Multiple Air Quality Monitoring Evidence of the Impacts of Large-scale Social Restrictions during the COVID-19 Pandemic in Jakarta, Indonesia

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ABSTRACT

Air pollution is a top contributor to global mortality. Air quality issues abound in developing Asian countries, but during COVID-19 lockdowns, urban air quality improved due to the reduction in public mobility and fuel consumption. In Indonesia, the Large-Scale Social Restriction (LSSR) program was implemented to prevent the wider spread of COVID-19, especially in large urban areas. It was not a total lockdown program but had the purpose of reducing urban public mobility. This study investigated the effects of social restrictions on air quality in Jakarta, Indonesia. Data were obtained from our long-term monitoring of fine (PM2.5) and coarse particulate matter (PM2.5-10) and compositions collected at a site in South Jakarta. Other data were obtained from the environmental protection agency’s (EPA’s) air quality monitoring station in Central Jakarta including PM10, PM2.5, SO2, NO2, CO, and O3. The aerosol optical depth (AOD) in Jakarta measured by a sun photometer and satellite data were used to assess the spatial distribution of AOD across Jakarta. During the first LSSR implementation period from 15 March to 30 May 2020, there were decreased average SO2, CO, NO, NO2, and NOx concentrations of 40 to 60% compared to the same period in 2019. However, O3 increased by 33% likely due to reduction in NOx emissions. The PM2.5 decline reached ~40%, but a similar decline was not observed for PM10. Elemental and black carbon concentration data showed reductions that ranged from 30% to more than 50%. Consistent with the PM observations, both ground and satellite based AOD showed reductions in the aerosol column burden over the city. The ground based AOD values showed moderate correlations with PM2.5. The results confirmed that significant reduction in public mobility was highly associated with the improvement of local air quality which useful to derive future control strategies.

Keywords: PM2.5, PM10, chemical composition, BC, AOD

1 INTRODUCTION

Air pollution has become an important global problem that requires serious attention because of its impacts on human health and environmental quality (Brauer et al., 2016). The deterioration of ambient air quality, especially in big cities needs major improvements. The Global Burden of Disease Project (Murray et al., 2020) reported that air pollution was globally responsible for more than 7.5 million deaths. Other studies showed that fine particulate pollution (PM2.5) was highly correlated with population mortality with morbidity (U.S. EPA, 2019; Murray et al., 2020). Long-term exposures to high PM2.5 concentration lead to various respiratory diseases such as
respiratory infections, asthma, chronic obstructive pulmonary disease (COPD) even lung cancer, since PM$_{2.5}$ contains various toxic substances and infectious agents that are able to penetrate into our lungs (Burnett et al., 2018; McGuinness et al., 2019; Alemayehu et al., 2020).

Other pollutants including SO$_2$, NO$_2$, CO and O$_3$ also affect human health and ecosystems (Rahman et al., 2019). SO$_2$ and NO$_2$ exposures in high concentrations will damage human health by facilitating respiratory tract infections or exacerbating respiratory illnesses such as asthma, COPD, etc. SO$_2$ and NO$_2$ can combine with rainwater creating acid rain that is hazardous for animals and plants (WHO, 2005, Chen et al., 2007; Rahman et al., 2019). Breathing high concentrations of O$_3$ can reduce the lung function and damage lung tissue. O$_3$ can also trigger a variety of responses such as eye irritation, throat irritation, coughing, and chest pain. The primary sources of emissions of SO$_2$ and NO$_2$ are anthropogenic activities involving the combination of burning fossil fuels, biomass, and the resulting emissions from vehicles and the power plants (Brunekreef and Holgate, 2002). Breathing elevated concentrations of CO can reduce O$_2$ transport in hemoglobin and cause health effects including headaches, chest pain, heart disease, etc. (Sharma et al., 1999) and moreover acute and chronic exposure in enclosed space will increase the risk for development of cardiopulmonary and cardiovascular events, including death (Chen et al., 2007).

In Indonesia, air pollution is one of many serious environmental problems encountered by major cities due to the high population growth, increasing economic activity, intensive transport, and industrial activities.

The capital city of Indonesia, Jakarta, is a megacity with a 2019 population of nearly 10.5 million people. Jakarta is also surrounded by industrial and sub-urban areas that are located within distance of 20–30 km from the center of the city. The city is characterized by a high mobility of transportation that in turn routinely causes traffic jams. High urban PM$_{2.5}$ pollution has been observed with frequent violations of the Indonesian annual mean national ambient air quality standard of 15 µg m$^{-3}$ (Santoso et al., 2013; Santoso et al., 2020). Twenty-four-hour PM$_{2.5}$ values at industrial sites ranged from 15 to 42 µg m$^{-3}$, while at residential sites, values ranged from 9 to 36 µg m$^{-3}$ (Santoso et al., 2011). The PM$_{2.5}$ concentrations measured in Jakarta at an arterial roadside were higher than those measured in the other Indonesian big cities. The mean concentrations of PM$_{2.5}$ and PM$_{10}$ were 25.76 and 75.20 µg m$^{-3}$, respectively. Another critical air pollutant that contributed to the dangerous air quality index in Jakarta was surface ozone (O$_3$) due to intensive build-up of local precursor emissions (e.g., NO$_x$) and the meteorological conditions favorable to photochemical reactions (Suhadi et al., 2005; Permadi and Kim Oanh, 2008). In Jakarta, the mean of NO$_2$ concentrations using passive samplers showed the weekly average of 20–70 µg m$^{-3}$, while the peak daily concentration was 446 µg m$^{-3}$. Annual average of NO$_2$ in Asian cities typically in the range of 23-74 µg m$^{-3}$ (WHO, 2005). Jakarta reported weekly average concentrations of SO$_2$ between 4 µg m$^{-3}$ and 24 µg m$^{-3}$ (WHO, 2005). This value does not differ significantly from those reported in central Jakarta in 2017. The reported mean SO$_2$ was 22.72 µg m$^{-3}$, while average NO$_2$ was 11.85 µg m$^{-3}$ (Rahman and Barus, 2019).

