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

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Abstract

Urbanization is an issue strongly emerging in southern West African (sWA). There is a lack of full understanding on chemical compositions and personal exposure (PE) levels to fine particulate matter ($PM_{2.5}$) and its health risks related to various anthropogenic sources in this region. In this study, PE $PM_{2.5}$ was for the first time studied in dry (January) and wet (July) seasons of 2016 to characterize the contributions of Domestic Fires (DF) to women and Waste Burning (WB) to students in Abidjan, Côte d'Ivoire, and Motorcycle Traffic (MT) to drivers in Cotonou, Benin.

The average PE $PM_{2.5}$ mass concentrations were 331.7±190.7, 356.9±71.9 and 242.8±67.6 μg m^{-3} at DF, WB and MT for the women, students and drivers, which were 2.4, 10.3 and 6.4 times of the ambient $PM_{2.5}$ concentrations, respectively. Elevated PE $PM_{2.5}$ levels in dry season were found at DF (358.8±100.5 μg m^{-3}), WB (494.3±15.8 μg m^{-3}) and MT (335.1±72.1 μg m^{-3}), on average 15% higher than that at DF and 55% higher at both WB and MT in 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%), strongly influenced by the waste burning emission. This results in the highest non-cancer risks on heavy metal for 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 exposure concentration to particulate polycyclic aromatic hydrocarbons (PAHs) at DF (77.4±47.9 ng m^{-3}) was 1.6 and 2.1 times, respectively, of that for students at WB (49.9±30.7 ng m^{-3}) and for drivers at MT (37.0±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 5 times exceed 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±335.2 ng m^{-3}) owing to obvious waste burning activities nearby. The drivers' exposures to fossil fuel combustions markers of hopanes in PE $PM_{2.5}$ at MT (50.9±7.9 ng m^{-3}) was 3.0-3.3 times of those for women at DF (17.1±6.4 ng m^{-3}) and students at WB (15.6±6.1 ng m^{-3}).

Overall, the current study shows that wood combustion, waste burning, fugitive dust and

motor vehicle emissions were the dominated sources for the PE $PM_{2.5}$ and mainly contributed to its toxicities. The exposure to heavy metals of Pb and Mn had high non-cancer risks to students at WB, while 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 standard of residents in this region.

Keywords: personal exposure to $PM_{2.5}$; domestic fires; waste burning; motorcycle traffic; southern West Africa

1. Introduction

The southern West Africa (sWA) region is experiencing an economic upturn. Anthropogenic emissions of air pollutants have been increasing since the last few years, leading to poor air quality to the areas (IMF, 2017; Norman et al., 2007). Fine particulate matter ($PM_{2.5}$ with equivalent aerodynamic diameters ≤ 2.5 μm) is one of the major concerns from international organizations and publics because of its high health impacts by personal exposures (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 pollutant, biomass burning and mixture aerosols are significantly associated with under-five and maternal mortality in Africa. However, studies on $PM_{2.5}$, especially direct personal exposure (PE) tests to $PM_{2.5}$ (non-stationary 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 Desert 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) among the 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 dry season were dust (26-59%), primary organic matters (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 that the increase of relative importance of particulate emissions from domestic fires and fossil fuel combustions in Africa. Uncertainty has been raised by the residents who live in urban areas as they do concern on the health impact from air quality. However, the works on PE to $PM_{2.5}$ 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 in 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, ecosystem and agricultural productivity. The information will be gathered and discussed with policy-makers, scientists, operational centers, students and general publics. The current work in the framework of the Work Package 2 "Air Pollution and Health" of DACCIWA tends to link emission sources, air pollution and health impacts over representative differentiated urban sources: domestic fires and waste burning in Abidjan (Ivory Coast) and two-wheel vehicle emission in Cotonou (Benin) for different groups of populations.

Smoking meat (e.g., fish and pork) by biomass fuels (wood) is an important diet pattern for residents of coastal countries in 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. Besides, urbanization leads explosive population growth and rural depopulation in sWA, generating a huge amount of urban domestic wastes. 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 and appropriate treatment method, a large amount of air pollutants was thus emitted from the combustion and stacking of waste. Such phenomenon damages the living environment and harm the residents' health (especially for children) in Abidjan (UNEP, 2015). Moreover, in most low-GDP countries, motorbike taxi is 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 working hours daily, 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 strong impact on $PM_{2.5}$ physicochemical characteristics, but also cause health risks. Typical trace toxic chemicals, such as heavy metals and polycyclic aromatic hydrocarbons (PAHs), in $PM_{2.5}$ have various health damages to humans (Cao et al., 2012; WHO, 1998; Xu et al., 2015). For instance, Pb is a neuro-developmental metal which affects children health and mental development (USEPA, 2006; Xu et al., 2017). Several PAHs are teratogenic and carcinogenic for humans (Tang et al., 2008). Up to now, only few studies have investigated chemical compositions of PE $PM_{2.5}$ samples, and little is known regarding the sources and health risks in sWA region. This poses a challenge on 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 sWA area in 2016. Study objectives include 1) to characterize the PE to $PM_{2.5}$ from different typical local anthropogenic sources by chemical component and $PM_{2.5}$ mass balance analysis;

2) to identify potential pollution sources to different exposed populations from fingerprint of organic markers; and 3) to evaluate the PE to $PM_{2.5}$ health risks by the United States Environmental Protection Agency (U.S. EPA) health risk assessment model. This information offers scientific understanding of the PE to $PM_{2.5}$ 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 participants selection

PE to $PM_{2.5}$ (hereafter defined as PE $PM_{2.5}$) filter samples were collected using portative devices in unique source-dominated environments for different target groups of humans, including Domestic Fires (DF) for women and Waste Burning (WB) for students in Abidjan, Côte d'Ivoire, and Motorcycle Traffic (MT) for drivers in Cotonou, Benin (Figure 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 a high level of industrialization and urbanization in 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 24-30ºC) and an average of relative humidity (RH) >80% throughout a year.

