Impact of COVID-19 Outbreak on the Long-Range Transport of Common Air Pollutants in KUWAMS

Hao Zhang,1 Lulu Zhang,2 Lu Yang,2 Quanyu Zhou,2 Xuan Zhang,2 Wanli Xing,2 Kazuichi Hayakawa,2 Akira Toriba,2 and Ning Tang*,2,3

1 Graduate School of Medical Sciences, Kanazawa University; Kakuma-machi, Kanazawa, Ishikawa 920–1192, Japan; 2 Institute of Nature and Environmental Technology, Kanazawa University; Kakuma-machi, Kanazawa, Ishikawa 920–1192, Japan; and 3 Institute of Medical, Pharmaceutical and Health Sciences, Kanazawa University; Kakuma-machi, Kanazawa, Ishikawa 920–1192, Japan.

Received August 30, 2020; accepted December 2, 2020

As a background sampling site in western Japan, the Kanazawa University Wajima Air Monitoring Station (KUWAMS) continuously observes the air pollutants, including PM1, PM2.5, organic carbon (OC) and element carbon (EC). Data for September 2019 to April 2020 were compared with data for September 2018 to April 2019. The mean concentrations of both PM1 and PM2.5 were 4.10 µg/m3 (47%) and 5.82 µg/m3 (33%) lower, respectively in the Coronavirus Disease 2019 (COVID-19) period (January to April) than in the same period in 2019. Notably, the average concentrations of both classes of particulate matter (PM) in the COVID-19 period were the lowest for that period in all years since 2016. OC and EC also considerably lower (by 69 and 63%, respectively) during the COVID-19 period than during the same period in 2019. All pollutants were then started to increase after the resumption of the work in 2020. The pollutant variations correspond to the measure implemented during the COVID-19 period, including the nationwide lockdown and work resumption. Furthermore, the reductions in the ratios PM1/PM2.5 and OC/EC during COVID-19 period indicate lighter pollution and fewer emission sources. This analysis of the changes in the pollutant concentrations during the epidemic and non-epidemic periods illustrates the significance of the dominant pollution emissions at KUWAMS and the impact of pollution from China that undergoes long-range transport to KUWAMS.

Key words common air pollutant; Coronavirus Disease 2019 (COVID-19); long-range transport; Kanazawa University Wajima Air Monitoring Station (KUWAMS)

Introduction

Air pollution is a global problem that has attracted considerable attention, health studies by many researchers had proven deleterious human health effects of atmospheric pollutants.1-3 Particulate matter (PM) including PM1 (aerodynamic diameter <1 µm) and PM2.5 is a ubiquitous air pollutant. According to Ava et al., PM2.5 has a long-term health impacts on a variety of physiological systems.4 After inhalation, PM2.5 enters lung cells (alveolar macrophages and epithelial cells, etc.), which leads to oxidative stress. Oxidative stress will induce a chain of the damaging events, or even the death of normally functioning cells.5 According to a WHO report, exposure to PM2.5 causes 4.2 million people earlier deaths per year worldwide.6 In the light of Yang et al., PM1 is more hazardous than PM2.5, especially for children. Continuous inhalation and accumulation of PM can cause cardiovascular diseases, lung cancer and chronic bronchitis, etc.7 The emission sources of PM, for example, steel mills, power plants, vehicles and large-scale ports are generally concentrated in the large cities.8 Carbonaceous aerosols, such as elemental carbon (EC) and organic carbon (OC) play a significant components of atmospheric aerosol pollution.9 OC can be emitted directly into the air or be formed via the reactions of reactive organic gases and then undergo gas-to-particle partitioning process. The incomplete combustion of OC will generate EC.10

As one of the world’s largest energy consumers, China consumed approximately 23.6% of the total global energy, ranking first energy consumption worldwide in 2018.11 High energy consumption leads to high levels of air pollution. To control the severe air pollution, strict air pollution control methods have been implemented since 2013.12 However, due to the high levels of air pollution, the air quality in China is still worse than that in other countries (Japan, European countries and United States, etc.).13,14

Transboundary pollution was detected at the Kanazawa University Wajima Air Monitoring Station (KUWAMS) in Japan, which has been in operation for 15 years. Studies have proven the transboundary pollutants at KUWAMS were mainly originate from China, via long-range transport.15-18 During the cold season, East Asia is in dominated by northeast monsoon weather patterns, which transport the air pollutants and greatly impact Asian countries.19 Studies have found that combustion-derived air pollutants emitted in Northeast China can strongly affect KUWAMS in the cold season.16 Additionally, the impact of air quality control measures implemented in China (e.g., for the 2008 Beijing Olympics) has been detected at KUWAMS, proving the sensitivity of this site for air pollution monitoring.18,20

