



The exposure of children to PM_{2.5} and dust in indoor and outdoor school classrooms in Kuala Lumpur City Centre

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ABSTRACT

It is important to assess indoor air quality in school classrooms where the air quality may significantly influence school children's health and performance. This study aims to determine the concentrations of PM_{2.5} and dust chemical compositions in indoor and outdoor school classroom located in Kuala Lumpur City Centre. The PM_{2.5} concentration was measured from 19th September 2017–16th February 2018 using an optical PM_{2.5} sensor. Indoor and outdoor dust was also collected from the school classrooms and ion and trace metal concentrations were analysed using ion chromatography (IC) and inductively couple plasma-mass spectrometry (ICP-MS) respectively. This study showed that the average indoor and outdoor 24 h PM_{2.5} was $11.2 \pm 0.45 \mu\text{g m}^{-3}$ and $11.4 \pm 0.44 \mu\text{g m}^{-3}$ respectively. The 8 h PM_{2.5} concentration ranged between 3.2 and $28 \mu\text{g m}^{-3}$ for indoor and 3.2 and $19 \mu\text{g m}^{-3}$ for outdoor classrooms. The highest ion concentration in indoor dust was Ca²⁺ with an average concentration of $38.5 \pm 35.0 \mu\text{g g}^{-1}$ while for outdoor dust SO₄²⁻ recorded the highest ion concentration with an average concentration of $30.6 \pm 9.37 \mu\text{g g}^{-1}$. Dominant trace metals in both indoor and outdoor dust were Al, Fe and Zn. Principle component analysis-multiple linear regression (PCA-MLR) demonstrated that the major source of indoor dust was road dust (69%), while soil dominated the outdoor dust (74%). Health risk assessment showed that the hazard quotient (HQ) value for non-carcinogenic trace metals was < 1 while the total cancer risk (CR) value for carcinogenic elements was below the acceptable limit for both indoor and outdoor dust through dermal and inhalation pathways, but not the ingestion pathway. This study suggests indoor contributions of PM_{2.5} concentrations are due to the activities of the school children while the compositions of indoor and outdoor dust are greatly influenced by the soil/earth source plus industrial and traffic contribution.

1. Introduction

Cities and urban areas that are developed for residential, commercial, industrial and infrastructure networks are increasingly the focus of trading and economic activities. Rapid economic development and the increase in human population in cities leads to environmental degradation and the production of high amounts of air pollutants (Jang et al., 2017; Joshi et al., 2009; Rodriguez et al., 2016). Air pollutants produced by stationary sources such as power plants, industries and heating purposes coupled with traffic sources contribute to urban air pollution where these pollutants are subsequently transported to the earth's surface in both particulate and vapor phases (Azimi et al., 2005; Monks et al., 2009; Schauer et al., 1999). Several studies have identified major contributions from traffic-related air pollution and particulate matter (PM) in urban aerosols (Amato et al., 2009; Quang et al., 2013;

Viana et al., 2011; Wu et al., 2013; Yang et al., 2005). PM with diameters of less than 2.5 μm (PM_{2.5}) has been the main interest of these studies, especially epidemiological studies, due to its effects on human health (Chen et al., 2015; Sarnat et al., 2008).

High concentrations of PM_{2.5} affect children's health, especially school children who spend much of their time in the school environment. Studies of PM_{2.5} concentrations in schools located in urban areas and cities with high traffic levels identified the negative influence of outdoor sources on school classroom indoor air quality (Chithra and Shiva, 2012; Guo et al., 2010; Madureira et al., 2012; Minguiñón et al., 2012; Mohamad et al., 2016; Pegas et al., 2012; Tippayawong et al., 2009; Wichmann et al., 2010). Urban schools in developing countries usually have natural ventilation systems and are in locations near to residential areas and busy roads. These areas have high concentrations of outdoor pollutants, and these pollutants can be transferred to the

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indoor environment via ventilation intake, open doors and windows and leaks in the building (Chen et al., 2013; Chithra and Shiva, 2012; Mohammadyan et al., 2017; Tippayawong et al., 2009). The infiltration of outdoor pollutants, especially $PM_{2.5}$, is also influenced by the micro-environment, ventilation type and air exchange rate (Bennett and Koutrakis, 2006; Liu and Nazaroff, 2001; Wichmann et al., 2010). During school hours, the air exchange rate was high due to open windows leading to the indoor to outdoor (I/O) ratio exceeding 1 (Guo et al., 2010). Another source of high $PM_{2.5}$ concentrations was the re-suspension of particles caused by movement in the occupied classroom (Amato et al., 2014; Morawska et al., 2017; Raysoni et al., 2013; Rim et al., 2017; Salthammer et al., 2016). School classrooms also accumulate other types of pollutants which can be emitted from school building surfaces or formed from physico-chemical processes produced by modern materials such as paint, glue for art classes or cleaning products (Salthammer et al., 2016).