DF site in Abidjan is located in the market of Yopougon-Lubafrique (5°19.7′ N, 4°6.4′ W) where is a large courtyard with about 25 fireplaces (Figure 2). The major fuel used is essentially hevea wood, which is a kind of local rubber trees. Several female adult workers were employed for grilling meat and/or roasting peanuts from 06:00 to 15:00 UTC (working time) in the working day. In this study, we selected two healthy, non-smoking female workers (an average age of 32.5 years old) to conduct personal exposure to $PM_{2.5}$ from domestic fire and related sources such as grilling (Figure 2). WB site in Abidjan is near the public landfill of Akouédo (5°21.2′ N, 3°56.3′ W), which has received all the wastes collected from Abidjan for the past 50 years (Figure 2). We selected two healthy and non-smoking primary school students (an average age of 11 years old) who live and study next to 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, 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 (Figure 2). It is largely dominated by a mass of emissions from motorcycle traffic (two-wheel vehicle powered by petrol, also named zemidjan in local

language) and a small quantity of other motor vehicles. We chose two healthy and non-smoking 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 by charcoal and butane gas as fuel (Figure S1abc) and daily household cleaning. One student participator (student A, boy, 8 years old) at WB did not involve in cooking activities at home [cooking energy is charcoal and liquefied petroleum gas (LPG)] (Figure S1ac), but another student (student B, girl, 14 years old) is usually responsible for household cooking with burning solid fuels (wood) (Figure 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 drove on road almost all the working time and returned home for meals. They did not participate any cooking at home (energy source for cooking is charcoal) (Figure S1a).

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

Twelve-hour time integrated (daytime: 07:30 to 19:30 UTC; nighttime: 19:30 to 07:30 on the next day UTC) PE $PM_{2.5}$ samples were collected in two major southwestern African cities (Figure 1) during dry season (from January 6^{th} to 11^{th}) and wet season (from July 5^{th} to 10^{th}) in 2016. PE $PM_{2.5}$ sampling was conducted for three consecutive days synchronously using the PEM (Personal Environmental Monitor) sampling devices with SKC pump (SKC Inc., USA) at a flow rate of 10 liter per minute (lpm). The PEM $PM_{2.5}$ sampling head worn in the breathing zone of participants in this study. $PM_{2.5}$ were collected on 37 mm pre-baked quartz filters (800°C, 3 hours, QM/A®, Whatman Inc., UK). A total of 72 PE samples, including 24 samples (12 pairs of diurnal samples, the same as thereafter) 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 not identical 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 compositions 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 Felix Houphouet Boigny Airport (Abidjan) and Cardinal Bernadin Gantin International Airport (Benin). The daily average air temperature, wind speed and rainfall accumulation are summarized in Table 1 as well.

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 hour. 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 by 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 mins) 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 in a pre-combustion at 340°C under O_2 for 2 h in order to remove OC; step 2 consisted in the oxidation of the remaining EC at 1200 °C under O_2. The difference (TC-EC) yielded 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 a separate 15 mL vials containing 10 mL distilled-deionized water (18.2 MΩ resistivity). The vials were placed in an ultrasonic water bath and shaken with a mechanical shaker for 45 min (15 min × 3 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 by an ion chromatograph (IC) analyzer (Dionex-600, Dionex, Sunnyvale, CA, USA), which is equipped with an AS11-HC anion column 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 ten heavy metals (i.e., V, Cr, Mn, Co, Ni, Cu, Zn, Sb, Ba and Pb) in PE $PM_{2.5}$ samples were determined by Energy Dispersive X-Ray Fluorescence (ED-XRF) spectrometry (the PANalytical Epsilon 5 ED-XRF analyzer, Netherlands) with a quarter of 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 5.2-13.9%. 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 shown in Table 5) by an in-injection port thermal desorption-gas chromatography/mass spectrometry (TD-GC/MS) method. The approach has the advantages of shorter sample preparation time (< 1 min), minimizing of contaminations from solvent impurities, and higher sensitivity, compared with the traditional solvent extraction-GC/MS method. The detail 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 trace contamination levels (<5.0%) of PE $PM_{2.5}$ samples 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; Wang et al., 2017a; Xu et al., 2018a). In this study, four heavy metals (Mn, Ni, Zn and Pb) and all measured PAHs and PAEs species in PE $PM_{2.5}$ were selected to determine the PE inhalation health risks (Xu et al., 2018a). The heavy metals 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 (USEPA, 2004, 2011). The average daily exposure dose (D) via inhalation was estimated to assess the risk by the equations (1) as follows:

$$D = (C \times R \times EF \times ED \times cf) / (BW \times AT) \qquad (1)$$

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

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

$$HQ = D / RfD \quad (2)$$

the threshold value of RfD indicates whether there is an adverse health effect during a certain period. Hazard index (HI) can be obtained by summing up the individual HQ to estimate the total non-cancer risks. If the HI < 1, then non-carcinogenic effect is impossible; HI ≥ 1, adverse health effect might likely 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 with D as equation (3):

$$ILCR = D \times CSF \quad (3)$$

for cancer risk, the value of 1×10^{-6} is an 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 nineteen PAHs, BaP has been used as an indicator of PAHs carcinogenicity (Wang et al., 2006). The carcinogenic health risk of PAH species can be assessed by $[BaP]_{eq}$ instead (Yassaa et al., 2001) by equation (4):

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

Additionally, the carcinogenic risk for PAEs was assessed by DEHP, which is identified as a possible carcinogen to humans by the International Agency for Research on Cancer (IARC) (IARC, 1982; Li et al., 2016). The definitions and recommended values of the parameters in equations (2-4) are also shown in Table 2 and 3.

2.5. Questionnaire and time-activity diary

Questionnaire (Supporting information (SI) A-C) and time-activity diary (SI 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, personal information, family status, dermatological, asthma symptoms, medical history, current health status and so on were first asked from each participant. Besides, 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 hour and daily travel mode/time); and (3) 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) 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) affected by the motorcycle emission and household air pollution sources.

The time-activity diaries requested the participants to mark on half an hour basis (sleeping time excluded) to assess each microenvironment time spending and detailed activities.