In December 2019, Coronavirus Disease 2019 (COVID-19) was discovered in a hospital in Wuhan, China. Subsequently, the number of infected people significantly increased and the spread across China.21 As the epicentre of the virus outbreak, Wuhan was heavily impacted by the disease. To reduce the speed and modes of disease transmission, On January 23rd, a
lockdown was enforced in Hubei Province, in which Wuhan is located. Following the initiation of the nationwide emergency response systems, other provinces and municipalities also entered lockdown on January 29th, 2020. Outside of the most basic survival activities, all factory operations, transportation and work and school activities were prohibited, leading to the significant decrease in air pollution. Compared to the level in the same period in 2019, the concentrations of PM\(_{10}\) and PM\(_{2.5}\) in 337 prefecture-level cities in China during the 2020 lockdown (February, 2020) relatively decreased by 30.0 and 27.3%, respectively. As mentioned above, during the cold season, fluctuations in pollutants in China significantly affect the level at the background sampling site KUWAMS in Japan. KUWAMS can be used to assess the impact of long-range of air pollutants transport from China during COVID-19.

The object of this paper is to document and analyze the impact of anti-epidemic measurements implemented in China during the COVID-19 outbreak on the long-range transport of air pollutants in East Asia. Changes in the concentrations of PM\(_{2.5}\), PM\(_{1}\), OC and EC at KUWAMS were observed from September 2019 to April 2020. The results of this study increase the understanding of the transport and variety of PM\(_{2.5}\), PM\(_{1}\), OC and EC in the atmosphere over East Asia.

**Experimental**

**Sampling Site** Figure 1 shows the location of the sampling site, where long-term observations of meteorological parameters and conventional air pollutants are made in Wajima located in the northern Ishikawa Prefecture, Japan. Unlike the other large metropolises in Japan, Wajima is located in a relatively remote area with only 26582 inhabitants live in 2019. Tourism, lacquerware and fisheries account for the vast majority of the industrial production in Wajima, and there are few sources of anthropogenic air pollutants were in the city.

Surrounded by dense forest, KUWAMS is located on the northwestern coast of Noto Peninsula, 2.1 km south of the Sea of Japan, and almost no sever local industrial emissions have been detected around the sampling site. Previous research proved that the air pollutants detected in Wajima during the cold season were mainly come from the region west outside of Japan, but the air pollutants detected in the hot season originate from mainland of Japan. As an observation station that has been in use for 15 years, KUWAMS has served in many studies as a background monitoring station to assess the long-range transport of air pollutants to Japan due to its unique geographic location. Reliable data was provided by KUWAMS in those studies of the long-distance transport of air pollutants, which offered strong evidence that pollutants generated in China become the primary air pollutants in Wajima after the long-range transport.

**Real-Time Monitoring of Air Pollution** An online air pollutants monitoring system provided by Kimoto Electric Co., Ltd. (Osaka, Japan) was used to monitor the air pollutants including PM\(_{2.5}\) and PM\(_{1}\) from September 2019 to April 2020 at KUWAMS. Specifically, a PM-714 particulate matter monitor with a flow rate of 16.7 L/min was uninterrupted measuring the PM\(_{1}\) and PM\(_{2.5}\) following the beta eta ray attenuation method. Additionally, an APC-710 automatic particulate carbon monitor with a flow rate of 16.7 L/min was used to measure the EC and OC in PM\(_{2.5}\) based on the near-IR/UV absorption method. Meteorological conditions including relative humidity (RH), temperature (T) and rainfall (rain) were monitored by a WXT530, the weather transmitter. The sampler automatically monitors air pollution in real-time, and the minimum temporal resolution of the data is on the order of minutes. The recorded data will be manually downloaded once per month. The concentration data of meteorological conditions and auto-monitored pollutants in this study were described in daily average. The calculation method was using the Manual for Continuous Monitoring of Air Pollution of Japan.

**Quality Control** For routine maintenance, the equipment was inspected by a specialist every month. Additionally, during the analysis, outliers caused by regular maintenance or other instrumental errors were removed.

