



Relationships between Outdoor and Personal Exposure of Carbonaceous Species and Polycyclic Aromatic Hydrocarbons (PAHs) in Fine Particulate Matter (PM_{2.5}) at Hong Kong

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ABSTRACT

Personal and ambient fine particulate matter (PM_{2.5}) samples were simultaneously collected at Hong Kong during winter in 2014. Mass concentration, organic carbon (OC), elemental carbon (EC), and polycyclic aromatic hydrocarbons (PAHs) relationships were analyzed. The correlations of personal and ambient concentrations of PM_{2.5}, OC, and EC indicated the ambient concentrations were the factors showing influences on the personal exposures. Personal to ambient (P/A) ratios in PM_{2.5}, OC, and EC were all > 1, suggesting influences between indoor sources and/or personal activities. Significant higher ambient ΣPAHs concentrations with P/A ratios were nevertheless < 1. The Σ15 U.S. EPA priority PAHs accounted for 50.6% and 70.8% of ΣPAHs in personal and ambient samples, respectively. The ratios of indicator compounds confirmed the origin of PAHs in personal PM_{2.5}, which were found to be associated predominantly with traffic emissions and the influence by the indoor sources.

Keywords: Personal exposure; Fine particulate matter; Carbonaceous aerosol; Polycyclic aromatic hydrocarbons.

INTRODUCTION

Fine particulate matter with an aerodynamic diameter less than or equal to 2.5 μm (PM_{2.5}) is an environmental issue subject to major health concern. The fine particles have been observed to be associated with numerous adverse human health effects (Ito *et al.*, 2006; Kim *et al.*, 2015). The fine particulate matter can penetrate into the deepest section of human lungs (alveolar) and diffuse to other target extrapulmonary organs causing notable symptoms, including cardiac and respiratory morbidity and mortality (Pope III *et*

al., 2002; Analitis *et al.*, 2006). Typically, epidemiological studies are based on ambient air quality data collected from outdoor stationary monitoring sites. However, people spend most of their time in indoor microenvironments (> 85%) at urbanized areas (Williams *et al.*, 2000; Jahn *et al.*, 2013); the adverse health effects of PM_{2.5} may not only be caused by ambient origin particles but also indoor pollutants (Cao *et al.*, 2005; Baumgartner *et al.*, 2011). In modern day individual's PM_{2.5} exposures and their relationships with the corresponding ambient concentrations have been studied in many developed countries (Janssen *et al.*, 1998; Williams *et al.*, 2000; Noullett *et al.*, 2006; Johannesson *et al.*, 2007) and a few Chinese cities (Du *et al.*, 2010; Jahn *et al.*, 2013).

Although positive associations between PM_{2.5} exposures and human health effects have generally been reported, the magnitudes of associations in different geographic locations can vary between locations (Jahn *et al.*, 2011). Variation in chemical components of PM_{2.5}, such as organic carbon (OC), elemental carbon (EC), and polycyclic aromatic hydrocarbons (PAHs) have been proposed as the links to different adverse

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human health outcomes (Kim *et al.*, 2013; Baumgartner *et al.*, 2014). OC and EC are the most important chemical components in PM_{2.5} especially in highly urbanized areas (Cao *et al.*, 2004). PAHs are products of incomplete combustion processes and ubiquitous in the atmosphere (Guo *et al.*, 2003). PAHs comprise only in a small fraction of composition in the PM_{2.5} mass; however, they are one of the most important pollutants of concerns due to their abilities to persist in environment, bioaccumulation properties, carcinogenic, and mutagenic effects (Machala *et al.*, 2001; Boström *et al.*, 2002).

Atmospheric fine particulate pollution in Hong Kong is mainly due to the emissions from motor vehicles, urban construction, industries, and trans-boundary pollution from the Pearl River Delta (PRD) region (Louie *et al.*, 2005; Ho *et al.*, 2006). Previous studies were mainly focused on characterization PM_{2.5} mass in indoors and outdoors (Naumova *et al.*, 2002; Cao *et al.*, 2005; Lazaridis *et al.*, 2008; Xu *et al.*, 2015), while studies with personal exposures targeting carbonaceous species and hazardous organic compounds (e.g., PAHs) are still very limited. Reliable scientific data on personal exposure to PM_{2.5} as well as their chemical characterization are essential to evaluate the potential human health effects in Hong Kong. The overall aim of the present work was to quantify personal exposure levels in different modes of conditions (outdoor and personal). This study was also targeted to enable reflection of factors specific to the individual modes.

The objectives of this study are to: (1) assess personal PM_{2.5} exposures (and their hazardous chemical components) and the associations between personal exposures and ambient concentrations in Hong Kong; (2) investigate PAHs abundance and specification in ambient and personal PM_{2.5}; (3) characterize the potential sources of PAHs in personal PM_{2.5}.

METHODS

Ambient and Personal PM_{2.5} Sampling

Ambient PM_{2.5} samples were collected on the rooftop (1.5 m above the ground) of Industrial Centre building at The Hong Kong Polytechnic University (PolyU campus), Hung Hom (HH) from 15th to 19th of February 2014 and on the rooftop of Shaw Auditorium at Prince of Wales Hospital, The Chinese University of Hong Kong, Shatin (ST) from 25th to 28th of February 2014. The Industrial Centre building at PolyU campus is a building equipped with a complete collection of engineering facilities such as additive manufacturing, digital manufacturing, electronics, intelligent automation, composites, fabrication, building services, safety, construction, design realization and aviation services all catered for research and project activities. It is assumed that the results are independent from the sampling locations. Twenty-four hour (24 h) integrated ambient PM_{2.5} samples were collected by mini-volume air samplers (Airmetrics, Eugene, OR, USA) on 47 mm quartz fiber filters. The sampler was equipped with a cyclone that separated the particles with a diameter less than 2.5 μm at a flow rate of 5 L min⁻¹.

Personal PM_{2.5} samples were collected along with the

above sampling schedule. Nine non-smoker adult subjects (6 females and 3 males, aged 21–42 years at recruitment) residing in different areas of Hong Kong (adjacent to HH and ST) participated in the personal sample collection. The nine recruited participants were college students and office workers. A Leland Legacy Pump (SKC, Inc., Eighty-Four, PA, USA) was connected with a PEM (Personal Environmental Monitor) loaded with one quartz filter (37 mm, Pall Tissuquartz Filter, Pall Corporation, Ann Arbor, MI, USA), which was carried by subjects during each sampling campaign. Flow check (before and after each sampling) was performed by connecting a PEM loaded with filter to a DryCal[®] air flow meter for calibration purpose. The air purge through the filters was set at a flow rate of 10 L min⁻¹ for the Leland/PEM samplers and collected a total air volume of 14.4 m³ after 24 h. Two to five samples were collected from each of the subjects and a total of 35 valid personal PM_{2.5} samples were obtained in this study.

A collocated sampling test was conducted to ensure the PEMs were comparable with mini-volume air samplers. Ambient PM_{2.5} samples were simultaneously collected with a mini-volume air sampler and three collocated PEMs. The average personal PM_{2.5} ($26.0 \pm 7.8 \mu\text{g m}^{-3}$) loaded with quartz filters showed comparable value with the ambient PM_{2.5} samples ($27.5 \pm 8.8 \mu\text{g m}^{-3}$). The deviation of the PEMs loaded with quartz filters expressed in coefficient of variance (CV = standard deviation/mean (%)) ranging from 0.7% to 5.7%.

Sample Preparation and Analysis

Gravimetric Analysis

All quartz fiber filters were pre-heated to 900°C for 3 h before sample collection in order to remove any organic contaminants. An average of triplicate filter weights ($\pm 3 \mu\text{g}$) were determined by a balance (Model MC-5; Sartorius AG, Goettingen, Germany, capacity range of 0.1 mg–5.1 g with sensitivity up to $\pm 1 \mu\text{g}$) before and after equilibration with no less than 24 h prior initial filter weighing and post-sample weighing under pre-conditioned temperature ($23 \pm 2^\circ\text{C}$) and humidity ($40 \pm 5\%$) in a controlled weighing room. All quartz fiber filters (47 and 37 mm) were stored in a drying box (relative humidity < 40%) prior to sample collection. After sample collection, loaded filters were stored in the refrigerator (-20°C) until further chemical analysis.