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±67.6 μg m^{-3} for women at Domestic Fires (DF), students at Waste Burning (WB) and drivers at Motorcycle Traffic (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, ~40% higher than that of the drivers. PE $PM_{2.5}$ ranged from 106.2 μg m^{-3} (nighttime in dry season, January 7th) to 1164.7 μg m^{-3} (daytime in wet season, July 5th) for women at DF; from 37.8 μg m^{-3} (nighttime in wet season, July 8th) to 1137.0 μg m^{-3} (daytime in dry season, January 11th) for students at WB; and from 65.0 μg m^{-3} (nighttime in wet season, July 11th) to 648.5 μg m^{-3} (daytime in dry season, January 15th) 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 lead to a drastic fluctuation for 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±72.1 μg m^{-3} in dry season (January), and 304.6±284.5, 219.5±71.3 and 150.6±10.4 μg m^{-3} in wet season (July) for women at DF, students at WB and drivers at MT, respectively (Table 4). Compared to dry season, the reduction rate of PE $PM_{2.5}$ for women at DF in wet season was approximately 15%, while the sharp reductions by more than 50% were observed for students and drivers. PE $PM_{2.5}$ concentrations reducing could be attributed to the occurrence of increased levels of rainfall in wet season in sWA (Table 1), which causes the large reduction of road dust exposed to drivers and limits the garbage spontaneous

combustion significantly around students. Moreover, large scale transport of mineral dust and combustion aerosols emitted by savannah wild fires contribute significantly to the aerosol load during the dry season (Djossou et al., 2018), which is more important at WB and MT than at DF (women worked in the 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 Figure 3). The 12-hour averaged PE $PM_{2.5}$ concentrations showed day/night (D/N) ratios of 3.4 (3.8 in dry season and 3.1 in wet season, the same sequence thereafter), 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 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 lower average in dry season but higher in wet season. Wet season high D/N ratios attribute to the increase in precipitation in Cotonou (Table 1), especially during nighttime (Sealy et al., 2003). This leads the lower PE $PM_{2.5}$ for drivers at night after aerosol scavenging. Shorter driving time in wet season is another explanation for the phenomenon, because of unfavorable weather occasionally (e.g., rain and storm).

The 5-h $PM_{2.5}$ average personal exposure concentration was 1574 μg 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 DF site in this study (1164.7 μg m^{-3}, daytime in wet season, July 5th), and was 4.7 times of the daily average PE $PM_{2.5}$ concentration in dry and wet seasons (331.7±190.7 μg m^{-3}). Student (10-17 years old) $PM_{2.5}$ exposures ranged from less than 10 μg m^{-3} to more than 150 μg m^{-3} (mean 56 μg m^{-3}) in four neighborhoods in Accra, Ghana (Arku et al., 2014), much lower than that for students at WB site (356.9±71.9 μg m^{-3}). It can be seen that the high exposure of students in this study is likely to be related to the waste burning emissions, while there was no obvious strong $PM_{2.5}$ emission source in the study of Arku et al. (2014).

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 similar sampling period. The average PE $PM_{2.5}$ were 3.0 and 2.0 times of the ambient values at DF, and 6.1 and 8.8 times 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 dry and 10.5 in wet seasons. Such large PE/A ratios are probably due to the impact of waste combustion affected the respiratory exposure of residents, especially on children; on the other hand, high PE/A ratios can be attributed to the fact that WB site is located in the lowest living quality region of Abidjan, where the simplest stove and non-qualified wood as fuel used in house (Figure S1d). These lead to an extremely high PE $PM_{2.5}$ indoors during the cooking time (especially for student B who was 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) that cause 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 $PM_{2.5}$ were 3.7 and 1.2 times of 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 fix-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 μg m^{-3}) and composition (24.4%) in women PE $PM_{2.5}$ samples were the highest among the three types of PE participants, due to their direct contact with the ignition, and close to the solid fuel (wood in this study) burning and meat roasting at the workplace and also their own residential units. However, the EC concentrations (8.4-10.5 μg m^{-3}) and compositions (3.0-3.5%) were very similar among the three different PE groups, representing 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; Cao et al., 2008; Li et al., 2009; Tian et al., 2017; Watson et al., 2001) summarized that average OC/EC characterizes 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 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 wet season than dry season, ascribed to the fact that the higher RH in 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. PE of women displays the higher (an average of 13.9) and more scattered OC/EC than those collected from students and drivers in wet season (Figure 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 for the participants were 1.2 and 2.5 times of the ambient OC/EC 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 individual's activities and potentially 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 on 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±5.0 μg 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 with 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, following 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 on 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 atmospheric oxidation of SO_2 emitted mainly from coal and diesel combustions (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 for lowering sulfur emissions and therefore decreasing 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, indirect 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; Zhang et al., 2014b) displayed a high absolute average concentration of 3.4 μg m^{-3} and composition of 14.5% in women' PE $PM_{2.5}$ samples, confirming their distinct exposure from biomass burning during the roasting at the workplace. To the best 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). Additional consideration includes 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.

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±0.2 μg 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 heavy metal for the students was 1.8 and 3.9 times of those for the women and drivers, mainly due to the emissions from garbage combustion at landfill (Wang et al., 2017b). 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 from the students and strong disturbances from landfill workers. Another reason is 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 of ~73% of total quantified elemental concentration in all samples. Ba took up a decisive advantage over other elements, having a contribution of >50% for students. It is usually added in 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 personal exposure elements for drivers at MT which are mainly derived from the motor oil additive, tyre wear and brake pads worn (Zhao and Hopke, 2006).

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

Calculation of 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 unresolved fraction (Figure 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. Unresolved fraction may include water and other undetected substances. For OM, since there is no full organic composition profiles for the PE $PM_{2.5}$, a conversion factor 1.4 (1.4 corrects the organic carbon mass for other constituent associated with the organic carbon molecule) is generally used (Turpin and Lim, 2001) to quantify OM by the equation (5):

$$OM = 1.4 \times OC \qquad (5)$$

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 for PE OC for women at DF. According to the information gathering 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% in 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 by equation (6):

$$GM = (1/4.0\%) \times Fe \qquad (6)$$

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 disturbance of motor vehicles and human

activities, construction dust from uncovered construction sites, and the dusts generated from burning, could be the dominant sources to PE $PM_{2.5}$ in this study. OM and GM showed the 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 for 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 in exceedingly low proportions to PE $PM_{2.5}$ for all groups. This reconfirms the limited contribution to PE $PM_{2.5}$ from secondary ionic formation again.

