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Assessment of Particulate Matter Concentrations at Bus Stations in Bucharest, Romania

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Abstract

Bus and tram stops expose commuters to traffic-related particulate matter, yet surface public transport microenvironments in Eastern Europe remain poorly characterized. This study assessed outdoor particulate matter concentrations at four bus and tram stops near educational facilities in Bucharest, Romania, and at one fixed comparison site between July and November 2025. Portable GRIMM Mini-LAS 11-E optical particle counters were used at the stops, at commuter breathing height, and a beta-attenuation monitor was used at the comparison site; one counter was harmonized to the other two on the basis of a side-by-side intercomparison, and the results were compared with nearby fixed urban monitoring stations. During the morning measurements in July, mean PM_{10} at three stops ranged from 44.4 to 61.3 $\mu g/m^3$, compared with 16.6–34.6 $\mu g/m^3$ reported by the nearest fixed stations for the same hours, while mean $PM_{2.5}$ ranged from 8.9 to 12.5 $\mu g/m^3$ and the $PM_{2.5}/PM_{10}$ ratio ranged from 0.19 to 0.28. At one tram stop, the afternoon (12:00–18:00) monthly mean $PM_{2.5}$ increased from 9.5 $\mu g/m^3$ in July to 30.2 $\mu g/m^3$ in November, and the $PM_{2.5}/PM_{10}$ ratio increased from 0.19 to 0.58. At a fourth stop, the 07:00–19:00 mean $PM_{2.5}$ over five days (38.7 $\mu g/m^3$) was on average about twice the 24-h mean at the comparison site (daily ratios 1.0–3.4). When the time actually spent at a stop is taken into account, these contrasts correspond to increments of less than 1 $\mu g/m^3$ in the 24-h mean exposure for waiting times of up to one hour per day. The results suggest that fixed stations may underestimate short-term concentrations at public transport stops, although the use of different instruments and averaging periods limits the precision of these contrasts.

Keywords: particulate matter; ground public transportation station; personal exposure; experimental measurements; Grimm particle counter

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1. Introduction

Numerous medical studies have demonstrated the harmful effects of particulate matter (PM) on human health [1–7]. $PM_{2.5}$ has been associated with increased respiratory mortality, cardiovascular events and all-cause mortality. Fine particles penetrate deep into the respiratory tract, reach the pulmonary alveoli and are difficult to eliminate [8,9]. Indoor PM has been studied extensively, since people spend most of their time indoors [10–18]; exposure in outdoor transport microenvironments is shorter but much more difficult to control.

Bucharest, the Romanian capital, is one of the most polluted cities in Europe, ranking eighth out of 105 in the overall European city ranking and fourth out of 35 among Eastern European cities, according to data collected in June 2026 [19]. Heavy vehicular traffic is a

primary source of particulate pollution in Bucharest, together with construction activities, while the limited area of green space reduces particle removal. Children and the elderly are among the groups most adversely affected by particulate pollution. Particles with a diameter smaller than 2.5 micrometers ($PM_{2.5}$) can reach the pulmonary alveoli, and the World Health Organization estimates that ambient and household air pollution together are associated with about seven million premature deaths each year [20]. Measures to reduce particulate pollution must therefore be identified and implemented not only at the city level but also locally.

A time–activity study in Belgium showed that, although people spent only about 6% of their time in transport, this time accounted for 21% of their personal exposure to black carbon and about 30% of the inhaled dose [21]. In Bogotá, the personal exposure of Bus Rapid Transit users was strongly influenced by the age and emission standard of the fleet [22], and Bus Rapid Transit systems alone carry about 32 million passengers per day in more than 160 cities worldwide [23]. Because of their proximity to traffic, bus and tram stops are among the places where many people are exposed to high concentrations of air pollutants during their daily activities. Although people spend only a small part of their time at stops, this time can contribute substantially to the inhaled dose of pollutants. The main sources of particulate matter at bus stops are vehicle exhaust, brake and tire wear, and the resuspension of road dust.

As the population and the number of private vehicles grow, raising air pollution levels in urban centers, people have been encouraged to use public transport such as buses, trams or the metro. However, significant levels of particulate matter have been shown to accumulate both inside and outside buses, exposing passengers directly to this pollutant while waiting at stops or traveling [24,25]. PM_{10} concentrations (particulate matter with an aerodynamic diameter of 10 μm or less) measured at bus stops during peak hours can be up to 2.5 times higher than the daily average concentrations in indoor environments [25–27].

Public transport is a key mode of urban mobility in Europe [28], and young people and, in urban areas, older people are among its most frequent users [29,30]; these are also the age groups most affected by particulate pollution. Public transport is widely used in Bucharest, where buses, trams and the metro are affordable and accessible across the metropolitan area [31]. In a 2020 evaluation by Greenpeace Romania, Bucharest ranked second among 14 European cities in the public transport category, owing to its low fares and high density of stops [32], a result also reported by the European Commission's Urban Mobility Observatory [33].

Recent studies have increasingly identified bus stops as hotspots of particulate matter pollution. The main contributors are vehicle emissions and traffic congestion, which considerably raise particle levels. This poses a substantial health risk to commuters, who are regularly exposed at these locations to elevated concentrations of particulate matter, particularly PM2.5 and PM10 [34,35]. Elevated particulate matter levels at bus stops, which often exceed health-based standards, can be predicted with models such as Artificial Neural Networks (ANNs) and Adaptive Neuro-Fuzzy Inference Systems (ANFISs). Such predictive tools can support public transport policies by providing reliable estimates of pollution levels, enabling better planning for passenger safety and lower exposure to harmful air pollutants [24]. In an urban area of central Slovakia, PM_{10} levels exceeded 83 $\mu g/m^3$ and $PM_{2.5}$ concentrations exceeded 16 $\mu g/m^3$ [36]. In Istanbul, Turkey, bus commuters and pedestrians were more exposed to coarse particles than metrobus or car users, and particle concentrations during walking reached peaks of up to 316 $\mu g/m^3$, indicating a considerably higher exposure for people walking or traveling by bus than for those using other modes of transport [37].

To mitigate particulate matter pollution in urban transport environments, cities such as Bucharest may electrify their public transport fleets, introduce hybrid buses, expand urban green infrastructure and optimize traffic flow around public transport stops. However, the effectiveness of these measures can only be assessed through direct field measurements with dedicated particulate matter monitoring instruments. Although particulate matter pollution has been investigated extensively worldwide, significant gaps remain regarding commuter exposure in surface public transport microenvironments, particularly in Eastern European countries. Available studies have focused predominantly on underground rail systems [38,39], whereas bus and tram stops have received considerably less attention [40]. In Europe, measurements during bus and tram commutes have been reported for Barcelona [41], and PM at bus stations has been measured in Žilina, Slovakia [42]; outside Europe, the exposure of passengers waiting at bus stops has been examined, for example, in Portland and Singapore [26,43]. To the best of the authors' knowledge, no previous study has experimentally assessed particulate matter concentrations at bus and tram stops in Bucharest.

The objective of this study was therefore to measure outdoor particulate matter concentrations at heavily trafficked bus and tram stops in Bucharest, Romania, most of which are located near educational facilities. By providing field measurements from surface public transport stops in an Eastern European capital, this study helps to address an important gap in the literature and provides data that may support future air quality management and public health strategies.

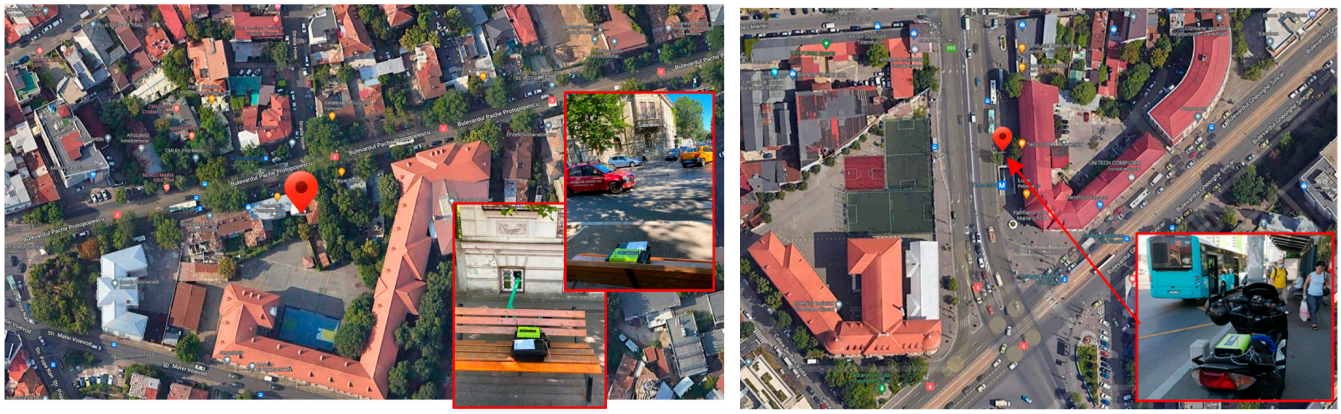
2. Materials and Methods

Field Study

Bucharest, the capital of Romania, is located in the southeastern part of the country. With a population of approximately 1.9 million, it is the largest urban center in Romania and the political, economic and cultural hub of the country. The city is crossed by the Dâmbovița River, whose course has been artificially modified to flow through the capital and supply water to its inhabitants. The climate of Bucharest is temperate-continental, with four distinct seasons. Winter temperatures can fall to −15 °C, with snowfall, while summer temperatures can reach around 42 °C, with little precipitation. Spring and autumn are transitional seasons, with moderate temperatures and more rainfall. In recent years, owing to global warming, winter precipitation has decreased significantly and the number of extremely hot summer days has increased. Surface public transport in Bucharest is operated mainly by buses and trams, which cover the entire metropolitan area with a fleet of approximately 1600 buses and 500 trams [44]. The bus system serves a large number of routes, and the tram network is about 332 km long. Fares are relatively low compared with other European capitals, and certain groups, such as retirees, students and schoolchildren, travel free of charge, which increases their use of public transport. This influenced the decision to select stops near educational institutions, since these locations reflect the exposure of groups that rely heavily on public transport and are particularly vulnerable to air pollution.