PM with diameters in the range of 1–10,000 μm that settles out of the atmosphere is collectively referred to as dust (Meza-Figueroa et al., 2007; Rodrigues et al., 2018). Soil dust, road dust, construction dust, re-suspended dust deposited on building surfaces, and urban dust are the types of dust that are usually studied by researchers around the world (Al-Khashman, 2007; Kong et al., 2011; Leung et al., 2008; Rodrigues et al., 2018; Wei et al., 2010; Yongming et al., 2006). Dust can contain various contaminants and trace metals, particularly Cr, Cu, Ni, Pb and Zn as these elements are the most prevalent and persistent in the environment (Joshi et al., 2009; Thorpe and Harrison, 2008). High levels of trace metals from outdoor sources can also accumulate in indoor dust as they can be transported into classrooms by school children who spend time in playgrounds, gardens and nearby homes (Liu et al., 2016). A study by Meza-Figueroa et al. (2007) found a dominance of the elements Zn, Pb, Cr and Cd for indoor and outdoor dust in schools located in an industrial-urban area with contributions from traffic and industrial sources. School children who have had significant exposure to high metal concentrations from indoor and outdoor dust will also have a tendency to accumulate these metals in fingernails and toenails (Rodrigues et al., 2018; Were et al., 2008).

Several studies have determined the concentrations and compositions of $PM_{2.5}$ in school environments in past decades; these significantly indicate the exposure of school children to high levels of $PM_{2.5}$ (Canha et al., 2014; Peng et al., 2017). Thus, investigations into fine particles ($PM_{2.5}$) and large particles (dust) in school environments are important to understand the impact of these particles on school children's health. Regarding the issues of the effects of urban air pollution on school children, this study aims to investigate the concentrations of $PM_{2.5}$ and dust chemical compositions in a school environment located in a highly urbanized area. To determine the origins of indoor and outdoor dust in the school environments, principle component analysis-multiple linear regression (PCA-MLR) was applied based on the ion and trace metal concentrations. The estimations of health risks posed by the elements/ions in indoor and outdoor dust were calculated using the United States Environmental Protection Agency (USEPA) Human Health Risk Assessment for school children via incidental dermal, ingestion and inhalation pathways for indoor and outdoor school classrooms.

2. Materials and methods

2.1. Study location and sampling site

Kuala Lumpur is located in central Peninsular Malaysia and is the national capital of Malaysia. With a population of 1.79 million people and an area of 243.65 km^2 (DOS, 2018), Kuala Lumpur has been affected by the rapid development of industries, infrastructure and residential areas which have contributed to the acceleration of the Malaysian economy. Kuala Lumpur's centre of business is at the Golden Triangle bounded by Jalan Ampang, Jalan Sultan Ismail and Jalan Bukit

Bintang (Adnan and Daud, 2010). The Golden Triangle is a central area where major shopping centres, entertainment facilities, offices and trade activities are located and where this study was performed.

The primary school chosen for this study is surrounded by roads with high density traffic, schools, hotels, a commercial lot and is near to the Kuala Lumpur Eco-Park forest. Landuse map of sampling location shown in Fig. S1. This school is located 200 m from a busy road. A classroom located on the first floor of the main school building facing the main road was selected for $PM_{2.5}$ measurement. The classroom characteristics were: wooden flooring covered with rubber mats; wooden tables and chairs; watercolour pigments; and both water- and oil-based paints covering the walls and ceilings. The ventilation is natural and no mechanical ventilation, such as air conditioning, is used. The description of the school classroom and building details are described in Table S1.