**Back-Trajectory Analysis** Based on data from National Oceanic and Atmospheric Administration (NOAA), the air pollutant transport scenarios were evaluated through window-based HYSPLIT 4 (Hybrid Single-Particle Lagrangian Integrated Trajectory) back-trajectory analysis from KUWAMS. HYSPLIT is often used to determine the orbit of air mass to track the source of air pollutants. According to the report of Yang et al., the initial height was set up to 1500 m.

**Statistical Analysis** The levels of auto monitored pollutants were portrayed as daily data. Given the outbreak of COVID-19 in January 2020, the sampling period was divided into three stages.

---

Fig. 1. Sampling Point Location
(KUWAMS: 37.4°N, 136.9°E; 60 m above sea level.)

---
into two parts, the normal period (September to December) and the COVID-19 period (January to April). Since the collected data did not present a normal distribution, the Mann-Whitney U test was used to compare the differences between the two periods during sampling campaign and between the same periods in the previous year (September 2018 to April 2019). The Spearman correlation was used to obtain the correlations between the samples. A p-value less than 0.05 demonstrates that the outcomes were statistically significant. SPSS 25.0 was used to evaluate the data (IBM Corp., Armonk, NY, U.S.A.).

Results and Discussion

**PM$_1$ and PM$_{2.5}$** Figures 2(a) and (b) show the concentrations of the mean value and the standard deviation of PM$_1$ and PM$_{2.5}$ at KUWAMS during the sampling period. For comparison, Fig. 2 also includes data from previous observations at KUWAMS made from September 2018 to April 2019. The concentration of PM$_1$ in this study was ranged from 0.5 to 14.1 µg/m$^3$, with a mean of 4.12 ± 2.33 µg/m$^3$. The concentration of PM$_{2.5}$ range from 3.3 to 35 µg/m$^3$, with a mean of 11.08 ± 4.53 µg/m$^3$, which is slightly higher than the annual mean guideline set by the WHO (10 µg/m$^3$). The PM$_1$ and PM$_{2.5}$ concentrations in 2019–2020 significant different (p < 0.001) from those in 2018–2019. The average concentrations of PM$_1$ and PM$_{2.5}$ were approximately 47 and 33% lower, respectively, in the 2019–2020 COVID-19 period than in the same period in 2018–2019. The decrease (36 and 19%, respectively) in PM$_1$ and PM$_{2.5}$ in the normal period compared to the same period in 2018–2019 was smaller than the decrease corresponding to the COVID-19 periods of the two cold seasons. Generally, the concentration of PM$_1$ shows an increase in January and then gradually decreases beginning in the end of February, which was the pattern in 2018–2019. However, in 2019–2020, the concentration of PM$_1$ dropped in January and then gradually increased from February to April. The temporal variation in PM$_{2.5}$ in 2019–2020 was similar to that of PM$_1$, decreasing in January and then gradually increasing over the next few months. The pollution trends during the normal period and the same period in 2018–2019 and 2019–2020, but the difference was smaller than that between the COVID-19 period and the same period in 2018–2019; this smaller difference might be a result of a series of air pollution control policies implemented in China in recent years (e.g. The Clean Air Action). The implementation of these policies continues to improve the air quality of China and surrounding areas year after year.

Figure 3(a) presents the variations in PM$_1$ and PM$_{2.5}$ from January 23–30, 2019. The trendlines in these plots have positive slopes: 0.79 and 1.24 for PM$_1$ and PM$_{2.5}$, respectively. Figure 3(b) shows the variations PM$_1$ and PM$_{2.5}$ for the same dates in 2020, and the slopes of the trendlines are opposite those for 2019: −0.62 and −1.05 for PM$_1$ and PM$_{2.5}$, respectively. These decreasing tendencies reflect the drastic measures taken to contain the spread of COVID-19, which also dramatically reduced air pollution. The pausing of heavy in-
The industry, for example, iron and petrol chemical, can significantly decrease the concentration of PM due to their high emission scaling factors (0.91 and 0.96, respectively). 32) It is also worth noting that Chinese New Year (CNY) occurs at the end of January and is the longest national holiday (4 weeks). During CNY, most industrial production is halted, which may also probably conduct the large-scale variation in PM concentration. 33,34 Therefore, at the end of January 2020, due to the simultaneous impacts of two factors, the concentrations of both PM₁ and PM₂.₅ dropped to their lowest level (0.5 and 3.3 µg/m³, respectively) in every January since 2016. Additionally, the average concentrations of both PM₁ and PM₂.₅ (4.64 and 11.75 µg/m³) at COVID-19 period fell to the lowest point since 2016. 20)