In late 2019, a contagious virus appeared in China that was identified as a novel strain of coronavirus belonging to the same family as acute respiratory syndrome (SARS) and Middle East respiratory syndrome (MERS) (Liu et al., 2020; Zhu et al., 2020). Then, the so-called coronavirus disease 2019 (COVID-19) was declared by the World Health Organization (WHO) as a worldwide pandemic in the mid-March 2020 based on the report of 118,319 cases and 4,292 deaths globally (WHO, 2020). Although this novel virus is close to the coronavirus found in animals, COVID-19 has been confirmed to be transmitted from human-to-human and has drawn significant attention globally (Hsiao et al., 2020; Guo et al., 2020; Shereen et al., 2020). Most of affected countries have been implementing restriction of activity (lockdown) measures with various level of stringency that were reported to require drastic reductions in public mobility and fossil fuel consumption. A global study using various satellite products showed that the air pollution in China, Italy, Spain and USA had declined by up to 30% while reductions were mobility was up to 90% (Muhammad et al., 2020). In Asia, many studies were conducted to investigate the impacts of various COVID-19 related measures on air quality in India, China, and Malaysia (Abdullah et al., 2020; Chen et al., 2020; Mahato et al., 2020), but more similar studies are required in other countries including in Indonesia. The results elsewhere showed substantial reductions in air pollutant concentrations (gases and PM) that somehow showed the effectiveness of the lockdown programs in reducing emissions.

The first case of COVID-19 in Indonesia was identified in February 2020. Since then, the virus...
has spread widely across Indonesia and still showing an increasing rate of new cases up to November 2020. Jakarta has become epicentral of the spread of COVID-19 in Indonesia with the highest number of confirmed cases of COVID-19 as well as deaths. On 16 March 2020, the Governor of Jakarta implemented a policy, named as Large-Scale of Social Restriction (LSSR) to prevent a wide spread of COVID-19. This action strictly limited the anthropogenic activities in Jakarta with only limited essential services such as health care, logistics, food supply, and banking being allowed to operate. Therefore, there were fewer vehicles on the roads, many cancelled flights, and restricted construction and industrial activities that led to decreased air pollutant concentrations.

The objective of the current study was to provide the evidence derived from various observation techniques on the impacts of the LSSR implementation on the air quality in Jakarta, Indonesia.

2 METHODS

We used three sources of ground-based monitoring data in Jakarta: a) automatic ambient air quality monitoring station (AQMS) for PM and gases, b) PM filter-based measurement (i.e., PM mass, carbonaceous and elemental compositions), and c) Aerosol Optical Depth (AOD) measured by sun-photometer equipment. In addition, we also retrieved Terra Moderate Resolution Imaging Spectroradiometer (MODIS) AOD to visualize the spatial distribution of column aerosol burden over the study area (https://giovanni.gsfc.nasa.gov/giovanni/).

The first of the three sites are shown in Fig. 1 as site (A). This site is located in the central business district (CBD) of Jakarta with a coordinate of 6°11’40.78’S/106°49’24.92’E. This air quality monitoring site is managed by the Environmental Protection Agency (EPA) of Jakarta Province with regular quality assurance and quality control (QA/QC) to provide continuous monitoring of PM10, PM2.5, CO, SO2, O3 and NO2. Continuous monitoring of PM mass uses the standard Federal Reference Method (FRM) Beta Attenuation (BAM) (Verewa F701-20 for PM10 and Horiba APDA-371 for PM2.5). Standard continuous gas analyzers have been used as follows: O3 (HORIBA APOA-370), CO...
(HORIBA APMA-370), SO$_2$ (HORIBA APSA-370), and NO$_x$ (HORIBA APNA-370). The data are averaged for every 30 minutes.

We conducted long-term PM monitoring (2nd site) at the provincial EPA office located on Casablanca Street, Central Jakarta 06°13’34.08"S, 106°50’04.42"E shown in Fig. 1 as site (B). This site is situated in the center of the city and is surrounded by an arterial road with heavy traffic especially in the morning and in the evening. The dichotomous sampler was located on the fourth-floor roof about 18 m above ground level. Integrated filter sampling was conducted using a Gent stacked filter unit (SFU) particle sampler capable of collecting particulate matter in PM$_{2.5-10}$ and PM$_{2.5}$ size fractions (Hopke et al., 1997). The samples were collected on an 8 µm pore nucleopore filter for the coarse fraction sample and on an 0.4 µm pore nucleopore filter for the fine fraction sample. The samples were collected from September 2019 until May 2020 for 24 h at least once per week. The aerosol masses of both PM$_{2.5-10}$ and PM$_{2.5}$ fractions were determined by weighing the filters before and after exposure, then divided by the volume of air passing through the filter to obtain the concentration of PM$_{2.5}$ and PM$_{2.5-10}$ (µg m$^{-3}$). The PM$_{10}$ mass concentration was obtained by summing up these two values.

Light absorbing carbon (black carbon, BC) in the samples was determined by reflectance measurement using an EEL model 43D smoke stain reflectometer that measures the reduction in reflected white light (Coulson and Ellison, 1963; Commins and Waller, 1967; Biswas et al., 2003). Secondary standards of known BC concentrations are used to calibrate the reflectometer (Biswas et al., 2003; Begum, et al., 2011). The light reflected or absorbed in the filter sample depends on particle concentration, density, refractive index and size. The sample filter is then placed on a white standard and then measured its reflection value by repeating it three times (Lestiani et al., 2008; Diffusion system manufacture, 2012). The reflectance value obtained from the filter sample is a value that is proportional to the number of BC in the filter using the assumption of an average of particle mass absorption coefficient of 5.7 m$^2$ g$^{-1}$ (Seneviratne, 2011).