In Figure 5, evident diurnal distinguishes can be observed on the two major chemical compositions of OM and GM. GM exhibited the 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 dry season because of the harmattan haze introduced mineral dusts and the lack of precipitation increasing road dust resuspension. Moreover, OM showed the equal or lower proportions in the daytime (25.0%) that nighttime (30.0%), relative to the meteorological parameters (i.e., factor affected the formation of secondary organic carbonaceous aerosol) and diurnal changes of combustion sources around subjects. An exception is that OM proportion of women PE $PM_{2.5}$ at daytime (50.8%) was much higher than nighttime (38.2%) in wet season, due to the influences from the damp wood burning at the working time. Burning biomass fuel with high moisture often results in low combustion efficiency, 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 the damp wood led to higher OC emission than dry wood, in-line with the observation for women PE results in this study.

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

Organic fingerprint markers can be used to indicate specific emission sources and further characterize the pollutions impacted on 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±1.0 ng m^{-3}, respectively, representing high organic pollutions in sWA region (Table 5). Dissimilar with the trend on $PM_{2.5}$ masses (students > women > drivers), the PE to target

organic compounds for 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 Figure 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 maker of low temperature combustion, such as wood burning) was 11.6±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±4.5 ng m^{-3}) and benzo[ghi]perylene (BghiP) (6.4±0.5 ng m^{-3}), respectively, indicating the contributions from the 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±54.8 ng m^{-3}) and drivers at MT (44.6±10.8 ng m^{-3}) in wet season were 326% and 52% higher than those in dry season (29.4±5.6 and 29.4±4.4 ng m^{-3} respectively), while ΣPAHs in wet season (36.8±15.7 ng m^{-3}) was 42% lower than that dry season (62.9±45.0 ng m^{-3}) for students at WB. The dramatic increase in PE to PAHs for women is mainly due to raise of moisture content in the wood used for grilling meat in wet season, promoting more PAHs emission from wood combustion processes (Shen et al., 2013). The restraint of waste combustion in wet season is the main factor for the lower of PE to $PM_{2.5}$-bound PAHs at landfill, in accordance with seasonal pattern on PE $PM_{2.5}$ mass. Fanou et al. (2006) measured the PE PAHs concentrations in Cotonou and found that the PAHs level associated with particles ranged from 76.21 to 103.23 ng m^{-3} for 35 taxi-moto drivers in March 2001. Our values for drivers at MT site was 50-64% lower than their values, suggesting that the exposure to PAHs for the motorbike drivers in this region has been improved.

In Figure 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 dry season but lower in wet season. Both the PAH profiles were featured with 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 ratio in dry and wet seasons were 0.8 and 0.3, respectively. The higher PE ΣPAHs concentrations at night and lower D/N ratios for drivers may be

explained by the potential combustion sources for PAHs close to the participants (e.g., sources nearby the drivers' homes) in Cotonou, Benin especially in wet season. This can be deduced by the combustion marker of BaP which was the highest PAH species at night in wet season, even though the drivers exposed to the traffic emissions during the night working time (18:30 to 21:00 UTC). Further studies are thus required to confirm the findings and figure out the reasons.

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, n = 3) for open wood fires in households in the Njombe district of Tanzania, which was much higher (~65 times) than the women exposure PAHs at DF site in the current research. The highest 12-h exposure PAHs for women at DF site in this study was 469.7 ng m^{-3} (daytime in wet season, July 6th), approximately one-tenth of the PAHs concentration from open wood fires in Tanzania mentioned above. The large PE PAH concentrations 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 (Figure 7), indicating 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 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 of 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 particles-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 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, respectively. This represents that more favorable meteorological conditions (i.e., higher light intensity) and more physical activities (i.e., time extending for particulate re-suspension) at daytime are

more conducive to the aging of $PM_{2.5}$ and its bounded PAHs. Moreover, IcdP/(BghiP+IcdP) 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 combustions, 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 combustions were more dominant for women at DF (0.52) and students at WB (0.52) (Figure 7). 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 from the matrix evaporation and combustion (Gu et al., 2010; Wang et al., 2017a). The PE levels of PAEs could be mainly attributed to the usage of the household products, painting material, plastic waste incineration and municipal sewage release (Zhang et al., 2014a). 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±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 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 (Figure 6B). The elevated ΣPAEs for students can be ascribed to the combustion of plastic products at landfill nearby. Our results are similar to the previous studies conducted in Xi'an and Tianjin, China (Kong et al., 2013; Wang et al., 2017a). The ΣPAEs ranged from 376.6 to 1074 ng m^{-3} outdoors, and from 469.2 to 1537 ng m^{-3} in classrooms (Wang et al., 2017a), where DEHP and DBP were also the most abundant PAEs with a sum of composition of 68% and 73% of the ΣPAEs outdoor and indoor, respectively.

The average concentrations of the ΣPAEs for women at DF, students at WB and drivers at MT were comparable in dry season. However, the average concentrations were 927.2±154.9, 1929.8±340.4 and 594.6±16.6 ng m^{-3} in wet season, 1.1, 2.3 and 0.7 times of the ΣPAEs in dry season (Figure 6B). A significant increase in PE ΣPAEs for student at WB can be attributed to the enhanced PAEs emission in the daytime with high RH (3173.6±1028.3 ng m^{-3}), consistent with the findings on PE $PM_{2.5}$. Dry and wet seasons had

similar PAEs profiles with different diurnal variations (Figure 6B). The average D/N ratios of the ΣPAEs in 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 wet season. Noticeably different diurnal D/N ratios for students at WB is interrelated with the human activities (specially the emissions from plastic materials) and the subdued waste combustion led by meteorological conditions (i.e., more precipitation at night in wet season), which had been mentioned in Section 3.1.1.

4.3. Hopanes

Hopanes are markers for fossil fuels (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±7.9 ng m^{-3}, 2.0 and 2.3 times higher than the women (17.1±6.4 ng m^{-3}) and students (15.6±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 consequence health issues. The Σhopanes showed unobvious seasonal variations for three types of PE participants. The Σhopane in dry season were 0.9, 1.8 and 0.7 times of those in wet season. Even though the Σhopane varied among three sites, their profiles on individual species were similar. 17α(H)-21β(H),30-norhopane (αβ-NH) and 17α(H)-21β(H)-hopane (αβ-HH) were two most abundant hopanes in all PE $PM_{2.5}$ samples, with the average concentrations of 6.0 and 6.5 ng m^{-3} and the compositions of 21.4% and 23.3% of the Σhopanes, respectively (Table 5 and Figure 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 DF, D/N ratio was both 2.0 in dry and wet seasons, with Σhopanes of 24.0±11.1 and 12.2±5.0 ng m^{-3} for daytime and nighttime in dry season, and 21.4±17.5 and 10.9±3.6 ng m^{-3} in 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±16.4 ng m^{-3} for daytime and nighttime in dry season, and 74.2±16.3 and 6.5±1.7 ng m^{-3} in wet season. It is notable that the daytime concentrations for drivers were comparable between the two seasons, while the nighttime hopanes in 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 the combustion activities, plastics emissions and fossil

fuel emissions (e.g., from gasoline vehicles), respectively, well matching to the potential air pollution sources impacted on 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 the environmental pollution sources and individual activities, but also prove reliable application of organic tracers on characterization of personal exposure study.

