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Air quality changes in a Central European city during COVID-19 lockdown

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ABSTRACT

A comparison of mobile and stationary air quality measurements in Lublin, Poland during the COVID-19 lockdown in 2020 and in a comparable period in 2017 has demonstrated that a substantial decrease of the traffic intensity by more than 50%, especially during certain times of the day in the lockdown period has only been partially reflected in the air quality improvement in the city. Mobile measurements carried out during six runs within a 24-hour period in 2017 and 2020 indicated a decrease of the average $PM_{2.5}$ and PM_{10} concentrations by ~ 30% and ~14%, respectively. In turn, stationary measurement results obtained for the same periods demonstrated their increase by respectively ~35% and ~106% and a decrease in the average NO_2, NO_x, C_6H_6 and CO concentrations. This could have been impacted by meteorological factors and emissions from other, non-traffic-related sources, mainly from residential coal burning. The changes in the vehicle fleet structure could also have played a role.

1. Introduction

Road traffic is considered as a major source of a wide variety of air pollutants in urban areas (Polednik et al., 2018; Xiang et al., 2020). Many studies have demonstrated clear connections between exposure to air pollution components and negative health effects for people living in urban areas and near roads and motorways (Baldauf et al., 2009; Font and Fuller, 2016). Therefore, it is common for various restrictions and measures to be introduced, aimed at the reduction of road traffic. In 2020, due to the outbreak of COVID-19 (Coronavirus Disease 2019), there has been an unprecedented decrease in traffic in almost all cities around the globe as well as in overall population mobility (Shakibaei et al., 2021; Beria and Lunkar, 2021). In the Central and Eastern Europe (CEE), similarly to other regions worldwide, traffic volume reduction as well as its pattern alteration has been observed. Significant changes in this respect were visible in Poland during the near month-long lockdown introduced on 25 March 2020 which included a ban on commuting (subject to certain exceptions) and resulted in a significant decrease of road traffic throughout the country. In accordance with the General Directorate for National Roads and Highways (GDDKiA, 2020), the daily traffic in April 2020 was on average approximately 42% less intense than the traffic recorded in an analogous period in 2019. Major changes in the fleet structure have also been observed. The share of passenger cars with mostly gasoline or LPG (Liquefied Petroleum Gas) engines decreased to approximately 61% (as compared to 71% before the pandemic) with a simultaneous increase of the share of trucks (with diesel engines) to approximately 24% (16% before the pandemic). In Lublin, which is the largest city in eastern Poland with a population of almost 350,000 inhabitants and an area of 147 km^2, the congestion level during the lockdown decreased by more than 50% (TomTom Traffic Index, 2020). The number of municipal transportation passengers has decreased as well. The Public Transport Authority in Lublin (ZTM, 2020) states that it was reduced by almost 90%. Lublin which is not considered as an industrial city is one of the most congested cities in Europe. This mainly results from the absence of adequate transportation infrastructure and high traffic intensity combined with a large number of relatively old, mainly gasoline- and diesel-powered vehicles. Therefore, vehicular emissions are a significant factor contributing to the total pollutant concentrations in Lublin (Polednik et al., 2018).

While comparing urban road traffic characteristics during the lockdown with the characteristics from before or after the lockdown one has to take into consideration that the traffic intensity depends on many factors (Duque et al., 2016; Farda and Balijepalli, 2018). For example, weather conditions or changes in the road network may be of importance as well as different calendars of public holidays or non-working days. Therefore, the analyses dealing with traffic intensity changes should strive to consider the greatest possible number of conditions that may impact the size of such changes.

Bearing in mind the extraordinary changes in urban traffic during the COVID-19 pandemic, one should anticipate a relatively large decrease of

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air pollutant levels. And indeed this is confirmed by research carried out in many cities worldwide which demonstrate clear evidence of lower air pollutant concentrations (Bauwens et al., 2020; Xu et al., 2020). However, there are also studies that report higher than expected air pollution levels. This is explained by the existence of non-COVID-19-related factors which primarily include different weather conditions in the compared periods before and during the lockdown (Zhao et al., 2020; Bekbulat et al., 2020).

Studies carried out to date paid relatively little attention to the changes of emissions from non-traffic related sources and the quality changes of traffic-related emissions. One can assume that apart from traffic intensity and the differences in the weather conditions, also varying air pollution sources and changes in the structure of the transportation fleet could have a significant impact on urban air quality.

The aim of this paper is to evaluate the impact of the COVID-19 lockdown on traffic-related air pollution in the center of Lublin. Such impact is specifically examined based on the comparison of mobile and stationary air pollutant levels measurements during the lockdown in the spring of 2020 and in a corresponding period in 2017.

2. Materials

Research included mobile and stationary air pollution measurements carried out in 2017 as well as during the lockdown introduced due to the COVID-19 outbreak in 2020. Mobile measurements were performed along a 2.1 km stretch of one of the busiest streets in the city which included three 4-way intersections with traffic lights (Fig. 1). Before the pandemic, the maximum traffic intensity was usually registered in the mornings and afternoons (7:00–9:00 and 15:00–17:00, respectively) with traffic volume at each of the intersections exceeding 2000 vehicles per hour (RBA, 2018).

Mobile Air Pollution Analytic Laboratory (MAPAL) was used to perform mobile measurements. The route as well as the location of the sampling points, instruments and measurement procedures have been described in detail in Polednik and Piotrowicz (2020).