Carbonaceous Species Analysis

OC and EC were analyzed (on a 0.526 cm² punch) by thermal analysis with optical detection following the IMPROVE protocol on a Desert Research Institute (DRI) Model 2001 Thermal/Optical Carbon Analyzer (Atmoslytic Inc., Calabasas, CA, USA) (Cao *et al.*, 2003). The method detection limits for OC and EC were 0.8 and 0.4 μg C cm⁻², respectively, with a precision better than 10% of total carbon (TC). More information about OC/EC analysis can be found in Cao *et al.* (2004).

Polycyclic Aromatic Hydrocarbons (PAHs) Analysis

The in-injection port thermal desorption (TD) gas chromatography/mass spectrometry (GC/MS) method was

used in this study with the advantage of high sensitivity, short sample preparation time (< 1 min), and minimal contamination (Ho *et al.*, 2008; Ho *et al.*, 2011). Briefly, a range of 0.526–2.630 cm² (actual sample size depended on aerosol loading at each sample) of quartz fiber filters were used to determine the concentration of PAHs. All filter strips were cut in pieces and inserted in a TD tube with the same dimension as the GC/MS injector liner. The injector port temperature was pre-conditioned to 50°C before analysis and subsequently raised to 275°C for desorption in splitless mode at the same time when the GC oven temperature was maintained at 30°C. The GC separation and collection of mass spectrum started after the injector temperature was reached to 275°C. The calibrations were established in a range of 0.01–1.0 ng for each of the PAHs per analysis. The replicate analysis for each of ten samples had the reproducibility of < 15%, indicating that the good precision on the chemical analysis. Twenty-six PAH compounds (parent- and alkyl-PAHs) including acenaphthylene (3-ring), acenaphthene (3-ring), fluorene (3-ring), phenanthrene (3-ring), anthracene (3-ring), fluoranthene (4-ring), pyrene (4-ring), benz[a]anthracene (4-ring), chrysene (4-ring), benzo[b]fluoranthene (5-ring), benzo[k]fluoranthene (5-ring), benzo[a]fluoranthene (5-ring), benzo[e]pyrene (5-ring), benzo[a]pyrene (5-ring), perylene (5-ring), indeno[1,2,3-cd]pyrene (6-ring), dibenz[a,h]anthracene (6-ring), benzo[ghi]perylene (6-ring), coronene (6-ring), dibenzo[a,e]pyrene (6-ring), 2-methylnaphthalene, 1-methylnaphthalene, 2,6-dimethylnaphthalene, 9-methylnaphthalene, methylfluoranthene, and retene were measured. For the discussion purpose, the ΣPAHs denotes the summation of the measured 26 PAH compounds in this study. Further information about the TD-GC/MS method used in this study can be found in Ho *et al.* (2008) and Ho *et al.* (2011).

Statistical Analysis

Concentrations for PM_{2.5}, OC, and EC were reported in μg m⁻³. Concentrations for particle-bound PAHs were reported in ng m⁻³. Two independent samples t-test was used to assess the mass difference between two variables (e.g., ambient concentrations, personal exposures). Spearman's rank correlation coefficients were used to investigate the associations between personal and ambient concentrations of PM_{2.5} and each variable. The analytical results were statistically processed by the IBM SPSS Statistics 21 program (SPSS Inc., USA). All *p*-values were derived from 2-tailed statistic tests and a value of less than 0.05 was considered statistically significant.

RESULTS AND DISCUSSION

Personal Exposure to PM_{2.5} and Carbonaceous Aerosol

The average mass concentrations of PM_{2.5} and carbonaceous aerosols (OC and EC) in personal samples and the associated ambient results are summarized in Table 1. Individual's exposure to PM_{2.5} from all sampling days ranged from 16.2 to 65.1 μg m⁻³ with an average of 46.2 ± 18.8 μg m⁻³ (Table 1). The mean residential indoor and

Table 1. Ambient concentrations and individual personal exposure to PM_{2.5}, organic carbon (OC), elemental carbon (EC), and Σ26PAHs.

Subject ID	PM _{2.5} (μg m ⁻³) ^a	OC (μg m ⁻³)	EC (μg m ⁻³)	Σ26PAHs (ng m ⁻³)	OC (%)	EC (%)	PAHs (%)	N ^b
Subject P1	49.4 ± 18.0	9.2 ± 5.5	2.0 ± 1.2	0.60 ± 0.09	18.7 ± 6.7	4.5 ± 2.8	0.0014 ± 0.0006	5
Subject P2	47.8 ± 20.9	12.6 ± 5.9	2.4 ± 0.7	0.80 ± 0.13	27.2 ± 6.7	5.4 ± 1.6	0.0019 ± 0.0007	5
Subject P3	55.1 ± 16.7	13.4 ± 5.3	3.8 ± 2.0	0.79 ± 0.12	23.9 ± 2.0	6.7 ± 1.8	0.0015 ± 0.0003	3
Subject P4	16.2 ± 4.8	4.7 ± 1.0	1.2 ± 0.6	0.54 ± 0.09	29.5 ± 3.0	7.0 ± 2.2	0.0035 ± 0.0007	4
Subject P5	31.9 ± 10.7	7.8 ± 0.9	1.7 ± 0.7	0.55 ± 0.06	26.1 ± 6.9	5.5 ± 1.6	0.0019 ± 0.0006	4
Subject P6	65.1 ± 30.1	14.8 ± 1.2	2.6 ± 0.4	0.75 ± 0.27	25.0 ± 9.8	4.6 ± 2.8	0.0012 ± 0.0001	2
Subject P7	45.4 ± 16.5	9.9 ± 3.2	2.4 ± 0.6	0.95 ± 0.10	22.1 ± 2.2	5.6 ± 1.9	0.0023 ± 0.0009	4
Subject P8	49.8 ± 15.4	11.3 ± 4.8	2.3 ± 0.9	1.87 ± 0.69	22.4 ± 3.5	4.7 ± 0.9	0.0037 ± 0.0004	4
Subject P9	34.2 ± 2.0	8.4 ± 1.3	2.1 ± 0.6	1.27 ± 0.18	24.5 ± 3.6	6.2 ± 1.7	0.0037 ± 0.0006	4
Personal exposure	42.6 ± 18.8	9.9 ± 4.5	2.2 ± 1.0	0.90 ± 0.48	24.3 ± 5.6	5.6 ± 1.9	0.0024 ± 0.0011	35
Ambient concentration	33.8 ± 10.2	3.1 ± 1.8	1.9 ± 1.1	3.46 ± 0.93	8.6 ± 2.8	5.3 ± 1.8	0.0110 ± 0.0059	9

Notes: ^aAverage ± standard deviation; ^bN denotes number of sampling days.

outdoor PM_{2.5} concentrations in Hong Kong were in a range of 45.0–69.5 $\mu\text{g m}^{-3}$ (Chao and Wong, 2002). Time series plots of daily personal exposure to PM_{2.5} are shown in Fig. 1(a). Daily mean personal PM_{2.5} exposures were in a range of 22.9–74.4 $\mu\text{g m}^{-3}$ during the sampling period. The highest individual PM_{2.5} exposure (86.4 $\mu\text{g m}^{-3}$) was observed in the 25th of February. A previous study conducted at Gothenburg, Sweden showed personal exposure and urban background mean PM_{2.5} concentrations were 11.0 and 10.1 $\mu\text{g m}^{-3}$, respectively (Molnár *et al.* 2014). Intra-individual coefficient of variation for each subject ranged from 5.8% to 43.8% for those sampled for no less than 3 days (Table 1). The CV of individual's exposures between days ranged from 4.4% to 71.9% with an overall mean of 29.9% (Fig. 1(a)). As in Fig. 1(a), the lowest (14.9 $\mu\text{g m}^{-3}$) and highest ambient PM_{2.5} concentration (55.1 $\mu\text{g m}^{-3}$) were both observed at HH. In general, average ambient PM_{2.5} concentrations ($33.8 \pm 10.2 \mu\text{g m}^{-3}$) were considerably lower ($p = 0.014$) than individual's exposures during the measurement period (Fig. 2(a)). The significant lower ambient concentrations compared to personal exposures was likely the consequence of higher baseline personal exposure for the subjects, which could be due to higher indoor exposures at either the subject's apartment, other indoor

microenvironments or possibly in proximity of local sources and personal activities. Personal exposure to PM_{2.5} complied with the 24 h Hong Kong Air Quality Objectives of 75 $\mu\text{g m}^{-3}$, while 77% of personal measurements with 24 h average PM_{2.5} levels exceeded World Health Organization (WHO) air quality guideline (25 $\mu\text{g m}^{-3}$).

