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Open waste burning causes fast and sharp changes in particulate concentrations in peripheral neighborhoods

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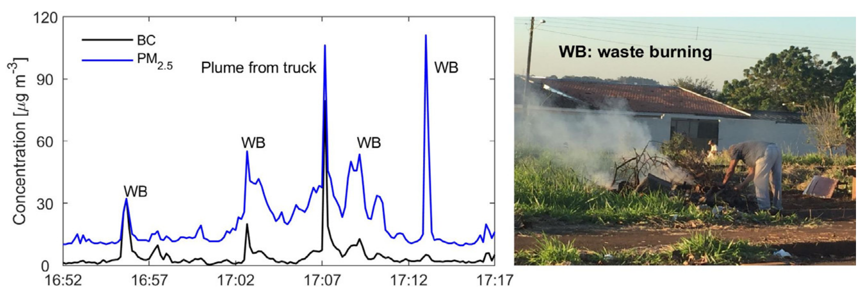
HIGHLIGHTS

- Open waste burning raises BC and $PM_{2.5}$ concentrations at the neighborhood level.
- $PM_{2.5}$ concentrations reach values higher than in heavily trafficked roads.
- Combined mobile and fixed-site monitoring unveiled air quality features at fine-scale resolution.
- A public opinion survey indicated that open waste burning occurs frequently in the area.
- Most people do not sort their waste properly, but dump and burn them in vacant lots.

GRAPHICAL ABSTRACT



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ABSTRACT

The open burning of municipal solid waste (MSW) –frequently observed in developing countries– emits harmful pollutants, including fine particulate matter ($PM_{2.5}$) and black carbon (BC), and deteriorates the air quality in urban areas. This work reports on $PM_{2.5}$ and BC measurements (fixed and mobile) conducted in a residential neighborhood on the outskirts of a Brazilian city (Londrina), complemented by a public opinion survey to understand the open burning in the context of waste management. Mean ($\pm$ standard deviation) BC concentration (1.48 ± 1.40 μg m^{-3}) at the fixed sites of the neighborhood was lower than downtown, while $PM_{2.5}$ (9.68 ± 8.40 μg m^{-3}) concentration was higher. The mobile monitoring showed higher mean $PM_{2.5}$ concentrations but lower BC/$PM_{2.5}$ ratios than downtown, with sharp and fast spikes (up to 317.87 and 565.21 μg m^{-3} for BC and $PM_{2.5,}$ respectively). The large spatial heterogeneity of particulate concentrations was associated with the occurrence of MSW burning events. Our observations were verified by the survey respondents who identified poor waste management practices: garbage in streets, waste burning, and illegal dump sites. Even though the area has a municipal waste collection service, the majority of the respondents (87%) had seen waste burning close to their homes on a weekly basis, and think that people burn waste out of habit (54%) and because they are not patient to wait for the collection services (67%). To combat this illegal practice, we suggest raising the public awareness through campaigns at local level, adopting education initiatives and economic incentives for correct waste segregation, and enforcing regular inspection of burning events by the authorities. Our research method proved to be a time- and cost-effective approach for mapping particulate concentrations and for identifying undesirable waste practices, and could be effectively applied to other global cities.

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1. Introduction

The world population generated 2.01 billion tons of municipal solid waste (MSW) in 2016, and it is estimated to reach 3.40 billion tons by 2050 under a business-as-usual scenario (Kaza et al., 2018). Brazil is the fifth largest waste producer on the planet (Wiedinmyer et al., 2014), with 62.8 million tons of MSW collected in 2018 (SNIS, 2019). Out of this total, 74% was disposed of in sanitary landfills, 11% was discarded in controlled landfills (with no leachate-control systems and/or gas collection), 13% in dump sites and 2% was treated in facilities for recycling materials (SNIS, 2019).

Even though the National Solid Waste Policy establishes sanitary landfills as safe disposal systems of non-recyclable waste since 2010 (Brazil, 2010), the law enforcement on illegal waste disposal has been delayed by 54% of the Brazilian municipalities (ABRELPE, 2019) because of financial constraints and infrastructure deficit. New deadlines for suppressing illegal disposal in Brazilian cities have been recently approved, with gradual enforcement in the period 2021–2024, depending on the city's population size (Brazil, 2020).

Dump sites can contaminate soil and water by leachate infiltration and run-off, deteriorate vegetation, contribute to global warming and to air pollution mainly through spontaneous fire or deliberate burning (ISWA, 2015; UN, 2019; Ferronato and Torretta, 2019). Open burning is a common practice to reduce volume and odors of dumped or uncollected waste, and can be observed at the municipal level (controlled landfills and dump sites) and at individual households in most developing countries (Estrellan and Iino, 2010; Lemieux et al., 2004). Reducing these environmental impacts is well aligned with the agenda of the United Nations through several Sustainable Development Goals (SDG 3, 11, 13 and14) (UN, 2019).

The open burning of MSW is a very inefficient combustion process due to its low temperature and limited oxygen supply. Moreover, since no air pollution control equipment is used, a variety of chemical species with different levels of toxicity is emitted into the atmosphere (Estrellan and Iino, 2010). Because the waste is highly heterogeneous and the burning conditions are uncontrolled, the number and share of the emitted species can vary largely (Lemieux et al., 2004). Typically, particulate matter (PM), carbon monoxide (CO), greenhouse gases (CO_2, N_2O, CH_4), polycyclic aromatic hydrocarbons (PAH), polychlorinated dibenzodioxin and dibenzofuran (PCDD/F) are injected in the atmosphere by the combustion of MSW (Estrellan and Iino, 2010; Lavric et al., 2004; Lemieux et al., 2004; Gullett et al., 2001; Zhang et al., 2017), which can cause harmful effects on human health. For example, the premature adult mortality per year due to the chronic exposure to fine particles ($PM_{2.5}$) from open waste combustion was estimated to be 270,000 worldwide (Kodros et al., 2016).

Black carbon (BC) is a component of the $PM_{2.5}$ resulting from the incomplete combustion of carbon-containing materials. It is associated with several deleterious effects on human health (e.g., Gan et al., 2010; Sunyer et al., 2015; Bové et al., 2019) and is an important short-lived climate pollutant because of its large global warming potential (Ramanathan and Carmichael, 2008). Within the municipal solid waste sector, open burning in landfills and dump sites represents a substantial source of BC, resulting in global emissions of 631 Gg BC per year (Wiedinmyer et al., 2014). Thus, curbing BC and $PM_{2.5}$ emissions and improving solid waste management in developing countries may bring significant co-benefits for health and the climate. Particularly, non-governmental organizations encourage initiatives to monitor, control and adopt measures to decrease BC emissions from the open burning of MSW. For example, the Climate and Clean Air Coalition (CCAC) municipal solid waste initiative provides assistance to municipalities to adopt integrated management systems to improve waste collection and to decrease the occurrence of garbage on streets or illegal dump sites, where open burning can occur (CCAC, 2020).

On a global scale, MSW consists mainly of organic waste (44%: food, yard and green waste) and dry recyclables (38%: plastic, paper, cardboard, metal, and glass), with the former accounting for up to 56% in low-income countries (Kaza et al., 2018). Unfortunately, the characterization of the open burning of MSW is less investigated, such as the amount and type of waste, the burning area and frequency of this practice. Nevertheless, knowledge of the type and composition of the MSW is vital to understand the impacts of uncontrolled waste burning on the air quality, human health and climate change and to improve atmospheric emission inventories (Reyna-Bensusan et al., 2019).

Because the open burning of MSW in developed countries is minimal, due to their financial capacity to manage waste and promote effective collection (Wiedinmyer et al., 2014), studies on the impact of this practice on urban air quality focus mostly on developing countries. For example, ambient concentration of several chemical species emitted by waste burned in landfills and households were measured in Indian (e.g., Agarwal et al., 2020; Dumka et al., 2018; Tiwari et al., 2015), Chinese (Liu et al., 2020), Lebanese (Baalbaki et al., 2016), and Nigerian (Oguntoke et al., 2019) cities. Even though Brazil is the third largest emitter of atmospheric pollutants by the open burning of MSW, just behind China and India (Wiedinmyer et al., 2014), to the best of the authors' knowledge, there are no studies on the environmental impact of this frequent practice in Brazil.

