mozaffar et al., 2020; Huang et al., 2020b), we divided VOCs into alkanes, alkenes, alkynes, aromatic hydrocarbons, halogenated hydrocarbons, and oxygenated organics; their average January–April concentrations in Wuhan in 2019 and 2020 are shown in Fig. 6. Due to the COVID-19 lockdown on January 23, 2020, the average January–April concentrations of TVOCs in 2020 (34.9, 19.2, 19.4, and 20.9 ppb, respectively) were significantly lower than those in 2019 (52.3, 37.5, 34.8, and 32.2 ppb, respectively), as shown in Fig. 6. Other regions in China also implemented similar lockdown measures, resulting in substantial TVOC emission reductions. Therefore, the February–March concentrations of alkanes, alkenes, alkynes, aromatic hydrocarbons, halogenated hydrocarbons, and oxygenated organics reduced by 35.2%, 45.8%, 20.3%, 61.4%, 68.5%, and 68.2%, respectively, from 2019 to 2020.

The atmospheric concentrations of alkanes, alkenes, alkynes, and aromatic hydrocarbons from February to April showed little change in
2020 (Fig. 6) but showed significant change in 2019. The low variability in 2020 was likely caused by the enforced shutdown of important VOC sources on January 23, 2020, including petrochemical (and related) industries and traffic sources.

According to Figs. 5 and 6, the reduction of both NO\textsubscript{2} and VOC concentrations from January to April 2020 did not cause a decrease in O\textsubscript{3} concentration. This phenomena has been observed in other studies and was attributed to inappropriate emission reductions (Wang et al., 2019b). Moreover, changes in the ratio of NO\textsubscript{x} to VOCs may also increase O\textsubscript{3} production (Lian et al., 2020). For example, a reduction in NO\textsubscript{x} decreased O\textsubscript{3} in rural Beijing but increased O\textsubscript{3} in urban Beijing (Shao et al., 2009). In traffic-intensive areas with high NO\textsubscript{x} emissions, the initial NO\textsubscript{x} reduction was found to first increase and then decrease the O\textsubscript{3} concentration with increasing airflow transport (Hashim et al., 2021). In addition, an increase in light flux from PM reduction may also increase O\textsubscript{3} concentrations (Wang et al., 2019b). Therefore, a recent study determined that both NO\textsubscript{x} and VOCs controlled O\textsubscript{3} production during the COVID-19 lockdown (Li, 2015; YAN, 2020). The O\textsubscript{3} generation mechanism in spring is different from that in summer and autumn in Wuhan, which is mainly controlled by VOCs. Both seasonal and weather changes can alter the form of O\textsubscript{3} production (Zhang et al., 2009). In the NO\textsubscript{x} reduction scenario, VOC reduction cannot compensate for the reduction in O\textsubscript{3} concentrations (Gualtieri et al., 2020).

If the production of O\textsubscript{3} was controlled by NO\textsubscript{x} and VOCs in April, the concentrations of NO\textsubscript{x} and VOCs must be reduced to an appropriate ratio to minimize the production of O\textsubscript{3}. However, the conditions during the COVID-19 period from January to April 2020 were unable to produce the appropriate combination of factors, which resulted in an increase in O\textsubscript{3} concentration.

3.2. Analysis of air quality impact factors

3.2.1. Meteorological factors

Air quality depends on both pollution emissions and regional meteorological conditions (Yuval et al., 2020). Changes in meteorological conditions can affect the physical processes and chemical reactions of pollutants. Wind speed and direction can affect the diffusion and transmission of pollutants (Yi et al., 2019), while temperature and humidity can affect pollutant reaction processes (Hong et al., 2018). Therefore, we produced wind rose charts for January to April 2017–2020 to illustrate the impact of residential lockdowns, traffic restrictions, and non-essential enterprise shutdowns on Wuhan’s air quality during the 2020 COVID-19 period. As shown in Fig. 7, northern wind was the dominant wind direction during January–April (which only changed slightly during 2017–2020), followed by the northeast and northwest winds. The wind speed was mainly in the range of 0–3.3 m/s. According to the wind speed classification, light winds (1.5–3.3 m/s) were dominant during this period (~40%), followed by soft winds (0.2–1.5 m/s; ~30%), and the proportion of wind with speeds greater than 3.3 m/s was approximately 20%. Therefore, wind speeds of less than 3.3 m/s dominated throughout winter and spring in Wuhan (HU et al., 2018), which is conducive to atmospheric pollutant accumulation. These conditions are the main cause for lower air quality during winter and spring in Wuhan (A D et al., 2015; M et al., 2017). High wind high speeds (>5 m/s) can also increase the external transmission of pollutants to Wuhan. According to Fig. 7, the wind speed in 2018 was generally greater than that in 2020, but the air quality in 2020 was better than that in 2017–2020.
in 2018. Moreover, as other cities adopted the same lockdown measures during the 2020 COVID-19 period, the impact of external transmission on Wuhan’s air quality was minimal. These findings suggest that wind direction and speed were not the main causes for the significant air quality improvement in Wuhan during the 2020 COVID-19 lockdown period.

