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Personal exposure to $PM_{2.5}$ emitted from typical anthropogenic sources in southern West Africa: chemical characteristics and associated health risks

Hongmei Xu[1,2,3,4], Jean-François Léon[2], Cathy Liousse[2], Benjamin Guinot[2], Véronique Yoboué[5], Aristide Barthélémy Akpo[6], Jacques Adon[2], Kin Fai Ho[7], Steven Sai Hang Ho[8], Lijuan Li[3], Eric Gardrat[2], Zhenxing Shen[1], and Junji Cao[3]

[1]Department of Environmental Science and Engineering, Xi'an Jiaotong University, Xi'an, China
[2]Laboratoire d'Aérologie, Université de Toulouse, CNRS, Toulouse, France
[3]SKLLQG, Key Lab of Aerosol Chemistry & Physics,Institute of Earth Environment, Chinese Academy of Sciences, Xi'an, China
[4]Collaborative Innovation Center of Atmospheric Environment and Equipment Technology, Jiangsu Key Laboratory of Atmospheric Environment Monitoring and Pollution Control (AEMPC), Nanjing University of Information Science & Technology, Nanjing, China
[5]Laboratoire de Physique de l'Atmosphère, Université Félix Houphouët-Boigny, Abidjan, Côte d'Ivoire
[6]Laboratoire de Physique du Rayonnement, Université Abomey-Calavi, Abomey-Calavi, Benin
[7]JC School of Public Health and Primary Care, The Chinese University of Hong Kong, Hong Kong, China
[8]Divison of Atmospheric Sciences, Desert Research Institute, Reno, NV 89512, USA

Correspondence: Hongmei Xu (xuhongmei@xjtu.edu.cn) and Cathy Liousse (cathy.leal-liousse@aero.obs-mip.fr)

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Abstract. Urbanization is an issue that is strongly emerging in southern West Africa (sWA). There is a lack of full understanding on chemical compositions and personal exposure levels to fine particulate matter (hereafter defined as PE $PM_{2.5}$) and its health risks related to various anthropogenic sources in this region. In this study, PE $PM_{2.5}$ was studied in dry (January) and wet (July) seasons of 2016 for the first time to characterize the contributions of a domestic fire site (DF) to the exposure of women and a waste burning site (WB) to that of students in Abidjan, Côte d'Ivoire, and a motorcycle traffic site (MT) to that of drivers in Cotonou, Benin.

The average PE $PM_{2.5}$ mass concentrations were 331.7 ± 190.7, 356.9 ± 71.9 and $242.8 \pm 67.6\ \mu g\ m^{-3}$ at DF, WB and MT sites for women, students and drivers, which were 2.4, 10.3 and 6.4 times the ambient $PM_{2.5}$ concentrations, respectively. Elevated PE $PM_{2.5}$ levels in the dry season were found at DF ($358.8 \pm 100.5\ \mu g\ m^{-3}$), WB ($494.3 \pm 15.8\ \mu g\ m^{-3}$) and MT ($335.1 \pm 72.1\ \mu g\ m^{-3}$) sites, on average 15 % higher than that at DF and 55 % higher at both WB and MT sites in the wet season. The seasonal variations were attributed to emission sources, meteorological factors and personal activities. In addition, the results show that geological material (35.8 %, 46.0 % and 42.4 %) and organic matter (34.1 %, 23.3 % and 24.9 %) were the major components of PE $PM_{2.5}$ at DF, WB and MT sites. It is worth noting that the contribution of heavy metals was higher at WB (1.0 %) than at DF (0.7 %) and MT (0.4 %) sites, strongly influenced by waste burning emission. This results in the highest non-cancer risks of heavy metals to students, 5.1 and 4.8 times the values for women and drivers, respectively.

By conducting organic speciation, fingerprints were used to access the exposure and identify the source contributions from typical local anthropogenic sources. The women's exposure concentration to particulate polycyclic aromatic hydrocarbons (PAHs) at DF ($77.4 \pm 47.9\ ng\ m^{-3}$) was 1.6 and 2.1 times, respectively, that of students at WB ($49.9 \pm 30.7\ ng\ m^{-3}$) and of drivers at MT ($37.0 \pm 7.4\ ng\ m^{-3}$). This can be associated with the higher contributions from solid fuels' burning and meat grilling activities to women, resulting in a level 5 times in exceedance of the cancer risk safety

threshold (1×10^{-6}). Phthalate esters (PAEs), commonly used as plasticizers in products, were in high levels in the student exposure $PM_{2.5}$ samples ($1380.4 \pm 335.2\,ng\,m^{-3}$), owing to obvious waste burning activities nearby. The drivers' exposures to fossil fuel combustion markers of hopanes in PE $PM_{2.5}$ at MT ($50.9 \pm 7.9\,ng\,m^{-3}$) was 3.0–3.3 times those for women at DF ($17.1 \pm 6.4\,ng\,m^{-3}$) and students at WB ($15.6 \pm 6.1\,ng\,m^{-3}$).

Overall, the current study shows that wood combustion, waste burning, fugitive dust and motor vehicle emissions were the dominant sources of PE $PM_{2.5}$ and mainly contributed to its toxicities. The exposure to the heavy metals Pb and Mn caused high non-cancer risks to students at WB, while the severe cancer risk of PAHs was found for women at DF via inhalation. The result of this study provides original data, initial perspective of $PM_{2.5}$ personal exposure and health risk assessment in the developing areas. The information encourages the governments to improve the air quality and living standards of residents in this region.

1 Introduction

The southern West Africa (sWA) region is experiencing an economic upturn. Anthropogenic emissions of air pollutants have been increasing in the last few years, leading to poor air quality in the region (IMF, 2017; Norman et al., 2007). Fine particulate matter ($PM_{2.5}$ with equivalent aerodynamic diameters $\leq 2.5\,\mu m$) is one of the major concerns of international organizations and the public because of its high health impacts related to personal exposure (Bruce et al., 2000; Chen et al., 2013; Owili et al., 2017). Owili et al. (2017) found that four types of ambient $PM_{2.5}$, including mineral dust, anthropogenic pollutants, biomass burning and mixed aerosols are significantly associated with under-five and maternal mortality in Africa. However, studies on $PM_{2.5}$, especially tests of direct personal exposure to fine particulate matter (hereafter defined as PE $PM_{2.5}$) (nonstationary sampling) and its health assessment, are very limited in these low gross domestic product (GDP) countries.

Since the 1990s, several international campaigns have been performed in Africa. Some of them were mainly focused on the particles or aerosols, such as DECAFE (Lacaux et al., 1995), EXPRESSO (Delmas et al. 1999; Ruellan et al., 1999), SAFARI-1992 (Lindesay et al., 1996), SAFARI-2000 (Swap et al., 2002), AMMA (Léon et al., 2009; Liousse et al., 2010; Marticorena et al., 2010) and INDAAF (Ouafo-Leumbe et al., 2017). In fact, Africa has the largest production of mineral dust particles from the Sahara and unpaved road surfaces (Laurent et al., 2008; Marticorena et al., 2010; Reeves et al., 2010) and carbonaceous aerosols originated from wild fires (mainly savannah fires) (Capes et al., 2008; Gaudichet et al., 1995) in the whole world. Therefore, these campaigns were more biased toward the natural sources of aerosols in Africa. In previous literature, the major contributions to the aerosol chemistry in northern Benin in the dry season were shown to be dust (26 %–59 %), primary organic matter (POC; 30 %–59 %), elemental carbon (EC; 5 %–9 %) and water-soluble inorganic ions (3 %–5 %) (Ouafo-Leumbe et al., 2017). Liousse et al. (2014) showed the increase of the relative importance of particulate emissions from domestic fires and fossil fuel combustion in Africa. Uncertainty has been raised by the residents who live in urban areas as they have concerns about the health impacts of air quality. However, works on PE $PM_{2.5}$ emitted from the typical anthropogenic sources in the emerging cities in Africa are still scarce.

The main anthropogenic emission sources of $PM_{2.5}$ in sWA include domestic wood burning, fossil fuel combustion, unregulated traffic and industries, waste burning and road dust. An ongoing project, Dynamics–Aerosol–Chemistry–Cloud Interactions in West Africa (Africa-DACCIWA), aims to quantify the influences of anthropogenic and natural emissions on the atmospheric pollutant composition over southern West Africa and to assess their impacts on human health, ecosystems and agricultural productivity. The information will be gathered and discussed with policymakers, scientists, operational centers, students and the general public. The current work in the framework of the Work Package 2 "Air Pollution and Health" of DACCIWA aims to link emission sources, air pollution and health impacts over representative differentiated urban sources: domestic fires and waste burning in Abidjan (Côte d'Ivoire) and two-wheel vehicle emission in Cotonou (Benin) for different groups of the population.

Smoking meat (e.g., fish and pork) by biomass fuels (wood) is an important diet pattern for residents of coastal countries in the sWA area. Many female workers are engaged in roasting activities without any personal health protection. They are directly exposed to the high $PM_{2.5}$ emissions from wood burning and smoking meat, which could cause serious health issues. In addition, urbanization has led to explosive population growth and rural depopulation in sWA, generating a huge amount of urban domestic waste. The biggest landfill in Abidjan involved in this study receives more than 1 000 000 t waste per year (Adjiri et al., 2015). Without any processing capacity or appropriate treatment methods, a large amount of air pollutants was thus emitted from the combustion and stacking of waste. Such a phenomenon damages the living environment and harms the residents' health (especially for children) in Abidjan (UNEP, 2015). Moreover, in most low-GDP countries, motorbike taxis are a major mode of local transportation (Assamoi and Liousse, 2010). In Benin, motorbike taxi drivers (mainly male) represented ~ 2.5 % of the total population in 2002 (Lawin et al., 2016). Due to long daily working hours, the drivers are exposed to traffic-related $PM_{2.5}$ emissions over years.

Major chemical components in $PM_{2.5}$ like OC, EC and ions not only have a strong impact on $PM_{2.5}$ physicochemical

Figure 1. Location of the sampling sites (white square) within the cities. **(a)** Domestic fire site (DF) at the Yopougon-Lubafrique market in Abidjan, **(b)** waste burning site (WB) at the landfill of Akeoudo in Abidjan and **(c)** motorcycle traffic site (MT) in the Dantokpa area in Cotonou.

characteristics but also cause health risks. Typical trace toxic chemicals in $PM_{2.5}$, such as heavy metals and polycyclic aromatic hydrocarbons (PAHs), cause various types of health damage to humans (Cao et al., 2012; WHO, 1998; Xu et al., 2015). For instance, Pb is a neurodevelopmental metal which affects children's health and mental development (U.S. EPA, 2012; Xu et al., 2017). Several PAHs are teratogenic and carcinogenic for humans (Tang et al., 2008). Up to now, only a few studies have investigated chemical compositions of PE $PM_{2.5}$ samples, and little is known regarding the sources and health risks in the sWA region. This poses a challenge for the formulation of strategies to mitigate $PM_{2.5}$ pollution and its health effects in this area.