This work reports on fixed-site and mobile measurements of BC and $PM_{2.5}$ concentrations in a residential area in the city of Londrina (Brazil) where the burning of MSW occurs frequently. It also presents the results of a public opinion survey with students attending two schools located in the study area in order to better understand waste burning (frequency, materials burned, motive) in the context of waste management.

2. Methodology

2.1. Study area

Londrina is a city of 569,733 inhabitants (IBGE, 2020) located in the northern part of the State of Paraná, southern Brazil (Fig. 1a). The climate of the region is temperate, wet throughout the year and with hot summers (Cfa in the Köppen-Geiger classification). The mean annual temperature is 21.6 °C and the mean cumulative annual rainfall is 1583 mm (INMET, 2019). The main local sources of air pollution are on-road traffic, industrial emissions, and illegal burning of MSW (Targino and Krecl, 2016; Charres, 2019). The region is also affected by long-range transport of pollutants emitted by biomass burning in other regions of Brazil and in neighboring countries, mainly from August to October (Targino et al., 2019).

Londrina has 54 neighborhoods and the collection of MSW occurs door-to-door for the residential sector, covering 100% of the urban area. One company collects organic and non-recyclable waste (up to 600 l per week and household) and seven waste picking cooperatives are in charge of collecting recyclable materials (CMTU, 2020). In 2017, 127,500 tons of organic and non-recyclable waste were collected in the city, while the cooperatives commercialized a total of 8766 tons of recyclable materials (CMTU, personal communication, 2020). The city has two voluntary delivery sites that receive up to 1 m^3 per household and month of construction debris, wood pieces, tree clippings, and large objects such as furniture. Disposal of organic, healthcare, industrial waste, as well as plaster, batteries, lamps and tires is prohibited at these locations. These delivery sites work Mon-Fri (08:00–16:45) and Sat (08:00–11:45). A composting initiative implemented by the municipality has been in place since 2011, using the organic waste collected in downtown neighborhoods separately from the non-recyclable waste. Finally, residents can drop off certain hazardous and non-hazardous waste (e.g., batteries, lamps, tires, e-waste) at specific sites (CMTU, 2020) for recovery or proper disposal, as part of the so-called "reverse logistic" initiative (the return of products from the end consumer back to the manufacturer).

The open burning of waste is illegal and subject to a fine, whose control and execution are responsibilities of the Municipal Environmental

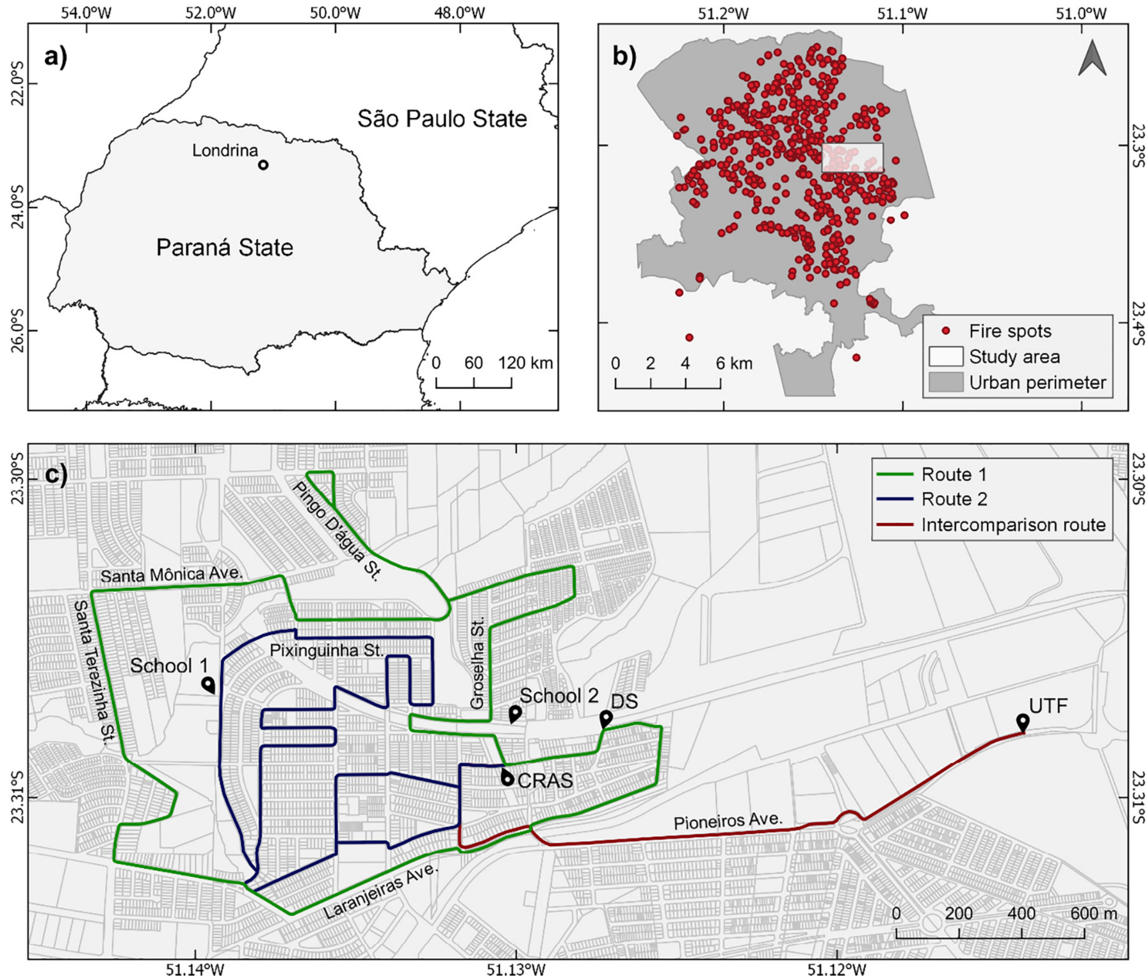

Fig. 1. Location of the city of Londrina (a), the study area and spots where the burning of MSW were reported (complaints registered at SEMA) in the period 2013–2016 (b), together with fixed-site monitoring, dump site (DS), mobile routes and schools where the survey was applied (c).

Secretariat (SEMA). Through a dedicated phone line, SEMA registered 565 anonymous complaints of illegal fires across the city (Fig. 1b) in the period 2013–2016. The burning of plastics, electronics, wood, tires, construction debris, tree clippings and mowing residues were most widely reported (SEMA, personal communication, 2017).

This study was conducted in a residential area in the east side of the city, incorporating Interlagos, Fraternidade and Ideal neighborhoods (Fig. 1c). We chose this area due to the large number of anonymous fire complaints (Fig. 1b), logistics (proximity to the sampling location for instrumental maintenance and frequent data downloading), security concerns (the equipment had to be left unattended at the fixed sites), and driving safely for mobile monitoring (routes 1, 2 and intercomparison). These neighborhoods consist of sparsely built, detached houses, gathering a population of 24,428 inhabitants with low-middle income and a literacy rate of 95% (IBGE, 2010). We covered an area of 1.29 km^2 (delimited by route 1, Fig. 1c) with the following land use: vegetation (40.5%), roof (36.6%), asphalt (20.0%), bare soil (2.8%) and water (0.1%). In these neighborhoods, organic and non-recyclable waste was collected three times per week and recyclable materials once per week. Even though dumping waste is unlawful, a dump site was identified within the study area (Fig. 1c, DS), where frequent open burning of MSW was reported by the local residents (SEMA, personal communication, 2017).

Table 1
Instrument set-up for mobile and fixed monitoring.

Sampling conditions	Fixed monitoring			Mobile monitoring		
	AE51	AE42	DustTrak	AE51	DustTrak	GPS
Flow rate (L min^{-1})	0.05	5.0	1.7	100	1.7	–
Frequency (s)	60	120	60	10	10	1

2.2. Field work

2.2.1. Fixed-site monitoring

Air quality instruments were deployed at two fixed sites: at the campus of the Federal University of Technology (UTF) and at the Social Assistance Office (CRAS) (Fig. 1c). BC and $PM_{2.5}$ concentrations were measured during 35 days (6 June through 11 July 2017) at CRAS site with a microaethalometer (model AE51, AethLabs, USA) and a photometer (DustTrak model 8520, TSI, USA), respectively. The instruments were installed in an office, with the sampling lines passing through a window to sample outside air at 2 m from the ground. BC concentrations were also monitored at the UTF campus with a seven-wavelength aethalometer (model AE42, Magee Scientific, USA) operated with a 2.5 μm cyclone and the inlet located at a height of 2 m. The details of the instrument setup are displayed in Table 1.