Measurement instruments installed in MAPAL included DustTrak DRX model 8533 (TSI Inc., USA) which was used to determine the mass concentrations of PM_1, $PM_{2.5}$, PM_{10}, RESP and TSP (particles with an aerodynamic diameter equal or less than 1, 2.5, 10 μm, respirable and total particles, respectively). Real time measurements of number concentrations of particles (PN) with the size ranging from 10 nm up to 32 μm as well as the mass concentrations of PM_1, $PM_{2.5}$, PM_{10} were performed using Grimm Aerosol Spectrometer 1.109 with Nano Sizer 1.321 (Grimm Aerosol, Germany). P-Trak model 8525 (TSI Inc., USA) was applied for measuring number concentrations of particles PN_1 within the size range from 0.02 to about 1 μm. The number concentrations of particles greater than 0.3 μm ($PN_{0.3–0.5}$, $PN_{0.5–1}$, $PN_{1–2}$, $PN_{2–5}$, $PN_{5–10}$ and $PN_{>10}$) were measured using optical spectrometer OPS 3330 (TSI Inc., USA). All instruments were calibrated by their manufacturers before the planned measurements and they were additionally subject to in-field calibration in accordance with the operation and service manuals. The logging interval for the instruments was set to 6 seconds. Air samples were provided to MAPAL instruments through tubes with endpoints located in the middle of the vehicle, on the left side, at the height of 1.7 m. Global Positioning System (GPS; Garmin Nuvi 2460LMT) was used to continuously record MAPAL's speed and position, while an HD 1080P Wide angle 170° camera was used to collect the traffic flow data. Air temperature and relative humidity outside and

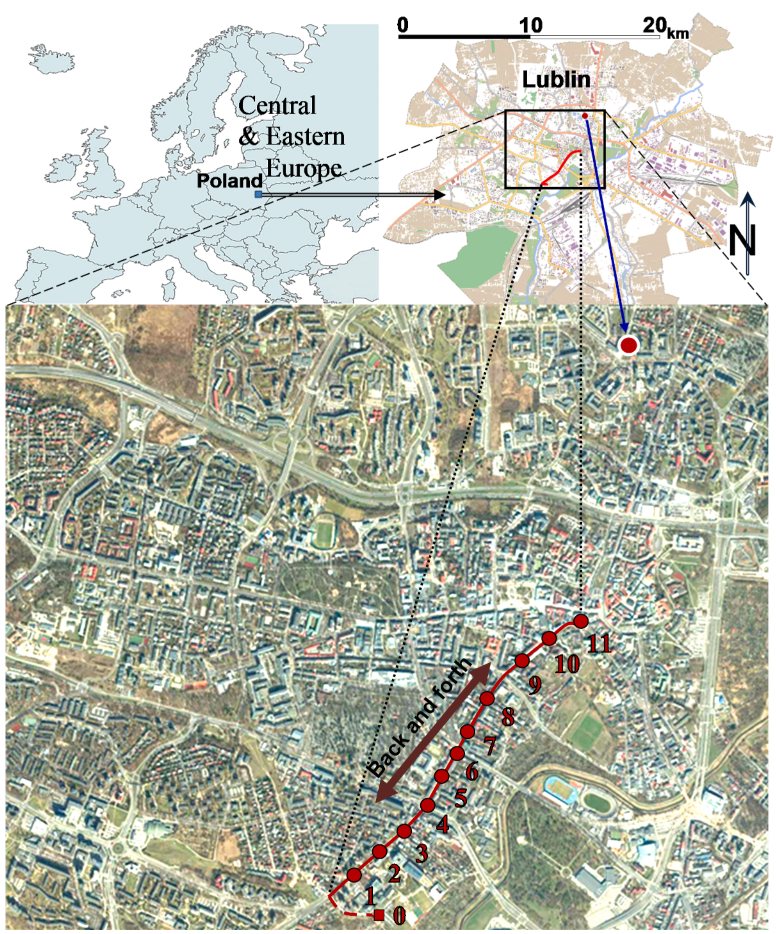

Fig. 1. Map of Europe with Lublin marked and satellite image (Geoportal.gov.pl) of the route for mobile monitoring with sampling points and the location of the air quality monitoring station.

inside the vehicle was measured by thermo-hygrometer LB-520 (LAB-EL, Poland).

The particle concentration results obtained during mobile measurements are considered as estimates of their actual values, as they were impacted by the type of the applied apparatus – its calibration and other related factors. For example, in previously conducted studies the average concentrations of PM_{10}, $PM_{2.5}$ and PM_1 measured by the Grimm were respectively 1.3, 1.7 and 2.1 times lower than the results determined using Dusttrak instrument. For PN_1 concentrations, the values obtained using Grimm were 1.8 times lower than the concentrations measured by P-Trak (Polednik et al., 2018).

The ambient air particulate sampling was performed during runs in both directions along the route. Six runs a day were made in 4-hour intervals. The route also included 11 stop points (with MAPAL parked on the sidewalk) at which measurements lasting 5 minutes each were carried out. The duration of a full run depended on the time of the day and the traffic intensity. In 2017 during peak hours it took 103 ±12 min, while at night 92 ±8 min. In 2020 the duration of test runs was significantly shorter – during the day they took about 90 minutes, while at night only about 70 minutes. It was assumed that the traffic conditions during a single run were the same.

Due to the absence of long-term measurements as well as measurements that would not be significantly impacted by various factors and that could be compared, this paper only analyzes results obtained during six consecutive runs in a 24-hour period on 4–5 April 2017 (Tuesday–Wednesday) and on 16–17 April 2020 (Thursday–Friday). The selected measurement days were characterized by similar mean ambient temperatures during the day/at night of approximately 17/8.5 °C (Table 1). They differed in terms of relative humidity and wind speed which amounted to 57/72% and 13/13 km h^{-1} in 2017 and 38/54% and 28/10 km h^{-1} in 2020 (https://www.weatheronline.com).

The stationary measurement results were collected from an air quality monitoring station located approximately 3 km from the monitored route in a residential area with predominantly 2-4-storey buildings heated mainly by means of coal stoves. The buildings are separated from one another with small green areas (gardens and trees). The monitoring station is located about 0.1 km from a street characterized by low traffic intensity near a hospital compound surrounded by trees. A more busy road is located approximately 0.6 km away from the monitoring station in the vicinity of an old wooded cemetery.