The average personal PM_{2.5} concentrations measured in the present study were compared with other studies (Table 2(a)). The average personal exposure to PM_{2.5} measured in the developed countries show lower values than in Hong Kong, such as in American cities (ranging from 8.4 to 44.8 $\mu\text{g m}^{-3}$) (Williams *et al.*, 2000; Larson *et al.*, 2004; Turpin *et al.*, 2007; Brinkman *et al.*, 2009), Canadian cities (varying from 18.0 to 22.0 $\mu\text{g m}^{-3}$) (Kim *et al.*, 2005; Noullett *et al.*, 2006), and European cities (winter = 25.1 $\mu\text{g m}^{-3}$; summer = 8.8 $\mu\text{g m}^{-3}$) (Zmirou *et al.*, 2000). In contrast, personal exposure to PM_{2.5} measured in Chinese cities usually have higher values (ranging from 45.4 to 122.4 $\mu\text{g m}^{-3}$) than that in Hong Kong.

Average individual OC and EC concentrations from all sampling days of personal monitoring are listed in Table 1. Time series plots of daily personal OC and EC exposures are shown in Figs. 1(b) and 1(c). The individual's exposure to OC and EC in PM_{2.5} varied from 3.4 to 22.7 $\mu\text{g m}^{-3}$ and 0.7

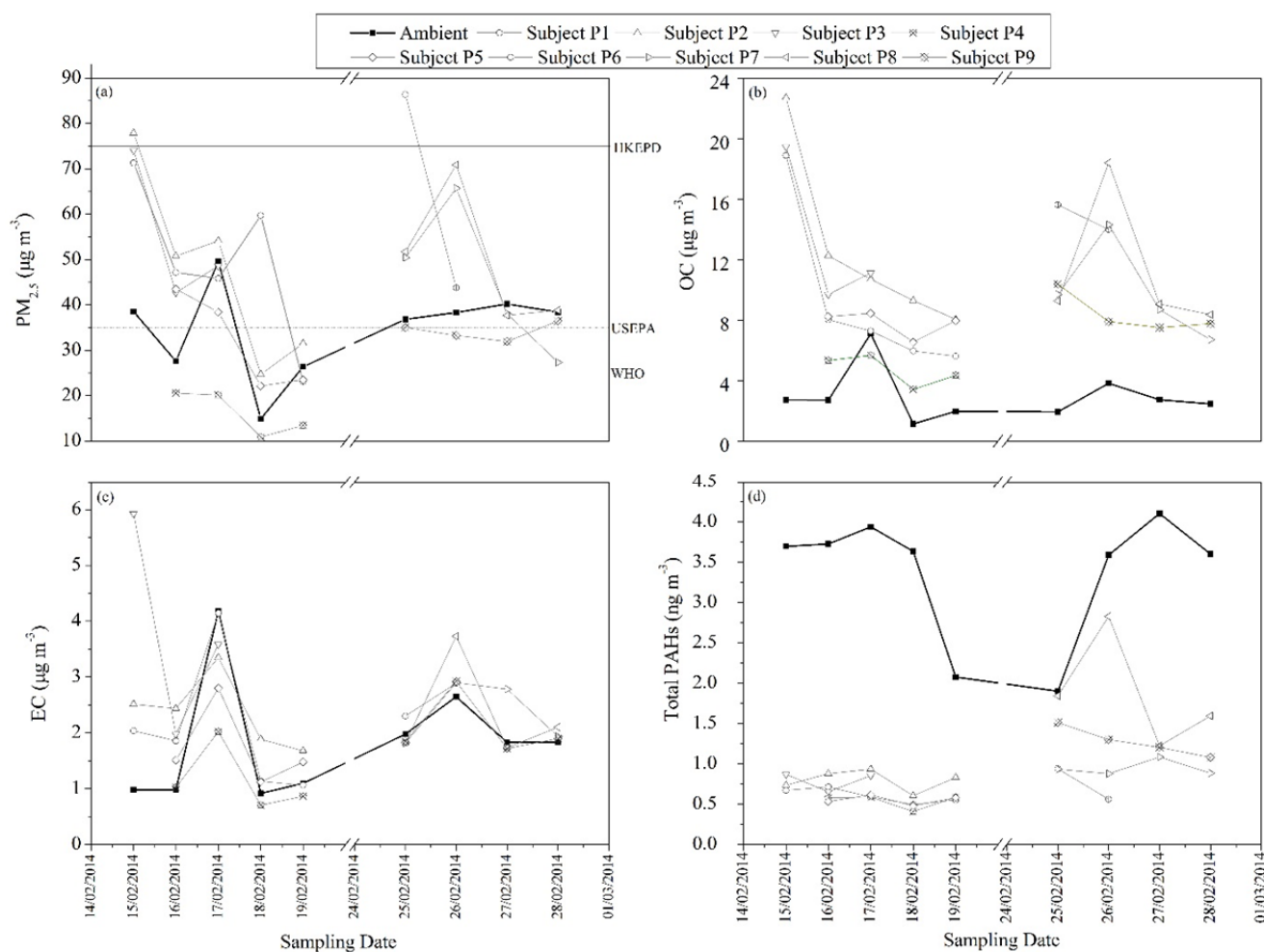


Fig. 1. Time series plots of daily personal exposure to (a) PM_{2.5}, (b) OC, (c) EC and (d) Σ PAHs.

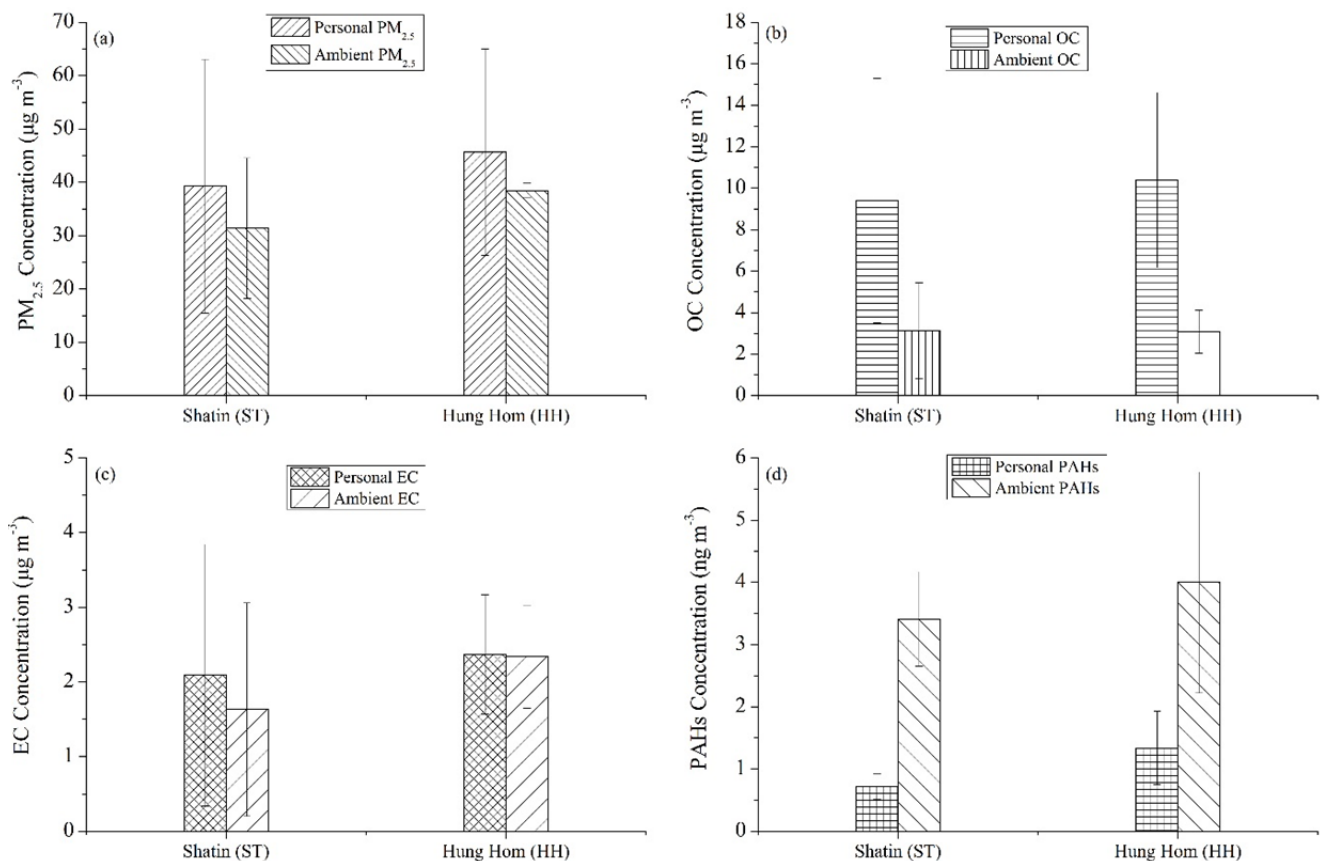


Fig. 2. Ambient and personal exposure to (a) $PM_{2.5}$, (b) OC, (c) EC, and (d) $\Sigma PAHs$ at Shatin (ST) and Hung Hom (HH) in Hong Kong.