Air quality measurements were conducted at five locations in Bucharest: four public transport stops along high-traffic roads (monitoring locations 1–4) and one fixed comparison site (monitoring location 5), which provided continuous measurements under stable operating conditions.

- Monitoring Site 1: The shared bus and tram stop on Pache Protopopescu Boulevard, in front of the "Mihai Viteazul" National College (Figure 1a), the "Iancului" Primary and Lower Secondary School and the Faculty of Building Services.



(**a**) (**b**)

Figure 1. Site 1, the shared bus and tram stop on Pache Protopopescu Boulevard, in front of the "Mihai Viteazul" National College, the "Iancului" Primary and Lower Secondary School and the Faculty of Building Services (**a**); Site 2, the bus stop on Dimitrie Cantemir Boulevard, next to the "Gheorghe Șincai" National College (**b**).

- Monitoring Site 2: The bus stop on Dimitrie Cantemir Boulevard, next to the "Gheorghe Șincai" National College (Figure 1b).
- Monitoring Site 3: The tram stop on Mihai Bravu Road, at the intersection with Iancului Road, close to Primary and Lower Secondary School No. 56.
- Monitoring Site 4: A tram stop at the intersection of Traian Street and Calea Moșilor, close to the "George Coșbuc" National Bilingual College.
- Monitoring Site 5: In front of the Faculty of Building Services Engineering, approximately 150 m from Site 1. The faculty is located in the same urban area as Sites 1, 3 and 4, all within approximately 2 km, so the comparison site is subject to similar urban-scale meteorological conditions. Site 5 is located on the same boulevard as Site 1, but it is not affected by buses and trams stopping, idling and accelerating at a stop; it should therefore be regarded as a roadside comparison site rather than an urban background site.

Sites 1 and 2 are shown in Figure 1a,b as examples of the roadside stops; Site 3 is shown as a tram platform between traffic lanes, and Site 4 is a tram stop next to a signalized intersection.

Particle concentrations were measured with GRIMM portable laser aerosol spectrometers (Model Mini-LAS 11-E; Grimm Aerosol Technik GmbH & Co. KG, Ainring, Germany), which measure particle number concentrations in 31 optical size channels. The mass concentrations of the PM_1, $PM_{2.5}$ and PM_{10} fractions were calculated by the instrument's internal algorithm [45], which converts the size-resolved counts into mass concentrations; according to the manufacturer, this conversion is optimized against the gravimetric reference methods for PM_{10} and $PM_{2.5}$ (EN 12341 and EN 14907). The instruments were operated with the factory settings (C-factor = 1.00), and no site-specific gravimetric calibration was performed. The optical-to-mass conversion depends on particle density, shape and refractive index, which vary with composition and source; ambient particle densities typically range from about 1.5 to 2.6 g/cm^3, and road dust and brake wear particles are denser than combustion particles. The calculated mass concentrations, particularly for the coarse fraction, are therefore uncertain; the effect of a possible multiplicative bias of the counters on the comparisons is evaluated in Section 3.1 (Table 1). Optical counters and beta-attenuation monitors are based on different measurement principles, so the PM concentrations they provide may differ, especially at high relative humidity: the GRIMM counters were operated without a dryer, and hygroscopic growth of particles may lead to an overestimation of the fine fraction.

The fixed stations B-3, B-13 and B-24 belong to the national air-quality monitoring network and use beta-attenuation monitors; validated hourly and daily data were downloaded from the network portal [46]. Comparisons with the fixed stations therefore indicate relative differences between microenvironments rather than exact differences in concentration. The reproducibility of the GRIMM instruments (±3%) refers to repeatability and does not include these systematic effects. The Mini-LAS 11-E covers optical diameters between 0.25 µm and 32 µm; particles smaller than 0.25 µm, mainly fresh combustion particles, are not counted, so PM_1 and, to a lesser extent, $PM_{2.5}$ may be slightly underestimated. Optical particle counters relate the amount of light scattered at an angle θ to the diameter of the particle passing the detector, according to the operating principle shown in Figure 2 [47]. The counters operate at temperatures between +4 °C and 40 °C and relative humidity below 95%, with a reproducibility of ±3% over the entire measuring range. Each counter is also equipped with a temperature and humidity sensor, with a temperature range of −40 to 80 °C, an accuracy of 0.2 K and a humidity resolution of 0.1% over the 0–100% range [45]. Three GRIMM 11-E counters (Grimm Aerosol Technik GmbH & Co. KG, Ainring, Germany) were used during the campaign. All three were calibrated before the campaign; this calibration concerns optical sizing and flow rate and is not a gravimetric mass calibration. Unit 114 was used at Site 1, unit 112 at Sites 2 and 4, and unit 116 at Site 3. An intercomparison of the three counters was carried out on 7 November 2025 in the laboratory, with the units placed side by side so that their inlets were close together, for 140 min (1-min values; Table S1, Figure S1). Units 112 (serial 11E15P06) and 116 (11E15P08) agreed closely, with the ratios of their period means being 0.97 for PM_{10}, 0.99 for $PM_{2.5}$ and 0.98 for PM_1, whereas unit 114 (11E15P07, firmware 12.40 instead of 7.80) measured 17–19% less PM_{10} and 36–37% less $PM_{2.5}$ and PM_1 than the other two, with a very high correlation ($R^2 = 0.99$ for $PM_{2.5}$ and PM_1 relative to the mean of the three units). Since unit 114 was used at Site 1, the Site 1 data were harmonized to the scale of units 112 and 116 by multiplying each 6-s value by the ratios of the period means, 1.22 for PM_{10}, 1.57 for $PM_{2.5}$ and 1.58 for PM_1. Harmonization removes only the differences between the counters, not their common bias relative to the beta-attenuation method, which was not quantified because the counters were not co-located with the beta-attenuation monitor. Data were recorded at a resolution of 6 s.

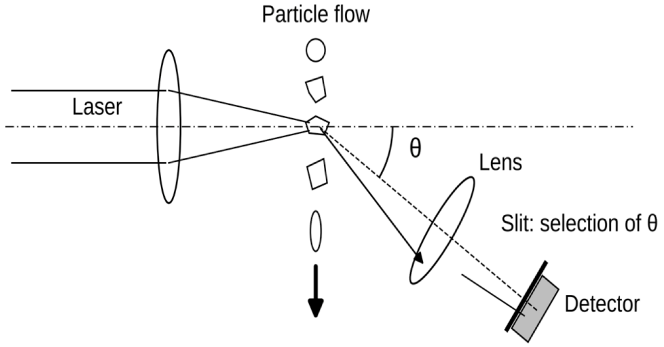


Figure 2. Operating principle of the optical particle counter (adapted from [47]).

Concentrations were measured simultaneously at two or three of the monitoring locations described above. Since in most cases the instruments could not be connected to an external power supply, the sampling duration was limited by battery life. In addition, the operating conditions specified by the manufacturer had to be strictly observed, as the instruments are sensitive to ambient temperature and relative humidity: the sampled air temperature must be between 4 and 40 °C and the relative humidity below 95%, and the instruments must not be exposed to direct sunlight during measurements.

Table 1. PM concentrations at the public transport stops and at the reference locations, averaging windows, concentration differences ($\Delta C = C_{stop} - C_{ref}$), concentration ratios ($CR = C_{stop}/C_{ref}$) and break-even bias factors ($k^* = C_{ref}/C_{stop}$).

Comparison (Period)	Window at Stop	Window at Reference	C_{stop} ($\mu g/m^3$)	C_{ref} ($\mu g/m^3$)	ΔC ($\mu g/m^3$)	CR (–)	k* (–)
PM_{10}: Site 1 vs. B-3 (July)	morning, ≈1.5 h	same hours	61.3	30.9	30.4	1.98	0.50
PM_{10}: Site 1 vs. B-13 (July)	morning, ≈1.5 h	same hours	61.3	34.6	26.7	1.77	0.56
PM_{10}: Site 3 vs. B-3 (July)	morning, ≈1.5 h	same hours	49.7	30.9	18.8	1.61	0.62
PM_{10}: Site 3 vs. B-13 (July)	morning, ≈1.5 h	same hours	49.7	34.6	15.1	1.44	0.70
PM_{10}: Site 2 vs. B-24 (July)	morning, ≈1.5 h	same hours	44.4	16.6	27.8	2.67	0.37
$PM_{2.5}$: Site 2 vs. B-24 (July)	morning, ≈1.5 h	same hours	8.9	6.4	2.5	1.39	0.72
$PM_{2.5}$: Site 3 vs. B-13 (July)	morning, ≈1.5 h	same hours	12.5	7.9	4.6	1.58	0.63
$PM_{2.5}$: Site 1 vs. B-13 (July)	morning, ≈1.5 h	same hours	9.2	7.9	1.3	1.17	0.85
$PM_{2.5}$: Site 4 vs. B-13 (19–23 Oct.)	07:00–19:00	24 h	38.7	15.3	23.4 (5.1–41.7)	2.53	0.40
$PM_{2.5}$: Site 4 vs. Site 5 (19–23 Oct.)	07:00–19:00	24 h	38.7	19.1	19.6 (1.1–38.1)	2.03	0.49
$PM_{2.5}$: Site 3 vs. Site 5 (Sep.)	12:00–18:00	24 h	14.5	16.1	−1.6	0.90	–
$PM_{2.5}$: Site 3 vs. Site 5 (Oct.)	12:00–18:00	24 h	24.9	17.1	7.8	1.46	0.69
$PM_{2.5}$: Site 3 vs. Site 5 (Nov.)	12:00–18:00	24 h	30.2	14.9	15.3	2.03	0.49

Site 1 values harmonized (Section 2). For Site 4, the value in parentheses is the 95% confidence interval of the mean paired daily difference ($n = 5$ days). k* is not given for $\Delta C < 0$. For the July comparisons, the network values are means for the same hours as at the stops; for Sites 3 and 4, the reference values are 24-h means.