Weather conditions (air temperature, relative humidity, wind direction and speed, and precipitation) were obtained from the meteorological station managed by the Paraná Meteorological System (SIMEPAR), located 7 km southwest of the UTF campus. We manually counted the motorized vehicles circulating in front of CRAS and UTF sites. Following

Targino et al. (2016), total volume per hour and per vehicle category (passenger cars, motorcycles, trucks and buses) were counted twice per day (08:00–09:00 and 17:00–18:00) on selected weekdays.

2.2.2. Mobile monitoring

The mobile samplings of BC and $PM_{2.5}$ were conducted on two routes, driven simultaneously with two passenger cars on eight selected weekdays during the field campaign (Fig. 1c). Each car contained a kit with an AE51, a DustTrak 8520 and a GPS receptor (model D-100, GlobalSat, Taiwan), operated under the conditions detailed in Table 1. The two routes consist of roads with reduced motorized traffic and no public transport bus service. This strategy allowed us not only to avoid sampling plumes emitted by vehicle exhausts, but also to drive slowly (vehicle speed below 20 km h^{-1}) and, thus, to increase the number of sampling points per km driven. Route 1 was 7.8-km long and comprised the outermost part of the neighborhoods, with a larger fraction of vacant lots, which may favor the occurrence of MSW dumping and burning. Route 2 covered the inner part of the study area with a total length of 5.6 km, and included a larger number of houses than route 1.

Because waste burning in the study area occurred more frequently in the late afternoons (Targino et al., 2016), mobile sampling started at 16:15. Both cars left the UTF campus driving on the same avenue (intercomparison route, Fig. 1c), and parked at the CRAS site for approx. 10 min to intercompare the instruments onboard the cars with those installed at that place. Finally, each car started its route, taking on average 45 (route 1) and 55 min (route 2) to close the loop. The researchers kept a logbook detailing departure and arrival times, presence of other vehicles driving close to the car, stops at traffic lights, smell of burned material and occurrence of MSW burning.

2.2.3. Public opinion survey

The main objective of this survey was to characterize the waste burning practice by assessing the students' perceptions of the frequency of fires, materials most commonly burned and reasons behind this practice. We also investigated the waste management practices at the residential level and the students' knowledge of the health and environmental consequences of MSW burning.

The survey was conducted through questionnaires containing a mixture of nine closed- and open-ended questions (Supplementary Material, Section C). The questionnaires were administered to students attending two schools located in the study area (Fig. 1c, School 1: Ana Molina Garcia State School, and School 2: Professor Carlos Zewe Coimbra Municipal School), with ages ranging from 11 to 17 years old at School 1 and from 6 to 10 years old at School 2. In Brazil, students must attend public school in the neighborhood where they live, hence we assume that their answers are representative of people living in that area. At least 394 questionnaires were required for a statistically representative sample, with a tolerable error of 5%, given a population of 24,428 inhabitants (Agresti and Finlay, 2008).

The questionnaire was previously tested with a small group of children for language comprehension and then approved by the teachers of the target students. Finally, the survey was administered to all students by the same researcher during three selected days in September 2017. The researcher explained the purpose of the work and the correct way to complete the questionnaire, and all students who were in the classroom that day participated.

2.3. Quality assurance and quality control procedures

We operated the instruments following the procedures recommended by their manufacturers. Flow rate checks on the DustTrak and AE51 were performed using a flowmeter (model 4100, TSI, USA).

The DustTrak units were zeroed daily at the fixed site and immediately before the mobile samplings and, if required, their impactors and chambers were cleaned. AE51 filters were replaced every day. We checked the instruments' batteries and memories and synchronized their clocks using the same laptop connected to the internet to ensure clock accuracy. These procedures were applied daily during weekdays for the fixed stations and before each mobile sampling.

The instruments were intercompared throughout the campaign to establish any discrepancies in the measurements caused by extended runtime and/or lack of calibration. Discrepancies in the concentrations were compensated by taking one of the instruments as reference to harmonize the datasets. The data were also inspected for inconsistencies such as negative, constant, extreme, rapidly changed and other suspicious data. In some occasions, the AE51 units installed at CRAS and onboard the cars showed negative concentrations. This behavior occurs in clean environments with concentrations below the AE51 limit of detection, and/or when the instrument operated at a high sampling rate, due to instrumental noise (Hagler et al., 2011). For example, negative concentrations were observed at CRAS (2.1% of the dataset) in the early hours when BC levels were low. Several correction algorithms (Optimized Noise-Reduction Algorithm (ONA), Local Polynomial Regression (LPR) and Centered Moving Average (CMA)) available on the manufacturer's site (https://aethlabs.com/) were applied to the AE51 data. However, the general outcome was unsatisfactory. The ONA algorithm zeroed the low concentrations; the LPR and CMA algorithms reduced the negative values (from 2.1% to 0.2% and 0.1%, respectively), but largely decreased the highest peaks in the data series (for example, from 111.0 μg m^{-3} to 82.0 μg m^{-3} and 66.1 μg m^{-3}, respectively). These artifacts smoothed out the time series and rendered unrealistic features, which could lead to misinterpretation of the datasets. Thus, we decided not to use any of these correction algorithms and opted to discard the negative data from further analysis. A compensation for the loading effect was applied to the BC data following Virkkula et al. (2007), but the data were not corrected for scattering artifacts due to the lack of concurrent measurements of the aerosol scattering coefficient.

The DustTrak 8520 provides mass measurements based on particle scattering and is factory calibrated to the respirable fraction of standard ISO 12103-1. This calibration usually overestimates the concentrations in urban environments, since urban aerosols are less dense than the one used in the calibration (Ramachandran, 2005). Thus, we adopted a correction factor of 2.01 obtained in a previous experiment by collocating the DustTrak 8520 with gravimetric measurements at UTF. This correction factor is consistent with values (between 1.65 and 2.78) reported in other urban studies (e.g. McNamara et al., 2011; Pitz et al., 2003; Wallace et al., 2011). Because aerosol optical properties are affected by water uptake in high humidity conditions, $PM_{2.5}$ measurements were corrected following the correction curve developed by Laulainen (1993).

2.4. Data processing

Hourly mean concentrations and descriptive statistics were computed for the fixed-site measurements (arithmetic mean, median, minimum, maximum, 5th and 95th percentiles, and standard deviation SD). Note that we reported BC concentrations at UTF only for the wavelength coincident (880 nm) with the AE51 monitors.

Black carbon particles from biomass burning have stronger absorption in the UV region than fossil fuel, which absorbs mostly in the IR (Kirchstetter et al., 2004), and this optical property allows to separate the contributions from these emission sources. The absorption Ångström exponent (Å) calculated at two separated wavelengths (UV and IR) can be used as an indicator of the origin of BC. The Å value for pure traffic conditions is low and less variable whereas the exponent for pure biomass burning has a higher and wider range, depending on the biomass type and combustion process (Kirchstetter et al., 2004; Sandradewi et al., 2008; Russell et al., 2010; Liu et al., 2018). In this study, we calculated Å using the 370 and 880 nm wavelengths ($Å_{370/880}$). We adopted the following criteria for aerosol characterization: i) $Å_{370/880}$ values smaller than 1.1 indicate fossil fuel combustion