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 way 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, Hazard Quotient (HQ) of Pb in wet season for students at WB was the highest (2.95×10^{-2}), which suggests that Pb non-carcinogenic risk to children is more severe in that area compared with other participants and metals. There was no consistent difference on the risks between dry and wet seasons, except Ni which showed much greater value in wet than dry season for both participants. Counting the total of four toxic heavy metals, Hazard Index (HI) 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 of 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 sWA region.

As shown in Table 6, the ILCRs of PAHs were all exceed the international acceptable level of 1×10^{-6} either in dry or wet season. Meanwhile, the ILCRs of PAEs were all below 1×10^{-6}, well within the safety limit of cancer risk. For all target participants, higher cancer risks of PE $PM_{2.5}$-bound PAHs and PAEs were found in wet season. The seasonal variations such as increase of RH could lead raise of PE cancer risks to toxic organics in $PM_{2.5}$. In dry season, the average ILCR values of PAHs for women and drivers were comparable, both ~50% lower than those for students, implying the high toxicity originated from the waste burning and high sensitivity to juveniles. In wet season, PAHs exhibited the highest ILCR for women at DF, 2.5 and 2.7 times of those for students and drivers, respectively. The domestic wood

burning and meat grilling can trigger nearly ten times the safety limit for PAHs. The cancer risks of PAEs showed the similar trend 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 was 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 the individual's health and subsequently alerting a need of effective emission control in sWA. Even though PAEs had low carcinogenic risks, the effects from waste burning to 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, there are a variety of emission sources impacted on the different degrees of impacts on the population groups in sWA region. Attention should be paid on health risks for chemicals via inhalation way, especially Pb and Mn for students at WB site as well as PAHs for women at DF in wet season.

6. Conclusions

This work can be regarded as a first attempt for the assessment of personal exposure to particulate matter originating from main sources of combustion aerosols in representative cities of southern West Africa. We targeted in this study different groups of people exposed to domestic fires, traffic and waste burning. Even though there are few drawbacks such as relatively short sampling period and limited number of participants, our findings provide a new insight on the health risk 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 strategy and more investigations on personal exposure and related potential health effects by cohort method will be considered in the future. In the short terms, developing and implementing appropriate preventive and control measures on anthropogenic combustion sources downtown such improving waste treatment equipment at landfill or efficient smoking equipment for domestic use, are appropriate.

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Author Contributions

H.X. and C.L. conceived and designed the study. H.X., J.-F.L., C.L. and B.G. contributed to the literature search, data analysis/interpretation and manuscript writing. J.-F.L., C.L., B.G., V.Y., A.A., K.H., S.H., Z.S. and J.C. contributed to manuscript revision. H.X., J.-F.L., E.G., J.A and L.L. conducted the particulate samples collection and chemical experiments, analyzed the experimental data.

Additional Information

Fig. S1 and SI A-D accompany this manuscript can be found in Supplementary Information.

Competing financial interests

The authors declare no competing financial interests.

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Figure Caption:

Figure 1. Location of the sampling sites (white square) within the cities. A: Domestic Fires (DF) site at the Yopougon-Lubafrique market in Abidjan; B: Waste Burning (WB) site at the landfill of Akeoudo in Abidjan; and C: Motorcycle Traffic (MT) site at Dantokpa area in Cotonou.

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

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

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, 1^{st}, 25^{th}, 50^{th}, 75^{th}, 99^{th} and max percentiles).

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.

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.

Figure 7. Correlations between PAHs diagnostic ratios (average ratio points of each type participant indicate day and night value respectively).

Figure 1.

Figure 2.

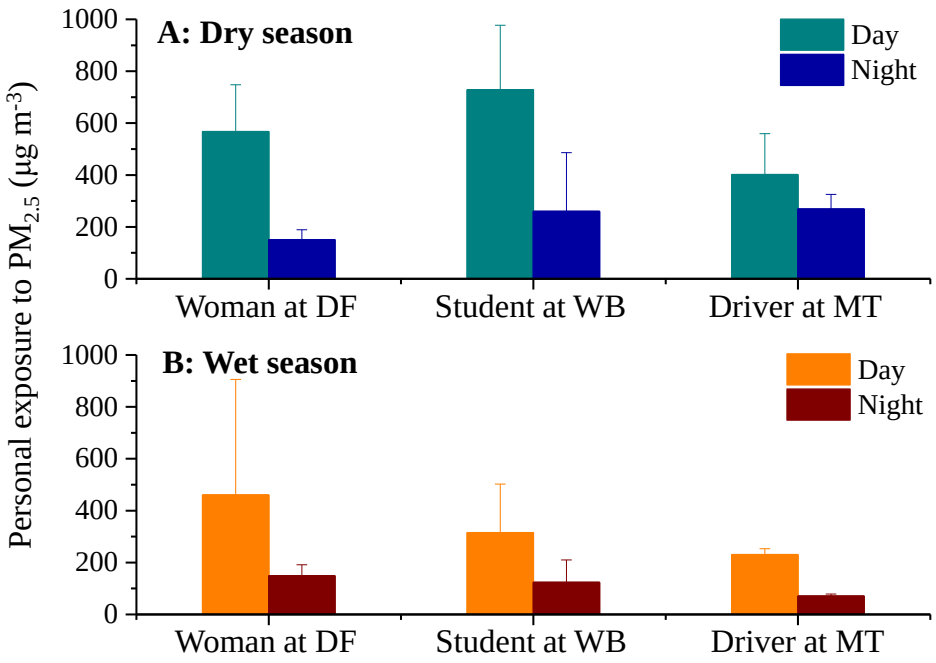

A: Dry season
Day
Night
1000
800
600
400
200
0
Personal exposure to PM2.5 (μg m-3)
Woman at DF
Student at WB
Driver at MT
B: Wet season
Day
Night
1000
800
600
400
200
0
Woman at DF
Student at WB
Driver at MT

Figure 3.