Fig. 8 shows the daily temperature and relative humidity in Wuhan from January to April 2017–2020. The temperature trend in Wuhan was consistent from 2017 to 2020 and generally showed an upward trend from January to April. The relative humidity trend was also consistent in all years. The average temperature in January, February, March, and April was 11.0, 10.2, 9.8, and 10.7 °C, respectively, in 2017–2020, and the average relative humidity was 76.4%, 76.5%, 79.8%, and 78.1%, respectively; this suggests that the January–April average temperature and relative humidity did not change significantly from 2017 to 2020. Based on these findings, we conclude that the changes in wind direction, wind speed, temperature, and relative humidity in the winter and spring in Wuhan were similar in all years (2017–2020). Therefore, meteorological conditions were not the main reasons for the significant air quality improvement in Wuhan during the 2020 COVID-19 lockdown period.

3.2.2. Pollution source emission control and primary pollutant concentration change

The control of air pollution source emissions is another major factor that affects air quality (Huneeus et al., 2020). The number of air pollution sources and emissions in Wuhan and its surrounding areas significantly reduced in response to strict home quarantine measures and industrial and transportation closures from January 23 to April 8, 2020. These control schemes significantly improved the air quality of Wuhan and increased the number of days with excellent air quality.

As shown in Fig. 1, the strict control measures increased the January–April excellent and good air quality rates in Wuhan by 17.6–20.1% in 2020 compared to that in 2017–2019. As shown in Fig. 3, lightly polluted days occurred in April 2020, while the air quality was always excellent or good in March. The reason for this change was due to the gradual lifting of strict control measures from April 8, 2020, with various industrial and transportation activities gradually returning to normal levels. This increased the air pollutant emissions from production and transportation sources, resulting in higher atmospheric TVOC concentrations in April than in March (Fig. 5). This indicates that the COVID-19 lockdown measures were closely related to the air quality changes in Wuhan from January to April 2020, which is consistent with the results of previous studies (Wang et al., 2020b).

According to the MEIC dataset (Multi-resolution Emission Inventory for China: https://www.meicmodel.org)—an inventory commonly used in air quality modeling studies in China—NOx emissions in Wuhan are predominantly sourced from traffic (~45.28%) and industrial sources (~41.64%). The largest share of PM-related (BC, OC, PMF, PMC) and VOC-related emissions is from industrial sources at ~52.01% and ~62.35%, respectively.

Due to the Wuhan COVID-19 lockdown, traffic emissions reduced by 80%. According to the data from the Wuhan Bureau of Statistics (http://tjj.wuhan.gov.cn/ztzl_49/xwfbh/202007/t20200722_1408370.}
Wuhan’s GDP decreased by 40.50% in the first quarter; however, as this value includes the tertiary sector, we estimate that industrial pollution emissions decreased by 30%. Combined with the aforementioned changes in pollutant concentrations, the decrease in NO\textsubscript{2} and PM concentrations can be mainly attributed to the substantial reduction in traffic emissions. The reduction in VOC concentrations was related to the reduction of industrial emissions, and the effect of traffic sources on VOC concentrations was not significant (Zhou et al., 2020). As a result, VOC concentrations were relatively stable. An in-depth analysis could not be conducted due to the limited data; however, this could be further investigated using air quality models.