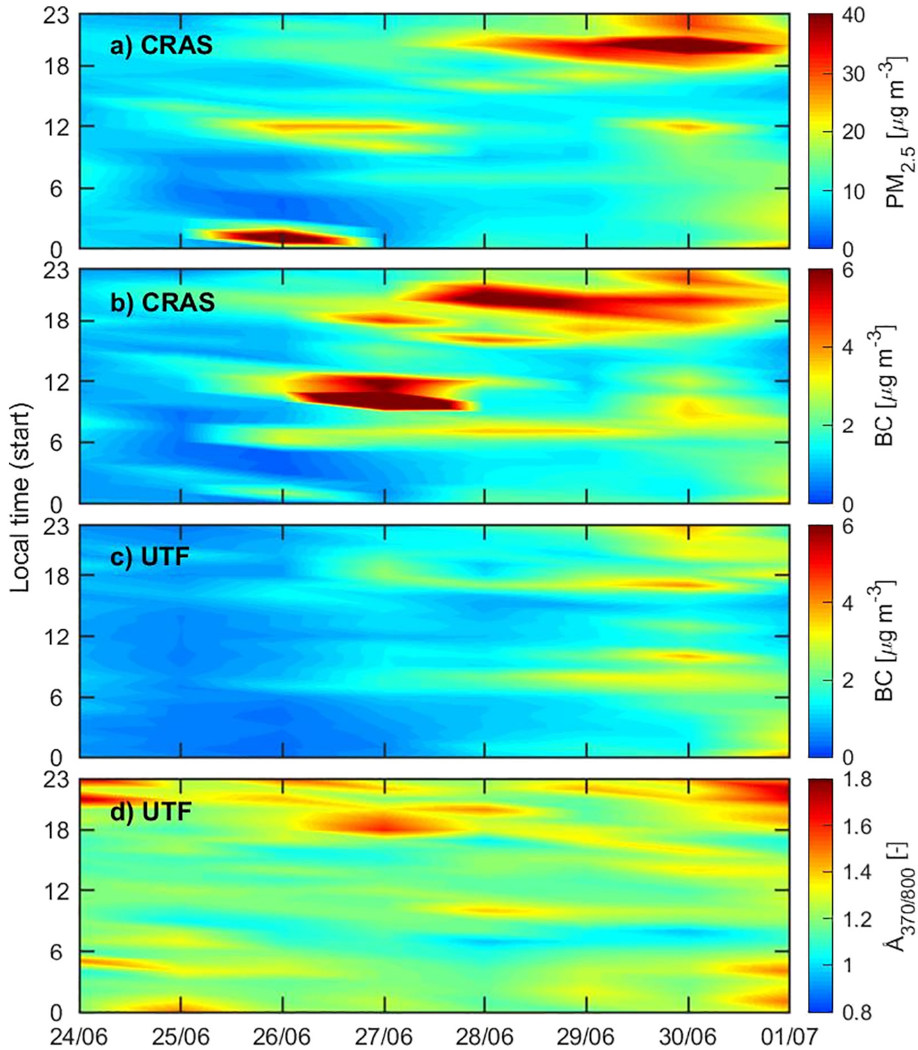

Fig. 2. Hourly values of $PM_{2.5}$ and BC, and $Å_{370/880}$ measured at the fixed sites over one week (24–30 June 2017).

(Sandradewi et al., 2008), ii) larger than 1.5 are mostly due to biomass burning (Liu et al., 2018), and iii) values in between (1.1–1.5) indicate that BC-rich aerosol was emitted by a combination of fossil fuel and biomass burning.

BC and $PM_{2.5}$ mobile measurements were georeferenced by matching their corresponding times with GPS times. The spatial distribution of the pollutants was calculated by taking median values of aggregated concentrations within 50–70 m buffers along the routes. The buffer size was based on a trade-off between data availability and overlapping buffers. Because the negative effect of peak pollution concentrations on human health is well documented (Dons et al., 2019; Jeong and Park, 2018), we also explored the spatial distribution of the 95th percentile concentrations aggregated within the same buffers used for the median calculations. Note that we only considered mobile data for routes 1 and 2, excluding the intercomparison section of the route to avoid influence from the exhaust plumes from the higher number of vehicles circulating on the latter.

Survey data were processed per school, coding closed-ended answers with a binary format (failure = 0, success = 1). In the case of open-ended questions (part of questions 6 and 9), all contributions from the respondents were grouped according to their similarity to facilitate the interpretation of the results. The response rate was also computed for each question. Finally, we calculated the mean percentage of respondents per question, considering both school together.

3. Results and discussion

3.1. Fixed-site monitoring

During the sampling campaign, the mean daily air temperature and relative humidity varied between 12.5 °C and 22.1 °C and between 57% and 96%, respectively. The mean wind speed was 2.4 m s^{-1} and the predominant wind directions were east (35%) and southeast (21%). The accumulated rainfall over the study period was 82.2 mm and it rained on five out of 36 days on weekdays mostly during the night, with the largest daily rainfall of 33.6 mm on 06 June 2017. The mean daily solar radiation dose was 11.7 MJ m^{-2}, varying between 1.4 (on 13 June) and 14.4 MJ m^{-2} (on 11 June).

To illustrate the spatio-temporal variability of the pollutant concentrations, Fig. 2 displays hourly values between 24 and 30 June 2017 (Saturday to Friday). Even though, the fixed sites were only 1.7 km apart, BC concentrations were higher at CRAS than at UTF (Fig. 2b,c), suggesting the occurrence of localized pollution events near the former. For example, a BC peak concentration of 13.7 μg m^{-3} was recorded at CRAS on 27 June at 10:00 whereas the value measured at UTF was 10-fold lower (1.36 μg m^{-3}), with wind conditions unfavoring the transport towards UTF (northeasterly at 3.6 m s^{-1}). The highest $PM_{2.5}$ peaks at CRAS were observed in the evening (28–30 June) and early hours (25–26 June), and did not coincide with the BC peaks, indicating that the contribution of BC to the $PM_{2.5}$ fraction largely varied with time (Fig. 2a,b).

Table 2
Descriptive statistics of hourly BC (880 nm), $PM_{2.5}$ and $Å_{370/880}$ measured at fixed sites.

Statistics	CRAS		UTF	
	BC [μg m^{-3}]	$PM_{2.5}$ [μg m^{-3}]	BC [μg m^{-3}]	$Å_{370/880}$ [−]
Mean	1.48	9.68	1.18	1.18
Median	1.11	7.57	0.99	1.16
SD	1.40	8.40	0.76	0.15
Minimum	0.18	0.12	0.31	0.68
5th percentile	0.35	1.80	0.41	0.96
95th percentile	3.63	25.56	2.59	1.45
Maximum	18.46	83.20	7.48	2.03
# samples	769	827	862	862

Note that values of $Å_{370/880}$ higher than 1.50, occurred in the evenings (Fig. 2d), indicating predominance of biomass smoke (Liu et al., 2018). Targino and Krecl (2016) also reported the increase of smoke and peak BC concentrations in the same area in the evenings.

Descriptive statistics of hourly concentrations of BC, $PM_{2.5}$ and values of $Å_{370/880}$ measured at the fixed sites are displayed in Table 2, considering the entire sampling campaign. Missing data corresponds to periods when the instruments were not operating due to maintenance, data download or electric power failure. BC concentrations were higher and more variable at CRAS (mean ± SD: 1.48 ± 1.40 μg m^{-3}) than at UTF (mean ± SD: 1.18 ± 0.76 μg m^{-3}). The Mann-Whitney *U* test at 5% significance level (p-value < 0.05) indicated that the BC concentrations at CRAS were significantly higher than at UTF. Because the CRAS site is located in a more densely populated area, it most likely that the higher BC concentrations are linked to frequent waste burning activities. $PM_{2.5}$ concentrations at CRAS also showed a large dispersion (mean ± SD: 9.68 ± 8.40 μg m^{-3}). The absorption Ångström exponent varied between 0.68 and 2.03, with 30% of the values smaller than 1.10 (interpreted as periods dominated by fossil fuel combustion), and 4% of the values higher than 1.50. Around 66% of the time, BC-rich aerosols were due to a combination of emissions from both fossil fuel and biomass burning.