to $5.9 \mu g m^{-3}$ with an average of $9.9 \pm 4.5 \mu g m^{-3}$ and $2.2 \pm 1.0 \mu g m^{-3}$, respectively. The average percentage of OC and EC in personal $PM_{2.5}$ were $24.3 \pm 5.6\%$ and $5.6 \pm 1.9\%$, which is consistent with the personal $PM_{2.5}$ exposures, as listed in Table 1. Intra-individual coefficient of variation for OC and EC for each subject ranged from 8.4% to 35.7% and from 18.2% to 61.7%, respectively. As listed in Table 2(a), personal OC and EC concentrations measured in other developed countries usually have considerably lower values than Hong Kong, such as personal OC and EC ranging from 5.4 to $8.3 \mu g m^{-3}$ and 0.2 to $1.4 \mu g m^{-3}$ in the United State. (Wilson *et al.*, 2000; Landis *et al.*, 2001; Turpin *et al.*, 2007; Brinkman *et al.*, 2009); personal EC varied from 0.6 to $1.0 \mu g m^{-3}$ in Toronto and Prince George, Canada (Kim *et al.*, 2005; Noullett *et al.*, 2006). Individual's OC and EC exposure in urban cities of China were considerably higher than that in Hong Kong (Du *et al.*, 2010; Zhang *et al.*, 2015).

As reported in Figs. 2(b) and 2(c), the average ambient OC and EC concentrations were 3.1 ± 2.3 and $1.6 \pm 1.4 \mu g m^{-3}$ at ST and 3.1 ± 1.0 and $2.3 \pm 0.7 \mu g m^{-3}$ at HH, respectively. It is noticed that OC concentrations in personal $PM_{2.5}$ (ranging from 4.7 to $14.8 \mu g m^{-3}$) observed from all subjects were significantly higher ($p < 0.001$) than OC ($3.1 \pm 1.8 \mu g m^{-3}$) in ambient samples (Figs. 1(b) and 1(c)). Relatively higher personal EC concentrations were observed compared to ambient EC while no significant mass difference

($p = 0.225$) was reported for EC. OC/EC ratio > 2 indicates the presence of secondary organic aerosol. Average ambient OC/EC ratio was 1.7 ± 0.6 during the same period when personal samples were collected (Table 1). Average OC/EC ratios varied from 4.2 to 5.9 for personal measurements with an average of 4.9 ± 1.2 , which suggested the presence of indoor sources of OC in personal samples. This is consistent with the result reported by Ho *et al.* (2004), which demonstrated the OC in indoor sources contributed 2 – $3 \mu g m^{-3}$ in the building near roadsides in Hong Kong.

Personal Levels of PAHs

A total of 26 particle-bound PAHs (parent- and alkyl-PAHs) were measured in this study. Daily personal exposure to $\Sigma PAHs$ ranged from 0.41 to $2.83 ng m^{-3}$ with an average of $0.90 \pm 0.48 ng m^{-3}$. As shown in Table 1, individual's exposure to $\Sigma PAHs$ from all sampling days varied between 0.54 and $1.87 ng m^{-3}$, and the average concentration of $\Sigma PAHs$ accounted for $0.0024 \pm 0.0011\%$ of personal $PM_{2.5}$ exposures. The results were all lower than workplace environment in natural rubber sheet factories contaminated by wood burning smoke (Choosong *et al.*, 2010). Intra-individual coefficient of variation for each subject ranged from 10.1% to 51.9% for those sampled no less than 3 days. The CV of personal exposure to PAHs between sampling days ranged from 9.6% to 71.9% with an overall mean of 29.9%.

Table 2. Concentrations of (a) personal exposure to PM_{2.5}, OC, EC, and Σ 26PAHs and (b) selected PAHs: comparison with other studies

Location (a)	Personal PM _{2.5}	Personal OC	Personal EC	Personal PAHs ^d	References
	$\mu\text{g m}^{-3}$	$\mu\text{g m}^{-3}$	$\mu\text{g m}^{-3}$	ng m^{-3}	
Shatin, Hong Kong ^e	39.3	9.4	2.1	0.721 (0.183 ^d)	This study
Hung Hom, Hong Kong ^e	45.7	10.4	2.4	1.338 (0.680 ^d)	This study
Guangzhou, China	45.4–92.5	N/A	N/A	N/A	(Jahn et al., 2013)
Beijing, China	118.5	N/A	11.0	N/A	(Du et al., 2010)
Tianjin, China (winter)	122.4	37.08	9.04	N/A	(Zhang et al., 2015)
Tianjin, China (summer)	74.7	30.07	2.99	N/A	(Zhang et al., 2015)
Baltimore, US	12.9	6.9 ^c	0.4 ^c	N/A	(Williams et al., 2000; Landis et al., 2001)
Boulder, US	8.39	8.38	0.15	0.69	(Brinkman et al., 2009)
Los Angeles, US	29.2	5.4 ^c	1.3 ^c	2.077 ^c	(Turpin et al., 2007)
Elizabeth, NJ, US	44.8	7.9 ^c	1.4 ^c	2.939 ^c	(Turpin et al., 2007)
Houston, US	37.2	7.2 ^c	0.7 ^c	3.081 ^c	(Turpin et al., 2007)
Seattle, US	10.5	N/A	1.44	N/A	(Larson et al., 2004)
Camden (WFS), NJ, US	N/A	N/A	N/A	2.25	(Zhu et al., 2011)
Camden (CDS), NJ, US	N/A	N/A	N/A	2.34	(Zhu et al., 2011)
Toronto, Canada	22.0	N/A	0.6	N/A	(Kim et al., 2005)
Prince George, Canada	18.0	N/A	1.0	N/A	(Noullet et al., 2006)
Berlin, Germany	N/A	N/A	N/A	3.98 ^c	(Fromme et al., 2004)
Baden, Germany	N/A	N/A	N/A	1.03 ^c	(Fromme et al., 2004)
Grenoble, France (Summer)	8.8	N/A	N/A	1.00	(Zmirou et al., 2000)
Grenoble, France (Winter)	25.1	N/A	N/A	8.46	(Zmirou et al., 2000)