The collected samples were analyzed for chemical composition using X-ray Fluorescence (XRF). Multielement analyses were performed using an Epsilon 5 ED-XRF (Panalytical, Ltd.) that has 9 secondary targets (Fe, CaF$_2$, Ge, Zr, CeO$_2$, Mo, Ag, Al and one Barkla polarizing target (Al$_2$O$_3$)). Single element MicroMatter standards were used to develop the calibration parameters, while for method validation, NIST SRM 2783 samples were periodically analyzed. These methods are described in detail in the previous studies (Landsberger and Creatchman (1999); Santoso and Lestiani (2014). XRF analysis of the APM filters determined the concentrations of Na, Mg, Al, Si, S, K, Ca, Ti, V, Cr, Mn, Fe, Co, Ni, Cu, Zn, Pb and As.

The third location of the monitoring site is shown in Fig. 1 as site (C) and is surrounded by office buildings and human settlements. Measurements of AOD using sun photometer were conducted by the National Bureau of Meteorology, Climatology and Geophysics (BMKG) under the support of The National Aeronautics and Space Administration (NASA), Aerosol Robotic Network (AERONET). This site is part of the largest sun photometer network in the world (Holben et al., 1998). The measurement system is a solar-powered CIMEL Electronique 318A spectral radiometer that measures sun and sky radiances at a number of fixed wavelengths within the visible and near infra-red (VNIR) spectrum.

The stations are situated at a distance, for example, between station A and B of about 3 km while between station A and C there is a distance of about 4 km. The length span between station C and B is more than 6 km. Therefore, the selected stations represent different city background.

### 2.1 Statistical Analyses

To determine the changes in pollutant concentrations resulting from the reduced anthropogenic activity during the COVID19 lockdown period, several non-parametric tests were performed to compare the data from 4 periods: Pre-LSSR 2020 (January 1–March 15, 2020), LSSR 2020 (March 16, 2020–May 31, 2020), Comparable Pre-LSSR 2019 (January 1–March 15, 2019), and Comparable LSSR 2019 (March 16–May 31, 2019). The distributions of each pollutant were found to not be normal distributions and thus, comparisons among the 4 periods were done using the Kruskal-Wallis ANOVA on ranks (Kruskal and Wallis, 1952). Individual pairwise comparisons were made using the Bonferroni procedure (Bonferroni, 1936). To further confirm these results, the data were also subjected to Mood’s Median Test (Mood, 1954). Mood’s median test was carried out to examine the hypothesis that the medians of all 4 samples are equal. It does so by counting the
number of observations in each sample on either side of the grand median over all 4 data subsets. A probability value is calculated based on a chi-square test.

3 RESULTS AND DISCUSSION

3.1 Continuous Ambient Air Quality Data

Particulate matter (PM10 and PM2.5) and gas measurement data (CO, SO2, O3, NO, NO2) were obtained from a continuous air quality monitoring station located in Central Jakarta that is located about 5 m from a major road. We compared the period average of PM10, PM2.5, SO2, CO, O3, and NO2 concentrations before (1st January–15th March, named as Pre-LSSR) and during the LSSR (16th March–31st May) for two years: 2019 and 2020. In addition, diel variations for Pre-LSSR and LSSR periods were analyzed for both particulate matter and gases.

The statistical summary for hourly values of each pollutant as well as hourly rainfall in each of the 4 periods is provided in Table 1. The results of the Kruskal-Wallis ANOVA on ranks and Bonferroni procedures for pairwise comparisons are summarized in Table 2. The detailed results of these analyses are presented in the Supplementary Material. There were significant differences among the 4 periods for each measured variable. The entries in the table indicate the statistically significant differences between pairs of periods. A plus (+) sign indicates that the first named period median which was greater than the second named period while a negative (−) sign indicates the opposite direction. A similar analysis was made for the rainfall data for each period.

3.1.1 Particulate matter

The distributions of PM10 and PM2.5 derived from hourly data for all 4 periods are presented as box and whisker plots in Fig. 2. The average values of PM10 for Pre-LSSR 2020 and LSSR 2020 were 41.0 ± 18.4 and 49.4 ± 20.3 µg m⁻³, respectively. The average values of PM2.5 for Pre-LSSR 2020 and LSSR 2020 were 23.8 ± 16.7 and 26.5 ± 15.5 µg m⁻³, respectively. PM10 and PM2.5 were higher in the LSSR 2020 period than in the Pre-LSSR 2020 period. This difference may be the result of much higher total rainfall in the Pre-LSSR 2020 period (836 mm) than in any of the other 3 periods (195, 290, and 207 mm, respectively, for the other 3 periods). Period average values of rainfall intensity (in mm) for different periods is presented in Fig. S1. Precipitation reduces PM2.5 concentrations at a lower extent than it reduces PM10 concentrations (Blanco-Becerra et al., 2015; Zhou et al., 2020). Comparing the LSSR period in 2019 with the LSSR 2020 period showed that the average PM2.5 values were 45.4 ± 21.3 and 26.5 ± 15.5 µg m⁻³, respectively, while PM10 concentrations were 46.1 ± 18.8 and 49.4 ± 27.3 µg m⁻³, respectively. PM2.5 in 2019 was substantially higher than in 2020. However, PM10 in 2019 was slightly lower than in 2020. This difference could be due to the somewhat higher rainfall intensity in 2019 (Fig. S1), showing that the LSSR in 2020 had a greater effect on PM2.5 than on PM10. Thus, there are other sources of PM10 that were unaffected by the LSSR.

To examine these temporal patterns in more detail, the hour-by-hour data have also been analyzed for differences over these 4 defined periods. Fig. 3 show the hour-by-hour box and whisker plots showing the distributions of PM2.5. The maximum diel average of PM2.5 also showed consistent result with the mean values with lower value during the LSSR period in 2020 (LSSR 2020) as compared to both periods in 2019. However, the Pre-LSSR 2020 value was lower than the LSSR 2020 period due to high rainfall (Fig. S1). For PM10, the maximum diel average value of LSSR 2020 period was higher as compared to all other periods (Fig. S2). Typical diel pattern of PM10 was seen for all periods with higher values during daytime. For PM2.5, higher values were seen at the late night may reflect the lower dispersion characteristics overnight when wind speeds and mixed layer heights would lead to lower dilution of ground level emissions. The higher daytime PM10 concentrations may reflect higher windspeeds to suspend coarse mode particles beginning after sunrise. The concentrations decreased in the afternoon as the mixed layer heights increase.