Table 3 shows a comparison with other studies conducted at several sites in Londrina. The mean BC concentrations at CRAS and UTF were lower than within a busy street canyon (south and north façades, being the latter close to a bus stop) and higher than at an urban background site (rooftop located downtown). Note that the street canyon site has a larger traffic volume at rush hours on weekdays (661 vehicles h^{-1}, 13% of which are heavy-duty diesel vehicles HDDV) (Krecl et al., 2019) than at UTF (234 vehicles h^{-1} 5% HHDV) and CRAS (45 vehicles h^{-1}, 2% HDDV). On the other hand, the mean $PM_{2.5}$ concentration at CRAS was higher than the values reported elsewhere, indicating a large contribution of other local emission sources than motorized traffic at this site. $Å_{370/880}$ values larger than 1.5 were observed at UTF (this study) and at an urban background site (rooftop) (Krecl et al., 2016), indicating influence of biomass burning events on the aerosol load over the city. Krecl et al. (2016) reported high $Å_{370/880}$ values between 23:00 and 04:00, which they attributed to the advection of pollutants emitted by biomass burning in suburban areas. They explained that the biomass plume was more easily identified downtown in this period than during the day, when the large traffic emissions masked the biomass signal.

Table 4 shows results from other studies conducted in global cities where the authors declared a contribution from burning of MSW to the particle load. The comparison is challenging because of the different characteristics of the sampling sites, sampling frequencies and reported statistics. In relation to BC, Indian cities (Delhi and Hyderabad) presented much higher BC concentration than our study. Note that both Indian studies were conducted in densely populated cities (29.4 and 10.0 million in 2020, respectively), with a variety of emissions sources and at sampling sites located downtown. These sites were largely dominated by combustion of fossil fuel (motorized traffic, industries) and waste burning was mentioned as a smaller contribution within the biomass combustion activities. The maximum $Å_{370/880}$ value at UTF site was higher than those reported by the Indian studies, indicating that eventually aerosols were more dominated by biomass burning in our case. For $PM_{2.5}$, the Indian studies again reported the highest concentrations in Chennai (population of 11.0 million in 2020) and Delhi. The studies by Weichenthal et al. (2015) and Baalbaki et al. (2016) showed that spikes in $PM_{2.5}$ concentrations (a high value for a short time period) due to the burning of MSW can be several times higher than the mean value. This was particularly relevant in the small city of Iqaluit (7700 inhabitants, North Canada) when the winds transported pollutants emitted by a spontaneous landfill burning at 2.4 km far from the receptor site.

Finally, the diurnal cycles (mean, median, 5th and 95th percentiles) of pollutant concentrations were plotted for weekdays, Saturday and Sundays and displayed in Fig. 3. Overall, the pollutant concentrations exhibited a different pattern compared to studies conducted in traffic-dominated environments. $PM_{2.5}$ concentrations were higher at weekends than on weekdays, with largest values between 17:00 and 22:00 (Fig. 3a) whereas they peaked on weekday mornings (07:00) and evenings (18:00) downtown (Krecl et al., 2019). BC concentrations were higher on weekdays than at weekends at both sites, with CRAS showing a larger variability with higher 95th percentiles in the afternoon (Fig. 3b, c). According to the evolution of the $Å_{370/880}$ values (Fig. 3d), the UTF site was largely impacted by biomass burning (including waste burning) every day in the late afternoon and evenings, particularly on Saturdays. These results match our visual observations of several fire foci in the neighborhood in the late afternoons which, allied to a more stable atmosphere in the evening, favored the accumulation of pollutants within the boundary layer.

Table 3
Comparison with other fixed-site studies conducted in Londrina.

Pollutant	Location	Mean	Year	Reference
BC [μg m^{-3}]	[a]Canyon, north	3.43–3.62	2014/2015; 2016	Krecl et al. (2016, 2019)
	Canyon, south	2.55–2.61	2014/2015; 2016	Krecl et al. (2016, 2019)
	[b]Rooftop	0.77–0.88	2014/2015; 2016	Krecl et al. (2016, 2019)
	UTF	0.69–1.38	2014/2015	Targino and Krecl (2016)
	UTF	1.46	2017	This study
	CRAS	1.48	2017	This study
$PM_{2.5}$ [μg m^{-3}]	Canyon, south	7.70	2016	Krecl et al. (2019)
	[b]Rooftop	5.25	2016	Krecl et al. (2019)
	UTF	5.06	2016	Krecl et al. (2019)
	CRAS	9.68	2017	This study
$Å_{370/880}$ [−]	[b]Rooftop	1.36 [c][0.30–2.52]	2014/2015	Krecl et al. (2016)
	UTF	1.18 [c][0.68–2.03]	2017	This study

[a] Close to a bus stop.
[b] Building rooftop located downtown (20 m height).
[c] Range.

3.2. Mobile monitoring

The average weather conditions during the mobile sampling were as follows: air temperature of 21.5 °C, relative humidity of 60%, wind speed of 3.0 m s^{-1}, prevailing easterly winds (69%) and no rainfall.

Table 4
Comparison with other studies conducted in global cities where MSW burning was reported.

Pollutant	City/country	Value	Year	Reference	Comments
BC [$\mu g\ m^{-3}$]	Londrina, Brazil	[a]1.48 ± 1.40	2017	This study	Suburban, influenced by MSW burning
	Delhi, India	[a]24.0 ± 12.2	2015/2016	Dumka et al. (2018)	Downtown, 28% biomass burning (including waste)
	Delhi, India	[a]12.1 ± 8.7	2011/2012	Tiwari et al. (2015)	Downtown, 6% biomass burning (including waste)
	Hyderabad, India	[b]14.0 [c]75.0	2004	Latha and Badarinath (2006)	Downtown, waste burning 500-m from sampling site
$PM_{2.5}$ [$\mu g\ m^{-3}$]	Londrina, Brazil	[a]9.68 ± 8.40	2017	This study	Suburban, influenced by MSW burning
	Delhi, India	[a]182.75 ± 114.5	2011/2012	Tiwari et al. (2015)	Downtown, 6% biomass burning (including waste)
	Iqaluit, Canada	[d]4.61 [e]85.0	2014	Weichenthal et al. (2015)	Site located 2.4 km from a smoldering landfill
	Beirut, Lebanon	[f]14.2–67.8 [g]665.0	2015	Baalbaki et al. (2016)	Rooftop close to a dump site with waste burning
	Chennai, India	[a]44.49 ± 2.0	2010	Karthikeyan et al. (2011)	Sampling at a dump site with open burning
$Å_{370/880}$ [−]	Londrina, Brazil	[d]1.18 [h][0.68–2.03]	2017	This study	Suburban, influenced by MSW burning
	Delhi, India	[d]1.29 [h][0.99–1.72]	2015/2016	Dumka et al. (2018)	Downtown, 28% biomass burning (including waste)
	Delhi, India	[d]1.09 [h][0.38–1.28]	2011/2012	Tiwari et al. (2015)	Downtown, 6% biomass burning (including waste)

[a] Mean ± SD.
[b] Nocturnal mean (no event).
[c] Nocturnal spike (event).
[d] Mean.
[e] Hourly spike.
[f] Range based on daily data.
[g] 10-min spike.
[h] Range based on hourly data.