Figure 4 shows the concentration value of PM₁ and PM₂.₅ in February, which reveals the changes caused by work resumption. On Approximately February 14th, significant increase in PM₁ and PM₂.₅ concentrations were observed. This

![Fig. 4. The Concentration of PM₁ and PM₂.₅ in February 2020](image)

![Fig. 5. The Tendency of the Concentration of PM₁ and PM₂.₅ (a) March 2020; (b) April 2020](image)
might have been due to the gradual resumption of work in the other provinces, and municipalities outside of Hubei Provinces beginning on February 10th. Additionally, by the effect of the northeast monsoon, three days will cost for the air masses transport from northeastern China to Japan which fit the time trajectory shown in Fig. 4. Finally, the peaks on February 22nd would be discussed in a later section.

The variations in PM$_1$ and PM$_{2.5}$ in March and April were shown in Figs. 5(a) and (b). In both two months, the slope of the concentration trend over time is positive, but the concentration increases are greater in March than in April. There is an apparent slowdown in the increases in PM$_1$ and PM$_{2.5}$ concentration in April, where the slope nearly decreased to zero. According to the Ministry of Industry and Information Technology of the Chinese government, the work resumption rate of small and medium-sized enterprises had increased to 76.8% by March. The work resumption rate slowed in April, reaching approximate 84% on April 15th, and this deceleration might have caused the rate of increase in the PM concentrations to slow. In addition, as the climate warms, air masses from domestic Japan will gradually become dominant, which might also slow the rate of increase.

To better comprehend about the trend during the COVID-19 period, the year-to-year change in the PM$_1$ and PM$_{2.5}$ concentration at KUWAMS was calculated (Table 1) and compared to Shenyang City, Liaoning Province, China. Comparing with Shenyang, the air quality condition in Shenyang 2019 was even worse. From September 2019 to January 2020, the concentration of PM$_{2.5}$ and PM$_{10}$ increased in average about 16.4 and 37% in Shenyang, respectively, in November 2018, higher than that in October (40, 75 µg/m$^3$) and December (47, 79 µg/m$^3$) 2018. Reversely, the PM$_{2.5}$ and PM$_{10}$ levels were 37 and 68 µg/m$^3$, respectively, in November 2019, lower than that in October (46, 89 µg/m$^3$) and December (62, 87 µg/m$^3$) 2019. Possible causes might be differences in their weather conditions and sources. The relatively high pollution in 2018 might leads to a decrease in the year-to-year contrast with 2019. In KUWAMS, the year-to-year change in PM$_1$ was −56% in January, which was the beginning of the COVID-19 pandemic, and remained at approximately −50% in the following two months. In April, the value increased to −23%, corresponding to the timeline of work resumption. The variation in PM$_{2.5}$ during COVID-19 period was similar to that in PM$_1$, and the year-to-year changes at Shenyang were similar with those at KUWAMS. The concentrations at Shenyang began to decrease in February instead of January as they did at KUWAMS.

The average ratio of PM$_1$/PM$_{2.5}$ for 2019–2020 was approximately 0.38, which was lower than the proportion (0.5) during the comparison period (September 2018–April 2019), and lower than the ratio which is reported by other studies, including Dandong station, China (0.86, 2006–2014), at the Gosan ABC superstation, Korea (0.608, 2017–2018); and in India (0.48, 2009–2010). High PM$_{1}$/PM$_{2.5}$ ratios indicate that the PM$_{2.5}$ levels are mainly caused by sources or processes that produce larger particles, rather than by mechanical processes. Biomass combustion, coal and fossil fuel combustion are the primary sources of PM$_{1}$, and contribute much more PM$_1$ than PM$_{2.5}$. Therefore, a higher PM$_1$/PM$_{2.5}$ ratio indicates a greater proportion of pollution from these sources. In this study, the PM$_1$/PM$_{2.5}$ ratio was much lower than the ratios which reported in the other periods and regions, which proves that the suspension of industrial and traffic activities caused by the outbreak of COVID-19 affected the long-range transport of PM$_1$ and PM$_{2.5}$ to KUWAMS.