Compound, Mean (ng m ⁻³)	(b) Boulder		Los Angeles ^c		Elizabeth ^c		Houston ^c		Camden (WFS)		Camden (CDS)		Berlin ^c		Baden ^c		Grenoble (Summer)		Grenoble (Winter)		Shatin		Hung Hom	
Pyrene	0.028	0.057	0.072	0.095	0.97	1.03	0.67	0.25	0.25	0.67	0.29	0.29	0.29	0.29	0.29	0.29	0.29	0.29	0.29	0.29	0.29	0.29	0.29	0.29
Benzo[a]anthracene	0.014	0.037	0.088	0.062	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Chrysene	0.061	0.16	0.28	0.46	0.47	0.53	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55	0.55
Benzo[b]fluoranthene	0.09	0.19	0.32	0.2	0.14	0.12	0.12	0.12	0.12	0.12	0.12	0.12	0.12	0.12	0.12	0.12	0.12	0.12	0.12	0.12	0.12	0.12	0.12	0.12
Benzo[k]fluoranthene				N/A	0.1	0.1	0.37	0.1	0.1	0.37	0.1	0.1	0.37	0.1	0.1	0.37	0.1	0.1	0.37	0.1	0.1	0.37	0.1	0.1
Benzo[e]pyrene	0.044	0.11	0.14	0.08	N/A	N/A	0.4	N/A	N/A	0.4	N/A	N/A	0.4	N/A	N/A	0.4	N/A	N/A	N/A	0.4	N/A	N/A	0.4	N/A
Benzo[a]pyrene	0.036	0.074	0.14	0.072	0.13	0.13	0.27	0.1	0.13	0.27	0.1	0.13	0.27	0.1	0.13	0.27	0.1	0.13	0.27	0.1	0.13	0.27	0.1	0.13
Perylene	0.005	0.019	0.029	0.022	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Indeno[1,2,3-cd]pyrene	0.032	0.28	0.32	0.29	0.08	0.08	0.33	0.08	0.08	0.33	0.08	0.08	0.33	0.08	0.08	0.33	0.08	0.08	0.33	0.08	0.08	0.33	0.08	0.08
Dibenz[a,h]anthracene	N/A	N/A	N/A	N/A	N/A	N/A	0.05	N/A	N/A	0.05	N/A	N/A	0.05	N/A	N/A	0.05	N/A	N/A	N/A	0.05	N/A	N/A	0.05	N/A
Benzo[ghi]perylene	0.057	0.34	0.37	0.25	0.07	0.07	0.35	0.07	0.07	0.35	0.07	0.07	0.35	0.07	0.07	0.35	0.07	0.07	0.35	0.07	0.07	0.35	0.07	0.07
Coronene	0.028	0.36	0.36	0.35	0.09	0.08	0.16	0.08	0.08	0.16	0.08	0.08	0.16	0.08	0.08	0.16	0.08	0.08	0.16	0.08	0.08	0.16	0.08	0.08
Dibenz[ace]pyrene	0.295	0.45	0.82	1.2	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Summation ^d	0.69	2.077	2.939	3.081	2.25	2.34	3.98	1.03	2.34	3.98	1.03	2.34	3.98	1.03	2.34	3.98	1.03	2.34	3.98	1.03	2.34	3.98	1.03	2.34

Notes: N/A denotes data not available; N.D. denotes data non-detectable; ^cDenotes these measurements were indoor concentrations, not personal exposure. ^ddenotes the summation of the selected PAHs listed in Table 2b; ^eThe samples were collected on February of 2014.

Time series plots of daily Σ PAHs for ambient concentrations and personal exposures are shown in Fig. 1(d). Considerably higher Σ PAHs were found in ambient samples than that in personal samples on each sampling day with significant mass difference ($p < 0.01$). The ambient site at HH is located approximately 100 m away from the main traffic road (Guo *et al.*, 2003). Spatial variations of ambient Σ PAHs were measured (Fig. 2(d)) with Σ PAHs concentrations observed at HH ($4.00 \pm 1.77 \text{ ng m}^{-3}$) and ST ($3.14 \pm 0.76 \text{ ng m}^{-3}$). It is noticed that subjects at HH exposed to higher personal Σ PAHs compared to the subjects at ST.

The personal exposure to PAHs (selected PAH compounds) measured in the present study were compared with the PAH compounds reported in other personal and/or indoor studies. Summation of the selected personal PAHs listed in Table 2(b) were found to be comparable with those reported in some of the urban cities, such as Boulder (0.69 ng m^{-3}) in the U.S. (Brinkman *et al.*, 2009) and Grenoble (1.00 ng m^{-3}) in France during summertime (Zmirou *et al.*, 2000). Generally, it was found that personal exposure to Σ PAHs in Hong Kong were considerably lower than those measured in America cities with heavy industries (e.g. refineries) nearby (varying from 2.077 to 3.081 ng m^{-3}) (Turpin *et al.*, 2007; Zhu *et al.*, 2011) and the cities in Europe (ranging from 3.98 to 8.46 ng m^{-3}) (Zmirou *et al.*, 2000; Fromme *et al.*, 2004). Overall, personal exposure to Σ PAHs in Hong Kong were considerably lower than those obtained in other studies in different countries. The differences in personal exposure to PAHs could be due to location of the sampling sites, different sampling times, in addition of different emission sources. Since the sampling sites in this study are located in university areas with minimal industrial actions, the lower Σ PAHs levels could be possibly attributed to these conditions. In addition, gas-particle partitioning is important in determining the transport, degradation and fate of organic contaminants in environment. The mass fraction for the particulate phase PAHs showed for molecular weights < 202 were less than 20%, whereas for molecular weights > 229 were over 90% (Huang *et al.*, 2014). The absent of gas phase sampling could have also affected the overall outcome of the analysis.

The concentrations of each quantified PAH compound in personal and ambient samples are shown in Table 3. The summation of particle-bound PAHs groups and their percentage contribution to Σ PAHs in personal samples are illustrated in Fig. 3. Among the 26 particle-bound PAHs measured, 2-Methynaphthalene (0.04 – 0.52 ng m^{-3}), chrysene + triphenylene (0.02 – 0.24 ng m^{-3}), and fluoranthene (0.02 – 0.19 ng m^{-3}) were the most abundant compounds and accounted for 25–50% of the concentration of Σ PAHs in personal samples (Table 3). The average concentrations of $\Sigma 15$ U.S. EPA priority PAHs in ambient samples were about four times higher than that in personal samples (Table 3), and the $\Sigma 15$ U.S. EPA priority PAHs accounted for $50.6 \pm 7.8\%$ and $72.8 \pm 2.0\%$ of Σ PAHs in personal and ambient samples, respectively.

High molecular weight PAHs (HMW-PAHs) are predominately associated with particulates while the low molecular weight PAHs (LMW-PAHs) resided in the

gaseous-phase (Ho *et al.*, 2009). Parent-PAHs are thermodynamically more stable than alkyl-PAHs. During combustion compounds formed at high temperatures, alkyl-PAHs are depleted as temperature increases; whereas alkyl-PAHs (e.g., 3-ring alkyl-PAHs) that generated at low temperatures, are abundant in petroleum (Douglas *et al.*, 1996; Saha *et al.*, 2009). On average, the concentrations of PAH compounds in personal samples were shown in Table 3. The average Σ HMW-PAHs/ Σ LMW-PAHs ratio in personal samples was 1.5, whereas significantly higher value (9.6) was reported in ambient samples (Table 3), indicating LMW-PAHs was easily transported from outdoors and accumulated in indoors. Moreover, this suggests higher contribution of vehicle emissions (with a higher percentage contribution of Σ HMW-PAHs to Σ PAHs) in ambient samples than personal samples.

Fig. 3(a) shows concentrations of the 15 U.S. EPA priority PAHs in the personal samples. Seven of these priority pollutant PAHs (termed Group 2B) are considered probable human carcinogens (IARC, 2012; Paolini *et al.*, 2016). As shown in Fig. 3(a), the highest Σ Group 2B PAHs and Σ HMW-PAHs exposures were observed in subject P8 ($0.66 \pm 0.24 \text{ ng m}^{-3}$) and subject P9 ($0.37 \pm 0.06 \text{ ng m}^{-3}$), both residing in HH. The concentrations of Σ LMW-PAHs were similar between ambient and personal samples in the recruited subjects (Fig. 3(a)).

Relationship between Personal Exposure and Ambient Concentrations

The personal/ambient (P/A) ratio is an indicator of the difference between personal exposure and ambient concentrations (Noullett *et al.*, 2010). P/A ratio greater than unity indicates personal exposure related sources (e.g., personal activities and/or indoor sources) were stronger than ambient sources, whereas weak personal exposure sources demonstrated the ratio less than unity (Wilson and Brauer, 2006). In this study, the P/A ratios for $\text{PM}_{2.5}$ and their chemical compositions (OC, EC, and particle-bound PAHs) were investigated to assess the difference between ambient concentrations and personal exposures. As shown in Fig. 4(a), daily P/A ratio of $\text{PM}_{2.5}$ ranged from 0.41 to 4.02 with an average of 1.31. The overall average P/A ratios of OC and EC were greater than unity (3.89 and 1.39, respectively). $\text{PM}_{2.5}$ and EC P/A ratios > 1 were reported in other studies (Williams *et al.*, 2000; Noullett *et al.*, 2006; Jahn *et al.*, 2013). It was calculated that residents in Hong Kong (personal activities data retrieved from a 40 subjects survey) spent an average of 71.5% ($\pm 24.5\%$) and 71.5% ($\pm 23.3\%$) of their daily time at home for students and office workers, respectively. A considerable portion of time was spent at school ($16.7\% \pm 17.5\%$) and in the office ($19.2\% \pm 18.7\%$) with the rest of the daily activities divided between transportation (2.4–3.9%) and outdoors (2.0–2.9%). Similar results were reported in the previous study in Hong Kong, on average, the subjects spent more than 86% of their time indoors, 3–7% in transit and 3–7% in outdoors (Chau *et al.*, 2002).