The station recorded the concentrations of $PM_{2.5}$, PM_{10}, as well as NO_2, NO_x, SO_2, C_6H_6, O_3 and CO in 1-hour intervals (https://powietrze.gios.gov.pl).

3. Results

Figure 2a presents the time series of mass concentrations of submicron (PM_1) and coarse (PM_{10}) particles measured using DustTrak and fine ($PM_{2.5}$) particles determined by the Grimm instrument in one-minute intervals obtained in mobile measurements during six consecutive test runs in 2017 and 2020. The time series of ultrafine ($PN_{0.1}$) and total (PN) particle number concentrations obtained using Grimm as well as submicron particle number concentrations (PN_1) determined by P-Trak in one-minute intervals are shown in Figure 2b.

Elevated particle concentrations in certain specific locations along the monitored route recorded by all the applied instruments during test runs in 2017 and, to a lesser extent, in 2020 are mostly related to the increase of traffic-related emissions. Such hotspots include mainly intersections with traffic lights.

Basic statistical information concerning the PM and PN concentrations measured respectively by Dusttrak and Grimm instrument in individual test runs w 2017 and 2020 is presented in Table 2. Significantly lower average PM concentration values in 2020 ($p < 0.001$) were obtained in test runs during the day and in the evening (at 8:00, 12:00, 16:00, 20:00). In turn, during test runs performed at night (at 0:00 and 4:00) they were significantly higher than in 2017. The average PN concentration values during each test run in 2020 were significantly lower ($p < 0.001$) than the average values obtained during corresponding runs in 2017. Overall average PM_1 and $PN_{0.1}$ concentrations obtained during mobile measurements in 2020 amounted to 24.1 ±10.9 μg m^{-3} and (4.1 ±1.4) x10^3 pt cm^{-3} (mean ±standard deviation) and they were lower by respectively ~31% and ~51% than in 2017. Similar changes were seen with respect to the overall average $PM_{2.5}$ and PN concentrations which in 2020 amounted to 24.7 ±11.0 μg m^{-3} and (4.9 ±1.6) x10^3 pt cm^{-3} while in 2017 were 35.5 ±20.3 μg m^{-3} and (10.0 ±6.5) x10^3 pt cm^{-3}, respectively. The overall average PM_{10} concentration during runs in 2020 was 36.5 ±11.3 μg m^{-3} and in 2017 was 42.3 ±43.6 μg m^{-3} which translated into a ~14% reduction. During mobile measurements in 2020 the average ratio of $PM_{2.5}$ to PM_{10} amounted to about 0.65 as opposed to 0.87 in 2017 and was approximately lower by 24%.

Figure 3 presents concentration changes of the main air pollutants ($PM_{2.5}$, PM_{10}, NO_2, NO_x, SO_2, C_6H_6, O_3 and CO) recorded by the air quality monitoring station during two days covering the mobile measurement test run periods (R) in 2017 and 2020. Average $PM_{2.5}$ and PM_{10} concentrations as well as average SO_2 and O_3 concentrations obtained during stationary measurements corresponding to the time of the test runs were generally higher in 2020 as compared to 2017. NO_2, NO_x, C_6H_6 and CO concentrations were distinctly lower. The mean hourly concentrations of $PM_{2.5}$ and PM_{10} obtained in stationary measurements in periods of the conducted test runs R amounted to 16.1 ±2.3 μg m^{-3} and 56.2 ±23.5 μg m^{-3} and were respectively ~35% and ~106% higher than in 2017. The average $PM_{2.5}$ to PM_{10} ratio amounted to about 0.35 in 2020 and to about 0.46 in 2017 i.e. it was lower by approximately 24% in 2017. The mean hourly concentrations of NO_x and CO (examples of pollutants for which concentration decrease was recorded) amounted to 9.5 ±5.0 μg m^{-3} and 117 ±58 μg m^{-3} in 2020 and 26.3 ±10.1 μg m^{-3} and 418 ±96 μg m^{-3} in 2017 which entails a respective reduction by approximately 64% and 72%. However, it needs to be clearly noted that the values are estimated based on relatively short-term measurements performed in a monitoring station which was not located in the direct proximity of the considered route and was situated in a substantial distance from traffic-intense streets. The obtained differences in the pollutant concentrations indicate that in 2020 (as compared to 2017) air pollution sources other than traffic-related ones were dominant in the vicinity of the monitoring station and presumably in the entire city. One of such probable sources is residential combustion for heating purposes, which could have been more intense due to less favorable weather conditions – greater wind speed.

Figure 4 presents average $PM_{2.5}$ and PM_{10} concentrations obtained in individual test runs during mobile measurements (data from Dusttrak) and results for the same periods obtained in stationary measurements in 2020 and 2017, average 2020/2017 ratios of such particle concentrations as well as average $PM_{2.5}$ to PM_{10} ratios obtained during measurements in both considered years.

The presented line graphs indicate that the change pattern of average $PM_{2.5}$ and PM_{10} concentrations obtained during the subsequent test runs in 2017 corresponds to the change pattern of those concentrations obtained in stationary measurements, with a reservation that higher values were obtained during mobile measurements. For both mobile and stationary measurements the highest values of the analyzed particle

Table 1
Weather conditions on measurement days in 2017 and 2020.

Date		T [°C]	RH [%]	w [km h^{-1}]	wd	p [hPa]	Description
2017	04.04	17/9	59/69	18/12	NW	1012	sunny/clear
	04.05	18/9	55/75	7/13	NW	1016	sunny/clear
2020	04.16	18/10	32/68	40/10	E	1001	sunny/cloudy
	04.17	14/6	43/39	15/9	NE	996	sunny/cloudy

Average day/night temperature (T), relative humidity (RH), wind speed (w), wind direction (wd), atmospheric pressure (p).