### Organic Carbon and Element Carbon

The range in the OC concentration in the 2019–2020 was 0.02–2.51 (µgC/m$^3$), with a daily average concentration of 0.58 ± 0.42 (µgC/m$^3$). The concentration of EC in this season ranged from 0.004 to 0.451 (µgC/m$^3$), with a daily mean concentration of 0.09 ± 0.06 (µgC/m$^3$). The total carbon values (OC + EC) ranged from 0.02–2.77 (µgC/m$^3$), and the average concentrations of OC and EC accounted for 84 and 16% of the total carbon, respectively. These proportion were similar to other research findings in Changchun, China (81 and 19%, respectively). Figures 6(a) and (b) show the concentrations of OC and EC at KUWAMS during the 2019–2020 period. The concentrations decreased in January then gradually increased in the following month. Compared with the same period in 2018–2019, although there was a difference trend at October and December, the trend of concentration in this period between these two years is almost the same, which the concentration tardily enhances from September to December. Still, according to the pervious report, the monsoon from China to Japan is not significant before winter (December), dominant wind from China is usually prevailing in winter and spring, which is the COVID-19 period. After December 2019, the trends were highly variable: the concentrations of both OC and EC more than doubled from January to February and then dramatically decreased over the next two months back to their levels before the doubling. Differently, the concentrations of both OC and EC slowly increased during the COVID-19 period. According to Liu et al., a lower standard deviation (S.D.) indicates relatively stable emissions. The S.D.s of OC and EC were 0.50 and 0.07 (µgC/m$^3$), respectively, during the COVID-19 period and 1.45 and 0.15 µgC/m$^3$ during the same period in 2019. Thus, the variability in OC and EC in 2020 was half that in 2019. The lower S.D. in the COVID-19 period might have been due to the suspension of unnecessary activities, which decreased
the variety in emission sources. Additionally, the use of central heating systems in Northeast China is one of the primary OC and EC emission sources during the cold period. Thus, as other primary sources were reduced or eliminated, the central heating was still employed, potentially reducing the S.D.

Figures 7(a) and (b) show the concentrations of OC and EC from January 23rd–31st in 2019 and 2020, respectively. The level of OC was below the detection limit on January 30th and 31st, 2020. The temporal variation in the OC concentration in 2020 was opposite that in the same period in 2019, similar to the patterns of PM$_{1}$ and PM$_{2.5}$ mentioned above. The mean concentrations of OC and EC in 2020 were 69 and 63% lower, respectively, than those in 2019. These differences might have been due to the decreased industrial and traffic activities that began on January 23rd, 2020.

The relationship between the EC and OC concentration during the normal and COVID-19 periods is shown in Figs. 8(a) and (b), respectively. A stronger correlation between OC and EC was observed during the COVID-19 period ($R^2 = 0.7280$, $p < 0.05$), than during the normal period ($R^2 = 0.4267$, $p < 0.05$). The strong correlation for the COVID-19 period indicates that carbonaceous aerosols in this period were primly from the joint source and were affected by similar transport procedures. There were fewer sources of OC and EC during this period, which might have been due to the curtailing of unnecessary activities, including industrial manufacturing and traffic activity etc. by the Chinese authorities. In the absence of such unnecessary activities, central
heating became the main pollution source during this period, potentially leading to the strong correlations between OC and EC.

The OC/EC ratio is an essential tool to study the formation processes and the sources of carbonaceous particles. Figure 9 shows the OC/EC ratio in 2019–2020 ranged from 5.55 (December) to 10.96 (October), with an average of 7.41. The ratio in the normal period (8.37) was higher than that in the COVID-19 period (6.45). The mean value in 2018–2019 was 6.8, higher than the ratio in the COVID-19 period but lower than value in the normal period and value for the whole period investigated in 2019–2020. EC is mostly from primary emissions; OC primarily forms Secondary Organic Carbons (SOC) by photochemical reactions. A value of OC/EC ratio higher than two can indicate the formation of SOC. Therefore, the SOC detected at KUWAMS primarily formed in the atmosphere during long-range transport. Contributions of SOC, changes in emission sources, and the temperature and the atmospheric conditions can all affect the OC/EC ratio. Newly formed SOC and biomass-burning burning for cooking etc. will also result in a high OC/EC ratio. Cachier et al. reported that OC/EC values of 9, 2.7, 1.1 demonstrate biomass combustion, coal combustion and vehicle sources, respectively. The greater variations in the OC/EC ratio during the normal period than during the COVID-19 period indicate the presence of additional sources of carbonaceous particles in the former, consistent with the events of COVID-19 outbreak.