After April 8, 2020, both the number of air pollution sources and emission concentrations gradually returned to pre-COVID-19 levels, which increased the concentration of atmospheric primary pollutants (pollutants that are directly or indirectly released to the environment from a specific source) by varying degrees. Using the typical pollutant

Fig. 8. Comparison of temperature and relative humidity in Wuhan from January to April 2017–2020.

Fig. 9. Comparison of pollutant concentrations from January to May in Wuhan in 2019 and 2020.
concentrations in May 2020 as an example, the typical primary pollutant changes before and after the COVID-19 controls are illustrated in Fig. 9.

As shown in Fig. 9, the April and May average concentrations of various pollutants in 2020 were similar to those in 2019. The average pollutant concentration difference of NO$_2$, PM$_{2.5}$, PM$_{10}$ and VOC between 2019 and 2020 in March was 32.2 g/m$^3$, 31.4 g/m$^3$, 14.5 g/m$^3$, and 15.4 ppb respectively; in April it was 6.0 g/m$^3$, 6.8 g/m$^3$, 3.7 g/m$^3$, and 11.3 ppb respectively; and in May it was 11.3 g/m$^3$, 18.1 g/m$^3$, 3.1 g/m$^3$, and 14.8 ppb respectively. This indicates that the concentrations of primary pollutants in April and May in 2020 slowly increased in response to the gradual resumption of work, reaching similar values observed in the same months in 2019. The monthly mean concentrations of PM$_{10}$, PM$_{2.5}$, CO, and VOCs in May 2020 increased by 3.8%, 1.6%, 10.9%, and 3.1%, respectively, compared to levels in April. This suggests that the resumption of urban transportation systems and enterprises after April 8, 2020, was the dominant cause for the increase in pollutant concentrations. The amount of pollutant emissions from various pollution sources therefore increased significantly with the resumption of enterprises and transportation, which in turn impacted Wuhan’s urban air quality (Sulaymon et al., 2021).

4. Summary and outlook

As a megacity in central China, Wuhan has experienced rapid urbanization and modernization, which has adversely impacted the city’s air quality. To promote the continuous improvement of air quality in urban areas, it is necessary to conduct further research on air quality impact factors. Based on pollution monitoring and meteorological data from January to May 2017–2020, we analyzed the dominant factors influencing urban air quality changes caused by the Wuhan COVID-19 lockdown in 2020. We found that excellent and good air quality rates in Wuhan from January to April 2020 reached 90.1%, which was 17.6–20.1% higher than that in the same months in 2017–2019. Moreover, the concentrations of NO$_2$, PM$_{10}$, PM$_{2.5}$, and TVOCs in Wuhan decreased by 38.23%, 30.25%, 32.92%, and 39.80% from January to April 2020 compared to the levels in 2019. By comparing the January–April wind direction, wind speed, temperature, and humidity data for 2017–2020, we found that meteorological factors were not the main cause for significant air quality improvement in Wuhan in 2020. Instead, the strict COVID-19 lockdown measures (non-essential industry shutdowns, restrictions on transportation, and strict home quarantine) were the main cause for significant air quality improvement. The lockdown therefore created a positive contribution to Wuhan’s improvement in air quality.

Regardless of the cost, numerous management and control measures were implemented to control the spread of COVID-19. However, policymakers are highly concerned with the cost and benefits of reducing emissions when considering appropriate strategies for pollution prevention and control. Therefore, future studies must also assess the cost-effectiveness of air pollution control measures. In addition, the management and control measures assessed in this study considered the entire city. However, the contribution rate of pollution emissions from different regions of the city varies significantly. Therefore, to effectively control pollution, it is necessary to perform a detailed spatial analysis of pollutant emissions.

Authors’ individual contributions

Chen Jiaxin, Conceptualization, Methodology, Formal analysis, Data curation, Writing – original draft, Supervision, Project administration. Hu Hui, Conceptualization, Methodology, Resources, Writing – review & editing, Supervision, Project administration, Funding acquisition. Wang Feifei, Writing – review & editing, Supervision, Project administration. Zhang Mi, Zhou Ting, Yuan Shicheng, Bai Ruojiao, Huang Hao, Methodology, Investigation, Data curation. Chen Nan, Xu Ke, Investigation, Data curation.

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.

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