Therefore, our study relies on the portative device sampling $PM_{2.5}$ PE samples in the sWA area in 2016. Study objectives are (1) to characterize the PE $PM_{2.5}$ from different typical local anthropogenic sources using chemical component and $PM_{2.5}$ mass balance analysis, (2) to identify potential pollution sources to different exposed populations from fingerprints of organic markers and (3) to evaluate the PE $PM_{2.5}$ health risks using the United States Environmental Protection Agency (U.S. EPA) health risk assessment model. This information offers scientific understanding of the PE $PM_{2.5}$ in sWA and arouses the government's attention to protect residents' health from various anthropogenic sources.

2 Materials and methods

2.1 Site description and participant selection

PE $PM_{2.5}$) filter samples were collected using portative devices in unique source-dominated environments for different target groups of humans, including a domestic fire site (DF) for women and a waste burning site (WB) for students in Abidjan, Côte d'Ivoire, and a motorcycle traffic site (MT) for drivers in Cotonou, Benin (Fig. 1). Abidjan (5°20′ N, 4°1′ W) is the economic capital of Côte d'Ivoire, with 6.5 million inhabitants in 2016. It is characterized by the highest level of industrialization and urbanization in the sWA area. Cotonou (6°21′ N, 2°26′ W) is the largest city and economic center of Benin, with about 1.5 million inhabitants in 2016. Both the cities experience a tropical wet and dry mixed climate, with relatively constant ambient temperatures (in a range of 24–30 °C) and an average of relative humidity (RH) > 80 % throughout a year.

The DF site in Abidjan is located in the market of Yopougon-Lubafrique (5°19.7′ N, 4°6.4′ W), which is a large courtyard with about 25 fireplaces (Fig. 2). The major fuel used is essentially hevea wood, which is a kind of local rubber tree. Several female adult workers are employed for grilling meat and/or roasting peanuts from 06:00 to 15:00 UTC (working hours) in the working day. In this study, we selected two healthy and nonsmoking female workers (an average age of 32.5 years old) to measure personal exposure to $PM_{2.5}$ from domestic fires and related sources such as grilling (Fig. 2). The WB site in Abidjan is near the public landfill of Akouédo (5°21.2′ N, 3°56.3′ W), which has received all the waste collected from Abidjan for the past 50 years (Fig. 2). We selected two healthy and nonsmoking primary school students (an average age of 11 years old) who live and study next to the WB site (within 100 m straight-line distance) to determine the personal exposure features to $PM_{2.5}$ from waste burning (spontaneous combustion at high ambient temperatures and irregular combustion by the landfill workers) emissions at landfill and other daily sources. Lastly, the MT site in Cotonou is located in the Dantokpa area (6°22.1′ N, 2°25.9′ E), one of the biggest markets in western Africa (Fig. 2). It is largely dominated by a mass of emissions from motorcycle traffic (two-wheeled vehicle powered by petrol, also named *zemidjan* in the local language) and a small quantity of other motor vehicles. We chose two healthy and nonsmoking male motorcycle drivers (an average age of 50 years old) to survey $PM_{2.5}$ personal exposure from motorcycle emission and related sources (such as road dust).

Two women (woman A and B) involved in this study at DF were both in charge of cooking at home, using charcoal

and butane gas as fuel (Fig. S1a, b, c in the Supplement), and daily household cleaning. One student participator (student A, boy, 8 years old) at WB was not involved in cooking activities at home (energy source for cooking is charcoal and liquefied petroleum gas, LPG) (Fig. S1a, c), but another student (student B, girl, 14 years old) is usually responsible for household cooking using solid fuels (wood) (Fig. S1d). Two motorcycle drivers (driver A and B) at MT work for a local motorcycle operation company whose working time is usually from 06:30 to 10:30, 12:00 to 17:00 and 18:30 to 21:00 UTC. They were on the road almost all the working hours and returned home for meals. They did not participate in any cooking at home (energy source for cooking is charcoal) (Fig. S1a).

2.2 Personal exposure to $PM_{2.5}$ sample collection and QA / QC

Twelve-hour time-integrated (daytime: 07:30 to 19:30 UTC; nighttime: 19:30 to 07:30 the next day UTC) PE $PM_{2.5}$ samples were collected in two major southwestern African cities (Fig. 1) during the dry season (from 6 to 11 January) and wet season (from 5 to 10 July) in 2016. PE $PM_{2.5}$ sampling was conducted for three consecutive days synchronously using the PEM (Personal Environmental Monitor) sampling devices with a SKC pump (SKC Inc., Fullerton, CA, USA) at a flow rate of 10 lpm (liters per minute). The PEM $PM_{2.5}$ sampling heads were worn in the breathing zone (within 20 cm of the nose and mouth area) of participants in this study. $PM_{2.5}$ was collected on 37 mm pre-baked quartz filters (800 °C, 3 h, QM/A®, Whatman Inc., UK). A total of 72 PE samples, including 24 samples (12 pairs of diurnal samples) for women at DF, 24 (12 pairs) for students at WB and 24 (12 pairs) for drivers at MT, were collected in this study. Moreover, 12 PE $PM_{2.5}$ field blanks (one field blank for each participant in one season collected on the second day of the three consecutive sampling days) were obtained as well.

In order to verify the comparability of PE samples and data caused by nonidentical sampling devices, 10 pairs of $PM_{2.5}$ samples were synchronously collected by two sets of actual PEMs with SKC pumps. The comparison results show a significant correlation between the $PM_{2.5}$ mass concentrations obtained from two sampling devices ($y = 0.986x + 0.189$, $R^2 = 0.974$, $P < 0.0001$). Identical membrane type (quartz fiber) and analytical treatments were applied in this study. After sampling, the filter samples were placed in Petri dishes, sealed with parafilm and stored in a freezer at −20 °C to prevent loss of mass through volatilization prior to analysis. Blank values from blank filter samples were used to account for any artifacts caused by gas absorption and subtract the background $PM_{2.5}$ and chemical composition concentrations in this area.

The meteorological observations during the dry (December 2015 to March 2016) and wet (April to July 2016) seasons at the sampling sites were shown in Table 1. The meteorological data were retrieved from the NOAA Global Surface Summary of the Day I (GSOD) at the airports of each cities, namely Félix Houphouët-Boigny Airport (Abidjan) and Cardinal Bernadin Gantin International Airport (Benin).

Table 1. Meteorological parameters of the studied two cities during the dry (December 2015 to March 2016) and wet (April to July 2016) seasons.

	Season	Abidjan	Cotonou
Mean daily air temperature (°C)	Dry	28.0	28.3
	Wet	27.5	27.7
Total rainfall (mm)	Dry	268	92
	Wet	626	558
Mean wind speed ($m\,s^{-1}$)	Dry	3.0	3.0
	Wet	3.4	4.3

2.3 $PM_{2.5}$ gravimetric and chemical analysis

PE $PM_{2.5}$ filter samples were analyzed gravimetrically for mass concentrations with a high-precision electronic microbalance (Sartorius MC21S, Germany) at Laboratoire d'Aérologie (Toulouse, France) before and after sampling in the weighing room after equilibration at 20–23 °C and RH of 35 %–45 % for at least 24 h. The absolute errors between replicate weights were less than 0.015 mg for blank filters and 0.020 mg for sampled filters.

Total carbon (TC) was determined on 0.5 cm^2 punch-out of the filters using a carbon analyzer (Ströhlein Coulomat 702C, Germany) at the Observatoire Midi-Pyrenees (OMP; Toulouse, France). The quartz filter samples were subjected to a thermal pretreatment step (kept at 60 °C for 20 min) in order to remove the volatile organic compounds (VOCs) and eliminate water vapor. Subsequently, the filters were combusted at 1200 °C under O_2 and detected as CO_2 in the carbon analyzer. EC was obtained using a two-step thermal method: step 1 consisted of a pre-combustion at 340 °C under O_2 for 2 h in order to remove OC. Step 2 consisted of the oxidation of the remaining EC at 1200 °C under O_2. The difference (TC − EC) yielded the OC concentration (Benchrif et al., 2018; Cachier et al., 2005).

To extract the water-soluble inorganic ions from the quartz filters, a quarter of the filter was placed in separate 15 mL vials containing 10 mL distilled–deionized water (18.2 MΩ cm resistivity). The vials were placed in an ultrasonic water bath and shaken with a mechanical shaker for 45 min (15 min, three times) to extract the ions. The extracts were filtered through 0.45 µm pore size microporous membranes. After that, three anions (Cl^-, NO_3^- and SO_4^{2-}) and five cations (Na^+, NH_4^+, K^+, Mg^{2+} and Ca^{2+}) in aqueous extracts of the filters were determined using an ion chromatograph (IC) analyzer (Dionex-600, Dionex, Sunnyvale, CA, USA), which is equipped with an AS11-HC anion column

Figure 2. Pictures showing the sampling sites and corresponding participants: **(a)** women at DF, **(b)** students at WB and **(c)** drivers at MT.

and a CS12 cation column for separation. Details of the IC measurement method are described in Bahino et al. (2018) and Cachier et al. (2005).

One element of Fe (representing earth's crust emission) and 10 heavy metals (i.e., V, Cr, Mn, Co, Ni, Cu, Zn, Sb, Ba and Pb) in PE $PM_{2.5}$ samples were determined using Energy Dispersive X-Ray Fluorescence (ED-XRF) spectrometry (the PANalytical Epsilon 5 ED-XRF analyzer, PANalytical B.V., the Netherlands) with a quarter of the filter. The relative errors for all measured elements were $< 6\,\%$ between the NIST Standard Reference Material (SRM) 2783 and our ED-XRF results, which is well within the required range of error, demonstrating the accuracy of method. Replicate analysis of one quartz-fiber filter sample (five times) yielded an analytical precision between 5.2 % and 13.9 %. Details of the ED-XRF measurements are shown in Brouwer (2003) and Xu et al. (2012).

Aliquot punches (0.1–1.0 cm^2) from the quartz filters were used to quantify organic compounds, including PAHs, phthalate esters (PAEs) and hopanes (details of target organic species and their abbreviations are shown in Table 5) using an in-injection port thermal desorption gas chromatography–mass spectrometry (TD-GC/MS) method. The approach has the advantages of a shorter sample preparation time ($<$ 1 min), minimizing contamination from solvent impurities, and a higher sensitivity, compared with the traditional solvent extraction GC/MS method. The detailed analytical procedures have been reported in previous publications (Ho and Yu, 2004; Ho et al., 2008, 2011; Xu et al., 2013, 2016a). The results of the blank analyses showed only traces of contamination ($< 5.0\,\%$) of PE $PM_{2.5}$ sample concentrations.

2.4 Health risk assessment model

A number of heavy metals and toxic organic species are associated with negative PE health effects (Škrbic et al., 2016; Val et al., 2013; J. Z. Wang et al., 2017; Xu et al., 2018a). In this study, four heavy metals (Mn, Ni, Zn and Pb) and all mea-

Table 2. Definitions and recommended values of the parameters in Eqs. (1)–(4) in this study.