2.2. $PM_{2.5}$ measurement

Sampling for $PM_{2.5}$ was conducted from the 19th September 2017 to the 16th February 2018. $PM_{2.5}$ measurements were performed in indoor and outdoor classrooms simultaneously and continuously during the sampling campaign which included school holidays (when the school was unoccupied) from the 27th November 2017 to the 1st January 2018. Two optical $PM_{2.5}$ sensors developed by Nakayama et al. (2018) were used for $PM_{2.5}$ measurement. The sensor operation is based on the detection of light scattering from particles and allows the detection of particles with diameters as small as 0.3 μm . The $PM_{2.5}$ mass concentration was obtained from the calculation of the particle size and particle number density (Nakayama et al., 2018). For indoor measurements, the sensor was placed 1 m from the ground in the centre of the classroom with details as followed by Massey et al. (2009) and Hassanvand et al. (2014), while for the outdoor measurements the sensor was placed in the corridor. The 8 h and 24 h average $PM_{2.5}$ concentrations were calculated from 1 min data intervals where 8 h averages were from 7.30 a.m. to 3.30 p.m. while 24 h averages were calculated from midnight to midnight as followed by Hoek et al. (2008).

2.3. Dust sample collection

Dust samples were collected using a soft paintbrush and clean polymethyl methacrylate shovel around the classroom surface (indoor) and corridor surface of the school (outdoor) areas three times during the sampling campaign. The samples were then sealed in seal polyethylene bags and brought to the laboratory. This sampling procedure is similar to those used in previous studies (Joshi et al., 2009; Latif et al., 2014). The samples were taken in areas where school children spend much of their time at school to determine the exposure of children to dust. As the soil chemical profile is also important when considering indoor and outdoor dust compositions, soil samples were also collected as followed by Balakrishna et al. (2011) and Othman and Latif (2013). Soil samples were collected after the removal of the top soil near to the main road of the school area and were then air-dried in the laboratory. The dust and soil samples were then sieved using a 63 μm sieve (Retsch, Germany).

The extraction of ions in dust and soil dust samples began with the digestion of 0.7 g of the sample in 15 mL ultrapure water by heating for 2 h and sonicating for 30 min in an ultrasonic bath (Jaradat et al., 2004). The samples were then diluted with 25 mL ultrapure water and filtered using Pall Acrodisc syringe filters and stored in polyethylene bottles. Trace metals in the dust samples were extracted using a mixture of nitric acid (Merck, Germany) and perchloric acid (Merck, Germany) (16:4) for each 1 g sample as described by Latif et al. (2014). For soil dust samples, digestion was carried out following the USEPA Method 3050B Acid Digestion of Sediments, Sludge and Soil (Othman and Latif, 2013). Filtered samples were then diluted with 100 mL ultrapure water and stored in polyethylene bottles at 4 °C.

The concentrations of ionic species (F^- , Cl^- , NO_3^- , SO_4^{2-} , Na^+ , Mg^{2+} , K^+ , Ca^{2+} , NH_4^+) were analysed using ion chromatography (IC) (Metrohm 850 IC Plus, Switzerland) while trace metal concentrations (Al, Pb, Cr, Fe, Cd, Co, Zn, V, Ni, As, Cu, Mn) were analysed using inductively coupled plasma-mass spectrometry (ICP-MS) (PerkinElmer Elan 9000, United States of America). The detection limits for ions were between 0.02 and 0.25 $ng\ g^{-1}$ while for trace metals they were 0.001–0.4 $\mu g\ g^{-1}$. Blanks, spiked samples and standard reference materials (NIST Urban Dust 1649a) were prepared in the same manner as the dust and soil samples before analysis with IC and ICP-MS. Spike recovery of the ions and trace elements ranged between 67% and 120% (Table S2). All glassware and containers used during the preparation and analysis procedures were soaked in a 10% acid nitric bath for more than 24 h and then washed with ultrapure water.

2.4. Indoor and outdoor relationship

The indoor and outdoor relationships for $PM_{2.5}$, ion and trace metal concentrations in the dust and soil dust samples were determined using the ratio of indoor to outdoor (I/O) values. The fraction of outdoor air pollution that penetrates the indoor environment is usually investigated using the I/O ratio (Huang et al., 2007; Li and Lin, 2003; Othman et al., 2016) and the ratio is used to provide information on the relationship between indoor and outdoor particles (Chen and Zhao, 2011). Moreover, the infiltration factor (F_{inf}), which represents the equilibrium fraction of the outdoor PM that penetrates indoors and remains suspended (Arhami et al., 2010; Rivas et al., 2015), was also determined in this study as expressed in Eq. (1):

$$C_{in} = F_{inf} \times C_{out} + C_{ig} \quad (1)$$

Where, C_{in} and C_{out} are the indoor and outdoor $PM_{2.5}$ concentrations; F_{inf} is the infiltration factor; and C_{ig} is indoor-generated particles.