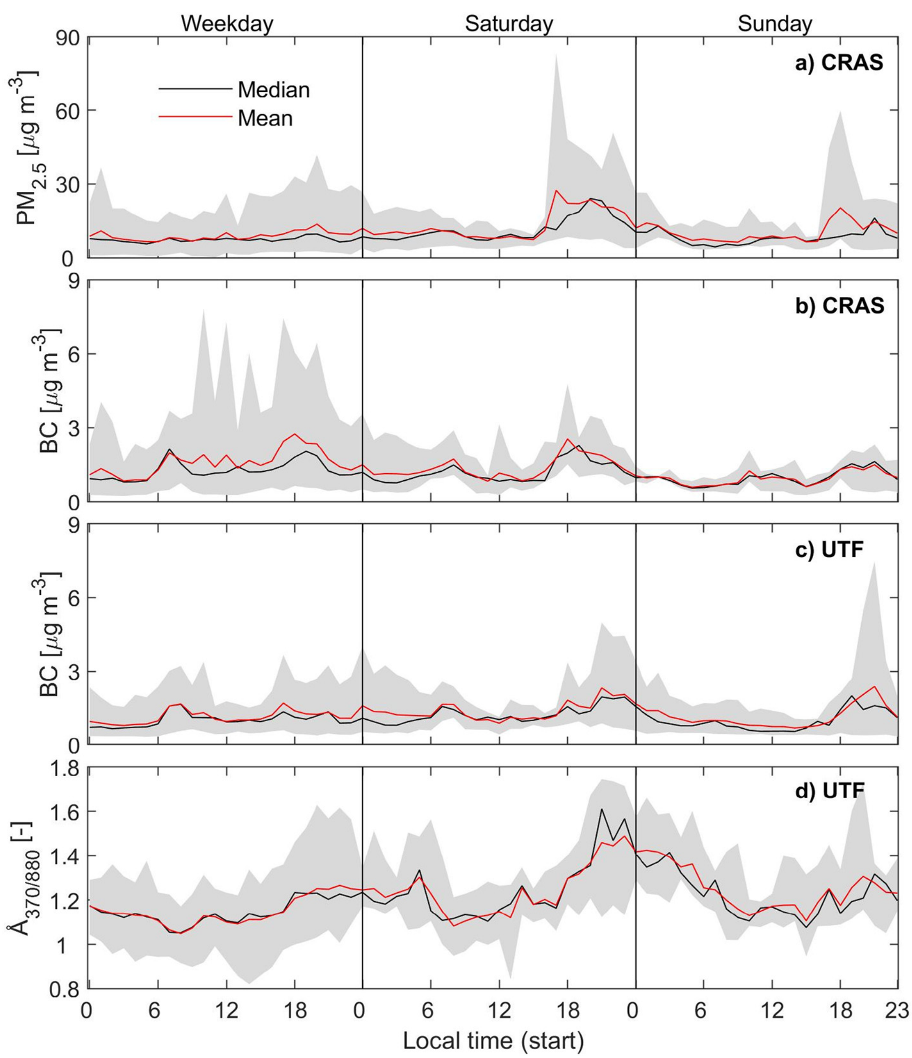

Fig. 3. Mean and median daily cycles of $PM_{2.5}$, BC and $Å_{370/880}$ at the fixed sites for the entire sampling period. The shaded area is delimited by the 5th and 95th percentiles.

As an example of mobile sampling, Fig. 4a illustrates the time series of BC and $PM_{2.5}$ concentrations (10-s) measured on route 1 on 22 June 2017, along with simultaneous measurements at CRAS (1-min). A 10-min intercomparison period (shaded area) in front of the CRAS building showed a very good agreement between instruments (Pearson's correlation coefficient of 0.91). Note that the particulate concentrations varied little at CRAS for the entire sampling period (BC = 1.81 ± 0.89 μg m^{-3}, $PM_{2.5}$ = 8.95 ± 1.48 μg m^{-3}), whereas several spikes occurred in the mobile BC and $PM_{2.5}$ data simultaneously, after the intercomparison period. Peak values varied between 18.14 and 79.34 μg m^{-3} for BC and between 18.50 and 111.21 μg m^{-3} for $PM_{2.5}$. All spikes but one were associated with observations of waste burning along the route, either in vacant lots between houses (Fig. 4b) or in empty land (Fig. 4c). The high pollution episode registered at 17:07 was due to a smoky truck passing by, yielding a high BC/$PM_{2.5}$ ratio (0.75), typical of heavy-duty diesel vehicles with poor engine maintenance (Lau et al., 2015). Note that the BC/$PM_{2.5}$ ratios presented a large range (0.05–0.98) for the waste burning spikes, most likely due to both different materials being burned and combustion phases (flaming or smoldering) (Santoso et al., 2019 and references therein).

The statistical description of mobile BC and $PM_{2.5}$ concentrations (10-s) combining both routes, along with simultaneous measurements at CRAS (1-min), is presented per sampling session in Fig. 5 and for all days together in Table A1 (Suppl. Material). Overall, the mean particulate concentrations were higher (BC: 4.46 vs. 1.58 μg m^{-3}, $PM_{2.5}$: 13.06 vs. 8.93 μg m^{-3}) and more variable (SD: 12.56 vs. 1.03 μg m^{-3} for BC, and 18.87 vs. 4.35 μg m^{-3} for $PM_{2.5}$) for the mobile monitoring than at CRAS. All mobile sampling sessions registered spikes (as illustrated by the high percentiles in Fig. 5a,c), while some transects presented even lower values than those registered at CRAS (Fig. 5a,c vs. b, d). This rapidly varying behavior along with high concentrations is typical of localized emission sources close to the measurement point. After carefully checking our logbooks, we found that these spikes were simultaneously observed with the burning of waste along the route in almost all cases (90%), and just few cases corresponded to diesel trucks emissions, as depicted in Fig. 4. Note that we were able to identify these spikes given that the total traffic was low in the study area, without buses and a low number of trucks (2 ± 1 per route).

Our mobile BC concentrations showed small differences (mean: 4.46 vs. 5.10 μg m^{-3}) when compared to mobile measurements conducted downtown in March–April 2015 (Targino et al., 2016), but $PM_{2.5}$ concentrations were higher in our study (mean: 13.06 vs. 8.43 μg m^{-3}) (Table A1, Suppl. Material). The higher $PM_{2.5}$ concentrations but lower BC/$PM_{2.5}$ ratios in the neighborhood might be explained by the different emission factors, fuel burned and combustion conditions for diesel engines and open waste burning, considering that the main emission sources are traffic (downtown) and waste burning (on the city's outskirts).

The spatial distribution of the aggregated median BC and $PM_{2.5}$ concentrations for all sampling sessions is displayed in Fig. 6 (panels a and b). The degree of spatial variability was assessed by the ratio of maximum to minimum median concentrations observed along the 50–70 m buffers of the monitored routes. This ratio was higher for BC (91.3) than for $PM_{2.5}$ concentrations (4.7). Thus, despite using the median values to represent the spatial distribution, the concentrations and variability were still high for the particulate pollutants, especially for BC. A large spatial heterogeneity of BC and $PM_{2.5}$ is typically reported in traffic environments (e.g., Hankey and Marshall, 2015; Targino et al., 2016; Krecl et al., 2020) due to a combination of various local factors (traffic volume, urban configuration) and meteorology. For example, Krecl et al. (2020) found ratios of 23.3 for BC and 7.4 for $PM_{2.5}$ in an area highly influenced by traffic emissions in downtown Curitiba (Brazil). This means that a neighborhood impacted by the smoke from the burning of MSW can present an even larger heterogeneity for BC than a traffic-impacted environment.

To better understand the presence of spikes along the routes, we also calculated 95th percentile concentrations aggregated in the same buffers (Fig. 6, panels c and d). Pollution peaks were observed along both routes, but the heterogeneity was higher for BC than for $PM_{2.5}$ (ratio of 1139.5 vs. 58.7), similar to the median spatial distribution. Route 1 presented sharper pollution spikes than route 2, in connection with a higher number of vacant lots where waste burning was observed. During every single sampling session, we observed waste and overgrown grass burning in several vacant lots and in the front of seemingly low-income houses on route 1 (Fig. B1, Supplementary Material). Smoke plumes dispersed along that street and over other houses

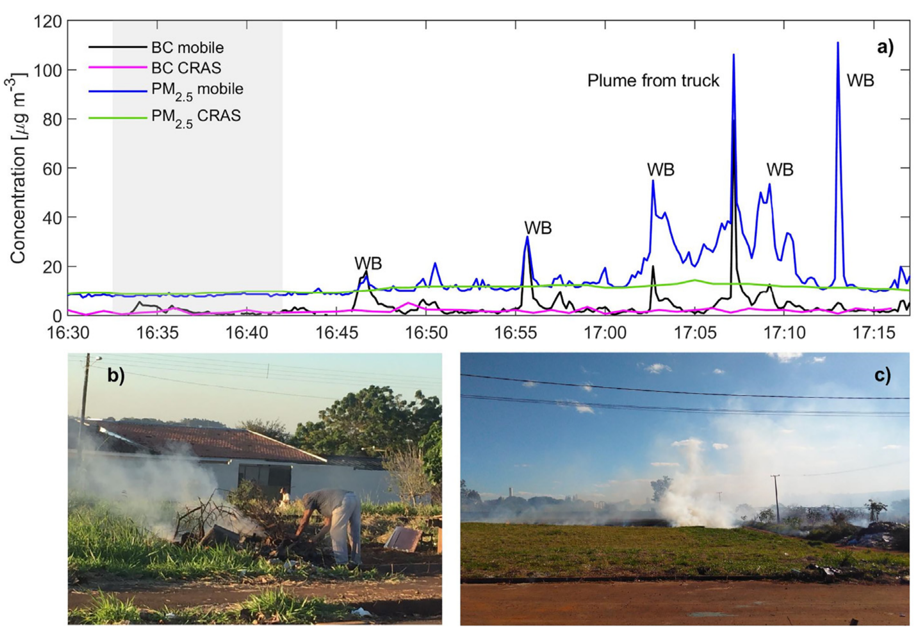

Fig. 4. a) Times series of BC and $PM_{2.5}$ measurements (10-s) conducted on route 1 on 22 June 2017, along with observations of waste burning (WB) and exhaust plume from a truck and concentrations simultaneously measured at CRAS site (1-min). The shaded area indicates the intercomparison period in front of CRAS. b) Burning of MSW in an empty slot between houses. c) Burning of MSW close to the dump site.