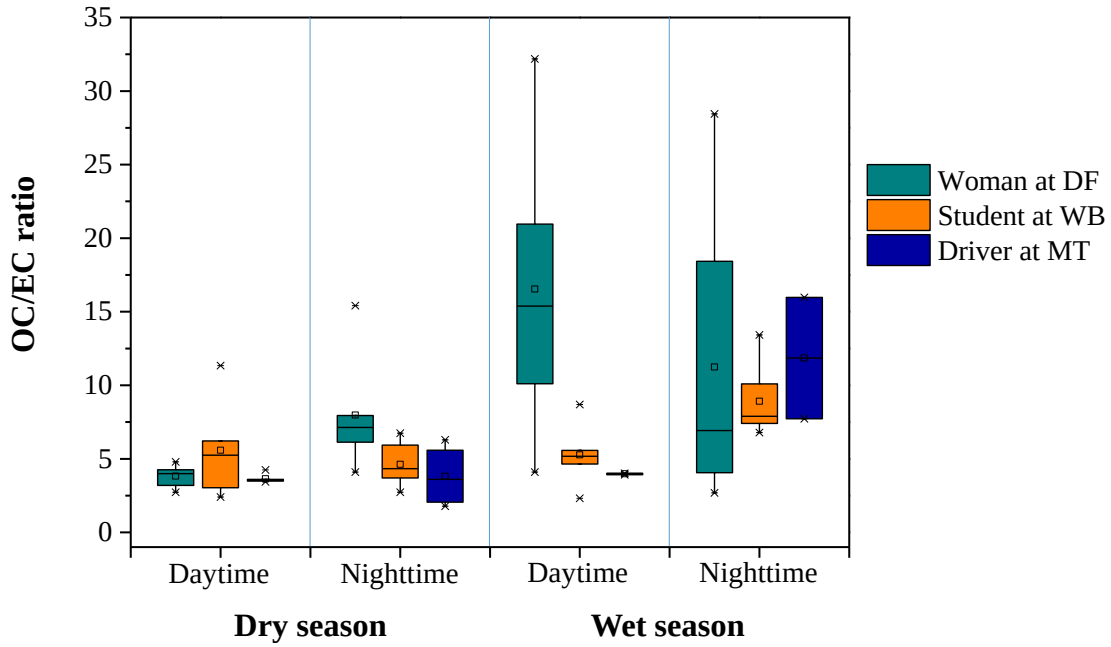

OC/EC ratio
35
30
25
20
15
10
5
0
Daytime
Nighttime
Daytime
Nighttime
Dry season
Wet season
Woman at DF
Student at WB
Driver at MT

Figure 4.