**Meteorological**

Meteorological parameters can affect the concentrations of particulate atmospheric pollutants. The Spearman correlation coefficient was used to clarify the correlations between such pollutants and the meteorological conditions. The correlation coefficients of each pollutant and meteorological data were calculated from daily averages. Table 2 shows the correlation between air pollutants and meteorological conditions. All meteorological conditions show a negative correlation with air pollutants. The correlation coefficients of OC, EC, PM$_1$ and PM$_{2.5}$ with temperature were $-0.195$, $-0.399$, $-0.105$ and $-0.054$, respectively; with rainfall were $-0.092$, $-0.098$, $-0.239$ and $-0.154$, respectively. In particular, relative humidity had a statistically significant negative correlation with all pollutants ($-0.184$, $-0.324$, $-0.274$ and $-0.181$ for OC, EC, PM$_1$ and PM$_{2.5}$, respectively). Basing on our previous study, the periods of low temperature, low humidity and less rainfall in KUWAMS are mostly prevailing the Asian winter monsoon. During this period, the long-range transport of pollutants from the Asian continent is the important cause of air pollution in KUWAMS. Nevertheless, during the prevailing period of Asian winter monsoon in this study, the expected sever pollution did not appear in the period, which can be explained as the impact of China’s economic stagnation during COVID-19.

**Back-Trajectory Analysis**

Seventy two hours back trajectories 1500 m above ground were used to evaluate the air samples at KUWAMS.

Three specific periods were chosen, for the calculation shown in Fig. 10. According to Figs. 10(a) to (c), the air masses that arrived at KUWAMS via long-range transport is mainly originated in Northeast Asia, especially Northeast China. Figure 10(a) shows the source of the contaminants detected at KUWAMS since the COVID-19 outbreak at the end of January. Almost 100% of the air masses were derived from northern China, travelling over North Korea and then the Pacific Ocean before finally arriving at KUWAMS. Figure

**Table 2. Spearman Correlation Among OC, EC, PM$_1$, and PM$_{2.5}$ with Meteorological Condition During September 2019 to April 2020**

|       | $T$   | RH     | Rain  |
|-------|-------|--------|-------|
| OC    | $-0.195^{**}$ | $-0.184^{**}$ | $-0.092$ |
| EC    | $-0.399^{**}$ | $-0.324^{**}$ | $-0.098$ |
| PM$_1$| $-0.105$    | $-0.274^{**}$ | $-0.239^{**}$ |
| PM$_{2.5}$ | $-0.054$ | $-0.181^{**}$ | $-0.154^{*}$ |

* $p<0.05$, ** $p<0.01$; $T$: temperature; RH: relative humidity; Rain (mm).
10(b) shows the orbits of the air masses from February 22nd, 2020 to February 25th, 2020. Approximately 87% of the air masses came from northeast China. Figure 10(b) shows the orbits of the air masses from February 22nd, 2020 to February 25th, 2020. Approximately 44% of the air masses passed through Beijing and Tianjin, then crossed the Pacific and finally reached KUWAMS. As mentioned before, Fig. 4 shows a high concentrations value of both PM$_{1}$ and PM$_{2.5}$ on February 22nd, possibly due to the long-range transported air masses from Beijing and Tianjin. From February 19th to 21st, 2020, Beijing and Tianjin were experiencing a mildly polluted weather, PM$_{2.5}$ of Beijing and Tianjin was approximate 100µg/m$^{3}$ during that period.50 Thus, these pollutants carried by monsoon and then reach KUWAMS, where the concentration of PM significantly increased on February 22nd. Figure 10(c) shows the trajectory of air mass in March and April in 2020, which China had started the work resumption. Although the pollutants reach the KUWAMS began to be gradually influenced by Japan, KUWAMS was still mainly affected by northeast China (approximately 88% of the air masses).

Conclusion
As a background station that has been in continuous use for more than 15 years, KUWAMS continues to provide reliable data on the long-range transport of air pollutants. Air pollution data were collected at KUWAMS from September 2019 to April 2020 and analyzed. Overall, the air pollutants decreased in COVID-19 period, which was different from the trends in the same periods in 2019. The results of the meteorological and back-trajectory analysis prove that China is still the primary source of air pollutants at this site, with the air mass carried by monsoons and transporting pollutants long range to KUWAMS. During the normal period, the concentrations of PM$_{1}$, PM$_{2.5}$, OC and EC slightly decreased. In contrast, there was a significant decrease in the concentrations of all the air pollutants during the COVID-19 period, because of the nationwide suspension of work and industrial production. The mean concentrations of both PM$_{1}$ and PM$_{2.5}$ at KUWAMS were the lowest that they had been in this period for each year since 2016. Moreover, the OC/EC ratio was lower in the COVID-19 period than in the same period of 2019, which might have been due to fewer sources of carbonaceous particles in 2020. Overall, the decrease in pollutant emissions during the epidemic in Northeast China caused a major reduction in the pollution at KUWAMS. This reduction in emissions was a result of considerable disruptions to normal daily life and caused considerable economic losses. Therefore, the emissions will soon return to their original level after the epidemic situation improves. Although China has achieved remarkable results from air pollution control measures implemented in recent years, to further enhance the air quality in China and surrounding countries, stringent air pollution control policies issued by the Chinese government should be continuously implemented.