Parameter	Definition (unit)	Value used in this study (reference)
D	average daily exposure dose ($mg\,kg^{-1}\,day^{-1}$)	/
C	heavy metal concentrations in equations ($ng\,m^{-3}$)	/
R	inhalation rate, air volume a participant inhaled each day ($m^3\,day^{-1}$)	16.0 for women and drivers; 15.2 for students (U.S. EPA, 2011)
EF	exposure frequency ($day\,year^{-1}$)	130 for women and drivers (half working days); 182 for students (half year)
ED	exposure duration (year)	30 for women and drivers (working years); 15 for students (before going to high school)
BW	body weight (kg)	62.5 for women*; 37.5 for students*;85.0 for drivers*
AT	averaging time (day)	30 or 15×365 (non-cancer); 70×365 (cancer)
cf	conversion factor ($kg\,mg^{-1}$)	10^{-6}
HQ	hazard quotient	/
RfD	reference dose, estimated as the maximum permissible risk to humans by daily exposure ($mg\,kg^{-1}\,day^{-1}$)	Table 3
HI	hazard index	/
ILCR	incremental lifetime cancer risk (ILCR)	/
CSF	cancer slope factor $(mg\,kg^{-1}\,day^{-1})^{-1}$	Table 3
$[BaP]_{eq}$	equivalent BaP toxicity concentration ($ng\,m^{-3}$)	/
C_i	individual PAH species concentration ($ng\,m^{-3}$) (i means target PAH species)	/
TEF_i	toxicity equivalency factor of each target PAH compound (i means target PAH species)	Nisbet and Lagoy (1992)

* Measured in this study.

sured PAH and PAE species in PE $PM_{2.5}$ were selected to determine the PE inhalation health risks (Xu et al., 2018a). The heavy metal non-carcinogenic risks and toxic organics carcinogenic risks of $PM_{2.5}$ via inhalation were calculated according to the U.S. EPA health risk assessment model (U.S. EPA, 2004, 2011). The average daily exposure dose (D) via inhalation was estimated to assess the risk using Eq. (1) as follows:

$$D = (C \times R \times \mathrm{EF} \times \mathrm{ED} \times \mathrm{cf})/(\mathrm{BW} \times \mathrm{AT}). \quad (1)$$

The definitions and recommended values of parameters are shown in Table 2.

A hazard quotient (HQ) for the non-cancer risk of heavy metals in PE $PM_{2.5}$ samples can be obtained from Eq. (2):

$$\mathrm{HQ} = D/\mathrm{RfD}. \quad (2)$$

The threshold value of RfD indicates whether there is an adverse health effect during a certain period. The hazard index (HI) can be obtained by summing up the individual HQs to estimate the total non-cancer risks. If $\mathrm{HI} < 1$, then a non-carcinogenic effect is impossible; if $\mathrm{HI} \geq 1$, an adverse health effect is likely to appear (Hu et al., 2012).

The incremental lifetime cancer risk (ILCR) of PAHs and PAEs in PE $PM_{2.5}$ samples can be calculated by multiplying the cancer slope factor (CSF) of PAHs and PAEs by D using Eq. (3):

$$\mathrm{ILCR} = D \times \mathrm{CSF}. \quad (3)$$

For cancer risk, the value of 1×10^{-6} is internationally accepted as the precautionary or threshold value above which the risk is unacceptable (Jedrychowski et al., 2015).

It is worth noting that, among the 19 PAHs, benzo[a]pyrene (BaP) has been used as an indicator of

PAH carcinogenicity (Wang et al., 2006). The carcinogenic health risk of PAH species can be assessed by $[BaP]_{eq}$ instead (Yassaa et al., 2001) using Eq. (4):

$$\Sigma[BaP]_{eq} = \Sigma(C_i \times TEF_i). \quad (4)$$

Additionally, the carcinogenic risk for PAEs was assessed by bis(2-ethylhexyl) phthalate (DEHP), which is identified as possibly carcinogenic to humans by the International Agency for Research on Cancer (IARC, 1982; Li et al., 2016). The definitions and recommended values of the parameters in Eqs. (2)–(4) are also shown in Tables 2 and 3.

2.5 Questionnaire and time-activity diary

A questionnaire (Supplement A–C) and time-activity diary (Supplement D) were collected from each participant during the sampling period, respectively, to fully grasp the basic information, potential personal exposure sources and activities of participants. In the questionnaire, each participant was asked for personal information, family status, dermatological and asthma issues, medical history, current health status and so on. In addition, the questions for women include (1) living habits and environment (past and current living conditions, general living habits, cooking habits and domestic fuel type/usage); (2) work environment and travel habits (workplace, work nature, working hours and daily travel mode/time); and (3) whether they are affected by the burning of domestic solid fuels and roasting meat. The questions for students include (1) living habits and environment (past and current living conditions, general living habits, participation in household duties, family cooking habits and domestic fuel type/usage, distance from home to WB site); (2) school environment and travel habits (school location and related environment and daily travel mode/time); and (3) whether they are affected by the burning of waste and household air pollution sources. The questions for drivers include (1) living habits and environment (past and current living environments, general living habits, participation in household duties, family cooking habits and domestic fuel type/usage); (2) working environment and travel habits (motorcycle power type, driving conditions, working hours and daily travel mode/time); and (3) whether they are affected by the motorcycle emission and household air pollution sources.

The time-activity diaries requested the participants to record information on a half an hour basis (sleeping time excluded) to assess in detail what time is spent in each microenvironment on different activities.

Table 3. Reference dose (RfD) ($mg\,kg^{-1}\,day^{-1}$) and cancer slope factor (CSF) $(mg\,kg^{-1}\,day^{-1})^{-1}$ via inhalation exposure as examined in this study.

	RfD	CSF	Reference
Mn	1.8×10^{-3}	/	Liu et al. (2015)
Ni	5.4×10^{-3}	/	Zhou et al. (2014); Liu et al. (2015)
Zn	3.0×10^{-1}	/	Zhou et al. (2014)
Pb	3.5×10^{-3}	/	Zhou et al. (2014); Hu et al. (2012)
BaP	/	3.140	U.S. EPA (2011)
DEHP	/	0.014	U.S. EPA (1997); J. Z. Wang et al. (2017)

3 Results and discussion

3.1 Personal exposure to $PM_{2.5}$ and its chemical compositions

3.1.1 PE $PM_{2.5}$ mass concentration

The average PE $PM_{2.5}$ mass concentrations were 331.7 ± 190.7, 356.9 ± 71.9 and $242.8 \pm 67.6\,\mu g\,m^{-3}$ for women at the domestic fire site (DF), students at the waste burning site (WB) and drivers at the motorcycle traffic site (MT), respectively, in this study. Among the three types of subjects, the average concentrations of PE $PM_{2.5}$ for women and students were quite similar, $\sim 40\,\%$ higher than that of the drivers. PE $PM_{2.5}$ ranged from $106.2\,\mu g\,m^{-3}$ (nighttime in the dry season, 7 January) to $1164.7\,\mu g\,m^{-3}$ (daytime in the wet season, 5 July) for women at DF; from $37.8\,\mu g\,m^{-3}$ (nighttime in the wet season, 8 July) to $1137.0\,\mu g\,m^{-3}$ (daytime in the dry season, 11 January) for students at WB; and from $65.0\,\mu g\,m^{-3}$ (nighttime in the wet season, 11 July) to $648.5\,\mu g\,m^{-3}$ (daytime in the dry season, 15 January) for drivers at MT. The ranges and standard deviations of PE $PM_{2.5}$ concentrations were extremely large, especially for women, because the direct combustion sources were close to the participants. The variations of physical activities and intensities of air pollution sources potentially led to a drastic fluctuation of PE $PM_{2.5}$.

The average mass concentrations of PE $PM_{2.5}$ were 358.8 ± 100.5, 494.3 ± 15.8 and $335.1 \pm 72.1\,\mu g\,m^{-3}$ in the dry season (January) and 304.6 ± 284.5, 219.5 ± 71.3 and $150.6 \pm 10.4\,\mu g\,m^{-3}$ in the wet season (July) for women at DF, students at WB and drivers at MT, respectively (Table 4). Compared to the dry season, the reduction rate of PE $PM_{2.5}$ for women at DF in the wet season was approximately 15 %, while sharp reductions by more than 50 % were observed for students and drivers. PE $PM_{2.5}$ concentrations reducing

Table 4. Statistical analysis (arithmetic mean ± standard deviation) of personal exposure to $PM_{2.5}$ mass concentrations and the chemical compositions (units: $\mu g\,m^{-3}$) during the sampling period in the sWA region.