At Site 5, located at the Faculty of Building Services Engineering, particulate matter concentrations were measured with a stationary Verewa F-701-20 monitor (DURAG GmbH, Hamburg, Germany) [48]. Unlike the portable instruments used at the four stops, this monitor was connected to the faculty's electrical grid, which allowed continuous measurements without battery limitations. Ambient air is sampled at a flow rate of 1 m^3 h^{-1} through a heated inlet equipped with a size-selective impactor and drawn through a glass-fiber filter tape, on which the particles are retained, while the volumetric flow rate is continuously monitored and recorded. After each sampling period, the mass of particles collected on the filter is determined by beta attenuation, using a carbon-14 beta source and a Geiger–Müller detector (DURAG GmbH, Hamburg, Germany). The beta radiation intensity is first measured through an unused section of the filter tape to establish a reference signal and is measured again through the loaded filter after particle collection. The natural logarithm of the ratio between the reference and measured intensities is proportional to the mass of particulate matter deposited on the measurement area of the filter, assuming a homogeneous particle distribution; since the measurement spot has a constant area, the deposited mass can be determined accurately. The particulate matter concentration, in micrograms per cubic meter, is then calculated as the ratio between the collected particle mass and the volume of sampled air. The inlet can be fitted with either a $PM_{2.5}$ or a PM_{10} impactor, so that particles with aerodynamic diameters below 2.5 µm or 10 µm, respectively, are measured continuously every 30 min; since only one impactor can be installed at a time, each measurement provides the concentration of a single size fraction. The inlet was positioned approximately 3 m above ground level to avoid disturbance by pedestrians while ensuring representative sampling of the ambient air. The $PM_{2.5}$ impactor was installed during the whole period analyzed here (12 September–21 November 2025). Daily $PM_{2.5}$ values at Site 5 were calculated as the arithmetic mean of the 30-min measurements.

Three sampling designs were used: (i) morning measurements at Sites 1–3 on 14, 22 and 29 July 2025 (a Monday and two Tuesdays), during the morning rush hours, with the duration limited by battery life (about 1.5 h per site and day: 09:03–10:41 at Site 1, 09:18–10:55

at Site 2 and 09:47–11:20 at Site 3, so that all three sites were measured simultaneously between about 09:47 and 10:41); (ii) afternoon measurements (12:00–18:00) at Site 3 on at least 10 days per month from July to November 2025; and (iii) daytime measurements (07:00–19:00) at Site 4 on five consecutive days, from Sunday 19 to Thursday 23 October 2025. All times are local time. For the July campaign, the 6-s values measured at each site on the three days were averaged for each time step (by clock time), and the mean, the 95th percentile (P95) and the proportion of time above 50 $\mu g/m^3$ were calculated from these averaged series. Averaging over three days smooths short concentration peaks, so the P95 values and the proportions above 50 $\mu g/m^3$ are lower than those of individual days. Because consecutive 6-s values are strongly autocorrelated, they were not treated as independent samples, and the small differences between sites were not tested statistically. Temperature and relative humidity were measured simultaneously with the sensors integrated into both types of particulate matter monitors. Of the four stops, only Site 2 was partially enclosed, with a roof and one closed side. However, the shelter was too small for all waiting passengers, and most commuters stood outside it; the monitoring equipment was therefore placed where passengers were actually waiting, to better represent their exposure. The sampling inlet was positioned 1.0–1.2 m above ground level to represent the breathing zone of commuters, approximately the breathing height of a child or of a seated adult, whereas the inlet at Site 5 was at about 3 m; vertical concentration gradients near the curb may therefore contribute to the differences between the stops and Site 5. Wind speed and direction were not measured, which is acknowledged as a limitation (Section 4.4). The experimental campaign was conducted from July to November 2025.

3. Results

The following sections present the experimental results: a comparison of particulate matter concentrations measured at the different monitoring sites, an assessment of seasonal commuter exposure at public transport stops, and an analysis of the relationships between particulate matter concentrations, ambient temperature and relative humidity.

3.1. Comparison of Particle Concentrations Measured at Selected Public Transport Stops

Figure 3 shows the temporal variation in PM_{10} and $PM_{2.5}$ concentrations at Sites 1, 2 and 3 (measured simultaneously at all three sites between about 09:47 and 10:41). The time series in Figure 3 were obtained by averaging, for each 6-s time step, the values measured on the three monitoring days (14, 22 and 29 July 2025), and the summary statistics reported below (means, P95 values and exceedance frequencies) were calculated from these averaged series (Section 2). During the three-day campaign, the average ambient temperature was 28.8 °C (range 26.6–33.1 °C) and the average relative humidity was 30.8% (range 25.2–42.6%). These values refer to the morning rush-hour measurements on these three days and are therefore not directly comparable with the monthly means presented in Section 3.2, which cover more days and a different time interval (12:00–18:00).

Both size fractions showed considerable short-term variability, with numerous transient peaks. Nevertheless, the three sites differed clearly in concentration level, temporal variability and the relative contribution of fine particles. For PM_{10}, Site 1 had the highest mean concentration (61.3 $\mu g/m^3$, after harmonization of the counter used at this site; Section 2) and the highest 95th percentile (P95 = 111.5 $\mu g/m^3$). Site 3 had a mean concentration of 49.7 $\mu g/m^3$ and a P95 of 87.3 $\mu g/m^3$, whereas Site 2 had the lowest values, with a mean of 44.4 $\mu g/m^3$ and a P95 of 80.3 $\mu g/m^3$. The time series also show several short PM_{10} peaks, particularly at Site 1. The proportion of time during which PM_{10} exceeded 50 $\mu g/m^3$ provides additional information on the persistence of elevated concentrations: it was 41.9% at Site 3, 30.5% at Site 2 and 55.1% at Site 1 (37.1% before harmonization). The difference

between Sites 2 and 3 (44.4 versus 49.7 μg/m^3) is small and, since the averaged series only partly overlap in time, should be interpreted with caution. The 50 μg/m^3 level is used here only as a contextual reference to the Romanian 24-h PM_{10} limit value; short-term values above this level should therefore not be interpreted as regulatory exceedances.

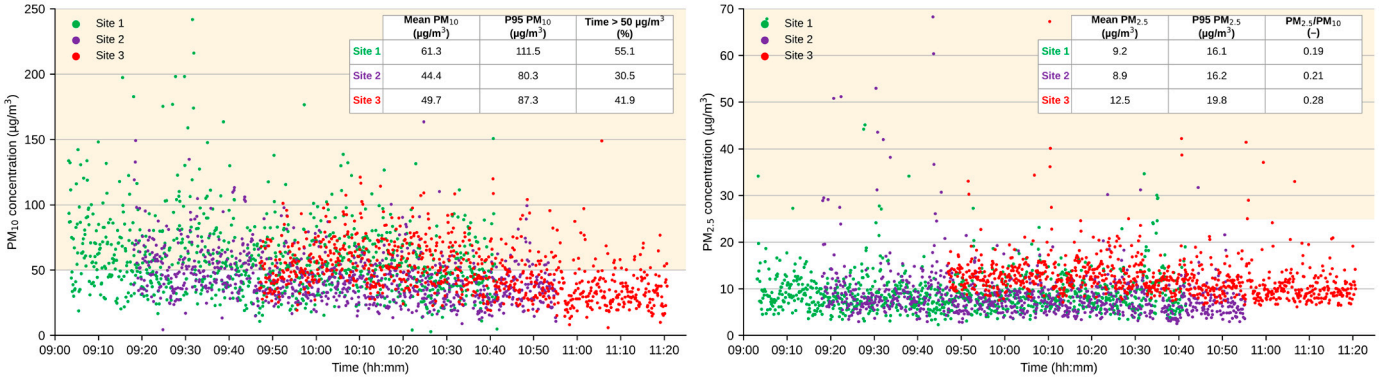


Figure 3. Temporal variation in PM_{10} and $PM_{2.5}$ concentrations measured at monitoring sites 1–3: mean of the three monitoring days (14, 22 and 29 July 2025) for each 6-s time step, plotted at the actual clock time of each site; Site 1 values harmonized (Section 2); the inset tables give the statistics of these averaged series. A few values above the axis limits are not shown.

$PM_{2.5}$ showed a different spatial pattern. Site 3 had the highest mean $PM_{2.5}$ concentration (12.5 μg/m^3), whereas Sites 1 and 2 had similar values (9.2 and 8.9 μg/m^3, respectively). The P95 values followed the same pattern (19.8 μg/m^3 at Site 3, 16.1 μg/m^3 at Site 1 and 16.2 μg/m^3 at Site 2). Without harmonization, the mean $PM_{2.5}$ at Site 1 would have been 5.9 μg/m^3, which shows the importance of intercomparing the counters.