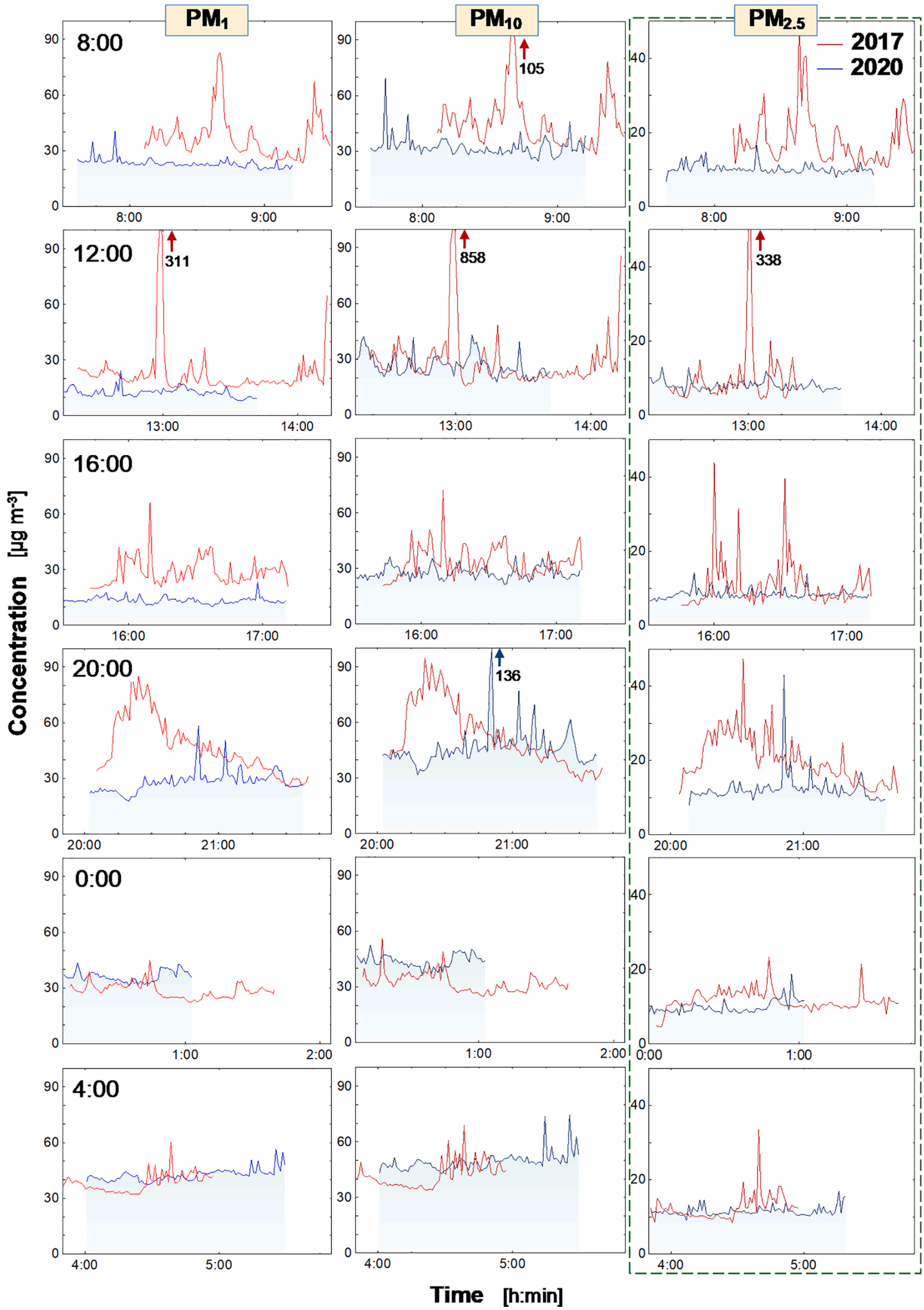

Fig. 2. (a). Time series of PM_1, $PM_{2.5}$ and PM_{10} concentrations during mobile measurements in 2017 and 2020 (PM_1, PM_{10} – data from Dusttrak, $PM_{2.5}$ – data from Grimm). (b). Time series of $PN_{0.1}$, PN_1 and PN concentrations during mobile measurements in 2017 and 2020 ($PN_{0.1}$, PN – data from Grimm, PN_1 – data from P-trak).