3.1.2 Gases (SO2, CO, NO, NO2, NOx, and O3)

Box and whisker plots for the gaseous pollutants are given in Fig. 4. SO2, CO, and the oxides of nitrogen (NO, NO2, and NOx) were substantially lower during the LSSR 2020 period compared to the prior months in 2020. The LSSR 2019 period had higher CO, SO2, NO, NO2, and NOx than in
2020. Thus, there were remarkable reductions of SO$_2$, CO, and NO$_2$ during the LSSR 2020 as compared to other periods. In Jakarta, some sectors such as transportation and industrial activities experienced a significant decrease since people was encouraged to stay at home due to the virus. Many sources reported that declines in transportation by rail (7%), sea public transportation (50.7%), air transportation (82.4%) and private car (19.3%). The mobility of the people going to the market and pharmacy fell by 67%. Those going to the mall/café were down by 77%. The decrease in air transportation (82.4%) and private car (19.3%). The mobility of the people going to the market had a significant decrease since people was encouraged to stay at home due to the virus. Many sectors reported that declines in transportation by rail (7%), sea public transportation (50.7%), air transportation (82.4%) and private car (19.3%). The mobility of the people going to the market and pharmacy fell by 67%. Those going to the mall/café were down by 77%. The decrease in air transportation (82.4%) and private car (19.3%). The mobility of the people going to the market and pharmacy fell by 67%. Those going to the mall/café were down by 77%. The decrease in air transportation (82.4%) and private car (19.3%).

| Table 1. Summary of the statistics for each pollutant for each of the 4 analysis periods. |
|---------------------------------------------------------------|
| **PM$_{10}$** | **PM$_{2.5}$** | **SO$_2$** | **CO** | **O$_3$** | **NO** | **NO$_2$** | **NO$_x$** | **Rain** |
| (µg m$^{-3}$) | (µg m$^{-3}$) | (µg m$^{-3}$) | (µg m$^{-3}$) | (µg m$^{-3}$) | (µg m$^{-3}$) | (µg m$^{-3}$) | (µg m$^{-3}$) | (mm) |
| Pre-LSSR2020 | Count | 1761 | 1155 | 1674 | 1758 | 1754 | 1752 | 1762 | 1760 | 1793 |
| Average | 41.01 | 23.78 | 22.22 | 1.31 | 38.84 | 38.85 | 48.45 | 86.30 | 0.47 |
| Standard deviation | 18.36 | 16.70 | 20.93 | 1.01 | 19.79 | 33.92 | 30.92 | 64.59 | 2.38 |
| Coeff. of variation (%) | 0.45 | 0.70 | 0.94 | 0.77 | 0.51 | 0.87 | 0.64 | 0.75 | 511% |
| Minimum | 9.00 | 0.30 | 0.11 | 0.01 | 2.66 | 0.19 | 1.91 | 1.91 | 0 |
| Maximum | 128.00 | 152.15 | 154.24 | 8.18 | 204.48 | 233.79 | 216.38 | 450.16 | 34.95 |
| Range | 119.00 | 151.85 | 154.13 | 8.17 | 201.82 | 233.60 | 214.47 | 448.25 | 34.95 |
| Std. skewness | 24.35 | 26.88 | 45.11 | 27.27 | 32.35 | 25.12 | 18.80 | 22.03 | 133.0 |
| Std. kurtosis | 22.94 | 43.83 | 98.68 | 31.27 | 57.50 | 20.77 | 11.13 | 15.70 | 609.8 |
| LSSR2020 | Count | 1847 | 1825 | 1810 | 1763 | 1821 | 1825 | 1824 | 1823 | 1848 |
| Average | 49.39 | 26.54 | 7.88 | 0.53 | 52.57 | 10.46 | 22.69 | 33.85 | 0.11 |
| Standard deviation | 20.21 | 15.54 | 7.17 | 0.44 | 31.31 | 11.25 | 13.66 | 24.41 | 0.891 |
| Coeff. of variation (%) | 0.41 | 0.50 | 0.91 | 0.83 | 0.60 | 1.08 | 0.60 | 0.72 | 844% |
| Minimum | 11.00 | 0.05 | 0.03 | 0.01 | 14.16 | 0.55 | 1.42 | 2.86 | 0 |
| Maximum | 161.02 | 130.04 | 76.10 | 4.36 | 212.89 | 109.71 | 118.62 | 228.33 | 18.5 |
| Range | 150.02 | 129.99 | 76.07 | 4.35 | 198.73 | 109.16 | 117.20 | 225.47 | 18.5 |
| Std. skewness | 19.66 | 24.18 | 45.32 | 43.92 | 28.37 | 55.81 | 32.61 | 42.95 | 251.6 |
| Std. kurtosis | 21.40 | 29.79 | 102.82 | 93.01 | 28.47 | 132.14 | 52.92 | 85.38 | 2118.5 |
| Pre-LSSR2019 | Count | 1760 | 1736 | 1696 | 1723 | 1630 | 1719 | 1562 | 1696 | 1766 |
| Average | 49.44 | 36.25 | 28.05 | 1.75 | 52.36 | 47.60 | 37.73 | 80.26 | 0.16 |
| Standard deviation | 27.27 | 22.79 | 12.34 | 1.05 | 48.35 | 35.52 | 24.36 | 46.69 | 1.00 |
| Coeff. of variation (%) | 0.55 | 0.63 | 0.44 | 0.60 | 0.92 | 0.75 | 0.65 | 0.58 | 612% |
| Minimum | 2.00 | 0.55 | 5.94 | 0.05 | 4.38 | 0.87 | 0.22 | 4.51 | 0 |
| Maximum | 173.67 | 406.22 | 131.54 | 6.23 | 355.46 | 231.22 | 148.49 | 307.28 | 19.6 |
| Range | 171.67 | 405.67 | 125.60 | 6.18 | 351.08 | 230.35 | 148.27 | 302.77 | 19.6 |
| Std. skewness | 19.42 | 58.16 | 43.16 | 18.93 | 31.66 | 24.51 | 15.36 | 13.48 | 193.7 |
| Std. kurtosis | 12.12 | 349.19 | 82.22 | 10.78 | 37.14 | 20.72 | 8.80 | 4.34 | 1388.8 |
| LSSR2019 | Count | 1843 | 1823 | 1811 | 1820 | 1819 | 1817 | 1813 | 1818 | 1848 |
| Average | 46.13 | 45.43 | 13.14 | 1.72 | 39.45 | 38.86 | 43.83 | 83.95 | 0.11 |
| Standard deviation | 18.77 | 21.29 | 8.54 | 0.95 | 34.32 | 29.10 | 20.96 | 36.93 | 0.87 |
| Coeff. of variation (%) | 0.41 | 0.47 | 0.65 | 0.55 | 0.87 | 0.75 | 0.48 | 0.44 | 776% |
| Minimum | 8.00 | 5.01 | 0.32 | 0.22 | 6.99 | 0.06 | 2.97 | 10.26 | 0 |
| Maximum | 151.54 | 197.80 | 63.30 | 6.36 | 221.54 | 202.42 | 142.72 | 230.74 | 17.9 |
| Range | 143.54 | 192.79 | 62.98 | 6.14 | 214.55 | 202.36 | 139.75 | 220.48 | 17.9 |
| Std. skewness | 20.37 | 15.86 | 27.30 | 23.04 | 32.09 | 25.82 | 13.36 | 10.90 | 219.7 |
| Std. kurtosis | 28.12 | 21.56 | 26.71 | 17.21 | 31.81 | 24.64 | 9.61 | 1.94 | 1710.5 |
Table 2. Results of the Kruskal-Wallis ANOVA on Ranks and Bonferroni Procedure for Pairwise Comparisons.$^a$