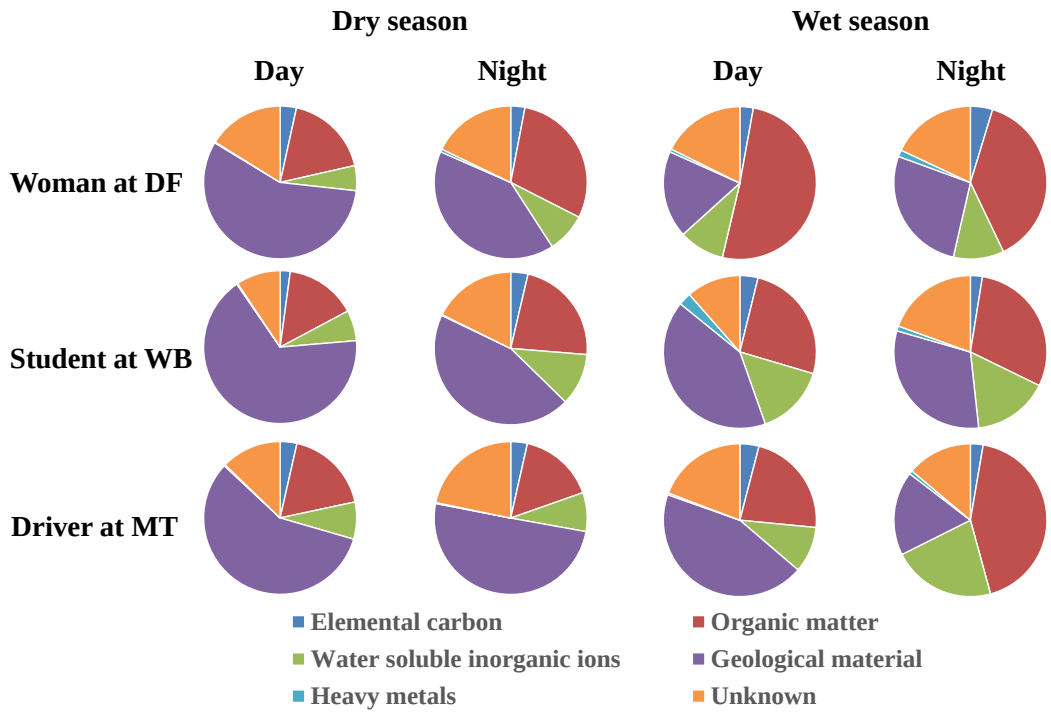
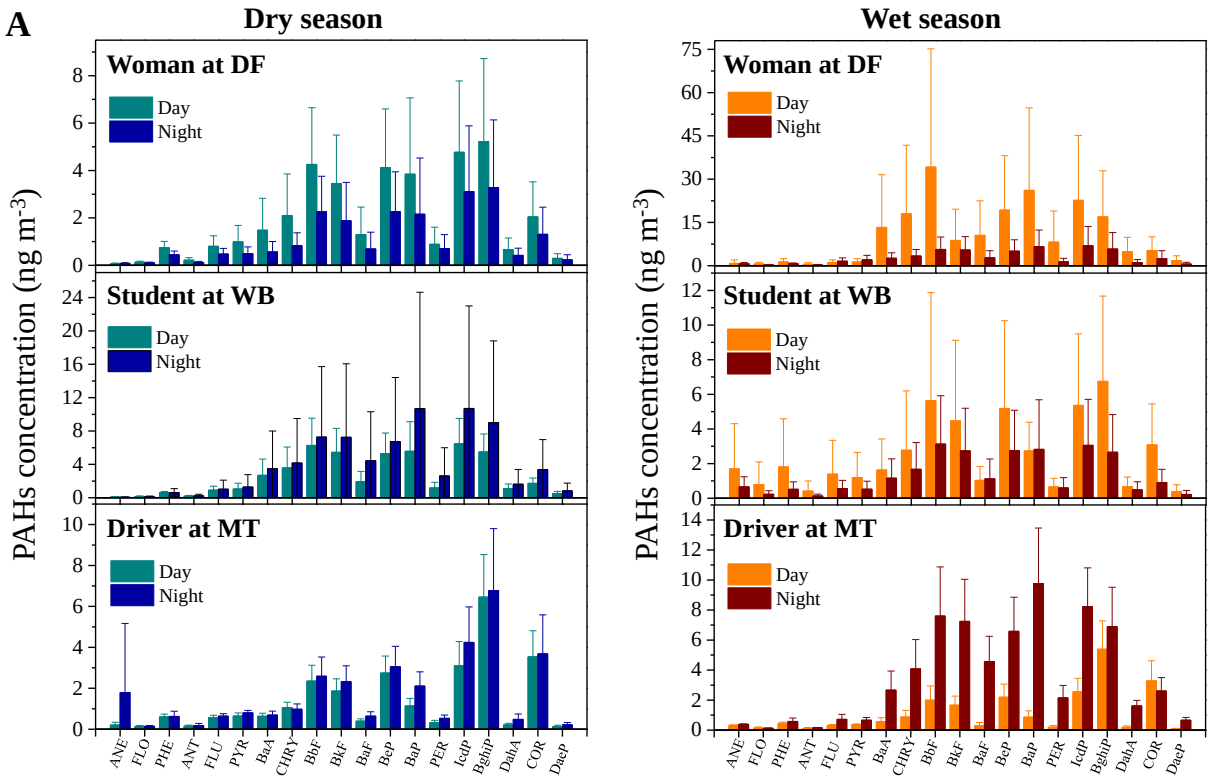
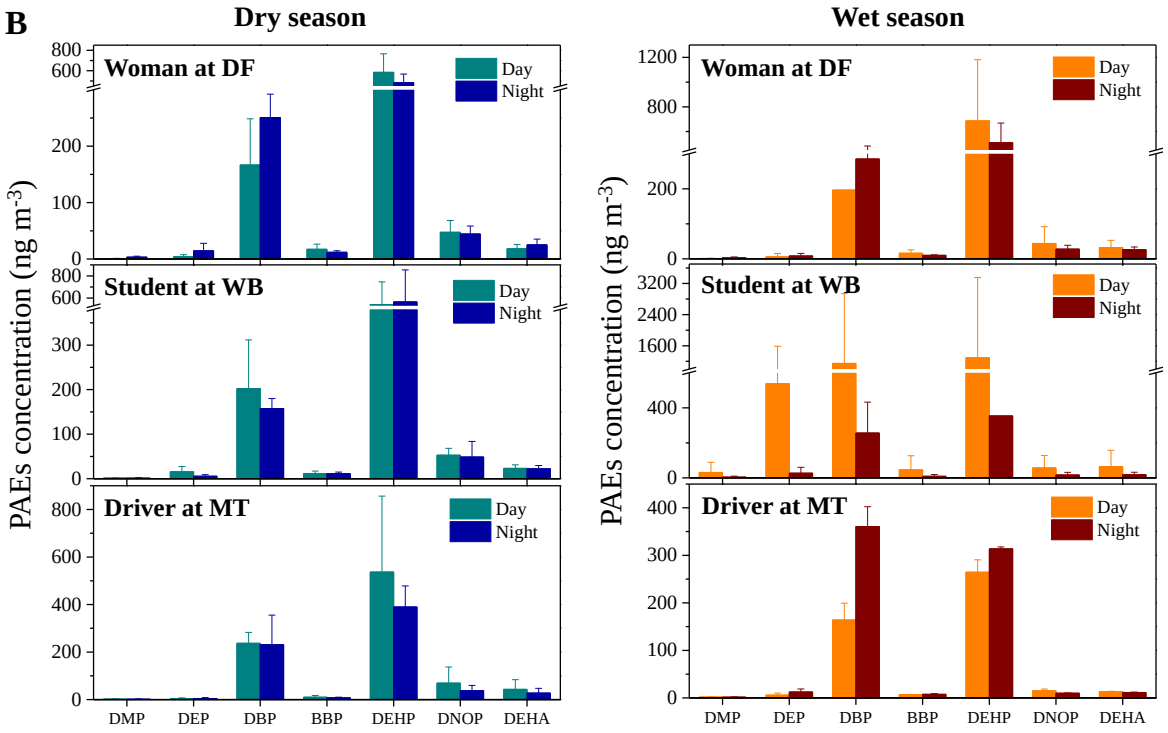

Dry season
Wet season
Day
Night
Day
Night
Woman at DF
Student at WB
Driver at MT
Elemental carbon
Organic matter
Water soluble inorganic ions
Geological material
Heavy metals
Unknown

Figure 5.