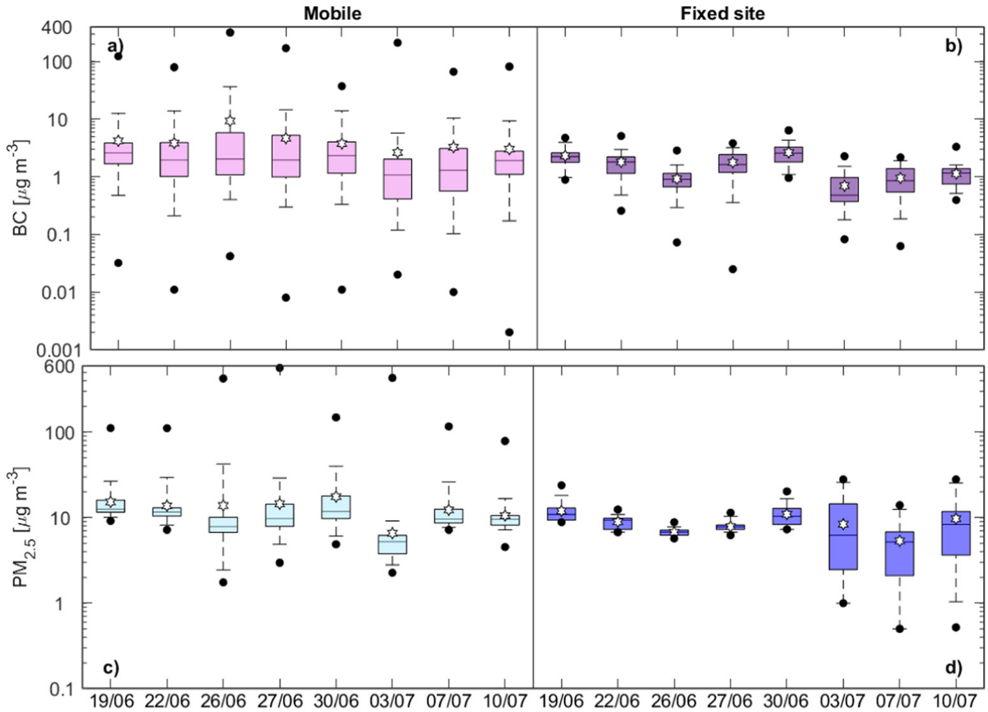

Fig. 5. Boxplots of BC and $PM_{2.5}$ concentrations for mobile (a,c, 10-s data) and fixed-site (b,d, 1-min data) monitoring matching the mobile sampling. Represented are 5th, 25th, 50th, 75th, and 95th percentiles, mean (white star) and extreme (black dots) values.

located several meters away. Note that the CRAS site was less affected by the particulate pollution during the mobile sampling (as shown in Fig. 5), whereas the peak concentrations occurred close to School 2, particularly on Groselha St. (route 1), where we observed smoke and flames from large vacant lots with waste and overgrown vegetation (Fig. B1, Supplementary Material).

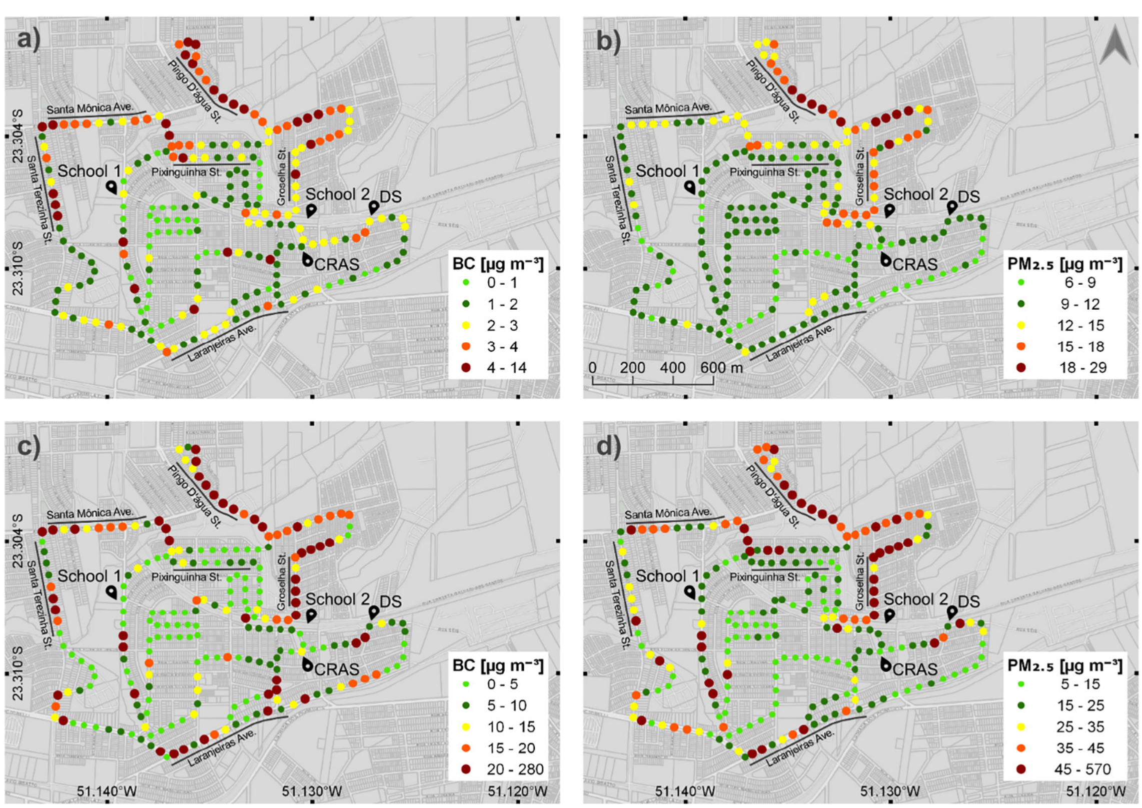

Fig. 6. Spatial distribution of median (a, b) and 95th percentile (c, d) aggregated concentrations.

3.3. Public opinion survey

The number of students that completed the survey was 399 (School 1: 212; School 2: 187), with an excellent response rate for all questions (lowest 99% for Question 7). For each question, a detailed summary of the responses per school and shift is provided in the Supplementary Material (Figs. C2–C10). Note that because students must attend public school in the neighborhood where they live, we can assume that their responses apply to our study area.

Considering all responses together, the students identified several environmental problems connected to waste management in their neighborhood: people throwing garbage in streets (85%) and burning waste (85%), and the presence of illegal dump sites (70%) (Question 1). In Londrina, it is estimated that there are 300 points of illegal waste disposal (CMTU, 2020) and the occurrence of waste burning was high during our mobile monitoring, particularly on route 1 (Section 3.2).

The survey revealed that for 76% of the students, the population disposes of their waste by using the door-to-door municipal collection services. For 27% of the children, the final destination of waste is burning (Question 2). Sorting waste for collection is not a strong habit in the study area, because 61% of the students believe that few people sort their household waste, whereas only 6% believe that everyone practices this activity (Question 3). A large fraction (87%) of respondents has already seen open waste burning close to their houses, and this occurs very frequently (one to seven days per week) (Question 4), according to 68% of the children (Question 5).