To further examine differences in particle-size distribution between the sites, the $PM_{2.5}/PM_{10}$ ratio was calculated for each pair of simultaneous $PM_{2.5}$ and PM_{10} values, and these instantaneous ratios were averaged with equal weight over the monitoring period at each site. The mean $PM_{2.5}/PM_{10}$ ratios were 0.19, 0.21 and 0.28 at Sites 1, 2 and 3, respectively, indicating an increasing relative contribution of fine particles to PM_{10} from Site 1 to Site 3.

The contrast between Sites 1 and 3 is particularly notable. Site 1 had a higher mean PM_{10} concentration than Site 3 (61.3 and 49.7 μg/m^3, respectively) but a lower $PM_{2.5}$ concentration (9.2 and 12.5 μg/m^3), and its mean $PM_{2.5}/PM_{10}$ ratio (0.19) was about two thirds of that at Site 3 (0.28). PM_{10} mass concentrations in different public transport microenvironments may therefore correspond to markedly different particle-size distributions: Site 1 was characterized by a larger relative contribution of coarse particles, whereas Site 3 had a substantially larger relative contribution of fine particles.

These differences may reflect differences in local conditions and particle-generation mechanisms. The relatively low $PM_{2.5}/PM_{10}$ ratio at Site 1, combined with its relatively high PM_{10} concentration, is consistent with a greater influence of mechanically generated coarse particles, including resuspended road dust. In contrast, the higher $PM_{2.5}/PM_{10}$ ratio at Site 3 may indicate a greater relative contribution from fine-particle sources. These interpretations are indicative rather than source-specific, since neither chemical composition nor source apportionment was analyzed in the present study.

Figure 4 shows the particle-size distribution at the three sites, separating the measured particulate matter into PM_1, $PM_{1–2.5}$ and $PM_{2.5–10}$ fractions. The $PM_{2.5}/PM_{10}$ and $PM_1/PM_{2.5}$ ratios reported below are means of the instantaneous ratios for each pair of simultaneous measurements and therefore differ slightly from the ratios of the mean concentrations. The coarse fraction ($PM_{2.5–10}$) was the dominant contributor to particulate mass at all three sites,

reaching 52.1 µg/m^3 at Site 1, 35.5 µg/m^3 at Site 2 and 37.2 µg/m^3 at Site 3. The fraction concentrations were computed from unrounded means, so sums of rounded values may differ by 0.1 µg/m^3. Consistent with the $PM_{2.5}/PM_{10}$ ratios discussed above, Site 3 also had the highest PM_1 concentration (7.4 µg/m^3) and $PM_1/PM_{2.5}$ ratio (0.62), compared with 0.54 at Site 1 and 0.53 at Site 2. Site 2 had an intermediate particle-size profile but the lowest mean PM_{10} (44.4 µg/m^3) and $PM_{2.5–10}$ (35.5 µg/m^3) concentrations. These differences appear consistent with the different urban settings of the sites. Site 3, a tram stop between traffic lanes, is the location closest to the surrounding traffic, which may explain the greater relative abundance of fine and submicron particles observed there. Sites 1 and 2 are both roadside stops and are therefore more comparable; however, unlike Site 1, which has no significant vegetation nearby, Site 2 is close to a large urban park. Compared with Site 1, Site 2 had approximately 28% lower mean PM_{10} and 32% lower $PM_{2.5–10}$ concentrations. This pattern may be consistent with an influence of the greener surroundings on local particle deposition, resuspension and dispersion, particularly for the coarse fraction. Nevertheless, the observed differences cannot be attributed solely to vegetation or traffic configuration, since these factors were not investigated independently and other local conditions may also have contributed.

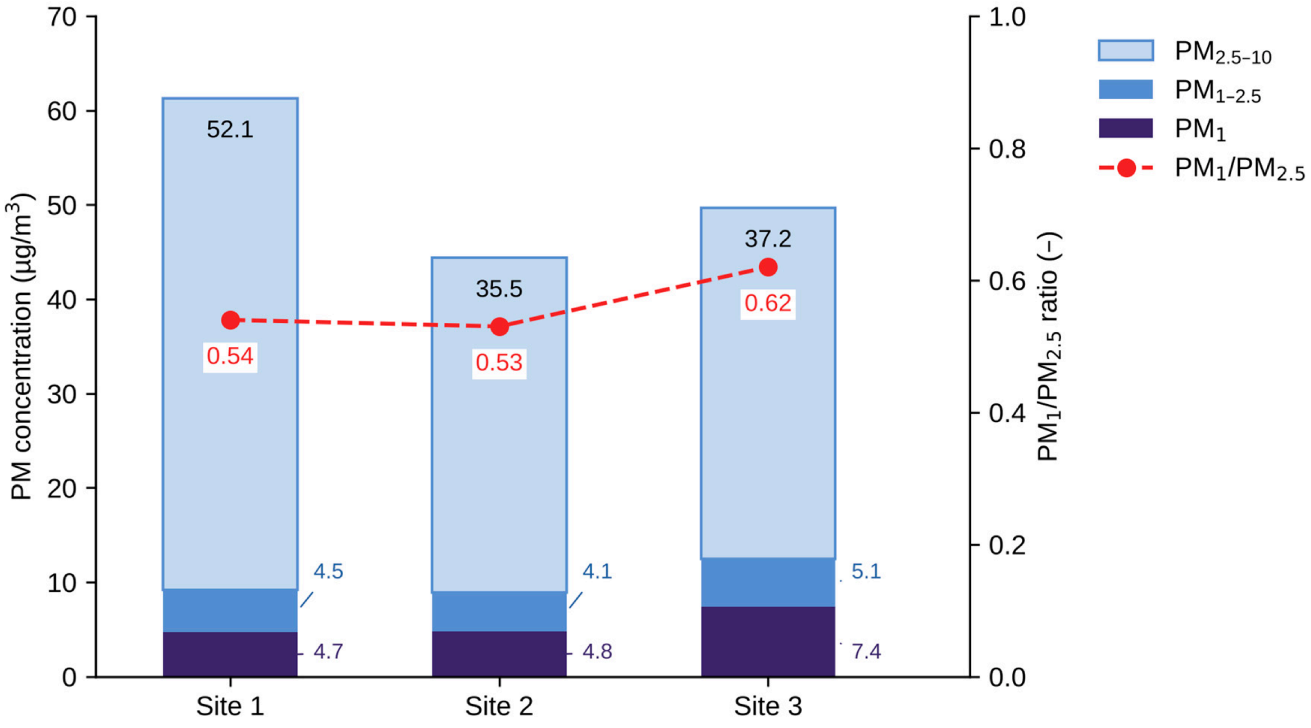


Figure 4. Particle-size distribution at Sites 1–3 (series averaged over 14, 22 and 29 July 2025 for each 6-s time step, morning measurements; Site 1 values harmonized, Section 2). $PM_1/PM_{2.5}$ is the mean of the instantaneous ratios.

Figure 5 shows a marked difference between the particulate matter concentrations reported by the fixed urban monitoring stations and those measured at the public transport stops in the breathing zone of commuters. The network values in Figure 5 are means of the hourly values for the same hours as the measurements at the stops on the three monitoring days, so these comparisons refer to identical time windows (Table 1). At Site 2, the mean PM_{10} concentration was 44.4 µg/m^3, compared with only 16.6 µg/m^3 at the nearby B-24 station, i.e., approximately 2.7 times higher. A similar but less pronounced difference was observed for $PM_{2.5}$ (8.9 versus 6.4 µg/m^3). For $PM_{2.5}$ at Sites 1 and 3, a comparison was possible only with station B-13 (7.9 µg/m^3), since $PM_{2.5}$ is not measured at B-3. The $PM_{2.5}$

concentration at Site 3 (12.5 μg/m^3) was approximately 1.6 times higher than at B-13, and that at Site 1 (9.2 μg/m^3) was about 17% higher. Concentrations at the stops were thus higher for PM_{10} at all three sites and, to a lesser extent, also for $PM_{2.5}$. For PM_{10} at Sites 1 and 3, mean concentrations of 61.3 and 49.7 μg/m^3 were measured, respectively, compared with 30.9 μg/m^3 at B-3 and 34.6 μg/m^3 at B-13. These differences are relevant because data from fixed monitoring stations are used to characterize and communicate urban air quality, whereas the present measurements show that substantially higher concentrations, particularly of PM_{10}, may occur locally in transport microenvironments, where people actually spend time waiting for public transport.