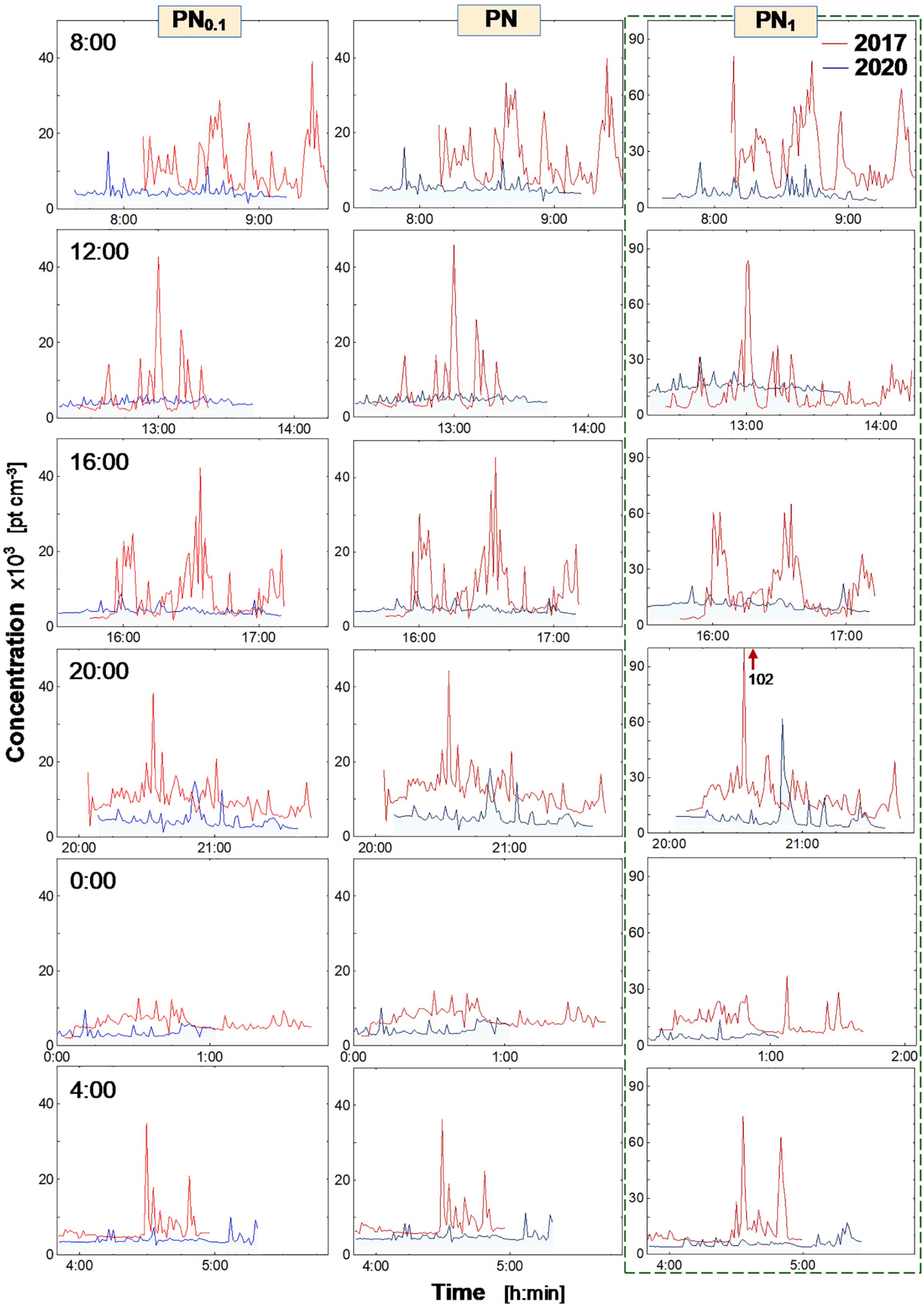

Fig. 2. (*continued*).

concentrations were recorded during the period of performing the test run at 20:00. In 2020 the average $PM_{2.5}$ and PM_{10} concentration change patterns for mobile and stationary measurement differ significantly. During mobile measurements the lowest concentration values of both particle fractions (below 30 μg m^{-3} and 35 μg m^{-3}, respectively) were obtained in test runs performed during the day (at 8:00, 12:00, 16:00). In turn, in all stationary measurements average $PM_{2.5}$ concentrations did not exceed 20 μg m^{-3} while the highest average PM_{10} concentration

Table 2
Descriptive statistics for particle mass (PM) concentrations (in μg m^{-3}) and particle number (PN) concentrations (in $x10^3$ pt cm^{-3}) obtained during mobile measurements in 2017 and in 2020 (PM data from Dusttrak and PN data from Grimm).

Run	Particles	2017	2020
8:00	PM_1	37/35 (25-83) [31.8]	23/22 (19-27) [7.0]
	$PM_{2.5}$	38/35 (25-84) [31.7]	23/23 (20-28) [7.0]
	PM_{10}	46/43 (30-105) [31.9]	31/31 (25-46) [12.2]
	$PN_{0.1}$	10.8/9.1 (3.0-28.8) [55.9]	4.2/3.9 (1.4-12.2) [32.3]
	PN	12.9/10.6 (3.7-33.4) [54.1]	5.0/4.8 (1.7-13.0) [28.1]
12:00	PM_1	26/21 (15-311) [132]	12/12 (8-24) [20.2]
	$PM_{2.5}$	27/21 (15-338) [140]	13/13 (8-25) [20.0]
	PM_{10}	41/24 (16-858) [235]	26/25 (18-43) [20.7]
	$PN_{0.1}$	6.5/3.5 (1.7-42.8) [111]	4.4/4.2 (3.1-6.3) [17.9]
	PN	7.5/4.4 (2.1-46.0) [104]	4.9/4.6 (3.6-7.1) [16.6]
16:00	PM_1	29/28 (20-66) [25.3]	14/13 (10-23) [14.2]
	$PM_{2.5}$	29/29 (20-67) [25,3]	14/14 (11-24) [13.7]
	PM_{10}	34/33 (21-72) [25.0]	28/27 (21-37) [12.5]
	$PN_{0.1}$	8.6/5.6 (1.8-42.3) [84.7]	4.2/4.0 (2.9-8.7) [24.9]
	PN	10/6.8 (2.1-45.4) [81.3]	4.8/4.6 (3.3-9.5) [23.8]
20:00	PM_1	48/44 (26-87) [32.2]	28/28 (18-58) [20.7]
	$PM_{2.5}$	49/44 (26-88) [32.1]	29/29 (19-62) [20.9]
	PM_{10}	54/49 (28-95) [31.8]	47/45 (32-136) [26.4]
	$PN_{0.1}$	10.1/9.3 (4.2-38.2) [44.2]	4.3/3.8 (1.3-14.8) [52.2]
	PN	12.4/11.5 (6.1-44.3) [40.8]	5.2/4.7 (1.7-18.2) [49.6]
0:00	PM_1	30/30 (22-45) [14.4]	36/36 (31-44) [8.6]
	$PM_{2.5}$	30/30 (22-45) [14.4]	37/36 (32-44) [8.5]
	PM_{10}	34/34 (25-56) [17.1]	44/43 (37-53) [8.1]
	$PN_{0.1}$	6.3/6.2 (2.0-12.8) [35.4]	3.5/3.0 (2.1-9.6) [38.3]
	PN	7.7/7.9 (2.5-14.6) [33.2]	4.4/3.8 (2.7-10.2) [35.3]
4:00	PM_1	38/37 (32-60) [14.1]	41/41 (37-45) [4.6]
	$PM_{2.5}$	39/37 (32-61) [14.5]	42/42 (38-45) [4.7]
	PM_{10}	42/40 (34-69) [17.9]	47/47 (43-53) [5.3]
	$PN_{0.1}$	6.6/5.3 (4.2-34.9) [66.3]	3.9/3.7 (2.5-7.2) [19.7]
	PN	8.0/6.7 (5.6-36.2) [56.2]	4.8/4.6 (3.4-8.4) [18.3]

Arithmetic average/median (range) [coefficient of variation].

values were recorded during the day and in the evening with a maximum value of above 80 μg m^{-3} at midday.