	Dry season						Wet season					
	Women at DF		Students at WB		Drivers at MT		Women at DF		Students at WB		Drivers at MT	
	Daytime	Nighttime	Daytime	Nighttime	Daytime	Nighttime	Daytime	Nighttime	Daytime	Nighttime	Daytime	Nighttime
PE $PM_{2.5}$	567.0 ± 180.6	150.6 ± 38.5	728.5 ± 248.5	260 ± 226.1	401.3 ± 158.0	269.0 ± 56.1	460.5 ± 445.2	148.6 ± 42.9	315.2 ± 186.9	123.7 ± 86.1	230.4 ± 22.8	70.7 ± 8.1
OC	72.4 ± 24.6	31 ± 5.0	85.0 ± 57.4	40.9 ± 34.4	49.5 ± 12.5	31.8 ± 14.2	189.3 ± 197.8	40.1 ± 9.3	65.2 ± 65.2	28.5 ± 26.8	37.0 ± 3.5	22.2 ± 10.6
EC	19.5 ± 7.3	4.7 ± 2.2	15.0 ± 4.7	8.6 ± 5.7	13.6 ± 3.6	9.0 ± 2.3	11.5 ± 10.8	6.3 ± 3.7	12.3 ± 11.4	3.6 ± 3.6	9.3 ± 0.8	1.9 ± 0.0
Total carbon	91.9 ± 31.1	35.7 ± 6.8	100.0 ± 60.1	49.5 ± 39.5	63.1 ± 16.0	40.8 ± 13.6	200.8 ± 207.1	46.3 ± 7.2	77.4 ± 76.2	32.1 ± 30.3	46.3 ± 4.2	24.1 ± 10.6
Cl^-	4.4 ± 1.3	1.6 ± 0.6	6.5 ± 3.6	6.4 ± 9.4	2.4 ± 0.8	2.2 ± 0.6	8.6 ± 8.4	1.9 ± 1.0	4.6 ± 5.4	1.9 ± 0.7	3.1 ± 0.2	2.3 ± 0.2
NO_3^-	2.7 ± 0.7	2.2 ± 1.4	5.5 ± 1.3	3.0 ± 0.7	3.7 ± 1.3	2.7 ± 0.5	2.2 ± 0.8	1.6 ± 0.7	5.0 ± 6.0	1.8 ± 1.3	1.6 ± 0.2	1.2 ± 0.1
SO_4^{2-}	4.0 ± 1.1	1.8 ± 0.6	7.5 ± 2.5	3.6 ± 0.9	7.5 ± 2.5	5.3 ± 0.6	6.8 ± 5.2	2.3 ± 0.8	6.4 ± 5.9	2.3 ± 0.4	5.2 ± 0.3	3.2 ± 0.5
Na^+	2.9 ± 0.4	1.6 ± 0.3	4.1 ± 1.1	1.9 ± 0.8	3.3 ± 1.1	2.4 ± 0.3	4.2 ± 2.2	4.4 ± 1.7	16.2 ± 17.3	3.3 ± 3.1	3.6 ± 0.2	2.6 ± 0.1
NH_4^+	0.6 ± 0.2	0.4 ± 0.5	1.4 ± 0.4	3.0 ± 4.1	1.1 ± 0.2	0.9 ± 0.2	0.6 ± 0.5	0.1 ± 0.0	0.6 ± 0.2	0.4 ± 0.3	0.7 ± 0.0	0.1 ± 0.0
K^+	3.2 ± 0.6	1.7 ± 0.6	5.8 ± 4.0	2.2 ± 0.8	1.9 ± 0.4	2.1 ± 0.9	7.6 ± 8.0	1.3 ± 0.8	3.3 ± 4.4	1.3 ± 0.6	1.1 ± 0.0	3.6 ± 1.5
Mg^{2+}	0.6 ± 0.2	0.2 ± 0.1	0.8 ± 0.3	0.3 ± 0.2	0.4 ± 0.2	0.3 ± 0.1	1.1 ± 1.2	0.3 ± 0.1	1.0 ± 0.9	0.3 ± 0.2	0.3 ± 0.0	0.2 ± 0.0
Ca^{2+}	11.0 ± 3.2	3.1 ± 0.9	14.9 ± 4.5	4.9 ± 3.2	10.6 ± 5.5	6.0 ± 1.2	6.6 ± 4.3	3.2 ± 0.8	17.3 ± 13.9	4.5 ± 3.8	6.8 ± 0.3	2.3 ± 0.1
Total ions	29.3 ± 6.6	12.5 ± 3.7	46.6 ± 15.4	25.2 ± 18.8	30.9 ± 11.9	21.9 ± 3.2	37.6 ± 29.5	15.1 ± 2.2	54.4 ± 50.0	15.8 ± 8.8	22.3 ± 1.0	15.5 ± 1.9
Fe	14.61 ± 5.25	2.64 ± 0.36	21.17 ± 4.64	4.85 ± 3.30	10.99 ± 6.50	5.90 ± 0.37	3.37 ± 3.34	1.87 ± 0.96	5.07 ± 1.74	1.76 ± 1.24	4.56 ± 0.64	0.57 ± 0.05
V	0.04 ± 0.02	0.00 ± 0.00	0.07 ± 0.02	0.02 ± 0.01	0.03 ± 0.02	0.01 ± 0.01	0.01 ± 0.01	0.00 ± 0.00	0.03 ± 0.03	0.01 ± 0.01	0.01 ± 0.00	0.01 ± 0.00
Cr	0.04 ± 0.02	0.01 ± 0.00	0.06 ± 0.02	0.01 ± 0.01	0.03 ± 0.03	0.01 ± 0.01	0.05 ± 0.02	0.06 ± 0.03	0.31 ± 0.35	0.04 ± 0.05	0.03 ± 0.00	0.03 ± 0.00
Mn	0.18 ± 0.06	0.04 ± 0.03	0.29 ± 0.08	0.07 ± 0.04	0.35 ± 0.12	0.21 ± 0.11	0.14 ± 0.16	0.04 ± 0.00	0.37 ± 0.36	0.06 ± 0.06	0.17 ± 0.02	0.04 ± 0.00
Co	0.05 ± 0.02	0.01 ± 0.01	0.09 ± 0.02	0.01 ± 0.01	0.05 ± 0.03	0.02 ± 0.02	0.02 ± 0.02	0.02 ± 0.02	0.04 ± 0.05	0.02 ± 0.02	0.02 ± 0.01	0.01 ± 0.00
Ni	0.02 ± 0.01	0.00 ± 0.00	0.02 ± 0.01	0.01 ± 0.01	0.02 ± 0.01	0.01 ± 0.01	0.02 ± 0.02	0.03 ± 0.02	0.12 ± 0.14	0.02 ± 0.03	0.02 ± 0.00	0.01 ± 0.00
Cu	0.04 ± 0.01	0.02 ± 0.01	0.14 ± 0.03	0.02 ± 0.01	0.05 ± 0.03	0.03 ± 0.01	0.13 ± 0.07	0.13 ± 0.07	0.67 ± 0.81	0.10 ± 0.09	0.07 ± 0.02	0.06 ± 0.01
Zn	0.40 ± 0.22	0.55 ± 0.73	0.49 ± 0.19	0.15 ± 0.12	0.33 ± 0.16	0.19 ± 0.07	0.51 ± 0.32	0.32 ± 0.17	1.41 ± 1.55	0.26 ± 0.27	0.29 ± 0.04	0.12 ± 0.00
Sb	0.02 ± 0.01	0.05 ± 0.02	0.02 ± 0.02	0.00 ± 0.00	0.02 ± 0.04	0.01 ± 0.01	0.12 ± 0.08	0.21 ± 0.18	1.16 ± 1.38	0.22 ± 0.29	0.07 ± 0.04	0.08 ± 0.09
Ba	0.19 ± 0.09	0.16 ± 0.12	0.25 ± 0.11	0.07 ± 0.09	0.22 ± 0.18	0.05 ± 0.07	0.47 ± 0.39	1.02 ± 0.60	6.80 ± 8.30	0.84 ± 1.41	0.18 ± 0.18	0.14 ± 0.01
Pb	0.07 ± 0.03	0.07 ± 0.07	0.17 ± 0.07	0.04 ± 0.03	0.07 ± 0.05	0.02 ± 0.03	0.14 ± 0.02	0.09 ± 0.03	0.92 ± 1.01	0.13 ± 0.18	0.05 ± 0.02	0.03 ± 0.01
Heavy metals	1.05 ± 0.28	0.91 ± 0.80	1.59 ± 0.51	0.40 ± 0.31	1.16 ± 0.66	0.56 ± 0.28	1.62 ± 0.65	1.93 ± 1.10	11.80 ± 13.91	1.69 ± 2.38	0.90 ± 0.26	0.53 ± 0.09

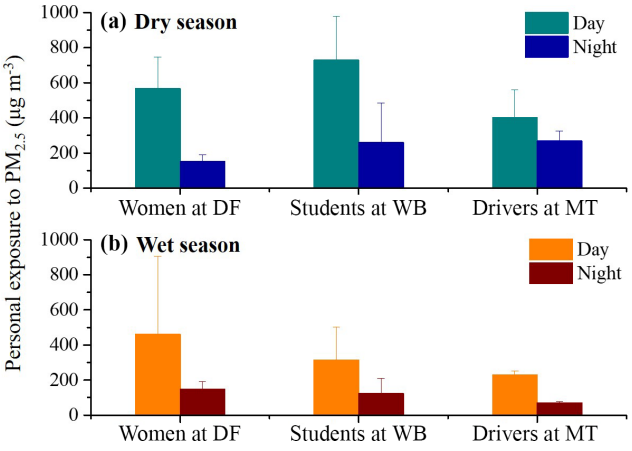

Figure 3. Personal exposure to $PM_{2.5}$ mass concentrations of women at DF, students at WB and drivers at MT in the dry season (January) and wet season (July) of 2016 in the sWA area.

could be attributed to the occurrence of increased levels of rainfall in the wet season in sWA (Table 1), which cause a large reduction of road dust being exposed to drivers and limit the spontaneous combustion of garbage significantly around students. Moreover, the large-scale transport of mineral dust and combustion aerosols emitted by savannah wild fires contributed significantly to the aerosol load during the dry season (Djossou et al., 2018), which is more important at WB and MT than at DF (as women worked in a crowded community environment).

The PE $PM_{2.5}$ mass concentrations in the daytime were much higher than those at nighttime in dry or wet seasons (Table 4 and Fig. 3). The 12 h averaged PE $PM_{2.5}$ concentrations showed day / night (D / N) ratios of 3.4 (3.8 in the dry season and 3.1 in the wet season), 2.7 (2.8 and 2.5) and 2.4 (1.5 and 3.3) for women at DF, students at WB and drivers at MT, respectively. Intensive human activities during the daytime, such as solid fuel combustion, waste combustion or motor vehicle emission influenced the different group subjects, elevating the exposure levels of $PM_{2.5}$. In the same case, lower PE $PM_{2.5}$ for students at WB in the nighttime can be explained by the fact that the participants usually spend most of their time indoors with limited physical activity, leading them to stay away and/or shelter from obvious emission sources (e.g., waste combustion) outdoors. Moreover, large fluctuations of D / N ratios for drivers were observed, with a lower average in the dry season but a higher average in the wet season. Wet season high D / N ratios are attributed to the increase in precipitation in Cotonou (Table 1), especially during nighttime (Sealy et al., 2003). This leads to the lower PE $PM_{2.5}$ for drivers at night after aerosol scavenging. Shorter driving time in the wet season is another explanation for the phenomenon because of occasion unfavorable weather (e.g., rain and storms).

The 5 h $PM_{2.5}$ average personal exposure concentration was 1574 $\mu g\,m^{-3}$ (± 287, $n = 3$) for open wood fires in households in the Njombe district of Tanzania (Titcombe and Simcik, 2011) and was comparable to the highest 12 h exposure level to $PM_{2.5}$ for women at the DF site in this study (1164.7 $\mu g\,m^{-3}$, daytime in the wet season, 5 July). It was 4.7 times the daily average PE $PM_{2.5}$ concentration in dry and wet seasons ($331.7 \pm 190.7\ \mu g\,m^{-3}$). Student (10–17 years old) $PM_{2.5}$ exposures ranged from less than 10 $\mu g\,m^{-3}$ to more than 150 $\mu g\,m^{-3}$ (mean 56 $\mu g\,m^{-3}$) in four neighborhoods in Accra, Ghana (Arku et al., 2015), much lower than that for students at the WB site ($356.9 \pm 71.9\ \mu g\,m^{-3}$). It can be seen that the high $PM_{2.5}$ exposure of students in this study was likely related to waste burning emissions, while there was no obvious strong $PM_{2.5}$ emission source in the study of Arku et al. (2015).

The average PE $PM_{2.5}$ levels are compared to the weekly ambient $PM_{2.5}$ concentrations (Djossou et al., 2018) in the same area during a similar sampling period. The average PE $PM_{2.5}$ levels were 3.0 and 2.0 times the ambient values at DF and 6.1 and 8.8 times the ambient values at MT in dry and wet seasons, respectively. The highest PE $PM_{2.5}$ to ambient (A) (PE / A) ratios were found at WB, i.e., 10.3 in the dry and 10.5 in the wet season. Such large PE / A ratios are probably due to the impact of waste combustion affecting the respiratory exposure of residents, especially children; on the other hand, high PE / A ratios can be attributed to the fact that the WB site is located in the lowest living quality region of Abidjan, where the simplest stoves and nonqualified wood as fuel are used in the house (Fig. S1d). This leads to an extremely high PE $PM_{2.5}$ indoors during the cooking time (especially for student B, who is in charge of cooking, recorded in the activity logging and questionnaire). Meanwhile, the ambient $PM_{2.5}$ sampling equipment at WB was neither fixed very close to nor located at the downwind direction of the landfill (Djossou et al., 2018), which causes the differences between the ambient and PE $PM_{2.5}$ concentrations.