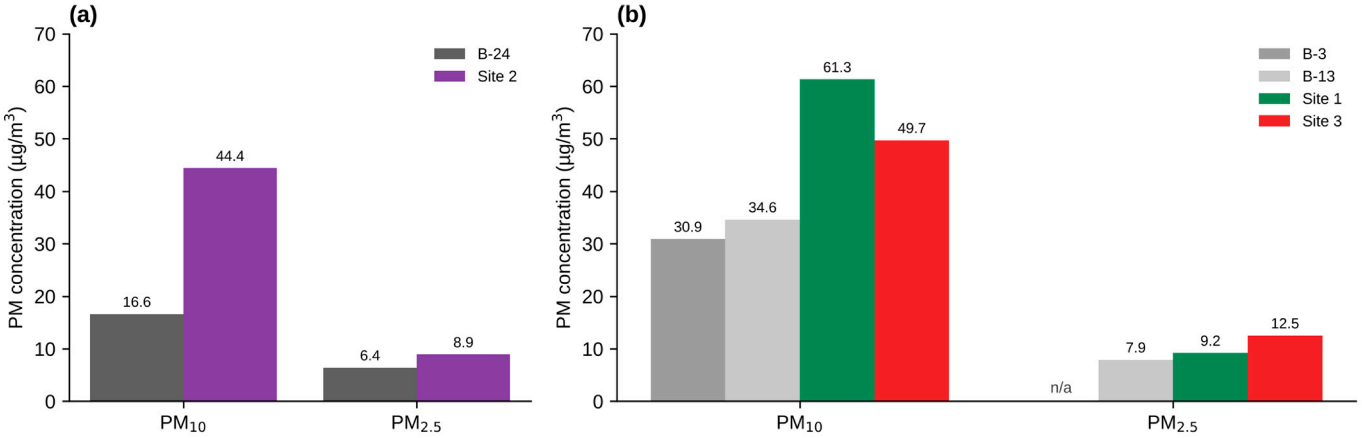


Figure 5. Mean PM_{10} and $PM_{2.5}$ concentrations measured at the public transport stops and at the nearest fixed air-quality monitoring stations in Bucharest on 14, 22 and 29 July 2025: (**a**) Site 2 and B-24; (**b**) Sites 1 and 3 and stations B-3 and B-13 (n/a—$PM_{2.5}$ not measured at B-3). Stop values: means of the series were averaged over the three days (about 1.5 h per site), with Site 1 harmonized (Section 2); network values: means of the hourly values for the same hours.

The discrepancy is most evident for PM_{10} and highlights the importance of the spatial representativeness of air-quality measurements. Fixed monitoring stations are designed to characterize air quality over a wider urban area and are generally less affected by highly localized emission and resuspension processes. In contrast, the present measurements were made in the commuters' breathing zone and close to heavily trafficked roads, and therefore capture short-range variations that the fixed network may not reflect. The results suggest that urban background or fixed-site measurements alone may underestimate the short-term particulate matter concentrations encountered by pedestrians and public transport users in traffic-intensive microenvironments. This does not mean that fixed stations characterize regulatory air quality inadequately; rather, regulatory monitoring and exposure-oriented measurements provide complementary information. Assessing human exposure in urban transport environments may therefore require measurements at locations and heights representative of where people actually breathe.

At Site 4, at the intersection of Traian Street and Calea Moșilor, $PM_{2.5}$ was measured continuously between 07:00 and 19:00 on five consecutive days, from Sunday 19 to Thursday 23 October 2025, with a GRIMM optical particle counter powered by a cable from a nearby building. Measurements were limited to daytime hours mainly because high relative humidity, particularly at night and in the early morning, could affect the operation of the instrument, and because the equipment had to be supervised continuously. To assess whether the concentrations at the stop were representative of wider urban $PM_{2.5}$ levels, the Site 4 measurements were compared with those at two fixed locations: Site 5, in front of the Faculty of Building Services Engineering, and the nearby B-13 station. For Site 4, $PM_{2.5}$ was averaged over the 07:00–19:00 interval, whereas the Site 5 and B-13 values are 24-h means.

The comparison therefore describes the relative magnitude and temporal variation in $PM_{2.5}$ rather than a direct comparison of identical averaging periods. As shown in Figure 6, $PM_{2.5}$ concentrations at Site 4 were higher than at both fixed locations on most days. The five-day mean concentration at Site 4 was 38.68 $\mu g/m^3$, compared with 19.06 $\mu g/m^3$ at Site 5 and 15.31 $\mu g/m^3$ at B-13, i.e., approximately 2.03 and 2.53 times higher, respectively. Considered day by day, the paired differences between Site 4 and Site 5 ranged from 0.4 to 35.3 $\mu g/m^3$ (mean 19.6 $\mu g/m^3$, 95% CI 1.1–38.1 $\mu g/m^3$, $n = 5$), and the daily ratios from 1.02 to 3.35 (geometric mean 1.89, 95% CI 1.09–3.29); relative to B-13, the paired differences ranged from 4.7 to 36.8 $\mu g/m^3$ (mean 23.4 $\mu g/m^3$, 95% CI 5.1–41.7 $\mu g/m^3$). With only five days, these estimates are imprecise. In contrast, the five-day means at the two fixed locations differed by only 3.75 $\mu g/m^3$ (Site 5/B-13 ratio of 1.25). The daily values provide further information on the spatial pattern of $PM_{2.5}$. Concentrations were relatively low at all three locations on 20 October, when Site 4 and Site 5 recorded almost identical values (17.5 and 17.1 $\mu g/m^3$, respectively) and B-13 recorded 12.84 $\mu g/m^3$. Concentrations then increased at all locations until 22 October. On 23 October, however, they diverged markedly: $PM_{2.5}$ remained high at Site 4 (50.3 $\mu g/m^3$) but decreased to 15.0 $\mu g/m^3$ at Site 5 and 13.52 $\mu g/m^3$ at B-13, so that Site 4 was approximately 3.35 times higher than Site 5 and 3.72 times higher than B-13. Similarly, on 19 October, the Site 4 concentration was approximately 3.46 times the B-13 value. From a health perspective, a 24-h mean may not adequately represent commuter exposure at public transport stops, since it includes periods with few passengers. For this microenvironment, the 07:00–19:00 mean, which covers the period of most intensive public transport use, is a more representative indicator of the $PM_{2.5}$ levels encountered by commuters. However, part of the contrast with the 24-h reference values may be due to the diurnal cycle of $PM_{2.5}$ itself, so the ratios given above are likely to overestimate the local enhancement.

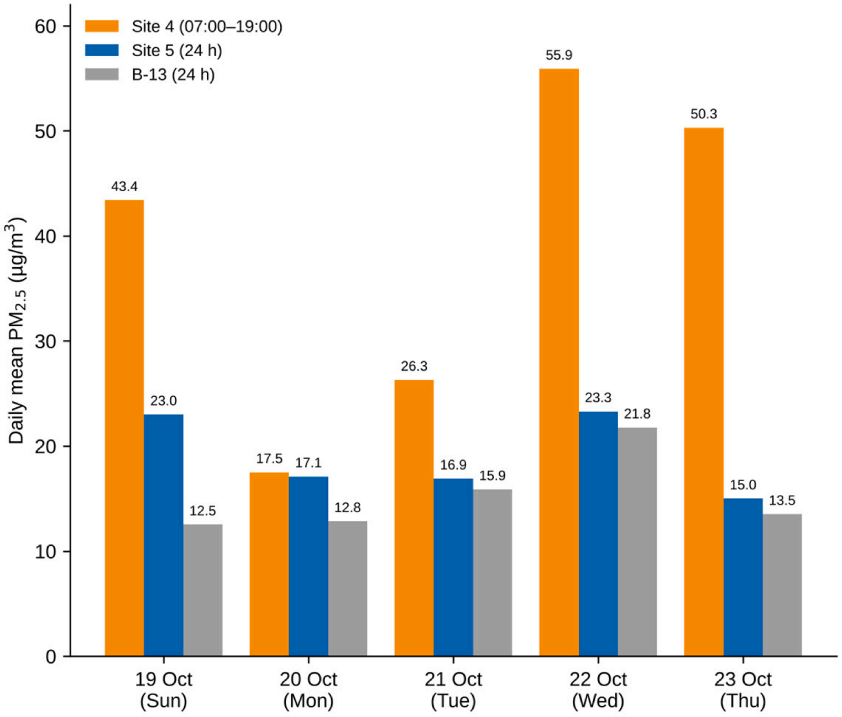


Figure 6. Daily mean $PM_{2.5}$ concentrations at Site 4 (07:00–19:00), Site 5 (24 h) and monitoring station B-13 (24 h), 19–23 October 2025.

The substantially higher $PM_{2.5}$ concentrations at Site 4, together with the much smaller differences between Site 5 and B-13, are consistent with the local enhancement of particulate

pollution at this stop, although the different averaging windows and instruments also contribute to the contrast. Site 4 had the highest $PM_{2.5}$ values measured in this study; however, since the stops were sampled in different periods (Site 4 only in October), this does not show that Site 4 is the most polluted stop. The intersection has heavy road traffic, with repeated stopping and acceleration of public transport vehicles. In addition, a considerable number of passengers were observed smoking while waiting during the campaign. Tobacco smoke may therefore have contributed to the locally elevated $PM_{2.5}$ concentrations, but the present measurements cannot separate its contribution from that of traffic and other local sources.

Table 1 summarizes the concentration contrasts between the stops and the reference locations, together with the averaging window of each value. These contrasts are affected by two sources of systematic uncertainty: the different averaging windows and a possible bias of the optical counters relative to beta attenuation. To assess the latter, the break-even factor $k^* = C_{ref}/C_{stop}$ was calculated, i.e., the factor by which the optical counter would have to overestimate the concentration for the contrast to vanish. For PM_{10}, k^* ranged from 0.37 (Site 2 versus B-24) to 0.70 (Site 3 versus B-13), so the higher PM_{10} at the stops would disappear only if the counters overestimated PM_{10} by more than about 40%. For $PM_{2.5}$ at Site 4 and at Site 3 in November, k^* was 0.40–0.49, whereas the smaller $PM_{2.5}$ contrasts in July (Sites 1–3) and at Site 3 in October ($k^* = 0.63$–0.85) could be explained by an instrumental bias of about 15–40%. Such a bias cannot be excluded, since the counters were not compared with the beta-attenuation monitor and the intercomparison of the counters themselves showed differences of up to 37% between units (Table S1). These smaller contrasts should therefore be regarded as uncertain.