A different outcome was shown for PAHs in this study, however, the Σ PAHs P/A ratio ranged from 0.12 to 0.87

Table 3. Ambient concentration and personal exposures of PAHs in Hong Kong.

	Personal (ng m ⁻³)		Ambient (ng m ⁻³)	
	Mean ± S.D. ^f	Range	Mean ± S.D. ^f	Range
<i>Parent-PAHs</i>				
Acenaphthylene#	0.013 ± 0.005	0.003–0.032	0.029 ± 0.010	0.019–0.066
Acenaphthene#	0.010 ± 0.005	0.001–0.040	0.002 ± 0.002	N.D. ^d –0.005
Fluorene#	0.012 ± 0.002	0.005–0.024	0.023 ± 0.013	0.013–0.053
Phenanthrene#	0.043 ± 0.013	0.020–0.097	0.101 ± 0.025	0.063–0.138
Anthracene#	0.067 ± 0.022	0.023–0.182	0.077 ± 0.034	0.039–0.138
Fluoranthene#	0.076 ± 0.030	0.023–0.194	0.268 ± 0.095	0.181–0.490
Pyrene#	0.040 ± 0.018	0.012–0.096	0.175 ± 0.068	0.103–0.327
Benz[a]anthracene#	0.015 ± 0.009	0.003–0.057	0.102 ± 0.040	0.051–0.176
Chrysene#	0.081 ± 0.044	0.017–0.243	0.384 ± 0.106	0.216–0.649
Benzo[b]fluoranthene#	0.047 ± 0.030	0.008–0.173	0.367 ± 0.173	0.058–1.161
Benzo[k]fluoranthene#	0.041 ± 0.027	0.008–0.154	0.387 ± 0.149	0.137–0.800
Benzo[a]fluoranthene	0.004 ± 0.003	N.D. ^d –0.013	0.053 ± 0.016	0.021–0.074
Benzo[e]pyrene	0.053 ± 0.036	0.008–0.225	0.445 ± 0.127	0.186–0.977
Benzo[a]pyrene#	0.013 ± 0.009	0.001–0.046	0.182 ± 0.067	0.081–0.353
Perylene	0.006 ± 0.003	0.001–0.027	0.025 ± 0.009	0.009–0.047
Indeno[1,2,3-cd]pyrene#	0.021 ± 0.018	N.D. ^d –0.100	0.145 ± 0.061	0.052–0.420
Dibenzo[a,h]anthracene#	0.003 ± 0.003	N.D. ^d –0.016	0.037 ± 0.017	0.015–0.157
Benzo[ghi]perylene#	0.038 ± 0.035	N.D. ^d –0.232	0.169 ± 0.095	0.051–0.437
Coronene	0.024 ± 0.033	N.D. ^d –0.239	0.098 ± 0.062	0.039–0.208
Dibenzo[a,e]pyrene	N.D. ^d	N.D. ^d	0.045 ± 0.045	0.013–0.314
<i>Alkyl-PAHs</i>				
2-Methylnaphthalene	0.187 ± 0.051	0.036–0.524	0.060 ± 0.017	0.042–0.215
1-Methylnaphthalene	0.014 ± 0.005	N.D.–0.049	0.011 ± 0.005	0.003–0.018
2,6-Dimethylnaphthalene	0.065 ± 0.018	0.027–0.142	0.074 ± 0.020	0.051–0.108
9-methylanthracene	0.003 ± 0.003	N.D. ^d –0.031	0.076 ± 0.026	0.046–0.129
Methylfluoranthene	0.106 ± 0.044	0.002–0.302	0.045 ± 0.016	0.024–0.071
Retene	0.027 ± 0.011	0.009–0.067	0.025 ± 0.008	0.012–0.057
Σ15 U.S. EPA Priority PAHs#	0.520 ± 0.228	0.163–1.552	2.448 ± 0.589	1.438–3.074
ΣLMW-PAHs ^a	0.144 ± 0.046	0.077–0.242	0.231 ± 0.048	0.165–0.165
ΣHMW-PAHs ^b	0.221 ± 0.232	0.039–1.203	2.215 ± 1.115	0.733–4.937
ΣComb-PAHs ^c	0.333 ± 0.273	0.087–1.348	2.706 ± 1.054	1.192–5.153
ΣAlkyl-PAHs ^c	0.385 ± 0.166	0.177–1.031	0.309 ± 0.069	0.225–0.475
Σ15 PAHs/Σ26 PAHs (%)	50.7 ± 7.8	36.9–66.9	72.8 ± 2.0	70.3–75.7
LMW-PAHs/Σ26 PAHs (%)	17.0 ± 6.7	7.4–34.0	7.0 ± 3.0	3.4–13.0
HMW-PAHs/Σ26 PAHs (%)	19.8 ± 10.3	6.6–41.8	55.0 ± 8.6	38.6–72.6
Comb-PAHs/Σ26 PAHs (%)	31.6 ± 11.0	16.9–61.9	71.7 ± 3.9	62.8–75.9
Alkyl-PAHs/Σ26 PAHs (%)	42.8 ± 10.2	18.6–60.3	8.8 ± 1.9	6.8–13.0

Notes: ^aΣLMW-PAHs: low molecular weight 3-ring PAHs; ^bΣHMW-PAHs: total concentration of higher molecular weight PAHs (including benzo[a]fluoranthene, benzo[e]pyrene, benzo[a]pyrene, perylene, indeno[1,2,3-cd]pyrene, dibenz[a,h]anthracene, benzo[ghi]perylene, coronene, and dibenzo[a,e]pyrene); ^cΣComb-PAHs: total concentration of nine combustion-derived PAHs (including fluoranthene, pyrene, benz[a]anthracene, chrysene, benzo[a]fluoranthene, benzo[e]pyrene, benzo[a]pyrene, indeno[1,2,3-cd]pyrene, benzo[ghi]perylene); ^dN.D.: denotes data non-detectable; ^eΣAlkyl-PAHs: total concentration of alkyl-PAHs (including 2-Methylnaphthalene, 1-Methylnaphthalene, 2,6-Dimethylnaphthalene, 9-methylanthracene, Methylfluoranthene, and Retene); ^fS.D.: denotes standard deviation.

with an average of 0.31 ± 0.20 . This was relatively lower compared to results with PAHs indoor/outdoor (I/O) ratios < 1 in Guangzhou (0.9–1.1) and Xi'an (0.8–0.9) (Li *et al.*, 2005; Xu *et al.*, 2015). Further examination of the data in Fig. 4(b) showed that the P/A ratios of PAH compounds > 1 were mostly observed in LMW-PAHs (3-ring) and alkyl-PAHs, indicating the presence of indoor sources. On the other hand, the P/A ratios less than unity were found in 4–6 ring PAHs. This result is similar to the finding in other

previous studies, such as, Zhu *et al.* (2011) found that 3-ring PAHs P/A ratios > 1 and the 4–6 ring PAHs P/A ratios were close to unity. Xu *et al.* (2015) reported that the indoor/outdoor ratios for 3-ring PAHs > 1 , while the I/O ratios for 5- and 6-ring PAHs were < 1 in fine particles (D_p : 0.5–1.0 μm) in the school classrooms in Xi'an.