Acknowledgments This work was supported by the Ministry of Education, Culture, Sports, Science and Technology, Japan (17K08388); the Environment Research and Technology Development Fund (5-1951) of the Environmental Restoration and Conservation Agency of Japan; the Sumitomo Foundation, Japan (183115); the CHOZEN Project of Kanazawa University, Japan; and the Institute of Nature and Environmental Technology, Kanazawa University, Japan (20015–20019).

Conflict of Interest The authors declare no conflict of interest.

References
1) Zhang L. L., Morisaki H., Wei Y. J., Li Z. G., Yang L., Zhou Q. Y., Zhang X., Xing W. L., Hu M., Shima M., Toriba A., Hayakawa K., Tang N., Sci. Total Environ., 705, 135840 (2020).
2) O’Brien R. L., Neman I., Rudolph K., Casey J., Venkataraman S., Soc. Sci. Med., 217, 92–96 (2018).
3) Jerrett M., Nature (London), 525, 330–331 (2015).
4) Ava O., Crishi A. L. M., Mary B., Christopher M., “Sustained Effects on Lung Function in Community Members Following Exposure to Hazardous PM2.5 Levels from Wildfire Smoke.”: https://assets.researchsquare.com/files/rs-35187/v1/4cfa41f6-4e6f-49d7-b58a-0a64a59cf82a.pdf/, cited 29 June, 2020.
5) Feng S. L., Gao D., Liao F., Zhou F. R., Wang X. M., Ecotoxicol. Environ. Saf., 128, 67–74 (2016).
6) WHO, “Ambient (outdoor) air pollution.”: https://www.who.int/news-room/fact-sheets/detail/ambient-(outdoor)-air-quality-and-health, cited 17 June, 2020.
7) Yang M., Guo Y. M., Bloom M., et al., Environ. Int., 145, 106092 (2020).
8) Nakatsubo R., Oshita Y., Aikawa M., Takimoto M., Kubo T., Matsumura C., Takaishi Y., Hiraki T., Sci. Total Environ., 702, 134744 (2020).
9) Huang L., Brook J., Zhang W., Li S. M., Graham L., Ernst D., Chi- valescu A., Lu G., Atmos. Environ., 40, 2690–2705 (2006).
10) Chu S., Atmos. Environ., 39, 1383–1392 (2005).
11) Guangming Daily (GD), “China’s energy consumption ranks first in the world for 10 consecutive years.”: http://difang.gmw.cn/gd/2019-12/15/content_33402523.htm, cited 19 June, 2020.
12) Cui L. L., Zhou J. W., Peng X. M., Ruan S. M., Zhang Y., Sci. Rep., 10, 5432 (2020).
13) Tang S. L., Zhou X. H., Zhang J. Z., Xue L. K., Luo Y. L., Song J., Wang W. X., Environ. Sci. Pollut. Res. Int., 27, 12122–12127 (2020).
14) Zhang L. L., Yang L., Zhou Q. Y., Zhang X., Xing W. L., Wei Y. J., Hu M., Zhang L. X., Toriba A., Hayakawa K., Tang N., J. Environ. Sci., 88, 370–384 (2020).
15) Pan X. L., Lu N., Hara Y., Kuribayashi M., Kobayashi H., Sugimoto N., Yamamoto S., Shimohara T., Wang Z. F., Geophys. Res. Lett., 42, 1593–1598 (2015).
16) Tang N., Hakamata M., Sato K., Okada Y., Yang X., Tatematsu M., Toriba A., Kameda T., Hayakawa K., Atmos. Environ., 120, 144–151 (2015).
17) Yang X., Okada Y., Tang N., Matsunaga S., Tamura K., Lin J. M., Kameda T., Toriba A., Hayakawa K., Atmos. Environ., 41, 2710–2718 (2007).
18) Tang N., Sato K., Tokuda T., Tatematsu M., Hama H., Suematsu M., Kameda T., Toriba A., Hayakawa K., Chemosphere, 107, 324–330 (2014).