Such short-term increases in $PM_{2.5}$ are relevant to human health. Epidemiological and panel studies have consistently associated short-term $PM_{2.5}$ exposure with adverse cardiovascular and respiratory responses. A recent global meta-analysis of 100 epidemiological studies reported an increased risk of cardiovascular mortality with short-term increases in $PM_{2.5}$, with a pooled risk ratio of 1.0090 (95% CI: 1.0074–1.0106) per 10 $\mu g/m^3$ increase in $PM_{2.5}$ [49]. At the physiological level, an updated meta-analysis of 33 panel studies showed that a 10 $\mu g/m^3$ short-term increase in $PM_{2.5}$ was associated with significant reductions in heart-rate variability, including a 0.92% decrease in SDNN and a 1.47% decrease in rMSSD, indicating altered cardiac autonomic regulation [50]. Short-term respiratory effects have also been demonstrated. The EAACI systematic review and guidelines reported that each 10 $\mu g/m^3$ increase in $PM_{2.5}$ was associated with an increased risk of asthma-related emergency department visits at lags of 0–3 days; for same-day exposure (lag 0), the pooled relative risk was 1.012 (95% CI: 1.001–1.023) [51]. Earlier meta-analytic evidence also showed more asthma-related emergency department visits after short-term $PM_{2.5}$ exposure, with particularly pronounced effects in children [52]. These findings suggest that the short-term $PM_{2.5}$ increments observed in public transport microenvironments may be of cardiovascular and respiratory relevance, even though such localized exposure episodes are not fully captured by the 24-h concentrations recorded at fixed stations. Their possible health relevance, taking into account the time actually spent at a stop, is discussed in Section 4.3.

3.2. Seasonal Variation in Particulate Matter Concentrations

Figure 7 shows the temporal evolution of $PM_{2.5}$ concentrations at Site 5, together with ambient temperature and relative humidity. $PM_{2.5}$ showed substantial short-term variability throughout the monitoring period, while air temperature generally decreased and relative humidity generally increased toward late autumn. Visual comparison of the three parameters does not reveal a simple relationship between $PM_{2.5}$ and either temperature

or relative humidity, suggesting that the variability of fine particulate matter reflects the combined effects of meteorological conditions, atmospheric dispersion and changes in local emission sources rather than a single meteorological parameter. To quantify these relationships, Spearman's rank correlation analysis was applied to the daily mean $PM_{2.5}$ concentrations, air temperature and relative humidity.

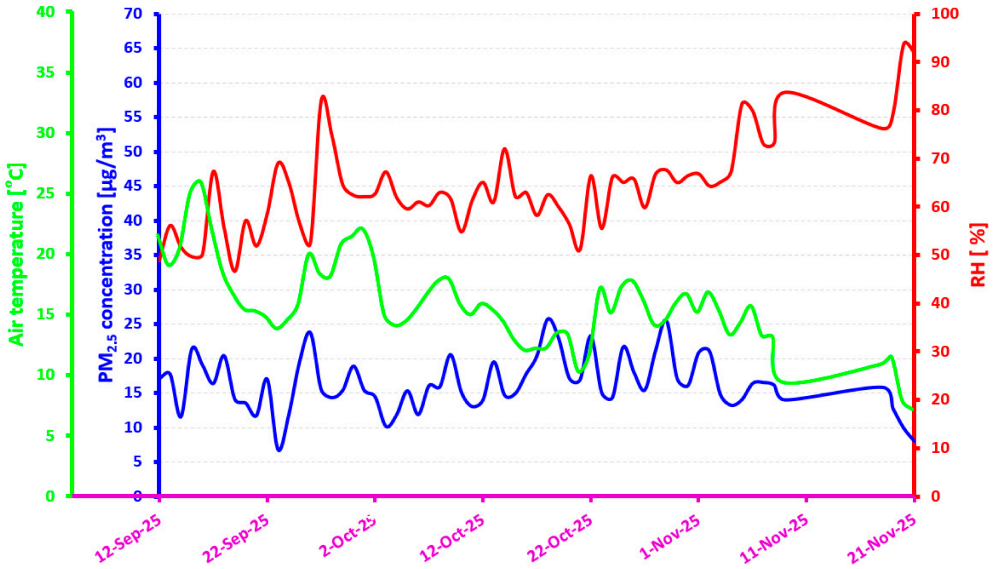


Figure 7. Daily mean $PM_{2.5}$ concentration, air temperature and relative humidity at Site 5, 12 September–21 November 2025.

Spearman's rank correlation analysis was performed on the daily mean values of the 63 days with valid data between 12 September and 21 November 2025. Spearman's rank correlation was chosen as a non-parametric measure of monotonic association that requires neither a linear relationship nor normally distributed variables. The coefficient was calculated as the Pearson correlation coefficient of the ranked observations, with average ranks assigned to tied values [53].

Spearman Rank Correlation

For two variables, X and Y, Spearman's rank correlation coefficient was calculated as follows:

$$\rho_s = \frac{\sum_{i=1}^{n}(R_{X,i} - R_X)(R_{Y,i} - R_Y)}{\sqrt{\sum_{i=1}^{n}(R_{X,i} - R_X)^2 \sum_{i=1}^{n}(R_{Y,i} - R_Y)^2}} \quad (1)$$

where $R_{X,i}$ and $R_{Y,i}$ are the ranks of the i^{th} observations of variables X and Y, respectively, R_X and R_Y are their mean ranks, and n is the number of paired observations ($n = 63$). This rank-based formulation is appropriate when tied observations may occur.

The statistical significance of the calculated Spearman coefficients was evaluated using the two-sided t-test approximation:

$$t = \rho_s \sqrt{\frac{n-2}{1-\rho_s^2}}, \quad (2)$$

with $n - 2$ degrees of freedom. The corresponding two-sided p-value was obtained from Student's t distribution. Since ambient temperature and relative humidity were themselves significantly correlated, a partial Spearman rank correlation was additionally calculated to evaluate the association between $PM_{2.5}$ concentration and temperature while controlling for

relative humidity [54]. For one control variable, the partial rank correlation was calculated as follows:

$$\rho_{PM,T\cdot RH} = \frac{\rho_{PM,T} - \rho_{PM,RH}\rho_{T,RH}}{\sqrt{\left(1-\rho_{PM,RH}^2\right)\left(1-\rho_{T,RH}^2\right)}}, \quad (3)$$

where $\rho_{PM,T}$, $\rho_{PM,RH}$, and $\rho_{T,RH}$ are the corresponding pairwise Spearman rank correlation coefficients. The statistical significance of the partial correlation coefficient was evaluated as follows:

$$t_{partial} = \rho_{PM,T\cdot RH}\sqrt{\frac{n-k-2}{1-\rho_{PM,T\cdot RH}^2}}, \quad (4)$$

where $k = 1$ is the number of controlled variables, resulting in 60 degrees of freedom for the present dataset.

The correlation analysis (Table 2) did not reveal a statistically significant monotonic association between daily $PM_{2.5}$ concentration and ambient temperature ($\rho_s = 0.202$, $p = 0.113$). The relationship between $PM_{2.5}$ and relative humidity was weakly negative and close to, but did not reach, the 0.05 significance level ($\rho_s = -0.238$, $p = 0.060$). Only the correlation between temperature and relative humidity was statistically significant ($\rho_s = -0.412$, $p < 0.001$), reflecting the transition to cooler and more humid conditions during the autumn. After controlling for relative humidity, the partial correlation between $PM_{2.5}$ and temperature remained weak and non-significant ($\rho_{\text{partial}} = 0.117$, $p = 0.365$). A non-significant coefficient does not demonstrate the absence of a meteorological influence; these results only show that, over this period, daily $PM_{2.5}$ at Site 5 did not follow a simple monotonic relationship with temperature or relative humidity. This is consistent with the monthly mean $PM_{2.5}$ concentrations, which remained within a narrow range (16.1 $\mu g/m^3$ in September, 17.1 $\mu g/m^3$ in October and 14.9 $\mu g/m^3$ in November) while the mean temperature decreased from 18.8 °C to 12.6 °C and the mean relative humidity increased from 59.0% to 76.6%. A weak association is plausible, since two opposing mechanisms may act simultaneously: higher temperatures may favor resuspension and secondary particle formation, whereas lower temperatures are accompanied by additional emissions from residential heating. Because these mechanisms act in opposite directions along the temperature gradient, a monotonic rank coefficient can be expected to be low. This interpretation could not be tested, since heating activity and atmospheric dispersion conditions were not measured. In addition, the daily values are serially correlated, so the *p*-values based on the t approximation are too small, and the coefficients in Table 2 are therefore interpreted descriptively, without conventional significance claims. Precipitation, wind and mixing-layer height were not included in the analysis. The correlation analysis identifies statistical associations only and does not imply causality; the observed $PM_{2.5}$ variability may also reflect seasonal changes in emission sources, atmospheric stability, precipitation and pollutant dispersion.

Table 2. Spearman and partial Spearman correlation analysis for $PM_{2.5}$ and meteorological parameters ($n = 63$).

Relationship	Correlation Coefficient	*t*	*p*-Value
$PM_{2.5}$—Temperature	$\rho_s = 0.202$	1.610	0.113
$PM_{2.5}$—RH	$\rho_s = -0.238$	−1.917	0.060
Temperature—RH	$\rho_s = -0.412$	−3.536	7.8×10^{-4}
$PM_{2.5}$—Temperature, controlling for RH	$\rho_{partial} = 0.117$	0.913	0.365

Figure 8 shows the seasonal variation in PM_{10} and $PM_{2.5}$ concentrations at Site 3, on the tram platform at Piața Iancului. For each month, the mean concentrations were calculated by averaging the daily 12:00–18:00 means over at least 10 monitoring days. The GRIMM counters were powered by their internal batteries and an external uninterruptible power

supply (UPS), which extended the monitoring period. The monthly $PM_{2.5}/PM_{10}$ ratio was calculated as the ratio of the corresponding mean $PM_{2.5}$ and PM_{10} concentrations. The July values in Figure 8 ($PM_{2.5}$ = 9.5 µg/m^3, $PM_{2.5}/PM_{10}$ = 0.19) represent the mean of all July monitoring days and differ from the three-day morning values in Section 3.1 (12.5 µg/m^3 and 0.28) because of the different days and time intervals and the different calculation of the ratio (ratio of the monthly means here, and mean of the instantaneous ratios in Section 3.1).