The patterns of ambient concentrations and personal exposures over the sampling period showed generally higher values in personal $\text{PM}_{2.5}$, OC, and EC. In contrast, average

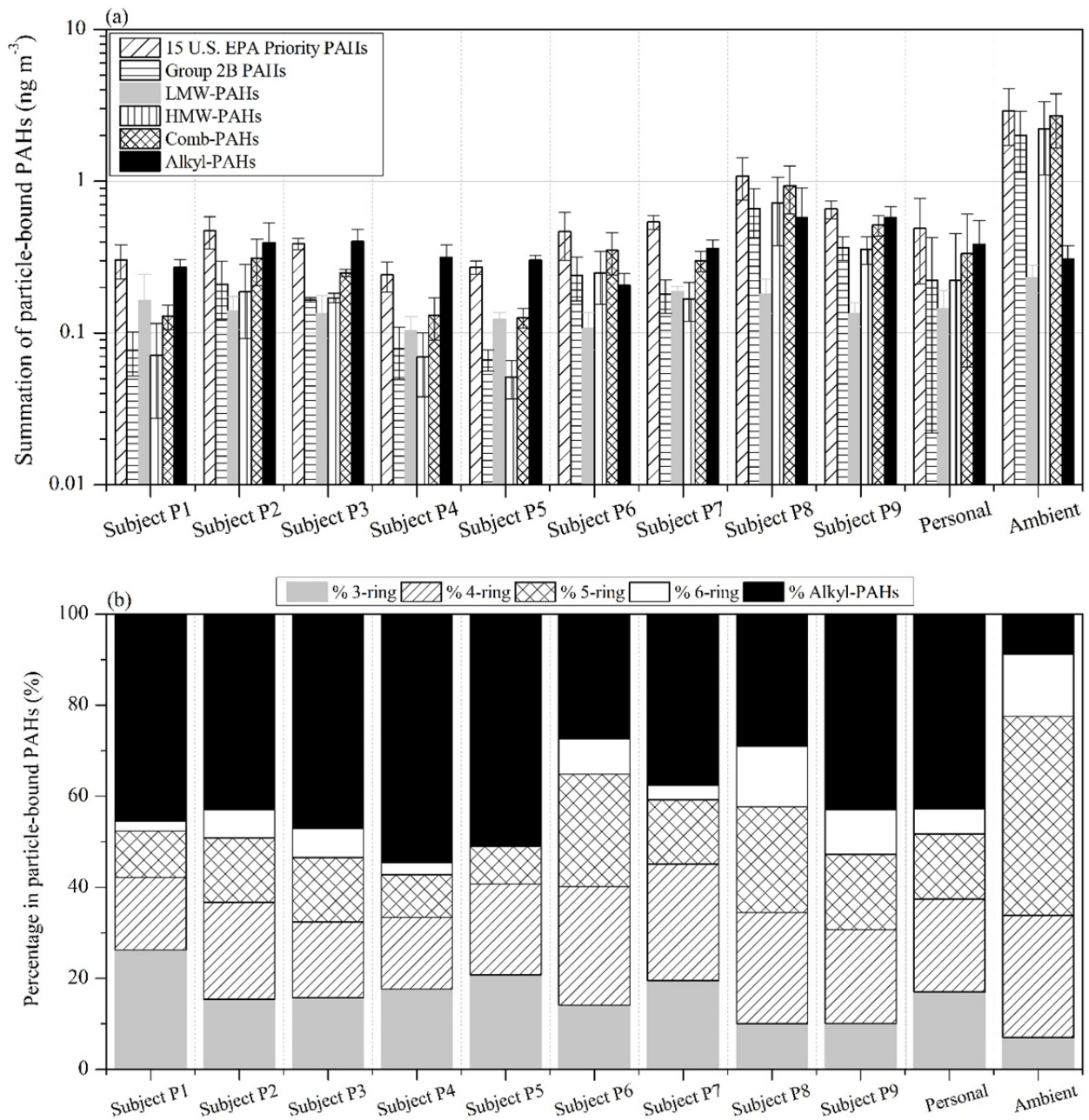


Fig. 3. Percentage contribution of different molecular weight PAHs groups to particle-bound PAHs in individuals' personal and ambient samples in Hong Kong.

personal exposure to Σ PAHs was significantly lower than ambient Σ PAHs on each sampling day. Spearman's correlation coefficients were applied to assess the associations between ambient and personal exposure to $\text{PM}_{2.5}$, OC, EC, and Σ PAHs. The weak correlation ($r_s = 0.328$, $p = 0.055$) between ambient and personal $\text{PM}_{2.5}$ and the intercept greater than zero from the regression analysis in Table 4 suggests that the non-ambient sources (e.g., indoor sources, personal activities) had a significant role in personal exposures. As reported in Brown *et al.* (2008) and Rivas *et al.* (2015), local traffic, indoor sources and/or personal activities can significantly affect the personal exposure to $\text{PM}_{2.5}$ and EC. Mohammadyan (2011) found that ambient PM and time

spent in polluted microenvironments (e.g., buses) are the most important determinants of personal exposure to $\text{PM}_{2.5}$. Several studies have measured the correlations between personal exposures and ambient concentrations in urban cities of the developed countries, with reported non-ambient $\text{PM}_{2.5}$ exposures ranged from 0.43 to $8.47 \mu\text{g m}^{-3}$ (Williams *et al.*, 2000; Wilson and Brauer, 2006; Noullett *et al.*, 2010).

As shown in Table 4, significant correlations ($r_s > 0.60$, $p < 0.01$) between OC and EC in personal $\text{PM}_{2.5}$ and in ambient samples were observed. Moderate significant correlation ($r_s = 0.580$, $p < 0.001$) between ambient and personal EC indicating that higher personal OC and EC exposures were likely due to indoor sources and/or personal

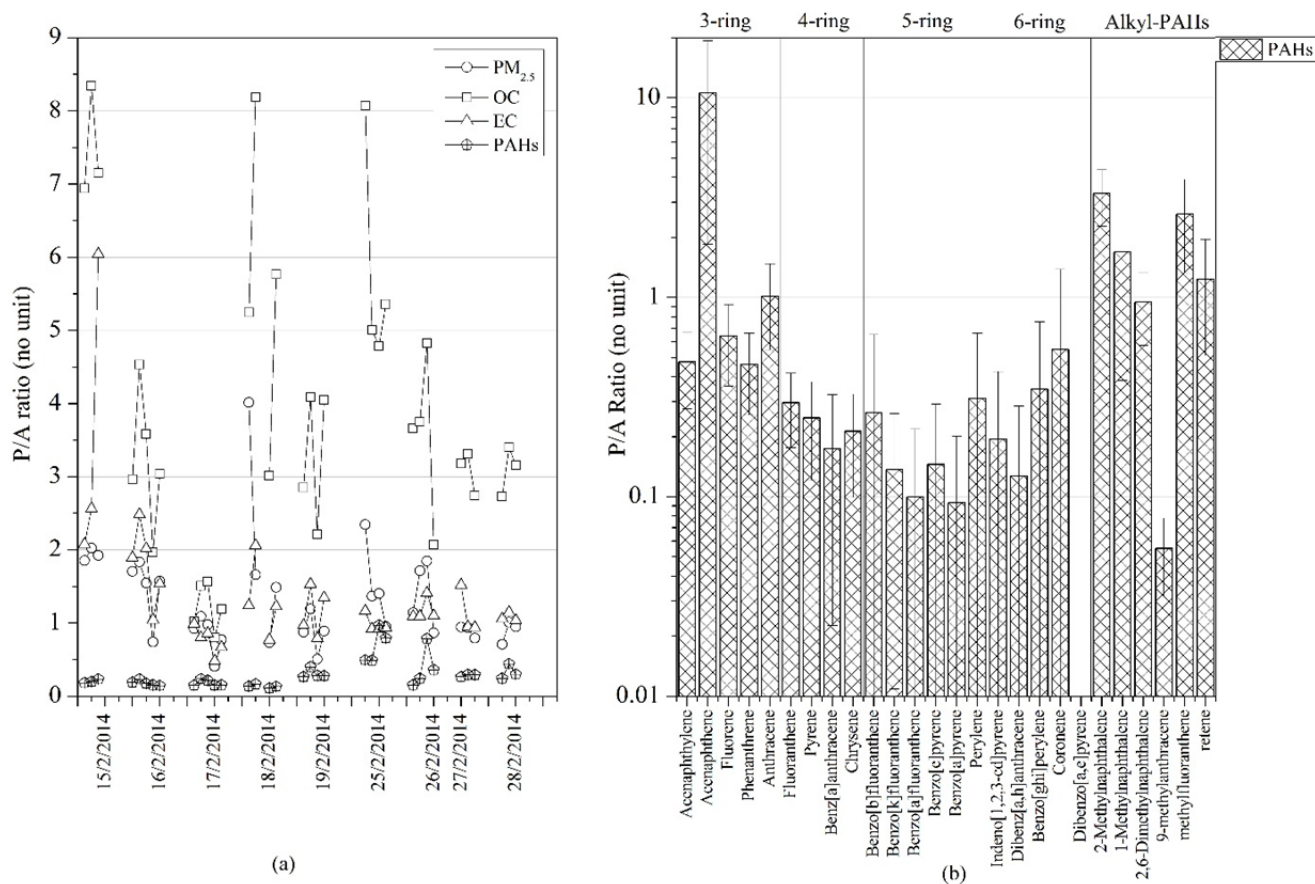


Fig. 4. Daily personal/ambient (P/A) ratios of (a) PM_{2.5}, OC, EC, and ΣPAHs during the sampling period and (b) P/A ratio of 3-, 4-, 5-, 6-ring, and alkyl-PAHs in Hong Kong.