Moreover, the daytime PE and ambient $PM_{2.5}$ mass concentrations on the same sampling dates were also compared. The average women daytime PE $PM_{2.5}$ levels were 3.7 and 1.2 times the ambient $PM_{2.5}$ at DF in dry and wet seasons, respectively, consistent with the finding from the weekly comparison mentioned above. However, for the students at WB and drivers at MT, the PE / A ratios were both much lower than those compared with the weekly ambient $PM_{2.5}$, with averages of 5.1 and 7.0 for the students at WB and 1.9 and 3.3 for the drivers at MT in dry and wet seasons, respectively. The PE / A ratios for students had the highest values, which is consistent with the results found earlier. The PE / A ratios all above 1.0 and large variability of $PM_{2.5}$ between PE and ambient concentrations imply that fixed-point sampling is likely to underestimate the PE $PM_{2.5}$ and consequent human health hazards. The results further confirm the importance of portative PE $PM_{2.5}$ sampling for health risk assessment.

3.1.2 PE $PM_{2.5}$ chemical compositions

Table 4 summarizes the average PE $PM_{2.5}$ chemical compositions, including carbon fractions (OC and EC), water-soluble inorganic ions and target heavy metals. TC was the highest composition in PE $PM_{2.5}$, accounting for 24.4 ± 4.5 %, 16.6 ± 2.0 % and 17.8 ± 4.9 % of PE $PM_{2.5}$ for women, students and drivers, respectively. High OC values suggest the strong contribution of combustion sources to PE $PM_{2.5}$ in sWA (Djossou et al., 2018; Ouafo-Leumbe et al., 2017). The average OC concentration ($83.2\,\mu g\,m^{-3}$) and composition (24.4 %) in women's PE $PM_{2.5}$ samples were the highest among the three types of PE participants, due to their direct contact with the ignition and closeness to the solid fuel (wood in this study) burning/meat roasting at the workplace or their own residential units. However, the EC concentrations ($8.4–10.5\,\mu g\,m^{-3}$) and compositions (3.0 %–3.5 %) were very similar among the three different PE groups, showing that EC was less affected by human activities related to combustion sources in this study.

The OC to EC ratio (OC / EC) has been used to determine emission and transformation characteristics of carbonaceous aerosols (Cao et al., 2008). The OC / EC averaged 9.9 ± 5.3 for women at DF, 6.1 ± 0.7 for students at WB and 5.8 ± 2.7 for drivers at MT. Previous studies (Cachier et al., 1989; Cao et al., 2005a, 2008; Li et al., 2009; Tian et al., 2017; Watson et al., 2001) characterized the average OC / EC ratio of 1.1 as motor vehicle exhaust, 2.7 as coal combustion and 9.0 as biomass burning from their source samples (i.e., fresh emissions/plumes). In the present study, the OC / EC suggests that biomass burning was the main contributor to PE carbonaceous aerosols for women at DF, while the mixed emissions of biomass and coal burning or/and motor vehicle exhaust were dominant for students at WB and drivers at MT. The OC / EC was mostly higher in the wet season than the dry season, ascribed to the fact that the higher RH in the wet season favors the formation of secondary organic carbon (SOC) (Huang et al., 2014). The daytime OC / EC for drivers' PE samples were relatively low (an average of 3.7) and constant between wet and dry seasons, promising that motor vehicle exhaust was the most dominant and stable pollution source in their working environment. The PE of women displays a higher (an average of 13.9) and more scattered OC / EC ratio than those measured for students and drivers in the wet season (Fig. 4). This was induced by particularly high and dramatic changes in individual exposure to obvious carbonaceous aerosol sources (e.g., wood burning and grilling).

With the data shown in Djossou et al. (2018), the PE OC / EC ratios for the participants were 1.2 and 2.5 times the ambient OC / EC ratios in dry and wet seasons at DF, 1.7 and 2.8 times at WB and 1.1 and 2.0 times at MT. Such higher OC / EC values in PE samples can be resulted from specific individuals' activities and potential contamination at microenvironments (Crist et al., 2008; Meng et al., 2009). In addition, the influences of precipitation and other meteorological factors on OC / EC in ambient samples were less than those in PE samples (i.e., dry season OC / EC was more comparable between the ambient and PE samples).

The average concentrations of total quantified water-soluble inorganic ions were 23.6 ± 12.8, 35.5 ± 18.3 and $22.7 \pm 5.0\,\mu g\,m^{-3}$ for women at DF, students at WB and drivers at MT, accounting for 8.5 ± 1.0 %, 12.1 ± 2.7 % and 11.9 ± 0.4 % of PE $PM_{2.5}$ masses, respectively. Dissimilar to the compositions in heavy polluted cities in China (SO_4^{2-}, NO_3^- and NH_4^+ were the most abundant ions in ambient or PE $PM_{2.5}$, accounting for 50 %–90 % of quantified ions and ~ 30 % of $PM_{2.5}$ masses) (Xu et al., 2016b, 2018b; Zhang et al., 2013), Ca^{2+}, a marker of fugitive dust, was the most abundant ion, accounting for ~ 28 % (in a range from 25.3 % to 29.3 %) of total quantified ions, followed by Cl^-, SO_4^{2-} and K^+ for women at DF, Na^+, SO_4^{2-} and Cl^- for students at WB and SO_4^{2-}, Na^+ and NO_3^- for drivers at MT. The profiles thus indicate that the particle resuspension by personal activities was the main contributor to the PE $PM_{2.5}$ in sWA (Chen et al., 2017; Xu et al., 2015). The diurnal variations in the composition of Ca^{2+} to total ions (i.e., daytime = 30.6 % and nighttime = 22.8 %) also illustrate this conclusion. Moreover, SO_4^{2-} forms primarily through the atmospheric oxidation of SO_2 emitted mainly from coal and diesel combustion (Seinfeld and Pandis, 2006; Xu et al., 2016b). As the second most enriched ion, the average proportion of SO_4^{2-} was 17.7 %, which implies that purification of raw coal and diesel (Wang et al., 2013) should be applied in this area to lower sulfur emissions and therefore decrease PE to SO_4^{2-} in $PM_{2.5}$. The SO_4^{2-} exposure levels for the drivers were 33 % and 40 % higher than the women and students, respectively, indirectly indicating that the emission of SO_2 might be higher in Cotonou or the participants are exposed to higher SO_2 or SO_4^{2-} from the diesel vehicle emissions.

Generally, Na^+ and Cl^- ranked the third and fourth abundant ions in the PE samples. The sampling sites in sWA cities in this study are all close to the sea and were affected by sea salt particles. It is also worth noting that biomass burning marker K^+ (Kang et al., 2004; T. Zhang et al., 2014) displayed a high absolute average concentration of $3.4\,\mu g\,m^{-3}$ and composition of 14.5 % in women's PE $PM_{2.5}$ samples, confirming their particular exposure to biomass burning during roasting in the workplace. To the best of the authors' knowledge, NO_3^- derives from NO_x emitted mainly from motor vehicle exhaust (especially gasoline vehicle), industry and power plants (Seinfeld and Pandis, 2006; Xu et al., 2016b). An additional consideration is that the industry is not well developed in this area (i.e., much less industry in Cotonou than Abidjan) and thus is not the main contributor to $PM_{2.5}$ (Ouafo-Leumbe et al., 2017). In comparison with the findings from the other two sites, motor vehicle emission obviously contributed to drivers' PE concentrations, consistent with the conclusion for SO_4^{2-} as discussed above.

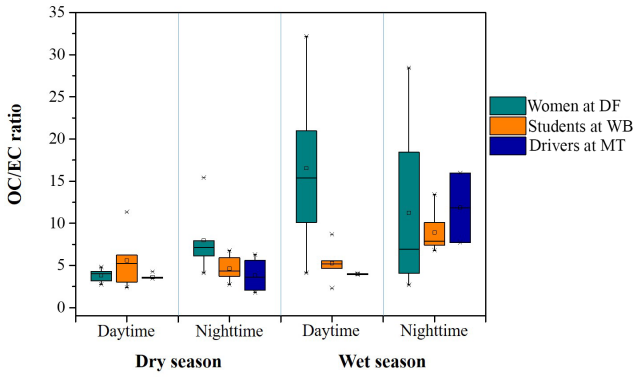

Figure 4. Variations of OC / EC ratios in personal exposure to $PM_{2.5}$ samples for women at DF, students at WB and drivers at MT. (The box plots indicate the average concentration and the min, 1st, 25th, 50th, 75th, 99th and max percentiles).

The concentrations of 10 targeted heavy metals, including V, Cr, Mn, Co, Ni, Cu, Zn, Sb, Ba and Pb, are also shown in Table 4. The total concentrations were 1.4 ± 0.3, 3.9 ± 6.5 and $0.8 \pm 0.2\,\mu g\,m^{-3}$ for women at DF, students at WB and drivers at MT, accounting for 0.7±0.4 %, 1.0±1.2 % and 0.4 ± 0.1 % of the PE $PM_{2.5}$, respectively. The PE to heavy metals for the students was 1.8 and 3.9 times those for the women and drivers, mainly due to the emissions from garbage combustion at landfill (Y. Wang et al., 2017). The D / N ratios ranged from 0.8 to 2.1 for women and drivers but averaged 4.0 and 7.0 in dry and wet seasons, respectively, for students. This can be explained by two reasons: the first is that there were intense physical activities by the students and strong disturbances by landfill workers. Another reason is the spontaneous combustion of waste occurring frequently during the day due to less precipitation and higher ambient temperature in the daytime. Ba, Zn and Mn were found to be the dominant heavy metals, accounting for ∼ 73 % of the total quantified elemental concentration in all samples. Ba had a decisive advantage over other elements, having a contribution of > 50 % for students. It is usually added to rubber and plastic products to improve acid and alkali resistance. Such products were main fractions of the garbage at landfill in this area (Feng et al., 2006). Zn and Mn ranked the first and second highest personal exposure elements for drivers at MT, which are mainly derived from motor oil additives, tyre wear and brake pad wear (Zhao and Hopke, 2006).

3.2 Mass balance of personal exposure to $PM_{2.5}$

The calculation of the mass balance of the PE $PM_{2.5}$ is an effective method to figure out the principal components in $PM_{2.5}$ and distinguish the pollution sources (Gokhale et al., 2008). PE $PM_{2.5}$ mass in this study can be classified into six parts: organic matter (OM), EC, water-soluble inorganic ions, geological material (GM), heavy metals and an unresolved fraction (Fig. 5). The first five main resolved fractions can explain 78.3 % to 90.6 % of total PE $PM_{2.5}$ mass concentrations in this study. The unresolved fraction may include water and other undetected substances. For OM, since there are no full organic composition profiles for the PE $PM_{2.5}$, a conversion factor of 1.4 (1.4 corrects the organic carbon mass for other constituents associated with the organic carbon molecule) is generally used (Turpin and Lim, 2001) to quantify OM using Eq. (5):

$$OM = 1.4 \times OC. \quad (5)$$

OM accounted for 34.1±6.3 %, 23.3±2.8 % and 24.9±6.9 % of the PE $PM_{2.5}$ mass for women at DF, students at WB and drivers at MT, respectively. The results show that there are distinct sources of PE OC for women at DF. According to the information gathered from the questionnaires, the combustion sources, such as roasting meat/peanuts and burning wood, are the major contributors to PE OC for women in this study.