For mobile measurements the ratios of $PM_{2.5}$ concentrations in 2020 and 2017 as well as the ratios of PM_{10} concentrations in 2020 and 2017 achieved the value greater than 1 only in test runs carried out at night. For test runs performed during the day, such ratios were lower than 1. For stationary measurements, those ratios were greater than 1 for both $PM_{2.5}$ and PM_{10} particle concentrations irrespectively of the time of the measurement. At 12:00 and 16:00 PM_{10} concentrations in 2020 were over 4 times higher than in 2017.

Average $PM_{2.5}$ to PM_{10} concentration ratios presented in Figure 4 demonstrate that greater values were obtained in both mobile and stationary measurements in 2017 and the greatest discrepancies as compared to 2020 are seen for measurements carried out during the daytime. For mobile measurements in 2017, the average $PM_{2.5}$ to PM_{10} concentration ratios exceeded 0.8 during all test runs. In 2020 this value was exceeded only for test runs carried out at night (at 0:00 and 4:00). The average $PM_{2.5}$ to PM_{10} concentration ratios in stationary measurements in 2017 and 2020 during the day amounted to approximately 0.4 and 0.2, respectively. At night those ratios had similar values and increased to about 0.5 at 0:00 and 0.7 at 4:00.

4. Discussion

Numerous studies conducted to date on the impact of COVID-19 lockdowns on air quality in urban areas failed to provide satisfactory explanations of the observed, often contradictory relations between decreased traffic intensity and air pollutant concentration (Nakada and Urban, 2020; Xiang et al., 2020; Kumar et al., 2020). The relationship between traffic flows and air pollution was unclear even before the pandemic. Although the majority of studies indicated positive correlations between traffic volumes and nitrogen oxide concentrations (Lähde et al., 2014; Shi et al., 2018), the results regarding particulate matter concentrations were not unequivocal. For example, some studies found that PM_{10} concentrations increased together with the increase of the traffic intensity (Grivas et al., 2004; Madrazo et al., 2019). In turn, other studies indicated either a poor or no such correlation (Kurz et al., 2020; Qu et al., 2019). A relatively low percentage of particle mass concentrations and high percentage of particle number concentrations from traffic emissions was also reported. For example, according to the Washington State Department of Ecology only 2% of annual statewide $PM_{2.5}$ emissions come from on-road mobile sources (Washington State Department of Ecology, 2018). However, it needs to be emphasized that in more populous counties such emissions are significantly higher, e.g. in King County they make up about 14% of total $PM_{2.5}$ emissions. With respect to particle number concentrations, Xiang et al., (2020) demonstrated that nucleation-mode particles are a leading contributor to roadside ultrafine particles. It was also noted that the obtained results are impacted by many factors which should be taken into account during their analysis. The most important factors include, among other things, differences in weather conditions such as wind speed and direction, air temperature, relative humidity, thermal inversions, and precipitation level as well as varying air pollutant emissions generated by residential and industrial sectors (Agudelo-Castañeda et al., 2013; Pan et al., 2016;). Attention was also paid to site-specific results that are characteristic for air quality measurements carried out by monitoring stations located away from roadways (Boogaard et al., 2010; Pasquier and André, 2017).

The results of this study suggest that in Lublin the traffic changes, especially in certain locations, and times of the day may not have decisive impact on the ambient air quality and indicate that factors such as varying other pollution sources, weather conditions and changes in the transportation structure may be of importance. The significantly higher particle concentrations (especially of PM_{10}) obtained in stationary and mobile measurements carried out at night during the COVID-19 lockdown in 2020 as compared to the corresponding measurements in 2017 may be attributed to intensive emissions from other non-traffic-related sources. In Lublin, similarly to many other CEE cities one can assume with a high degree of certainty that such sources include residential coal burning for heating purposes which have a dominant impact e.g. on mass particle concentrations during certain periods of the day (Polednik, 2013). One also has to draw attention to the fact that the air quality monitoring station where the stationary measurements were carried out is located in residential area where traffic-related emissions usually have a lower impact on the air quality than the emissions related to residential activities. Due to the absence of industrial emission sources, their impact should be excluded. The risk of pollutant transportation is also minimal as the areas around the city are rather rural (Geoportal.gov.pl).

Naturally, apart from non-traffic-related emissions, the reported pollutant levels can be impacted by weather conditions. The considered measurements were carried out on non-rainy days, however, both measurement periods in 2017 and 2020 differed in terms of wind speed, its direction and air relative humidity. Such weather conditions e.g. wind speed in studies carried out by Xiang et al., (2020) were generally negatively correlated with urban air pollution levels. On the other hand, in studies carried out by e.g. Rossi et al. (2020) wind speed also had a negative impact on the ambient air quality, however, the temperature – similarly as in several other studies (Wang et al., 2020; Tobías et al., 2020) had a positive effect. The increased particle mass concentrations in measurements carried out in 2020 may therefore be partially attributed to higher wind speed that could result in more intense resuspension of particles deposed all types of surfaces, including the sidewalks of the monitored road. The study carried out by Zhao et al. (2020) that applied the WRF model is an example of research in which an attempt was made to eliminate the impact of weather conditions on the changes of air pollutant concentrations during the COVID-19 pandemic. Xiang et al., (2020) neutralized the impact of metrological factors by adjusting them to the traffic changes with the use of multivariate autoregressive models.