19) Griffith S., Huang W. S., Lin C. C., Chen Y. C., Chang K. E., Lin T. H., Wang S. H., Lin N. H., Sci. Total Environ., 741, 140214 (2020).
20) Zhang X., Zhang L. L., Yang L., Zhou Q. Y., Xing W. L., Toriba A., Hayakawa K., Wei Y., Tang N., Int. J. Environ. Health Res., 17, 957 (2020).
21) WHO, “Coronavirus disease (COVID-19) pandemic.”: https://www.who.int/emergencies/diseases/novel-coronavirus-2019, cited 19 June, 2020.
22) Yu Z., “Timeline: The 76-day memorabilia of lockdown in Wuhan.”: http://news.sina.com.cn/c/2020-04-08/doc-iimxxsth4244405.shtml, cited 19 June, 2020.
23) Xu K. J., Cui K. P., Young L. H., Hsieh Y. K., Wang Y. F., Zhang J. J., Wan S., Qual. Res., 20, 915–929 (2020).
24) Zhang R., X., Zhang Y. Z., Lin H. P., Feng X., Fu T.-M., Wang Y., H., Atmos. Environ., 11, 433 (2020).
25) Ministry of Ecology and Environment of the People’s Republic of China (MEE), “City Air Quality Monthly Report.”: http://www.mee.gov.cn/hjzl/dqkj/csqaqlzkyb/, cited 17 June, 2020.
26) Wajima city, “City introduction.”: http://honyaku.jserver.com/LUCWAJIMA/cdata/lucwajima0_jaen.html, cited 15 June, 2020.
27) Yang L., Tang N., Matsuki A., Takami A., Hatakeyama S., Kaneosen/manual_6th/index.html, cited 16 October, 2020.
28) Ministry of the Environment, "Manual for Continuous Monitoring of Air Pollution of Japan, Version 6.0.": http://www.env.go.jp/air/access.html, cited 15 June, 2020.
29) Zhang R. X., Zhang Y. Z., Lin J. M., Kameda T., Toriba A., Hayakawa K., Atmos. Environ. 152, 354–361 (2017).
30) Tamamura S., Sato T., Ota Y., Wang X. L., Tang N., Hayakawa K., Atmos. Environ., 41, 2580–2593 (2007).
31) Zheng B., Tong D., Li M., Liu F., Hong C. P., Geng G. N., Li H. Y., Li X., Peng L. Q., Ota Y., Wang X. L., Tang N., Hayakawa K., Atmos. Chem. Phys., 18, 14095–14111 (2018).
32) Liu T., Wang X. Y., Hu J. L., Wang Q., An J. Y., Gong K. J., Sun J., Sun Y., Chen Q., Zhao H., Wang L., Tao R., Atmos. Environ. Sci. Pollut. Res. Int., 264, 128427 (2021).
33) Liu B., Song N., Dai Q., Mei R., Sui B., Bi X., Feng Y., Atmos. Res., 170, 23–33 (2016).
34) Wu X., Chen B., Wen T., Habib A., Shi G., J. Environ. Sci., 87, 1–9 (2020).
35) Cheng M. C., You C. F., Cao J., Jin Z., Atmos. Environ., 60, 182–192 (2012).
36) Liu B., Bi X., Feng Y., Dai Q., Xiao Z., Li L., Wu J., Yuan J., Zhang Y., Atmos. Res., 181, 20–28 (2016).
37) Cong Z., Kang S., Kawamura K., Liu B., Wan X., Wang Z., Gao S., Fu P., Atmos. Chem. Phys., 15, 1573–1584 (2015).
38) Cachier H., Brémond M.-P., Buat-Ménard P., Nature (London), 340, 371–373 (1989).
39) Li Y., Chen Q., Zhao H., Wang L., Tao R., Atmosphere, 6, 150–163 (2015).
40) Tang N., Sato K., Tokuda T., Tatamatsu M., Hama H., Suematsu C., Kameda T., Toriba A., Hayakawa K., Chemosphere, 107, 324–330 (2014).
41) Zhang L. L., Yang L., Zhou Q. Y., Zhang X., Xing W. L., Zhang H., Toriba A., Hayakawa K., Tang N., Qual. Res., 20, 2035–2046 (2020).
42) AQistudy. “Air quality historical data query.”: https://www.aqistudy.cn/historydata/, cited 28 October, 2020.