In addition, Fe has been widely used to estimate the upper limit of GM (Taylor and McLennan, 1985). Fe constitutes ∼ 4.0 % of dust of the earth's crust (Cao et al., 2005b; Hao et al., 2007; Kabata-Pendias and Mukherjee, 2007; Sun et al., 2014; Wu et al., 2012; Xu et al., 2016b). The amount of GM is calculated using Eq. (6):

$$GM = (1/4.0\,\%) \times Fe. \quad (6)$$

It is found that GM contributed 35.8±2.1 %, 46.0±3.7 % and 42.4 ± 4.7 % of PE $PM_{2.5}$ mass concentrations for women at DF, students at WB and drivers at MT, respectively. Fugitive dusts, including road dust resuspension from the disturbance of motor vehicles and human activities, construction dust from uncovered construction sites and the dusts generated from burning, could be the dominant sources of PE $PM_{2.5}$ in this study. OM and GM showed similar proportions (34.1 % and 35.8 %, respectively) of PE $PM_{2.5}$ mass for women at DF. The fractions of GM in PE samples for students and drivers were approximately 10 % and 7 % higher than that for women. Therefore, the fugitive dust was the most important source of PE $PM_{2.5}$ in this less developed area, shown by nearly 50 % contribution for students and drivers, attributable to human physical activities and a large amount of covered land. It is surprising to note that the secondary formed ions (i.e., SO_4^{2-}, NO_3^- and NH_4^+) and the total quantified water-soluble inorganic ions were shown to have exceedingly low proportions of PE $PM_{2.5}$ for all groups. This reconfirms the limited contribution to PE $PM_{2.5}$ from secondary ionic formation.

In Fig. 5, evident diurnal distinctions can be observed in the two major chemical compositions of OM and GM. GM exhibited a lower proportion at nighttime (35.3 %) than daytime (47.5 %), suggesting its close relationship with human activities. Higher GM was found for all groups in the dry season because of harmattan haze which introduced mineral dusts and the lack of precipitation which increased road dust

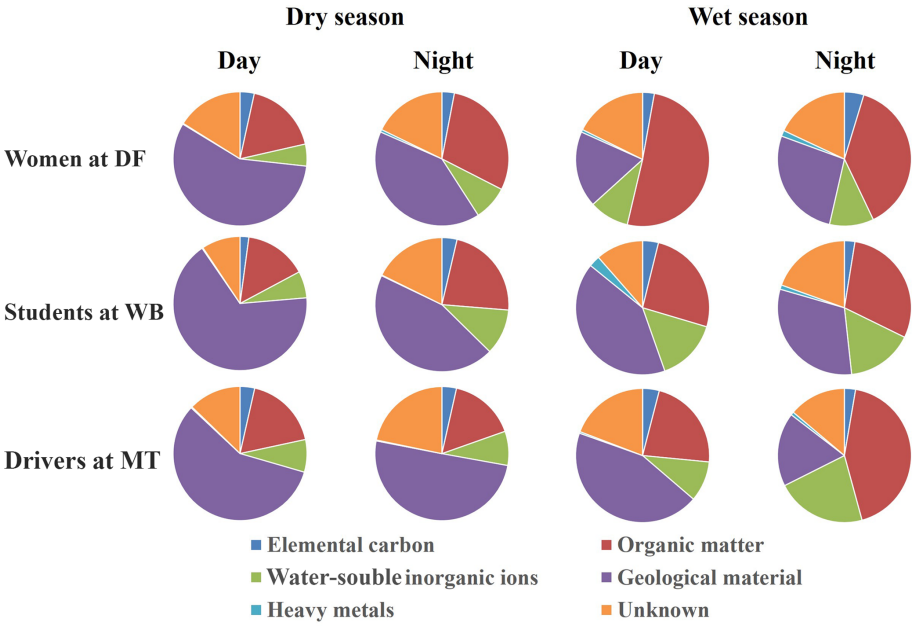

Figure 5. Personal exposure to $PM_{2.5}$ mass concentration closures for women at DF, students at WB and drivers at MT in different sampling seasons.

resuspension. Moreover, OM showed equal or lower proportions in the daytime (25.0 %) than nighttime (30.0 %), relative to the meteorological parameters (i.e., factors that affected the formation of secondary organic carbonaceous aerosol) and diurnal changes of combustion sources around subjects. An exception is that OM proportion of women's PE $PM_{2.5}$ in the daytime (50.8 %) was much higher than nighttime (38.2 %) in the wet season, due to influences of damp wood burning during working hours. Burning biomass fuel with high moisture often results in low combustion efficiency, a long smoldering period and high air pollutant emissions (Grandesso et al., 2011; Shen et al., 2012, 2013). The emission factor of OC usually increases with the fuel moisture content (Chen et al., 2010; Keita et al., 2018). Therefore, burning damp wood led to higher OC emission than dry wood, in line with the observation for women's PE results in this study.

4 Organic species fingerprints of personal exposure to $PM_{2.5}$

Organic fingerprint markers can be used to indicate specific emission sources and further characterize the pollution that impacts different populations. The average PE concentrations of $PM_{2.5}$-bound PAHs, PAEs and hopanes were 54.8 ± 20.3, 986.8 ± 82.2 and $27.9 \pm 1.0\,ng\,m^{-3}$, respectively, representing high organic pollution in the sWA region (Table 5). Dissimilar to the trend in $PM_{2.5}$ masses (students > women > drivers), the PE to target organic compounds of different groups were varied, with a descending order of women > students > drivers for PAHs, students > women > drivers for PAEs and drivers > women > students for hopanes (Table 5 and Fig. 6).

4.1 PAHs

Benzo[b]fluoranthene (BbF) was the most abundant PAH for women at DF, followed by benzo[a]pyrene (BaP) and indeno[1,2,3-cd]pyrene (IcdP). The average concentration of BbF (a marker of low temperature combustion, such as wood burning) was $11.6 \pm 19.2\,ng\,m^{-3}$, accounting for approximately 15.0 % of the ΣPAHs for women (Wang et al., 2006) (Table 5). The most abundant PAH species for students at WB and drivers at MT were IcdP ($6.4 \pm 4.5\,ng\,m^{-3}$) and benzo[ghi]perylene (BghiP) ($6.4 \pm 0.5\,ng\,m^{-3}$), respectively, indicating the contributions from waste incineration and/or high temperature combustion of fuel (e.g., gasoline vehicle emission) (Baek et al., 1991; Wang et al., 2006). The average ΣPAHs of women at DF ($125.4 \pm 54.8\,ng\,m^{-3}$) and drivers at MT ($44.6 \pm 10.8\,ng\,m^{-3}$) in the wet season were 326 % and 52 % higher than those in the dry season (29.4 ± 5.6 and $29.4 \pm 4.4\,ng\,m^{-3}$, respectively), while ΣPAHs in the wet season ($36.8 \pm 15.7\,ng\,m^{-3}$) were 42 % lower than that dry season ($62.9 \pm 45.0\,ng\,m^{-3}$) for students at WB. The dramatic increase in the PE of women to PAHs is mainly due to an increase in moisture content in the wood used for grilling meat in the wet season, promoting more PAH emission from wood combustion processes (Shen et al., 2013). The restraint of waste combustion in the wet season is the main factor for the lower PE $PM_{2.5}$-bound PAHs at landfill, in accordance with seasonal patterns of PE $PM_{2.5}$ mass. Fanou et

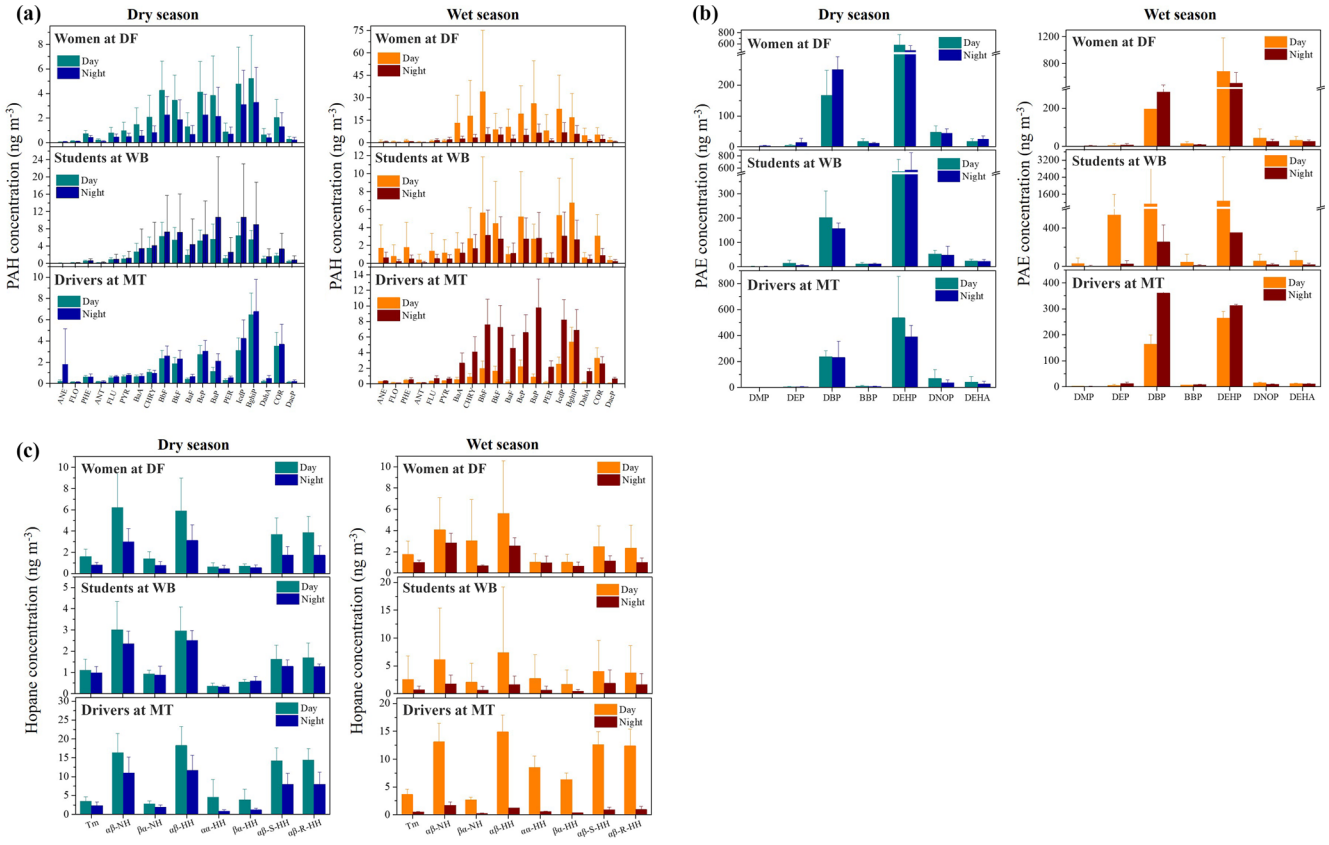

Figure 6. Distributions of **(a)** PAHs, **(b)** PAEs and **(c)** hopanes in $PM_{2.5}$ personal exposure samples for women at DF, students at WB and drivers at MT in dry and wet seasons of 2016.

al. (2006) measured the PE PAH concentrations in Cotonou and found that the PAH level associated with particles ranged from 76.21 to 103.23 ng m^{-3} for 35 motorbike taxi drivers in March 2001. Our values for drivers at the MT site were 50 %–64 % lower than their values, suggesting that the exposure to PAHs for the motorbike drivers in this region has been improved.