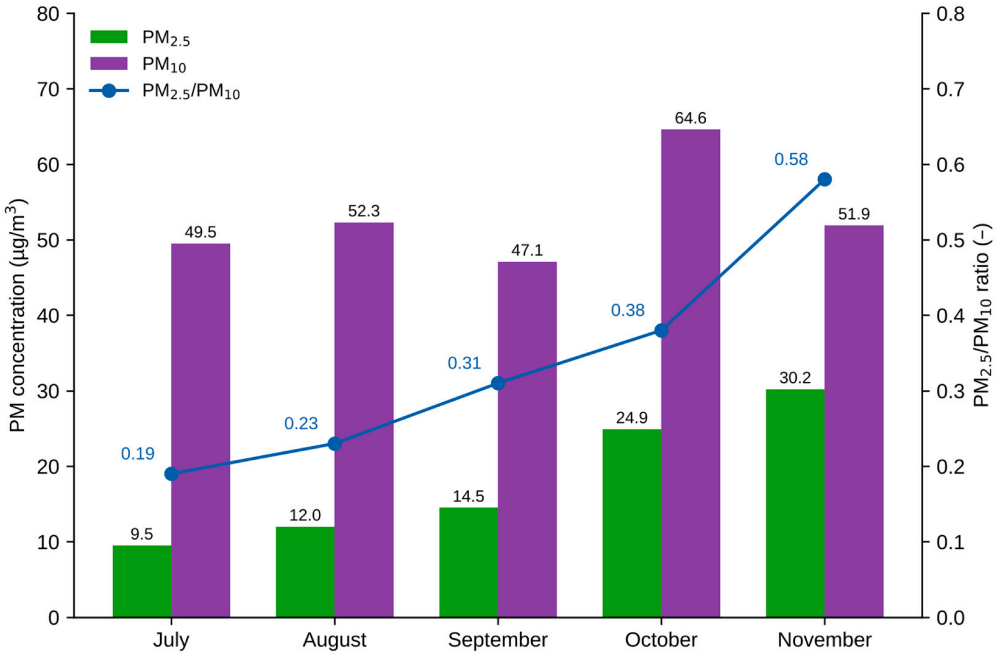


Figure 8. Mean PM_{10} and $PM_{2.5}$ concentrations and $PM_{2.5}/PM_{10}$ ratio at Site 3. Monthly means of the daily 12:00–18:00 means (at least 10 days per month); ratio = mean $PM_{2.5}$/mean PM_{10}. Mean RH during the sampling hours: 22.9, 36.5, 41.0, 39.6 and 39.7% from July to November.

The mean $PM_{2.5}$ concentrations at Site 3 during the 12:00–18:00 interval differed from the daily means at Site 5, in front of the Faculty of Building Services Engineering, by −9.9%, +45.6% and +102.7% in September, October and November, respectively (difference between Site 3 and Site 5 concentrations divided by the Site 5 concentration). The inlet was at approximately 1.2 m above street level at Site 3 and at 3 m at Site 5. While the concentrations were similar in September, the differences became substantial in October and particularly in November. In late autumn, daytime $PM_{2.5}$ concentrations at the stop were therefore higher than the 24-h means at the comparison site; part of this difference may be due to the different averaging windows, since the 12:00–18:00 values were compared with 24-h means. From an exposure perspective, the difference is relevant because the 12:00–18:00 interval is a period of active public transport use, whereas a 24-h mean also includes nighttime hours with much lower human activity and potentially different emission and dispersion conditions.

The two particle fractions showed distinct July–November patterns during the monitored afternoons. Mean PM_{10} concentrations remained within a relatively narrow range, from 47.1 to 64.6 µg/m^3, without a consistent increase from summer to late autumn. In contrast, $PM_{2.5}$ concentrations increased continuously from 9.5 µg/m^3 in July to 12.0 µg/m^3 in August, 14.5 µg/m^3 in September, 24.9 µg/m^3 in October and 30.2 µg/m^3 in November, i.e., by a factor of approximately 3.2. The $PM_{2.5}/PM_{10}$ ratio shows this change even more clearly, increasing continuously from 0.19 in July to 0.23 in August, 0.31 in September, 0.38 in October and 0.58 in November; $PM_{2.5}$ thus accounted for approximately 19% of PM_{10} in July but approximately 58% in November. Since PM_{10} did not increase comparably over

the same period, the results indicate a progressive change in the particle-size distribution, with fine particles becoming increasingly important during autumn. This increase in the fine fraction may be associated with seasonal changes in the dominant urban emission sources, such as combustion-related emissions from road traffic and, particularly in the colder months, the additional contribution of heating. The relatively limited seasonal variation in PM_{10} may reflect the different behavior of the coarse fraction ($PM_{2.5–10}$), which is more strongly influenced by mechanical processes, resuspension and atmospheric removal. The weak association between $PM_{2.5}$ and relative humidity at Site 5 cannot be used to exclude a humidity artifact at Site 3, because the beta-attenuation monitor samples through a heated inlet, whereas the optical counters were operated without a dryer. During the Site 3 sampling hours (12:00–18:00), the mean relative humidity recorded by the counter's sensor was 22.9% in July, 36.5% in August, 41.0% in September, 39.6% in October and 39.7% in November, well below the range of about 70–80% above which hygroscopic growth becomes important for undried optical counters [55]. In particular, RH remained practically unchanged from September to November, while $PM_{2.5}$ increased from 14.5 to 30.2 $\mu g/m^3$ and the $PM_{2.5}/PM_{10}$ ratio from 0.31 to 0.58; hygroscopic growth is therefore unlikely to explain the increase. The afternoon RH at Site 3 was much lower than the 24-h mean RH at Site 5 (59.0–76.6%), which also includes the more humid night and morning hours. No humidity correction was applied, since no correction validated for these instruments was available. The July–November increase is therefore described as a pattern observed during the monitored afternoons, and the contributions of traffic and residential heating remain hypotheses that would require source-apportionment analysis to be confirmed.

4. Discussion

4.1. Exposure at Public Transport Stops Compared with Fixed Monitoring Stations

The measurements show that particulate matter concentrations at the bus and tram stops were generally higher than those reported by the fixed urban stations, especially for PM_{10}. The mean PM_{10} concentrations at Sites 1–3 (44.4–61.3 $\mu g/m^3$) were 1.4–2.7 times higher than at the nearest fixed stations (16.6–34.6 $\mu g/m^3$). For $PM_{2.5}$, the contrasts were smaller: concentrations at Sites 1–3 exceeded those at the reference stations by factors of 1.2–1.6. The largest contrast was observed at Site 4, where the 07:00–19:00 mean $PM_{2.5}$ (38.68 $\mu g/m^3$) was about 2.0–2.5 times the 24-h means at Site 5 and B-13. These results agree with earlier studies reporting higher particulate levels at bus stops than in the surrounding urban background, with PM_{10} up to 2.5 times higher during peak hours [24–27], and support the view that bus and tram stops act as local hotspots [34,35]. Higher concentrations during bus and tram commutes than at nearby fixed sites were also reported in Barcelona [41], and in Žilina, Slovakia, PM at bus stations increased markedly during the morning peak [42]. However, part of the contrasts observed here may result from the different averaging windows and instruments (Table 1). Our mean concentrations are lower than the PM_{10} values above 83 $\mu g/m^3$ reported for an urban area in Slovakia [36], but our PM_{10} 95th percentiles (80.3–111.5 $\mu g/m^3$) are of similar magnitude, and the Site 4 $PM_{2.5}$ means exceed the $PM_{2.5}$ levels above 16 $\mu g/m^3$ reported there. Instantaneous PM_{10} values at the stops occasionally exceeded 200 $\mu g/m^3$, below the peaks of 316 $\mu g/m^3$ measured for pedestrians in Istanbul [37]; the two values are not directly comparable, however, as they were obtained with different methods and averaging times.

Relative to the health-based guideline values of the World Health Organization [20], the mean PM_{10} at the stops was close to or above the 24-h guideline value of 45 $\mu g/m^3$, and the daily $PM_{2.5}$ at the comparison site (Site 5) exceeded the 24-h guideline of 15 $\mu g/m^3$ on 40 of the 63 monitored days. These comparisons are only indicative, since our measurements at the stops covered a few hours per day, and the 50 $\mu g/m^3$ reference used in Section 3.1 is

not a regulatory exceedance criterion. Nevertheless, the findings suggest that fixed-station data may underestimate the concentrations experienced by commuters in traffic-intensive microenvironments, which is particularly relevant for the schoolchildren, students and elderly people who use these stops.

4.2. Particle Size Distribution and Seasonal Behavior

The coarse fraction ($PM_{2.5–10}$) dominated the particulate mass at all three sites during the July campaign (35.5–52.1 $\mu g/m^3$), and the $PM_{2.5}/PM_{10}$ ratios were low (0.19–0.28). This pattern is consistent with the main sources mentioned in the Introduction, such as road-dust resuspension and brake and tire wear, although no source apportionment was carried out. The differences between sites suggest an influence of local configuration: Site 3, between traffic lanes, had the highest fine-particle fraction, whereas Site 2, near a large park, had about 28% lower PM_{10} and 32% lower coarse-particle concentrations than Site 1 (after harmonization of the Site 1 counter). These associations remain indicative, since vegetation and traffic were not examined independently.