|       | LSSR2020-PreLSSR2020 | Pre2019-PreLSSR2020 | LSSR2019-PreLSSR2020 | PreLSSR2019-LSSR2020 | LSSR2019-LSSR2020 | LSSR2019-PreLSSR2019 |
|-------|----------------------|---------------------|----------------------|----------------------|-------------------|----------------------|
| PM$_{10}$ | +                     | +                   | +                    | -                    | -                 | -                    |
| PM$_{2.5}$ | +                     | +                   | +                    | +                    | +                 | +                    |
| SO$_2$ | -                     | +                   | -                    | +                    | +                 | -                    |
| CO    | -                     | +                   | +                    | +                    | +                 | +                    |
| O$_3$ | +                     | -                   | -                    | -                    | -                 | -                    |
| NO    | -                     | +                   | +                    | +                    | +                 | -                    |
| NO$_2$ | -                     | -                   | +                    | +                    | +                 | +                    |
| NO$_x$ | -                     | +                   | +                    | +                    | +                 | +                    |
| Rain  | -                     | +                   | -                    | -                    | -                 | +                    |

$^a$ Entries are significant at the 95th recent confidence level.

Fig. 2. Box and whisker plots of the distributions of PM$_{10}$ (Top) and PM$_{2.5}$ (Bottom) in µg m$^{-3}$ for the periods: 0–Pre-LSSR 2020, 1–LSSR 2020, 2–Pre-LSSR 2019, 3–LSSR 2019.

CO, and NO$_2$ into the atmosphere are combustion of carbon containing fuels (fossil fuels, coal, wood, etc.) and the resulting emissions from cars, trucks, buses, and power plants (WHO, 2005; Rahman et al., 2019).

Box and whisker plots for the O$_3$ is given in Fig. 5. The average ozone concentration (secondary pollutant) was higher in the LSSR 2020 compared to the Pre-LSSR 2020 and LSSR 2019 periods that can be understood by changes in urban photochemistry. The reduction in primary NO$_x$ emissions led to reduced titration of the O$_3$ and increased the O$_3$ concentration. Thus, the O$_3$ concentrations were highest during the LSSR 2020 period. Meteorological factors also affected
the measured ozone concentrations as well as the VOC emissions from local and regional non-traffic sources. This result is similar to what has been observed in many other locations around the world (e.g., Abdullah et al., 2020; Chen et al., 2020; Mahato et al., 2020; Huang et al., 2021; Qiu et al., 2021). The increased ozone suggests that Jakarta is in a VOC-limited regime for ozone formation (Seinfeld and Pandis, 2016) so that reducing NO emissions led to increased ozone. There are relatively small differences in the monthly average photoperiods ranging from 12.4 h in January to 11.8 h in June so that variations in photochemical activity and temperature are much less than in many other locations observing increased ozone concentrations.

We also compared the diel patterns between LSSR 2020 period and corresponding period in 2019 and the results are presented in Fig. 6. Even though the patterns are typical, but the concentrations of SO2, CO, and NO in 2019 are far higher than those measured during the LSSR 2020 period especially for the maximum diel values. For ozone, the diel average concentrations are in a comparable magnitude for both periods. Non-day time ozone concentrations in LSSR 2020 period are slightly higher than the corresponding period in 2019 may be due to less NOx emissions.

Fig. 6 shows the O3 diel patterns with sharp rises beginning at 10:00 in both LSSR periods, but higher values in 2020. The NO during the morning rush hour was significantly reduced from the comparable time of day in the other 3 periods (see Supplemental Material S5 for the detailed statistical analyses). A clearer evening rush hour can be observed in the LSSR 2020 period, but the values are generally reduced compared to the others suggesting a substantial reduction in vehicular traffic that would be the major NOx source. The morning rush hour in the LSSR 2019 period is quite pronounced showing the extent of traffic emissions under more typical traffic volumes that occurred in 2019. The LSSR 2020 NOx and NO2 distributions are uniformly lower across the 24-hourly distributions. The CO morning rush hour peak was distinctly lower during LSSR 2020 than in any of the other 3 periods suggesting that the reduced mobility period was
Fig. 4. Box and whisker plots of the distributions of (a) SO₂, (b) CO, (c) NO, (d) NO₂, and (e) NOₓ for different periods.