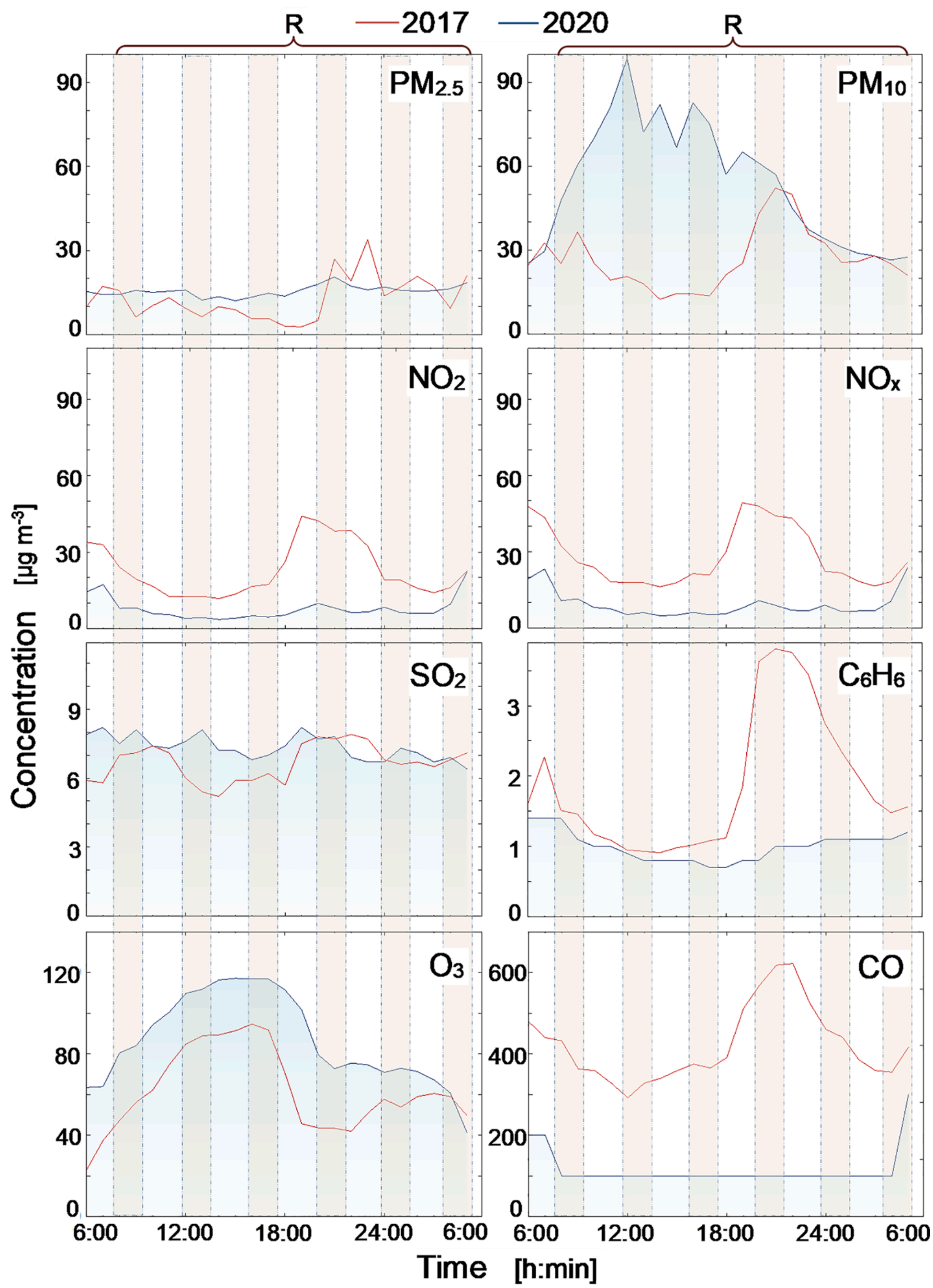

Fig. 3. Air pollutant concentration changes during stationary measurements in 2017 and 2020. R – periods of test runs during mobile measurements.

The obtained results related to the concentrations of the remaining measured non-particulate air pollutants are generally consistent with those obtained in previous studies in which e.g. lower nitrogen oxide concentrations during the COVID-19 lockdowns was reported (Lian et al., 2020; Connerton et al., 2020; Rossi et al., 2020). The decrease in NO_2 concentration which reflected the progressively reduced traffic intensity in the region of Lombardy in Italy has been shown in studies performed e.g. by Piccoli et al. (2020). O_3 concentration increases, CO decreases and negligible changes in SO_2 have been observed in studies performed in different regions of India by Sharma et al. (2020). Tobías et al. (2020) reported O_3 concentration increases by around 50% in in the city of Barcelona. Drastic reductions in NO, NO_2 and CO concentrations and increases in O_3 concentrations in the urban area in São Paulo were reported by Nakada and Urban (2020). Substantial reductions in NO_2 and other air pollutants as well as O_3 increases were also observed in Malaysia (Latif et al., 2021). Studies carried out by Wang and Li (2021) indicated that in selected eight cities (Wuhan, New York, Milan, Madrid, Bandra, London, Tokyo and Mexico City) COVID-19 lockdowns reduced NO_2 concentrations by 40-50% with a simultaneous increase of O_3 concentrations by 17-20%. In turn, while considering VOCs in ambient air Altuwayjiri et al. (2021) and Sannino et al. (2021) have reported decreased C_6H_6 concentration at residential sites respectively in Milan and in the area of Naples. The above-presented observations depart form the assumed pattern in which NO_2 and VOCs concentrations are closely related to O_3 concentration in ambient air (Biswas et al., 2019). NO_2 and VOCs under sufficient solar radiation and high humidity act as precursors of O_3. Xu et al. (2020) in studies carried out in three cities in central China (in Wuhan, Jingmen, and Enshi) explain the observed O_3 increases and NO_2 decreases with constraints on photochemical reactions and the related less effective O_3 removal.