At Site 3, $PM_{2.5}$ increased more than threefold between July (9.5 $\mu g/m^3$) and November (30.2 $\mu g/m^3$), and the $PM_{2.5}/PM_{10}$ ratio rose from 0.19 to 0.58, while PM_{10} showed no clear trend. In contrast, the monthly mean $PM_{2.5}$ at the comparison site was nearly constant (16.1, 17.1 and 14.9 $\mu g/m^3$ in September, October and November). As a result, the difference between the stop and the comparison site grew from −9.9% in September to +45.6% in October and +102.7% in November. Several factors may contribute: additional combustion emissions during the heating season, which would be expected to affect both sites; a stronger local traffic contribution at the stop; the different time windows (12:00–18:00 versus 24 h); and measurement-related effects, such as the loss of volatile material in the heated inlet of the beta-attenuation monitor. A humidity artifact of the optical counter is unlikely, since the mean RH during the Site 3 sampling hours remained at about 40% from September to November (Section 3.2), even though the 24-h mean RH at Site 5 increased from about 59% to 77%. The present data do not allow the remaining explanations to be distinguished.

The Spearman analysis at Site 5 showed only weak, non-significant rank correlations of daily $PM_{2.5}$ with temperature ($\rho_s = 0.20$, $p = 0.113$) and relative humidity ($\rho_s = -0.24$, $p = 0.060$). This is compatible with the coexistence of opposing mechanisms (more resuspension and secondary formation at higher temperatures, additional heating emissions at lower temperatures) that a monotonic coefficient cannot capture. Wind speed and direction were not measured, which limits the interpretation of these relationships.

4.3. Potential Health Implications

The concentration contrasts in Table 1 refer to the hours spent at a stop, whereas the concentration–response functions available in the literature were derived from daily (24-h) concentration increments in population time-series studies; these functions therefore cannot be applied directly to the contrasts measured at the stops. To illustrate the order of magnitude, the contribution of waiting at a stop to the 24-h mean exposure was estimated as follows:

$$\Delta C_{24h} = \Delta C \cdot t_w/24, \quad (5)$$

where ΔC is the $PM_{2.5}$ contrast between the stop and the reference location (Table 1), and t_w is the daily waiting time (h). Three scenarios were considered: 15, 30 and 60 min per day, corresponding to one short wait, a return trip with two waits and a longer commute with transfers, respectively. The calculation assumes that the concentration during waiting equals the mean measured over the sampling window at the stop, that the person would otherwise be exposed to the reference concentration, and that the exposure–response relationship remains log-linear for such small increments. ΔC_{24h} was combined with two concentration–

response functions derived from same-day (lag 0) 24-h $PM_{2.5}$ in population studies: hospital admission or mortality for stroke, RR = 1.011 (95% CI 1.011–1.012) per 10 µg/m^3 [56], and asthma-related emergency department visits, RR = 1.012 (95% CI 1.001–1.023) per 10 µg/m^3 [51]. The excess relative risk was calculated as ERR = 100·[$RR^{\Delta C24h/10} - 1$]. The lower limit of the stroke interval equals the central estimate because the published values are rounded. Both are population-level relative risks: they describe the expected change in the daily rate of events in a population and cannot be interpreted as individual risks.

For the largest contrasts (Site 4), waiting 30 min per day increased the 24-h mean exposure by about 0.4–0.5 µg/m^3, corresponding to an excess relative risk of about 0.05% for both endpoints; for the other stops, the values were smaller (Table 3). These intervals reflect only the uncertainty of the concentration–response coefficients; the uncertainty of the measured contrasts (Table 1) and of the waiting time is larger, and the Site 4 contrasts are likely to be overestimated because of the different averaging windows. The comparison with a negative ΔC (Site 3 in September) was not converted into health indicators, since a lower concentration at a stop than at the reference location cannot be interpreted as a protective effect. Panel studies have also associated short-term increases in $PM_{2.5}$ with reduced heart-rate variability [50] and with small reductions in FEV_1 in healthy adults, −7.63 mL (95% CI −10.62 to −4.63 mL) per 10 µg/m^3 in environmental studies [57], and a global analysis reported increased cardiovascular mortality after short-term increases in $PM_{2.5}$ [49]; these endpoints were not quantified here because personal exposure and waiting times were not measured. Overall, the time-weighted increments are small for an individual commuter, and no clinical or individual risk can be inferred from these measurements; however, stops near schools are used repeatedly by many children, students and elderly people, so reducing local concentrations at such stops remains important.

Table 3. Illustrative time-weighted scenario: increment of the 24-h mean $PM_{2.5}$ exposure (ΔC_{24h}, Equation (5)) for daily waiting times of 15, 30 and 60 min, and excess relative risk (ERR, 95% CI) for a waiting time of 30 min per day. Only comparisons with ΔC > 0 are shown.

Comparison	ΔC (µg/m^3)	ΔC_{24h} for 15/30/60 min (µg/m^3)	ERR Stroke, 30 min (%)	ERR Asthma ED Visits, 30 min (%)
Site 1 vs. B-13 (July)	1.3	0.01/0.03/0.05	0.003 (0.003–0.003)	0.003 (0.000–0.006)
Site 2 vs. B-24 (July)	2.5	0.03/0.05/0.10	0.006 (0.006–0.006)	0.006 (0.001–0.012)
Site 3 vs. B-13 (July)	4.6	0.05/0.10/0.19	0.010 (0.010–0.011)	0.011 (0.001–0.022)
Site 4 vs. B-13 (Oct.)	23.4	0.24/0.49/0.97	0.053 (0.053–0.058)	0.058 (0.005–0.111)
Site 4 vs. Site 5 (Oct.)	19.6	0.20/0.41/0.82	0.045 (0.045–0.049)	0.049 (0.004–0.093)
Site 3 vs. Site 5 (Oct.)	7.8	0.08/0.16/0.33	0.018 (0.018–0.019)	0.019 (0.002–0.037)
Site 3 vs. Site 5 (Nov.)	15.3	0.16/0.32/0.64	0.035 (0.035–0.038)	0.038 (0.003–0.073)

The intervals reflect only the uncertainty of the concentration–response coefficients; they do not include the uncertainty of ΔC (instrument bias, averaging windows) or of the waiting time.

4.4. Limitations and Future Research

Several limitations should be considered. The comparison of Sites 1–3 is based on morning measurements on three days in July, averaged for each time step and only partly overlapping in time, and the Site 4 results on five days in October; the measurements at different sites and in different seasons cover different time windows, and instruments based on different principles (optical counters and a beta-attenuation monitor) were used without a parallel comparison between them. The three optical counters were compared with each other on one day only (Table S1), and the harmonization factors applied to the Site 1 data rely on this single comparison. The mass concentrations from the optical counters rely on the manufacturer's conversion algorithm and were not calibrated against gravimetric measurements at the sites; in addition, the counters were operated without a dryer, so although the mean relative humidity during the stop measurements did not exceed 42%, short episodes of higher humidity may have caused a slight overestimation

of $PM_{2.5}$. Waiting times and personal exposure were not measured, so the health-related estimates in Section 4.3 are scenario-based. Meteorological measurements were limited to temperature and humidity; chemical composition and traffic volumes were not determined; and the correlation analysis is affected by the serial correlation of the daily data. Future work should include longer parallel measurements at the stops and at fixed stations, together with wind measurements and traffic counts, chemical speciation to separate road dust, exhaust, tobacco smoke and heating sources, personal exposure monitoring, and the evaluation of mitigation measures such as fleet electrification, shelter design, vegetation and stop location.

5. Conclusions

To the authors' knowledge, this study provides the first experimental assessment of particulate matter at surface public transport stops in Bucharest. Mean PM_{10} concentrations at three stops (44.4–61.3 $\mu g/m^3$) were 1.4–2.7 times those reported by nearby fixed monitoring stations for the same hours, and $PM_{2.5}$ at a fourth stop (38.68 $\mu g/m^3$) was about two to 2.5 times higher than at the comparison site and the fixed station, suggesting that fixed-station data may underestimate the short-term concentrations to which commuters are exposed while waiting at these stops. The coarse fraction dominated the particle mass in July, with $PM_{2.5}/PM_{10}$ ratios of 0.19–0.28, while at one tram stop, $PM_{2.5}$ rose from 9.5 to 30.2 $\mu g/m^3$ and the $PM_{2.5}/PM_{10}$ ratio from 0.19 to 0.58 between July and November during the monitored afternoons, a pattern that cannot be explained by the relative humidity during the sampling hours, which remained at about 40% from September to November. Daily $PM_{2.5}$ at the comparison site showed only weak rank correlations with temperature and relative humidity. When the time spent at a stop is taken into account, the excess concentrations correspond to increments of less than 1 $\mu g/m^3$ in the 24-h mean exposure for waiting times of up to one hour per day. The results support the need for exposure-oriented monitoring at public transport stops, combined with measures to reduce local emissions and resuspension near schools and other busy locations.

Supplementary Materials: The following supporting information can be downloaded at https://www.mdpi.com/article/10.3390/toxics14111013/s1. Table S1: Intercomparison of the three GRIMM Mini-LAS 11-E counters operated side by side in the laboratory on 7 November 2025; Figure S1: One-minute PM_{10}, $PM_{2.5}$ and PM_1 concentrations measured by each counter versus the mean of the three counters, 7 November 2025.

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