Fig. 5. Box and whisker plots of the distributions of O₃ for different periods.
Fig. 6. Diel averages of \( \text{SO}_2 \), \( \text{CO} \), \( \text{O}_3 \) and \( \text{NO} \) (not presented in the same scale): (a) LSSR 2020, (b) LSSR 2019.
successful in reducing the light duty traffic volume. The diel pattern of SO2 distributions in the Pre-LSSR periods showed relatively uniform hourly values over the whole day. In both LSSR periods, there were small increases from 8:00 to noon and a drop into the afternoon. Diesel fuel in Indonesia has a high sulfur content (3000 ppm) so they represent a local SO2 source. However, heavy-duty diesel trucks are restricted to overnight hours. Thus, this daytime rise seems likely to be the result of downmixing of emissions from the stacks of coal-fired power plants and oil refineries in West Java (Santoso et al., 2020). These results strongly suggest that the restricted mobility rules reduced motor vehicle traffic and some industrial activities resulting in decreased concentrations of CO, SO2, and NOx. However, there were increasing O3 concentration that can be understood in terms of decreased NO titration and sufficient reactive hydrocarbon concentrations to support ozone formation.

3.2 PM2.5-10 and PM2.5 Mass and Multi-elemental Compositions

This section presents the results of our long-term PM and composition monitoring to investigate the impact of LSSR on PM air quality in Jakarta (Santoso et al., 2020). Both coarse and fine PM monitoring results are presented and discussed separately.

3.2.1 Coarse PM (PM2.5-10)

We compared the period average (derived from daily average concentrations) of PM2.5-10 and compositions between pre-LSSR and LSSR period and the results are presented in Table 3 and Fig. S3. The average PM2.5-10 concentrations during the LSSR 2020, pre-LSSR 2020 and LSSR 2019 periods were 11.38 µg m\(^{-3}\), 22.16 µg m\(^{-3}\), and 28.05 µg m\(^{-3}\), respectively. Compared to the other periods, the implementation of LSSR in 2020 reduced the concentration of PM2.5-10, as well as reduced the concentrations of its constituents. Crustal elements such as Al, Si, Ca, and Fe had a dominant contribution to PM2.5-10 (Santoso et al., 2020). The concentrations of the crustal elements decreased by more than 50%. There was also a substantial decrease in sulfur (particulate sulfate) compared to both the LSSR 2019 and pre-LSSR2020 periods, indicating a decrease in traffic intensity. The elements of K, Zn, and Pb are key indicators of anthropogenic activity and also showed decreased by more than 40% compared to the same period in 2019. During the LSSR period, there were restrictions on both commercial and social activities, so it was expected that there would be emissions reductions from street cooking, construction, and other commercial activities.

|         | Jan–15 March 2020 | 16 March–May 2020 | 16 March–May 2019 |
|---------|-------------------|-------------------|-------------------|
|         | Mean | SD   | Min  | Max  | Mean | SD   | Min  | Max  | Mean | SD   | Min  | Max  |
| PM2.5-10 | µg m\(^{-3}\) | 22.16 | 7.17 | 10.18 | 36.30 | 11.38 | 4.56 | 4.90 | 18.20 | 28.05 | 12.78 | 11.13 | 59.36 |
| Na      | ng m\(^{-3}\)   | 250  | 79   | 159  | 443  | 164  | 112  | 46   | 414  | 708  | 408  | 81   | 1521 |
| Mg      | ng m\(^{-3}\)   | 51   | 34   | 14   | 129  | 49   | 29   | 16   | 99   | 108  | 58   | 13   | 201  |
| Al      | ng m\(^{-3}\)   | 472  | 149  | 226  | 703  | 194  | 83   | 66   | 334  | 608  | 182  | 273  | 962  |
| Si      | ng m\(^{-3}\)   | 1195 | 387  | 555  | 1840 | 515  | 199  | 227  | 857  | 1503 | 436  | 679  | 2313 |
| S       | ng m\(^{-3}\)   | 279  | 98   | 147  | 468  | 195  | 93   | 50   | 321  | 597  | 307  | 186  | 1332 |
| Cl      | ng m\(^{-3}\)   | 510  | 229  | 190  | 1075 | 213  | 101  | 83   | 401  | 238  | 134  | 32   | 562  |
| K       | ng m\(^{-3}\)   | 133  | 41   | 74   | 220  | 87   | 35   | 37   | 137  | 192  | 82   | 76   | 434  |
| Ca      | ng m\(^{-3}\)   | 1012 | 352  | 483  | 1663 | 521  | 231  | 174  | 937  | 1077 | 332  | 497  | 1919 |
| Ti      | ng m\(^{-3}\)   | 48.47| 15.33| 20.97| 77.96| 23.13| 8.14 | 9.78 | 36.49| 63.47| 18.80| 26.90| 91.56|
| V       | ng m\(^{-3}\)   | 1.30 | 0.63 | 0.42 | 2.47 | 0.67 | 0.36 | 0.18 | 1.29 | 1.77 | 0.90 | 0.01 | 3.42 |
| Cr      | ng m\(^{-3}\)   | 4.25 | 1.41 | 2.30 | 7.22 | 1.58 | 1.06 | 0.27 | 3.04 | 4.85 | 1.71 | 2.03 | 8.23 |
| Mn      | ng m\(^{-3}\)   | 11.57| 5.80 | 2.55 | 23.48| 6.85 | 1.69 | 5.02 | 9.77 | 24.02| 11.56| 7.56 | 54.07|
| Fe      | ng m\(^{-3}\)   | 657  | 210  | 297  | 1018 | 284  | 105  | 112  | 457  | 908  | 265  | 423  | 1303 |
| Ni      | ng m\(^{-3}\)   | 0.55 | 0.45 | 0.11 | 1.32 | 0.64 | 0.42 | 0.09 | 1.24 | 1.80 | 0.64 | 0.72 | 3.14 |
| Cu      | ng m\(^{-3}\)   | 13.34| 4.90 | 6.64 | 21.75| 5.05 | 1.94 | 2.63 | 7.69 | 14.19| 6.16 | 5.31 | 31.95|
| Zn      | ng m\(^{-3}\)   | 79.45| 32.80| 38.96| 134.26| 38.89| 17.24| 6.27 | 60.44| 267.90| 197.50| 53.28| 753.16|
| Pb      | ng m\(^{-3}\)   | 17.75| 22.80| 2.65 | 83.83| 12.19| 13.88| 0.98 | 42.40| 23.96| 23.03| 1.69 | 85.74|
Fig. 7. Comparison of monthly average (a) PM$_{2.5-10}$ and (b) PM$_{2.5}$ concentration between 2020 and 2010–2019.