In Fig. 6a, PE PAHs showed fluctuating diurnal variations for the three kinds of subjects. For the women at DF, the daytime concentrations in wet and dry seasons were both higher than those at nighttime due to the intensive roasting meat and burning wood during working hours. For the students at WB, PE PAHs at night were higher in the dry season but lower in the wet season. Both the PAH profiles showed high combustion markers of BbF and benzo[e]pyrene (BeP) and high gasoline vehicle emission markers of dibenzo[a,h]anthracene (DahA) and BghiP (Baek et al., 1991; Wang et al., 2006). For the drivers at MT, the average D / N ratios in dry and wet seasons were 0.8 and 0.3, respectively. The higher PE ΣPAH concentrations at night and lower D / N ratios for drivers may be explained by the potential combustion sources of PAHs close to the participants (e.g., sources nearby the drivers' homes) in Cotonou, Benin, especially in the wet season. This can be deduced by the combustion marker of BaP, which was the highest PAH species at night in the wet season, even though the drivers were exposed to traffic emissions during the night working hours (18:30 to 21:00 UTC). Further studies are thus required to confirm findings and figure out reasons for them.

In the study of Titcombe and Simcik (2011), the authors found that the 5 h average total PAH personal exposure concentration was 5040 ng m^{-3} (±909 ng m^{-3}, $n = 3$) for open wood fires in households in the Njombe district of Tanzania, which was much higher (∼ 65 times) than women's exposure to PAHs at the DF site in the current research. The highest 12 h exposure to PAHs for women at the DF site in this study was 469.7 ng m^{-3} (daytime in the wet season, 6 July), approximately one-tenth of the PAH concentrations from open wood fires in Tanzania mentioned above. The large PE PAH concentration difference between these two studies may be influenced by many factors such as wood type, combustion state, stove structure and sampling time.

Diagnostic ratios of PAHs have been widely used in source identification (Tobiszewski and Namiesnik, 2012; Yunker et al., 2002). In our study, the average values of BeP / (BeP + BaP) and IcdP / (IcdP + BghiP) were 0.47 and 0.52 for women at DF, 0.51 and 0.52 for students at WB and 0.64 and 0.34 for drivers at MT, respectively (Fig. 7), indicat-

Table 5. Mass concentrations of PE $PM_{2.5}$-bound PAH, PAE and hopanes species for women at DF, students at WB and drivers at MT ($ng\,m^{-3}$).

Specific species (abbreviations)	Women at DF		Students at WB		Drivers at MT	
	Average	SD	Average	SD	Average	SD
acenaphthene (ACE)	0.4	0.5	0.6	1.2	0.7	1.7
fluorene (FLO)	0.3	0.3	0.3	0.6	0.1	0.0
phenanthrene (PHE)	0.8	0.4	0.9	1.2	0.6	0.1
anthracene (ANT)	0.3	0.2	0.2	0.2	0.2	0.0
fluoranthene (FLU)	1.0	0.4	1.0	0.7	0.6	0.1
pyrene (PYR)	1.2	0.5	1.0	0.5	0.6	0.1
benzo[a]anthracene (BaA)	4.5	8.5	2.2	1.5	1.1	0.5
chrysene (CHR)	6.1	11.2	3.0	1.6	1.8	0.8
benzo[b]fluoranthene (BbF)	11.6	19.2	5.6	2.7	3.6	1.2
benzo[k]fluoranthene (BkF)	4.9	4.2	5.0	2.9	3.3	1.1
benzo[a]fluoranthene (BaF)	3.8	5.3	2.1	2.4	1.5	0.8
benzo[e]pyrene (BeP)	7.7	8.1	5.0	2.5	3.6	0.7
benzo[a]pyrene (BaP)	9.7	12.5	5.5	5.7	3.5	1.6
perylene (PER)	2.8	5.0	1.3	1.4	0.8	0.4
indeno[1,2,3-cd]pyrene (IcdP)	9.4	9.3	6.4	4.5	4.5	0.7
benzo[ghi]perylene (BghiP)	7.8	6.1	6.0	3.6	6.4	0.5
dibenzo[a,h]anthracene (DahA)	1.8	2.2	1.0	0.6	0.6	0.1
coronene (COR)	2.8	1.6	2.3	1.4	3.3	0.4
dibenzo[a,e]pyrene (DaeP)	0.7	0.7	0.5	0.3	0.3	0.1
ΣPAHs	77.4	47.9	49.9	30.7	37.0	7.4
dimethyl phthalate (DMP)	2.2	1.0	9.6	27.9	1.9	0.5
diethyl phthalate (DEP)	8.3	4.1	146.5	517.0	6.8	1.4
di-*n*-butyl phthalate (DBP)	224.8	90.6	440.7	848.4	248.2	42.1
benzyl butyl phthalate (BBP)	13.8	4.3	19.7	37.3	8.1	2.9
bis(2-ethylhexyl) phthalate (DEHP)	566.4	181.4	688.0	899.1	376.3	144.5
di-*n*-octyl phthalate (DNOP)	40.9	16.9	43.8	26.2	33.0	31.0
bis(2-ethylhexyl) adipate (DEHA)	25.6	6.0	32.0	41.8	23.8	19.0
ΣPAEs	882.0	193.3	1380.4	335.2	698.1	192.4
17α(H)-22,29,30-trisnorhopane (Tm)	1.3	0.5	1.3	1.9	2.5	0.5
17α(H)-21β(H),30-norhopane ($\alpha\beta$-NH)	4.0	1.2	3.3	4.1	10.6	1.9
17β(H)-21α(H),30-norhopane ($\beta\alpha$-NH)	1.5	1.8	1.1	1.5	1.9	0.3
17α(H)-21β(H)-hopane ($\alpha\beta$- HH)	4.3	1.9	3.6	5.4	11.5	2.2
17α(H)-21α(H)-hopane ($\alpha\alpha$-HH)	0.8	0.2	1.0	2.0	3.6	2.1
17β(H)-21α(H)-hopane ($\beta\alpha$-HH)	0.7	0.2	0.8	1.2	2.9	1.2
17α(H)-21β(H),(22S)-homohopane ($\alpha\beta$-S-HH)	2.3	0.7	2.2	2.4	8.9	1.3
17α(H)-21β(H),(22R)-homohopane ($\alpha\beta$-R-HH)	2.2	0.8	2.1	2.1	8.9	1.3
Σhopanes	17.1	6.4	15.6	6.1	50.9	7.9

ing the unique impacts on the PE $PM_{2.5}$ from different atmospheric pollution sources. The average BeP / (BeP + BaP) ratios ranged from 0.47 to 0.64, comparable with those reported in the Chinese megacities of Guangzhou (0.41–0.72) and Xi'an (0.59–0.73) (Li et al., 2005; Xu et al., 2018c), but lower than the value measured in Shanghai (> 0.70) (Feng et al., 2006). This implies the low oxidability of the PAHs in the less-developed cities in sWA. PAHs in drivers' PE samples were more prone to aging (i.e., the average ratio was 1.3–1.4 times those for women and students) because of their re-suspension onto road dusts (i.e., longer residence lifetime) and longer outdoor activity time (i.e., exposure to more sunlight). Fine and ultra-fine particle-bound PAHs are emitted in high-temperature combustion from motor vehicular engine, which are more easily photochemically oxidized in the atmosphere (Baek et al., 1991; Lima et al., 2005). The differences of BeP / (BeP + BaP) ratios between dry and wet seasons were not obvious, without a general pattern. However, the ratio exhibited a significant day–night variation, with an average of 0.59 and 0.49 in the daytime and nighttime, re-

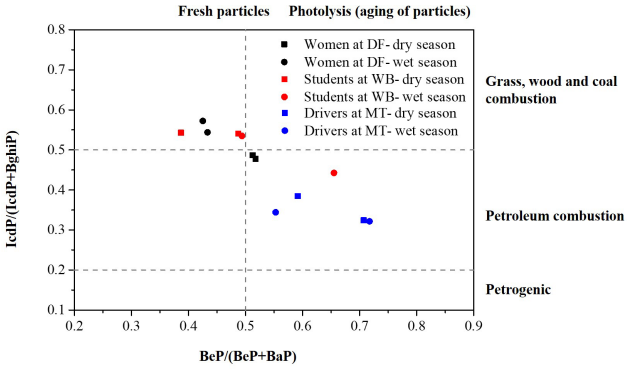

Figure 7. Correlations between PAH diagnostic ratios. (Average ratio points of each type participant indicate day and night values, respectively.)

spectively. This shows that more favorable meteorological conditions (i.e., higher light intensity) and more physical activities (i.e., extended time for particulate resuspension) in the daytime are more conducive to the aging of $PM_{2.5}$ and its bounded PAHs. Moreover, IcdP / (BghiP + IcdP) ratios of < 0.2, 0.2–0.5 and > 0.5 were used to identify petrogenic, petroleum combustion and a mix of grass, wood and coal combustion, respectively (Yunker et al., 2002). The relatively low ratio for drivers at MT (0.34) demonstrates that the PAHs were mainly produced from motor vehicles, while grass, wood and coal combustion was more dominant for women at DF (0.52) and students at WB (0.52) (Fig. 7). The IcdP / (IcdP + BghiP) ratio did not show significant seasonal variation.

4.2 Phthalate esters (PAEs)

Phthalate esters are widely used as plasticizers in materials and can be released into the air through matrix evaporation and combustion (Gu et al., 2010; J. Z. Wang et al., 2017). The PE levels of PAEs could be mainly attributed to the usage of household products, painting material, plastic waste incineration and municipal sewage release (L. B. Zhang et al., 2014). The total concentrations of six phthalate esters (the first six species of PAEs in Table 5) and one plasticizer (bis(2-ethylhexyl) adipate, DEHA) (abbreviated as ΣPAEs for the total seven species) were 882.0 ± 193.3, 1380.4 ± 335.2 and $698.1 \pm 192.4\,ng\,m^{-3}$, respectively, for women at DF, students at WB and drivers at MT (Table 5). Bis(2-ethylhexyl) phthalate (DEHP) was the most dominant PAE species, followed by di-n-butyl phthalate (DBP) for all the three groups of participants. DEHP is mainly used as a plasticizer for the manufacture of polyvinyl chloride (PVC); and together with DBP, they are the most widely used PAEs globally (Meng et al., 2014). The average DEHP and DBP concentrations were 543.6 and $304.6\,ng\,m^{-3}$, accounting for approximately 55.1 % and 30.9 % of the ΣPAEs, respectively (Fig. 6b). The elevated ΣPAEs for students can be ascribed to the combustion of plastic products at landfill nearby. Our results are similar to previous studies conducted in Xi'an and Tianjin, China (Kong et al., 2013; J. Z. Wang et al., 2017). The ΣPAEs ranged from 376.6 to $1074\,ng\,m^{-3}$ outdoors and from 469.2 to $1537\,ng\,m^{-3}$ in classrooms (J. Z. Wang et al., 2017), where DEHP and DBP were also the most abundant PAEs, with a sum of composition of 68 % and 73 % of the ΣPAEs outdoors and indoors, respectively.