More than 60% of the students have seen plastics, paper, leaves/branches, sofas and tires being burned in empty lots in their neighborhood. Burning of wire cables and grass was identified by 50% and 44% of the respondents, respectively. We frequently observed these illegal practices close to our university campus (UTF) to recover copper by unauthorized waste pickers and to clean lots with overgrown grass (Figs. C11 and C12, Supplementary Material). The open-ended question about other types of burned waste was answered by 100 students (Question 6), who identified clothes/shoes (28%), old furniture (17%, mainly matrasses and beds), and organic and non-recyclable waste (9%) as the main waste groups. Other groups that were mentioned at least by five respondents were in the following order: car wreckage, wood pieces, rubber/styrofoam, electronic waste (TV, mobile phones) and construction debris. Note that several items identified by the students as burned material should have been collected by the municipal service (organic and non-recyclable wastes), picked up by the cooperatives for recycling (plastics, paper) or discarded in the voluntary delivery points (furniture, wood pieces, green residues and construction debris). Tires should have been disposed of through reverse logistics. However, this operation has been proved unsuccessful in Brazil (Milanez and Bürhs, 2009).

Almost all students believed that waste burning affects their heath (95%) and the environment (96%) (Questions 7 and 8). The students answered that the main motivation for burning waste are that people are not patient to wait for the collection service (66%), burning waste is a habit (54%), and people do not know that this activity is harmful to their health (41%) (Question 9). Even though organic and non-recyclable waste is collected three times per week, the warm climate (which speeds up the organic decomposition, creates odor nuisance and attracts insects), household layout (inappropriate space for waste storage), and animals scavenging through the garbage might explain why people want to dispose of their waste rapidly. In the case of recyclable material, the collection occurs once per week resulting in a large volume of materials stored at home. Any failure in the service (change in collection times and/or no service that week) might imply longer wait for the final disposal of the waste.

In the open-ended section of Question 9, 66 students added two other important reasons for waste burning: people do not mind doing this (39% of the respondents) and are too lazy to sort waste (32% of the respondents). As previously commented, once waste accumulates at home, some people might either dump it in vacant lots on top of pre-existing fire or burn it in their backyards. Other highlighted motives were problems with waste collection services such as the truck not collecting certain items (most likely because they are not sorted, or they are not organic) or residents not having time/resources to transport construction debris, electronics and wood pieces to the voluntary delivery sites, or exceeding the volume of 1 m^3 per month and household. Some students also pointed out that people start fire as a form of vandalism.

3.4. Study strengths and limitations

The biggest strength of this study is the characterization of the airborne particulate matter (BC and $PM_{2.5}$) in a residential neighborhood frequently impacted by municipal solid waste burning. As far as we know, this is the first work in South America that particularly focuses on this underestimated emission source, typical of urban areas in developing countries. Another strength is the choice of the pollutants: $PM_{2.5}$ (a frequently used indicator of particulate exposure with well-documented health effects) and BC (largely associated with incomplete combustion and a potent short-lived climate pollutant).

This field campaign, and particularly the mobile monitoring, occurred over a limited number of sampling days in the winter season. Thus, further measurements covering different seasons are needed for capturing changes in emission patterns, and dispersion conditions under other meteorological settings. Other chemical species, highly toxic and typically emitted by MSW combustion, such as PAH and PCDD/F, should be included in a future campaign. Mobile measurements with microaethalometers operating at multiple wavelengths (including UV) might be an asset to separate the biomass burning contribution from traffic emissions, regardless of the traffic volume.

4. Summary and conclusions

We combined fixed-site and mobile BC and $PM_{2.5}$ measurements to characterize the particulate pollution in a residential neighborhood impacted by waste burning smoke. We supplemented the air pollution data with a public opinion survey to understand the population's behavior in relation to the disposal and burning of MSW in the study area.

On average, particulate pollution concentration at the residential fixed sites were lower than downtown for BC, but higher for $PM_{2.5}$. According to the analysis of the absorption Ångström exponent, the residential area was impacted by a combination of traffic and biomass burning emissions during 66% of the sampling time, which might also explain the increase in $PM_{2.5}$ concentrations. Different from traffic-impacted environments, $PM_{2.5}$ concentrations were highest in the late afternoon/evening of weekends, when biomass burning dominated the emission sources ($Å_{370/880} > 1.50$). While fixed-site sampling was conducted over a longer period, allowing us to identify the weekly pollution pattern, mobile monitoring covered a larger area in a short-time frame and much closer to the waste burning sites. By comparing our mobile measurements with a similar study conducted downtown, we concluded that: i) mean $PM_{2.5}$ concentrations were higher in our study but BC/$PM_{2.5}$ ratios were lower, most likely due to differences in emission factors, fuel burned and combustion conditions for open waste burning (neighborhood) and diesel engines (more abundant downtown), and ii) the larger spatial heterogeneity in the median and 95th percentile concentrations was linked to the occurrence of fast and sharp spikes, associated mostly with the burning of MSW events, according to our logbooks.

The public opinion survey showed that the respondents identified poor waste management (people throwing garbage in streets, waste burning, among others) as an environmental problem in their neighborhood. Most respondents have seen these practices close to their home very frequently. The most commonly burned materials were plastics, paper, branches/leaves that could otherwise have been managed by the city's collection service. The vast majority of respondents knows that burning waste is harmful to their health and the environment, and that the waste disposal habit of the population and lack of patience to wait for the collection services would be the main reason for burning. Thus, it can be concluded that the study area is significantly impacted by the burning of MSW, and that despite the regular waste collection service, the population often opts for burning, either out of force of habit or for their convenience.

Even though almost all students were aware of the deleterious effects of waste burning on their health and the environment, 41% of them reported that the residents who burn waste do not know the associated health risks. Thus, we suggest raising awareness of the population through public campaigns at local level. We also identified opportunities for environmental education initiatives with residents in relation to waste segregation, since sorting MSW at home is crucial to reduce cross-contamination and maintain the quality of recycling materials. Moreover, proper segregation reduces both the amount of waste at home and the possibilities of burning as a final method of disposal. Municipal programs in suburban neighborhoods could be implemented on a regular basis aiming to increase waste segregation through economic incentives, and to also expand the composting initiative to other areas of the city. We also emphasize that two voluntary delivery sites (green residues, furniture, construction debris) and the few sites for reverse logistics are scarce for the size of the population, and the allowed volume for waste disposal should be revised. Other solutions should be implemented for residents who either cannot afford or do not have time to transport their waste to those points. Finally, regular inspection by the authorities is also essential to eliminate/reduce this practice, and more budget and staff should be allocated to this end to increase enforcement.

Brazil has legislation and a national strategy to develop a solid waste management system, which all cities should adhere to. However, the implementation and enforcement are often lacking. Even though this study was carried out in a relatively small neighborhood compared to the dimensions of this continent-sized country, it paves the road for further studies in other parts of Brazil (or South America) where MSW collection systems are yet to be implemented or proven to be inefficient, leading to the open burning of household waste. Our combined methodology (fixed-site and mobile monitoring complemented by public opinion survey) proved to be a time- and cost-effective approach to map out ambient air pollutant concentrations and identify undesirable waste practices. This approach can be applied to other cities around the world where the open burning of MSW is identified as an environmental problem.

Our pioneering work reveals that this topic is very relevant, given the high particulate pollution levels in the areas near the fire spots. In closing, after searching for similar studies in other countries we identified a large scientific gap in this underresearched field. We focused on the spatio-temporal characterization of two particulate pollutants (BC and $PM_{2.5}$), but other toxic substances should be targeted in further research as well as the personal exposure of local residents.

CRediT authorship contribution statement

Patricia Krecl: Conceptualization, Methodology, Software, Formal analysis, Investigation, Data curation, Visualization, Writing - original draft, Writing - review & editing. **Caroline Hatada de Lima:** Formal analysis, Investigation, Data curation, Visualization. **Tatiane Cristina Dal Bosco:** Methodology, Formal analysis. **Admir Créso Targino:** Investigation, Resources, Funding acquisition, Writing - original draft, Writing - review & editing. **Elizabeth Mie Hashimoto:** Methodology, Formal analysis, Data curation. **Gabriel Yoshikazu Oukawa:** Software, Formal analysis, Visualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2020.142736.

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