Monthly PM$_{2.5-10}$ concentrations during the implementation of the LSSR 2020 (March–May 2020) were compared to the long-term monthly averages from 2010–2019 (Santoso et al., 2020) for the same months. The results are presented in Fig. 7(a). The monthly averages of PM$_{2.5-10}$ during the LSSR period were lower than the long-term averages. For example, the March concentration was only 10 $\mu$g m$^{-3}$ while the long-term average value was 27 $\mu$g m$^{-3}$. Similar results were seen for April and May where the long-term average concentrations were more than double of the 2020 measurements confirming the effect of the LSSR on coarse PM at this site in Jakarta.

3.2.2 Fine PM (PM$_{2.5}$)

Period averages of PM$_{2.5}$ and elements were compared for pre-LSSR and LSSR periods and the results are presented in Table 4 and Fig. S3. In addition, measurement of BC was also reported as an indicator of diesel traffic. There was a clear reduction in PM$_{2.5}$ concentrations from the pre-LSSR 2020 (14.19 $\mu$g m$^{-3}$) to the LSSR 2020 period (10.45 $\mu$g m$^{-3}$) and compared to LSSR 2019 (18.76 $\mu$g m$^{-3}$). PM$_{2.5}$ is mainly derived from anthropogenic sources such as fossil fuel combustion. Thus, it is likely related primarily to the reductions in the traffic activity and to a lesser extent on reductions of other combustion sources. Compared to 2019, the average S concentration during LSSR 2020 decreased by more than 50% from 1333 ng m$^{-3}$ to 543 ng m$^{-3}$, but the concentration did not change compared to pre-LSSR 2020. Average S concentration was similar between the LSSR 2020 period and the pre-2020. The value was also lower than in the LSSR 2019 period. BC concentrations declined by more than 30% due to the reduction of on-road transport. Compared to the 2019 period, there was a decrease in the concentration of the elements Pb, Zn, Cu, Fe in the LSSR 2020 period by more than 50%, while K and Ca declined by ~30%. The decrease in
Table 4. Period average of mass concentrations (in µg m⁻³) and multi-elemental concentration (in ng m⁻³) of PM₂.₅ at Jakarta from September 2019 to May 2020.

| Element | Jan–15 March 2020 | 16 March–May 2020 | 16 March–May 2019 |
|---------|-------------------|-------------------|-------------------|
|         | Mean   | SD      | Min     | Max     | Mean   | SD      | Min     | Max     | Mean   | SD      | Min     | Max     |
| PM₂.₅  | µg m⁻³  | 14.19   | 3.83    | 9.19    | 22.15   | 10.45   | 4.50    | 5.00    | 18.30   | 18.76  | 10.26   | 1.85    | 37.10   |
| BC      | µg m⁻³  | 3.07    | 0.96    | 1.05    | 4.66    | 2.12    | 0.57    | 1.07    | 2.63    | 3.86   | 2.16    | 0.41    | 6.99    |
| Na      | ng m⁻³  | 169     | 50      | 106     | 270     | 121     | 41      | 47      | 170     | 553    | 603     | 19      | 2205    |
| Mg      | ng m⁻³  | 43      | 20      | 18      | 76      | 21      | 8       | 7       | 30      | 23     | 20      | 1       | 89      |
| Al      | ng m⁻³  | 56      | 31      | 2       | 118     | 52      | 12      | 33      | 69      | 125    | 68      | 45      | 300     |
| Si      | ng m⁻³  | 166     | 65      | 80      | 307     | 81      | 26      | 49      | 132     | 167    | 115     | 17      | 414     |
| S       | ng m⁻³  | 547     | 230     | 172     | 955     | 543     | 289     | 100     | 885     | 1333   | 1110    | 113     | 4094    |
| Cl      | ng m⁻³  | 10.19   | 4.91    | 2.58    | 16.88   | 9.29    | 3.10    | 4.41    | 12.95   | 16.84  | 18.35   | 1.45    | 76.65   |
| K       | ng m⁻³  | 137     | 43      | 83      | 216     | 145     | 65      | 80      | 267     | 215    | 158     | 15      | 697     |
| Ca      | ng m⁻³  | 142     | 54      | 66      | 256     | 65      | 31      | 32      | 131     | 101    | 78      | 9       | 291     |
| Ti      | ng m⁻³  | 8.01    | 2.62    | 4.83    | 14.44   | 4.27    | 1.55    | 2.09    | 7.35    | 8.56   | 6.68    | 0.23    | 20.93   |
| V       | ng m⁻³  | 0.64    | 0.33    | 0.13    | 1.16    | 0.24    | 0.19    | 0.02    | 0.54    | 0.66   | 0.43    | 0.12    | 1.87    |
| Cr      | ng m⁻³  | 0.99    | 0.54    | 0.38    | 2.29    | 0.41    | 0.23    | 0.17    | 0.67    | 1.79   | 1.20    | 0.15    | 3.98    |
| Mn      | ng m⁻³  | 2.18    | 1.49    | 0.68    | 6.13    | 2.46    | 1.33    | 0.52    | 5.01    | 6.62   | 6.71    | 0.65    | 25.31   |
| Fe      | ng m⁻³  | 114     | 36      | 69      | 197     | 55      | 21      | 26      | 99      | 148    | 110     | 13      | 361     |
| Ni      | ng m⁻³  | 0.71    | 0.35    | 0.11    | 1.18    | 0.21    | 0.14    | 0.03    | 0.38    | 0.75   | 0.46    | 0.02    | 1.74    |
| Cu      | ng m⁻³  | 14.00   | 8.76    | 2.52    | 34.07   | 2.59    | 1.27    | 1.00    | 4.93    | 9.26   | 6.70    | 0.45    | 27.20   |
| Zn      | ng m⁻³  | 62      | 24      | 25      | 98      | 35      | 16      | 3       | 53      | 206    | 290     | 16      | 1328    |
| Pb      | ng m⁻³  | 35.08   | 52.40   | 4.10    | 196.56  | 19.44   | 24.77   | 0.93    | 76.84   | 54.52  | 71.36   | 0.13    | 328.41  |