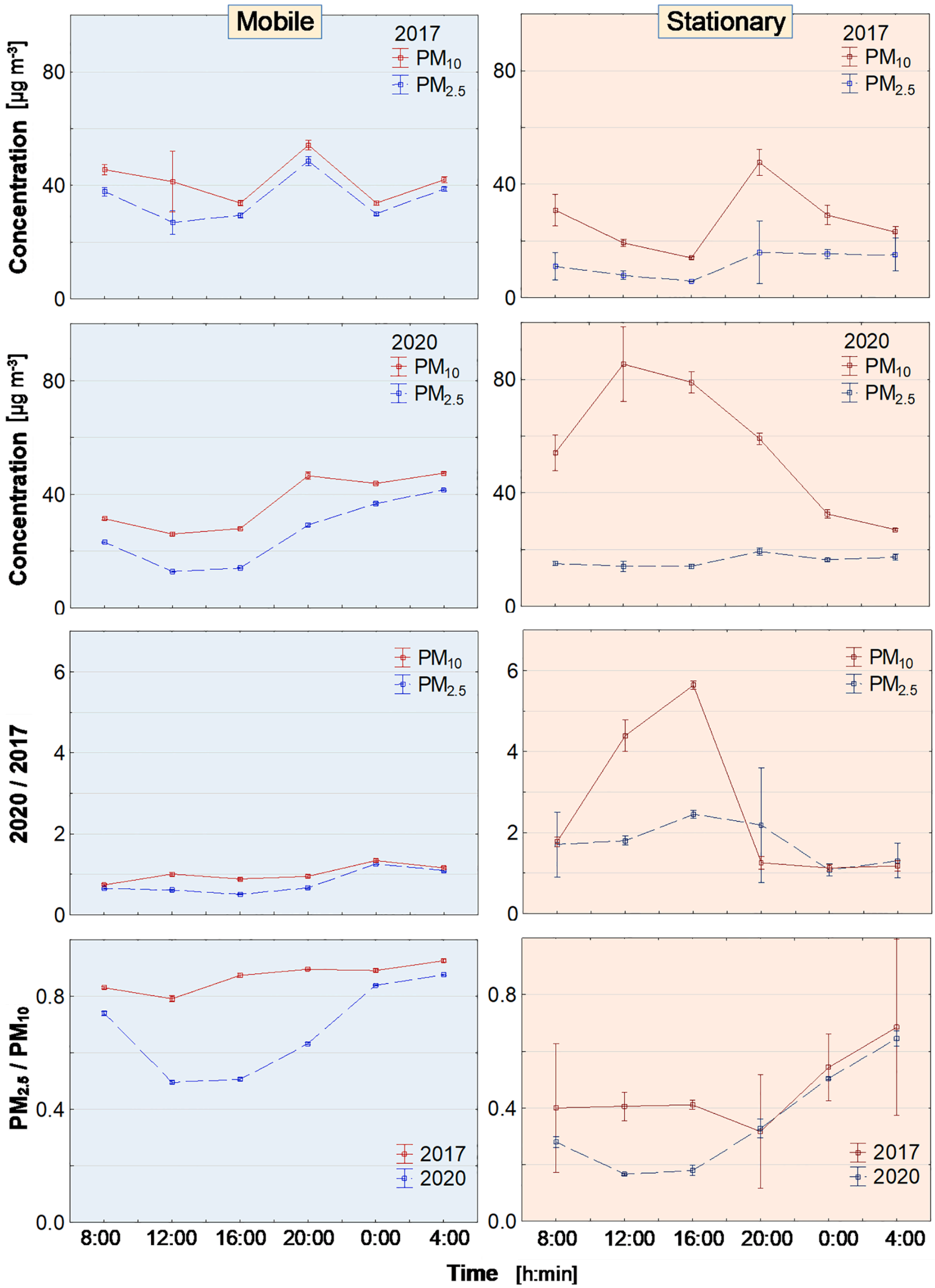

Fig. 4. $PM_{2.5}$ and PM_{10} concentrations and their ratios during mobile and stationary measurements in 2017 and 2020.

Quantitative results concerning the impact of the COVID-19 outbreak on the urban air quality should be interpreted with great caution, all the more that the studies may not consider all influencing factors, while the factors that are examined may not be stable. For example, the observed differences in the air pollutant concentrations in Lublin in 2020 measurements during the COVID-19 lockdown could, apart from the decreased traffic intensity and the differences in weather conditions, be also attributed to the changes of the vehicle fleet structure and other non-stable traffic characteristics. Due to the uncertainty of the latter factors, it is difficult to assess their impact on the urban air quality.

Further, more detailed research concerning the impact of road traffic changes on the urban air quality should take into consideration quantitative pollutant emission variations from all major sources. This includes the impact related to traffic and meteorological parameters that may play a decisive role in the dispersion of the pollutants. Although the presented preliminary results of studies carried out in a typical CEE city can be considered as representative for this region, more comprehensive research covering also other locations is advisable to be able to eliminate or control and effectively minimize the specific factors deteriorating urban air quality.

5. Conclusions

The carried out mobile and stationary measurements of the air pollutant levels in Lublin in 2020 during the COVID-19 lockdown characterized by a significant decrease of the traffic flow and in a comparable period in 2017 indicated that traffic-related pollutions, especially during certain times of the day are not the main factor impacting the quality of the urban air. Emissions from other non-traffic-related sources and weather conditions may be of importance and may lead to substantial changes of the air pollutant levels. For the considered periods and locations in which mobile and stationary measurements were carried out residential coal burning and wind speed could have been of importance. Changes in the vehicle fleet structure and in other traffic characteristics during the COVID-19 lockdown could also be significant contributors. Further, more comprehensive research is necessary to effectively control, eliminate or minimize factors that adversely affect urban air quality.

Declaration of Competing Interest

The author declares that he has no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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