Table 4. Correlation analysis between ambient and personal exposure to PM_{2.5}, OC, EC, and ΣPAHs during the sampling period.

Spearman's Rho (r_s)		PM _{2.5}	OC	EC	ΣPAHs
Personal (P)	PM _{2.5}	1	0.803**	0.617**	0.356*
	OC		1	0.657**	0.444**
	EC			1	0.434**
	PAHs				1
Ambient (A)	PM _{2.5}	1			
	OC	0.816**	1		
	EC	0.691**	0.637**	1	
	PAHs	0.525**	0.594**	0.003	1
Personal vs. ambient	r_s	0.328	0.279	0.580**	-0.108
	p -value	0.055	0.105	< 0.001	0.537
	Slope	0.61	0.64	0.61	-0.05
	Intercept	21.7	8.1	1.1	1.1
	N	35	35	35	35

Notes: ^aN denotes the number of the valid data; * Correlation is significant at the 0.05 level (2-tailed); ** Correlation is significant at the 0.01 level (2-tailed).

activities (e.g., transportation). Fair correlations (r_s : 0.307–0.319*, $p < 0.05$) were observed for 3–4 ring PAHs between ambient and personal samples in this study. However, no significant correlation was found ($r_s = -0.108$) for ΣPAHs or 5–6 ring PAHs between ambient and personal samples. Poor correlation has been observed for BghiP (6-ring PAHs) between ambient and personal samples in Zhu *et al.* (2011).

Different from this finding, Li *et al.* (2005) reported significant correlations for 5–7 ring PAHs between indoor and outdoor concentrations.

Source Identification of PAH Compounds

Among various particle-bound PAHs emission sources, the vehicle emissions have been known to be the most

Table 5. Diagnostic ratios of PAHs species in ambient and personal PM_{2.5}

	Personal		Ambient	
	Mean ± S.D. ^a	Median	Mean ± S.D. ^a	Median
PHE/PHE + ANT	0.42 ± 0.13	0.43	0.58 ± 0.11	0.60
BbF/BkF	1.19 ± 0.26	1.14	1.12 ± 0.56	1.02
BeP/BeP + BaP	0.78 ± 0.09	0.79	0.71 ± 0.05	0.71
IcdP/IcdP + BghiP	0.36 ± 0.06	0.37	0.48 ± 0.06	0.49

Notes: ^aS.D.: denotes standard deviation; PHE-phenanthrene; ANT-anthracene; BbF-benzo[b]fluoranthene; BkF-benzo[b]fluoranthene; BeP-benzo[e]pyrene; BaP-benzo[a]pyrene; IcdP-indeno[1,2,3-cd]pyrene; BghiP-benzo[ghi]perylene.

important contributor in urban cities of different countries (Guo *et al.*, 2003; Li *et al.*, 2005; Zhu *et al.*, 2011). Guo *et al.* (2003) found that vehicle emissions (e.g., gasoline, diesel) were the predominant sources of airborne PAHs in Hong Kong. Gasoline vehicle exhaust contained more HMW-PAHs (e.g., benzo(a)pyrene, dibenzo(a,h)anthracene), whereas diesel truck was the major source of lighter PAHs (e.g., 4-ring PAHs) (Miguel *et al.*, 1998). It has been suggested to use PAH markers and their ratios in distinguishing emission sources (Guo *et al.*, 2003; Zhu *et al.*, 2011). Table 5 lists the diagnostic ratios for individual PAHs, e.g., PHE/PHE + ANT, BbF/BkF, BeP/BeP + BaP, IcdP/IcdP + BghiP, in personal and ambient samples, which were used to investigate their origin or as indicators showing aging of air samples. The values were compared with those reported in other previous studies.

Khalili *et al.* (1995) found that the ratio of PHE/PHE + ANT was 0.50 for gasoline and 0.65 for diesel engines, respectively. The mean PHE/PHE + ANT ratio was 0.58 ± 0.11 in ambient PM_{2.5}, and considerably lower values 0.42 ± 0.13 were observed in personal samples. The results indicated that the sources of PAHs dominated in ambient samples were fresh emission from gasoline and diesel engines, and aging particles could be the main sources of personal PAHs (Grimmer *et al.*, 1983; Galarneau, 2008; Katsoyiannis *et al.*, 2011; Tobiszewski and Namieśnik, 2012; Zhang *et al.*, 2005). The BbF/BkF in ambient (1.12) and personal samples (1.19) were similar to those reported for automobile exhaust (1.26) (Dickhut *et al.*, 2000). Zhu *et al.* (2011) reported similar median ratio of BbF/BkF (1.00) in an urban area in the U.S.

Most of the fresh exhausts have similar contents of benzo(e)pyrene (BeP) and benzo(a)pyrene (BaP) (Grimmer *et al.*, 1983). BaP can be easily decomposed by light and oxidants, thus, BeP/BeP + BaP ratio is regarded as an index of the aging of particles. Relatively higher BeP/BeP + BaP ratio was observed in personal samples (0.78 ± 0.09) than in ambient samples (0.71 ± 0.05), indicating that emission of these two PAHs was not concurrent and BaP could be affected by vigorous activity in indoors. BeP/BeP + BaP ratios reported in this study were higher than those measured in indoors in Guangzhou (0.41–0.72) (Li *et al.*, 2005). The ratio of IcdP/IcdP + BghiP was found to be 0.18 for gasoline emissions and 0.37 for emission from diesel engines (Grimmer *et al.*, 1983; Guo *et al.*, 2003). The average IcdP/IcdP + BghiP ratio in personal samples was 0.36 ± 0.06 indicating the contribution of both gasoline and diesel emissions. Significantly higher IcdP/IcdP + BghiP ratio

(0.48 ± 0.06) was observed at the ambient site, which indicated that diesel emissions were the main sources. This result is similar with the finding in other studies (Li *et al.*, 2005; Ho *et al.*, 2009; Saha *et al.*, 2009).

CONCLUSIONS

Personal and ambient PM_{2.5} samples were simultaneously collected in Hong Kong during the winter of 2014. PM_{2.5} mass, OC, EC, and ΣPAHs concentrations were determined and their correlations were investigated. Relatively higher personal exposure to PM_{2.5} (42.6 ± 18.8 μg m⁻³) were reported compared to ambient concentrations (33.8 ± 10.2 μg m⁻³) and the mass difference was statistically significantly different ($p < 0.05$). Consistent with the PM_{2.5} mass, considerably higher OC and EC concentrations were reported in personal samples (9.9 ± 4.5 and 2.2 ± 1.0 μg m⁻³, respectively) compared to ambient samples (3.1 ± 1.8 and 1.9 ± 1.1 μg m⁻³, respectively). Personal to ambient (P/A) ratios > 1 for PM_{2.5}, OC, and EC were found in this study. Weak to moderate correlation coefficients were reported for PM_{2.5}, OC, and EC, suggesting that personal exposure was less influenced by ambient sources nevertheless showed greater influences by indoor sources and/or personal activities. A different pattern was shown for ΣPAHs with significantly higher concentrations ($p < 0.01$) observed from ambient samples (3.46 ± 0.93 ng m⁻³) than personal samples (0.90 ± 0.48 ng m⁻³). The measured PAHs compounds showed considerably lower fractions of ΣHMW-PAHs (from benzo[b]fluoranthene to dibenzo[a,e]pyrene) in personal samples (19.8 ± 10.3%) than ambient samples (55.0 ± 8.6%), indicated that concentrations of these PAHs compounds in personal PM_{2.5} were dominated by ambient sources (e.g., vehicle emissions). Higher percentages of ΣLMW-PAHs and ΣAlkyl-PAHs to ΣPAHs were observed in personal samples compared to ambient samples, indicating the importance of contribution of personal activities in indoors.

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DISCLAIMER

The authors declare no conflict of interest.

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