Elemental concentrations during the LSSR 2020 period was a result of the reduced anthropogenic activities during the LSSR implementation in Jakarta.

Long term monthly average PM₂.₅ concentrations were calculated for March, April and May during the period of 2010–2019. We then compared the values with those calculated for the year of 2020 when the LSSR was implemented and the result is presented in Fig. 7(b). Typical reduction of concentration during the LSSR were also clearly seen. In all months, the PM₂.₅ concentration reduction ranged from 31% to 38%. This reduction was comparable to the values reported in the other studies conducted in India and China (Chen et al., 2020; Mahato et al., 2020).

3.3 Aerosol Optical Depth Observation

Fig. 8. shows the monthly average AOD during the pre-LSSR and LSSR periods. Overall, the monthly average AOD observed during the LSSR period were higher than those measured from October–December 2019, but lower than in January and February 2020. The period average values showed a clear AOD reduction during the LSSR period compared to the Pre-LSSR (normal).

AOD measurement estimates the aerosol burden in a column of atmosphere therefore it is not only affected by the ground emission but also the upper air meteorology. The average values were affected by the number of daily data available hence data completeness is important. The AOD data were correlated with the ground PM measurements at the same site. The data provided an indication that the improvement of PM air quality during the LSSR period was also captured by the ground based AOD observations.

The AOD was compared with the PM₂.₅ concentrations obtained from the two sites presented above (filter-based (daily) and continuous monitoring) for the period of Oct 2019–May 2020 (See Fig. S4). Comparison of more than 25 data pairs for the second site showed relatively good correlation between AOD and our PM₂.₅ data (showed by coefficient of determination value, r² of 0.596). A comparison for more than 100 data pairs between AOD and PM₂.₅ measured at EPA’s automatic monitoring station also showed a moderate correlation with r² = 0.453. Therefore, there was a consistent PM reduction as shown by these data.

The MODIS Terra AOD (for both land and ocean) were retrieved for periods: a) March–May 2018, b) March–May 2019, c) Nov 2019–Feb 2020, and d) March–May 2020 (LSSR implementation). The spatial distribution of the MODIS Terra AOD for the selected periods are presented in Fig. 9.
The observed AOD value over the Jakarta area for the period of March–May 2020 were consistently lower than the values observed in other 3 periods. The values ranged from 0.3 to 0.4 during the LSSR period while in other periods consistently ranged from 0.4 to 0.5.

These findings supported the results from the ground AOD observations. Larger scale observation by satellite observed similar AOD reduction patterns especially above the Jakarta area. An exceptional situation was seen in Fig. 9(c) since in the September to February period, a forest fire hotspot was seen over the provinces of Riau and Jambi (Sumatera Island). However, it would not affect Jakarta's air quality due to the southwest monsoon synoptic winds. The reduction was also seen for other areas especially in the western part of Java Island.
4 CONCLUSIONS

Various air quality observations combining ground-based and satellite derived data were utilized to investigate the impact of the LSSR 2020 on urban air quality in Jakarta, Indonesia. Continuous monitoring data from the AQMS installed in Central Jakarta showed reductions in PM$_{2.5}$, NO$_2$, CO, and SO$_2$ concentrations during the LSSR period as compared to the period before (normal). However, ozone increased in the LSSR 2020 period. Our long-term PM monitoring at another site located in the southern part of the city showed consistent substantial reductions of coarse and fine PM as well as major elements during the LSSR period. The findings were enhanced by the ground-based and MODIS Terra AOD observations which showed exceptional lower AOD values during the period where traffic and other anthropogenic activities were reduced. While the results indicated that the government’s program was rather successful as seen from the air quality monitoring data, the situation maybe different in other areas where the data do not exist. It is suggested that this study should be followed up by the epidemiological research to investigate the potential short-term health benefits.

ACKNOWLEDGMENTS

The authors acknowledge the National Nuclear Energy Agency (BATAN) for the financial support throughout this work. The authors also gratefully acknowledge to all the staffs in Radiometry Analytical Techniques group in Center for Applied Nuclear Science and Technology BATAN and Jakarta Provincial EPA for their technical assistance. Besides that, the authors also thank the International Atomic Energy Agency for financial support through a research project no. TCINS7007. AERONET and BMKG are highly acknowledged for providing the AOD data for this study. Muhayatun Santoso, Philip K Hopke, and Didin Agustian Permadi are the main contributors to this article.

DISCLAIMER

Reference to any companies or specific commercial products does not constitute an endorsement by the authors, the Center for Applied Nuclear Science and Technology BATAN, the Jakarta Provincial EPA, or the International Atomic Energy Agency.

SUPPLEMENTARY MATERIAL

Supplementary material for this article can be found in the online version at https://doi.org/10.4209/aaqr.200645

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