The average concentrations of the ΣPAEs for women at DF, students at WB and drivers at MT were comparable in the dry season. However, the average concentrations were 927.2 ± 154.9, 1929.8 ± 340.4 and $594.6 \pm 16.6\,ng\,m^{-3}$ in the wet season, 1.1, 2.3 and 0.7 times the ΣPAEs in the dry season (Fig. 6b). A significant increase in PE ΣPAEs for students at WB can be attributed to the enhanced PAE emission in the daytime with high RH ($3173.6 \pm 1028.3\,ng\,m^{-3}$), consistent with the findings on PE $PM_{2.5}$. Dry and wet seasons had similar PAE profiles with different diurnal variations (Fig. 6b). The average D / N ratios of the ΣPAEs in the dry season demonstrate constant concentrations, with an average of 1.0, 1.0 and 1.3, respectively, for women, students and drivers, while much larger variations of 1.1, 4.6 and 0.7 were found for the wet season. Noticeably different diurnal D / N ratios for students at WB are interrelated with human activities (especially the emissions from plastic materials) and the decreased waste combustion influenced by meteorological conditions (i.e., more precipitation at night in the wet season), which was mentioned in Sect. 3.1.1.

4.3 Hopanes

Hopanes are markers for fossil fuel (e.g., petroleum) combustion (Simoneit, 1999; Wang et al., 2009). The average PE to the sum of eight quantified hopanes (Σhopanes) for the drivers was $50.9 \pm 7.9\,ng\,m^{-3}$, 2.0 and 2.3 times higher than the women ($17.1 \pm 6.4\,ng\,m^{-3}$) and students ($15.6 \pm 6.1\,ng\,m^{-3}$), respectively (Table 5). The results indicate an extremely high personal respiratory exposure contribution from the motor vehicle emissions (e.g., gasoline combustion) for the drivers. It is important to note that number of automobiles is rapidly increasing in sWA cities, further exacerbating the air pollution and consequential health issues. The Σhopanes showed unobvious seasonal variations for three types of PE participants. The Σhopanes in the dry season were 0.9, 1.8 and 0.7 times those in the wet season. Even though the Σhopanes varied among the three sites, their profiles on individual species were similar. 17α(H)-21β(H),30-norhopane ($\alpha\beta$-NH) and 17α(H)-21β(H)-hopane ($\alpha\beta$-HH) were the two most abundant hopanes in all PE $PM_{2.5}$ samples, with average concentrations of 6.0 and $6.5\,ng\,m^{-3}$ and compositions of 21.4 % and 23.3 % of the Σhopanes, respectively (Table 5 and Fig. 6c).

Compared with D / N ratios of ΣPAHs and ΣPAEs, Σhopanes exhibited a more stable diurnal trend, higher in the daytime due to heavier traffic emissions. For women at

Table 6. Non-cancer risks of heavy metals and cancer risks of PAHs and PAEs via inhalation exposure in PE $PM_{2.5}$ of women at DF, students at WB and drivers at MT in dry and wet seasons.

	Dry season			Wet season		
	Women	Students	Drivers	Women	Students	Drivers
Non-cancer risk						
Mn	5.71E-03	2.02E-02	1.09E-02	4.83E-03	2.31E-02	4.26E-03
Ni	1.44E-04	5.60E-04	1.77E-04	4.49E-04	2.59E-03	2.00E-04
Zn	1.45E-04	2.15E-04	6.16E-05	1.24E-04	5.45E-04	5.05E-05
Pb	1.75E-03	5.98E-03	9.33E-04	2.97E-03	2.95E-02	7.75E-04
HI	7.74E-03	2.70E-02	1.21E-02	8.37E-03	5.57E-02	5.29E-03
Cancer risk (ILCR)						
PAHs ($[BaP]_{eq}$)	3.13E-06	6.43E-06	3.22E-06	9.33E-06	3.68E-06	3.42E-06
PAEs (DEHP)	2.92E-07	3.36E-07	1.86E-07	3.15E-07	4.86E-07	1.16E-07

DF, the D / N ratio was both 2.0 in dry and wet seasons, with Σhopanes of 24.0 ± 11.1 and $12.2 \pm 5.0\,ng\,m^{-3}$ for daytime and nighttime in the dry season and 21.4 ± 17.5 and $10.9 \pm 3.6\,ng\,m^{-3}$ in the wet season. The D / N ratio of Σhopane for drivers at MT had the highest value of 11.5, with 78.0 ± 19.1 and $44.9 \pm 16.4\,ng\,m^{-3}$ for daytime and nighttime in the dry season and 74.2 ± 16.3 and $6.5 \pm 1.7\,ng\,m^{-3}$ in the wet season. It is notable that the daytime concentrations for drivers were comparable between the two seasons, while the nighttime hopanes in the wet season were mostly washed away by rainfall, resulting in a very large decline in its concentrations.

Even though these organic groups are not major fractions in PE $PM_{2.5}$, their fingerprints can more accurately illustrate the contributions of air pollution sources to $PM_{2.5}$. PAHs, PAEs and hopanes are source markers for combustion activities, plastics emissions and fossil fuel emissions (e.g., from gasoline vehicles), respectively, well matching the potential air pollution sources that impact the PE $PM_{2.5}$ for participants in this study. Our results not only indicate that the $PM_{2.5}$ respiratory exposure can be strongly contributed from environmental pollution sources and individual activities but also prove the study to be a reliable application of organic tracers for the characterization of personal exposure.

5 Health risk assessment of personal exposure to $PM_{2.5}$

Non-cancer risks of four heavy metals (i.e., Mn, Ni, Zn and Pb) and cancer risks of PAHs and PAEs via inhalation exposure for women at DF, students at WB and drivers at MT are shown in Table 6. In general, the non-carcinogenic risks of Mn and Pb were relatively higher than those of Ni and Zn but still well below the international threshold value of 1.0. Among those four metals, the hazard quotient (HQ) of Pb in the wet season for students at WB was the highest (2.95×10^{-2}), which suggests that the Pb non-carcinogenic risk to children is more severe in that area compared with other participants and metals. There was no consistent difference in the risks between dry and wet seasons, except Ni, which showed a much greater value in the wet than the dry season for all participants. Counting the total of four toxic heavy metals, hazard index (HI) values for participants are also shown in Table 6. The dry / wet season ratios of HI were 0.9, 0.5 and 2.3 for women, students and drivers, respectively, suggesting that the non-cancer risk of PE to metals in $PM_{2.5}$ for drivers was significantly higher in dry than wet seasons, owing to a mass of fugitive dust on the road at low RH. Moreover, the average HI levels were 8.06×10^{-3}, 4.13×10^{-2} and 8.68×10^{-3} for the women, students and drivers, respectively. The highest non-cancer health risks of the heavy metals in PE $PM_{2.5}$ for students were 5.1 and 4.8 times those for women and drivers. Overall, Mn, Zn, Ni, Pb and HI were all below the safety limit for the populations involved in this study, representing negligible non-cancer risks of heavy metals in PE $PM_{2.5}$ in the sWA region.

As shown in Table 6, the ILCRs of PAHs all exceeded the international acceptable level of 1×10^{-6}, both in dry and wet seasons. Meanwhile, the ILCRs of PAEs were all below 1×10^{-6}, well within the safety limits of cancer risk. For all target participants, higher cancer risks of PE $PM_{2.5}$-bound PAHs and PAEs were found in the wet season. The seasonal variations such as an increase of RH could lead to an increase in cancer risks of toxic organics in PE $PM_{2.5}$. In the dry season, the average ILCR values of PAHs for women and drivers were comparable, both $\sim 50\,\%$ lower than those for students, implying the high toxicity originated from the waste burning and the high sensitivity of young people. In the wet season, PAHs exhibited the highest ILCR for women at DF, 2.5 and 2.7 times those for students and drivers, respectively. The domestic wood burning and meat grilling can trigger nearly 10 times the safety limit for PAHs. The cancer risks of PAEs showed similar trends in dry and wet seasons (Yang et al., 2011), with the descending order of students > women> drivers. The carcinogenic risks of PAEs for

the drivers were the lowest, much lower (45 % and 76 % for dry and wet seasons) than those for students who live close to waste incineration. In a word, the ILCRs of PAHs exceeded the threshold value of 1×10^{-6} for all the participants, indicating that the carcinogenic PAHs are a threat to individuals' health and subsequently alert to the need for effective emission control in sWA. Even though PAEs had low carcinogenic risks, the effects from waste burning on students should not be ignored, and proper control measures for both $PM_{2.5}$-bound heavy metals and toxic organic must be established.

In addition, it should be noted that both non-cancer and cancer risks could be potentially underestimated since many toxic chemical components were not involved in this study. Based on the current study, there are a variety of emission sources that impact population groups in the sWA region to different degrees. Attention should be paid to the health risks of chemicals via inhalation, especially Pb and Mn for students at the WB site as well as PAHs for women at DF in the wet season.

6 Conclusions

This work can be regarded as a first attempt at the assessment of personal exposure to fine particulate matter originating from main sources of combustion aerosols in representative cities of southern West Africa. We targeted different groups of people exposed to domestic fires, traffic and waste burning in this study. Even though there are few drawbacks such as a relatively short sampling period and a limited number of participants, our findings provide a new insight into the health risks due to $PM_{2.5}$ exposure in areas with scarce observations. Developing countries of southern West Africa are facing a great challenge regarding air pollution mitigation strategies, and more investigations on personal exposure and related potential health effects using a cohort method will be considered in the future. In the short term, developing and implementing appropriate preventive and control measures on anthropogenic combustion sources downtown, such as improving waste treatment equipment at landfill or making smoking equipment for domestic use more efficient, are appropriate actions.

Data availability. Requests for data sets and materials should be addressed to Hongmei Xu (xuhongmei@xjtu.edu.cn).

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Author contributions. HX and CL conceived and designed the study. HX, J-FL, CL and BG contributed to the literature search, data analysis and interpretation and writing of the paper. J-FL, CL, BG, VY, AA, KH, SH, ZS and JC contributed to revision of the paper. HX, J-FL, EG, JA and LL conducted the particulate sample collection and chemical experiments and analyzed the experimental data.

Competing interests. The authors declare that they have no conflict of interest.

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