1129



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C

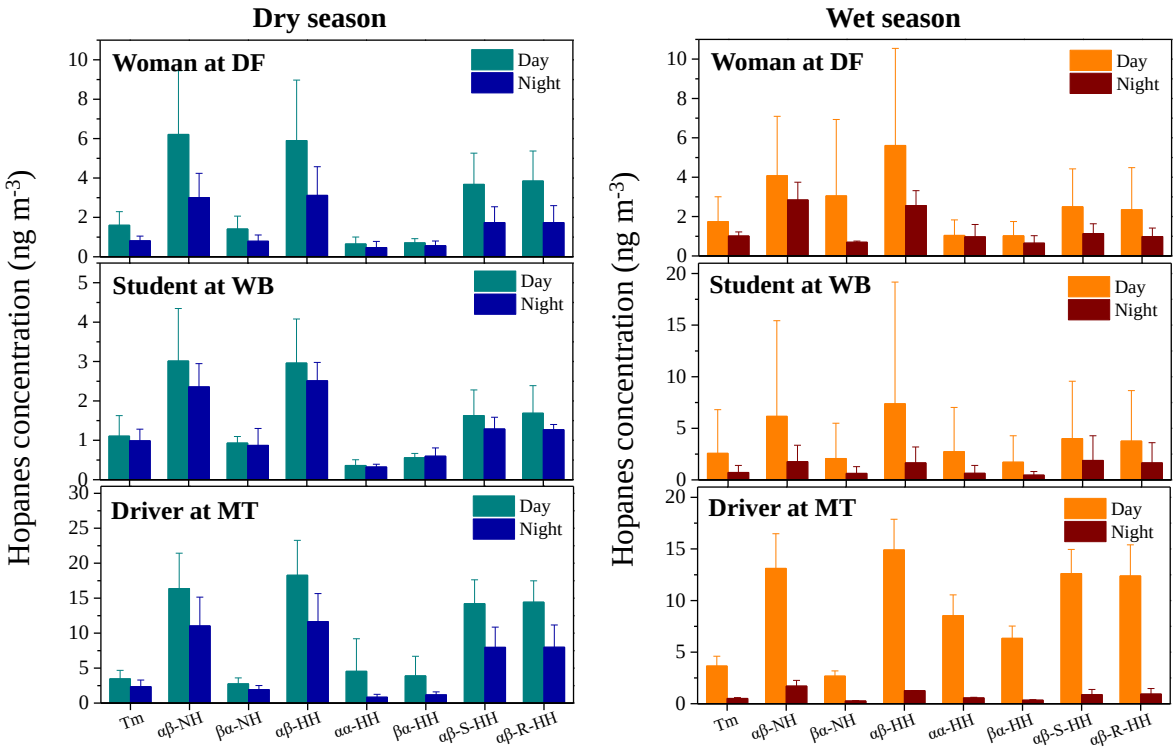
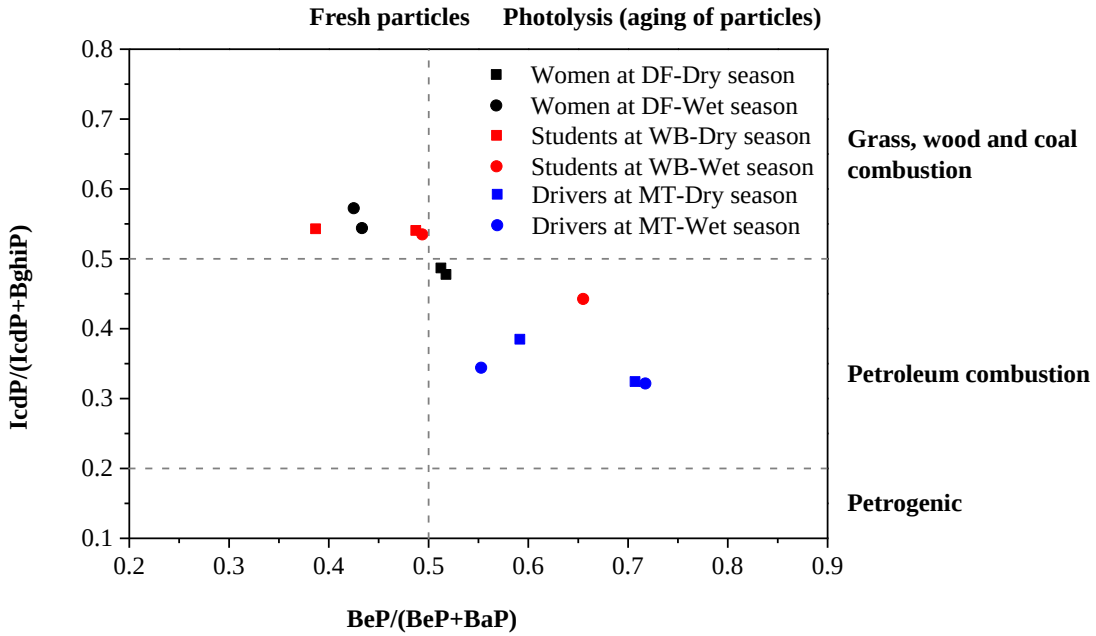

Dry season
Wet season
Woman at DF
Student at WB
Driver at MT
Day
Night
Hopanes concentration (ng m-3)
Tm
αβ-NH
βα-NH
αβ-HH
αα-HH
βα-HH
αβ-S-HH
αβ-R-HH

Figure 6.



1135

1136 **Figure 7.**

1137

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

Table 2 Definitions and recommended values of the parameters in equations (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 metals concentrations in equations (ng m^{-3})	/
R	inhalation rate, air volume a child inhaled each day (m^3 day^{-1})	16.0 for women and drivers; 15.2 for students (USEPA, 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[a]; 37.5 for students[a]; 85.0 for drivers[a]
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 on human 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)

a: Measured in this study.

Table 3 Reference dose (RfD) (mg kg^{-1} day^{-1}) and cancer slope factor (CSF) (mg kg^{-1} $day^{-1})^{-1}$ via inhalation exposure way used 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	USEPA, 2011
DEHP	/	0.014	USEPA, 1997; Wang et al., 2017a

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

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

Specific species (abbreviation)	Women at DF		Students at WB		Drivers at MT	
	Average	Stdev*	Average	Stdev*	Average	Stdev*
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 (αβ-NH)	4.0	1.2	3.3	4.1	10.6	1.9
17β(H)-21α(H),30-norhopane (βα-NH)	1.5	1.8	1.1	1.5	1.9	0.3
17α(H)-21β(H)-hopane (αβ- HH)	4.3	1.9	3.6	5.4	11.5	2.2
17α(H)-21α(H)-hopane (αα-HH)	0.8	0.2	1.0	2.0	3.6	2.1
17β(H)-21α(H)-hopane (βα-HH)	0.7	0.2	0.8	1.2	2.9	1.2
17α(H)-21β(H),(22S)-homohopane (αβ-S-HH)	2.3	0.7	2.2	2.4	8.9	1.3
17α(H)-21β(H),(22R)-homohopane (αβ-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**

*: standard deviation

Table 6 Non-cancer risks of heavy metals and cancer risks of PAHs and PAEs via inhalation exposure way 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