2.5. Airway particle deposition in school children

Particle deposition in the respiratory tracts of school children was calculated using the Multiple-Path Particle Dosimetry (MPPD) model (Applied Research Associates, United States of America). The MPPD model is used for particle dosimetry calculations where monodisperse and polydisperse aerosol deposition can be estimated in the respiratory tracts of rats and humans based on both single-path and multi-path routes (Ji and Yu, 2012; Othman et al., 2018). The single-path route calculation includes the particle deposition per airway generation while the multi-path calculation includes the lung structure constructed from actual airway measurements (Asgharian et al., 2001; Othman et al., 2018).

For this study, the typical exposure scenarios for indoor and outdoor classrooms were simulated with different body orientations, breathing frequencies and $PM_{2.5}$ concentration. The particle properties were set to be count median mean (CMB) with a diameter of 2.5 μm and a geometric standard deviation (GSD) of 1.0. For the indoor exposure scenario, a 'leaning forward' body orientation was selected to demonstrate the learning conditions in the classroom, with an aerosol concentration of 10 $\mu g\ m^{-3}$ while for outdoor scenario, an upright body orientation was selected with an aerosol concentration of 9 $\mu g\ m^{-3}$. Breathing frequency of 26 breaths min^{-1} for heavy exercise was selected as followed by Gangwal et al. (2011), where high activities were performed by school children such as running, jumping and other physical activities outdoors. List of all input parameters used in MPPD model was listed in Table S3.

2.6. Geoaccumulation index, pollution assessment and enrichment factor

In order to characterize dust metal pollution, geoaccumulation index (Igeo), pollution index and enrichment factors (EF) were calculated. Igeo and pollution index calculations can provide the degree of

metal pollution enrichment in dust (Lu et al., 2009) while EF can provide the degree of anthropogenic influence (Chen et al., 2014). Pollution index were evaluated by comparing metal concentrations with the equivalent guideline/background levels (Luo et al., 2012) while Igeo compared metal concentrations with background matrices multiplied by a correction factor to account for lithogenic effects (Ghrefat and Yusuf, 2006). Details of the calculations of Igeo, pollution index and EF are shown in Supplementary 1.

2.7. Health risk assessment of dust exposure

Health risk assessments are widely applied to determine the human health risk of exposure to trace metals (Ferreira-Baptista and De Miguel, 2005; Jiang et al., 2016; Shao et al., 2018; Zheng et al., 2010). In this study, the exposure of school children to both non-carcinogenic and carcinogenic trace metals was calculated based on the Guideline for Human Exposure Assessment (USEPA, 2009), where the classification of both trace metal groups was based on the Integrated Risk Information System (IRIS) (USEPA, 2017). Non-carcinogenic elements were Mn, Cu, Zn, while carcinogenic elements were Ni (refinery dust), Cr (VI), Pb, and Cd. Due to only Cr (VI) of the chromium oxidation states being classified as a carcinogen, this study determined the human health impact of Cr (VI) using the calculation of one seventh of the total Cr concentration (Mancuso, 1975; Othman et al., 2016).

Exposure to non-carcinogenic and carcinogenic trace metals expressed as daily dose is calculated separately for each element for the inhalation, dermal and ingestion pathways. Average daily dose (ADD) for each pathway is expressed in Eqs. (2)–(4) while Lifetime Average Daily Dose (LADD) for carcinogens were calculated as in Eq. (5) as followed by Ferreira-Baptista and De Miguel (2005), Han et al. (2017) and Shao et al. (2018).

$$ADD_{inh} = C \times \frac{InhR \times EF \times ED}{PEF \times BW \times AT} \quad (2)$$

$$ADD_{ing} = C \times \frac{IngR \times EF \times ED}{BW \times AT} \times 10^{-6} \quad (3)$$

$$ADD_{dermal} = C \times \frac{SA \times SL \times ABS \times EF \times ED}{BW \times AT} \times 10^{-6} \quad (4)$$

$$LADD = \frac{C \times EF}{AT \times PEF} \times \left(\frac{InhR_{child} \times ED_{child}}{BW_{child}} + \frac{InhR_{adult} \times ED_{adult}}{BW_{adult}} \right) \quad (5)$$

where C is the concentration of non-carcinogenic and carcinogenic trace metals ($mg\ kg^{-1}$); IngR is ingestion rate where the value is 200 $mg\ day^{-1}$; InhR is the inhalation rate where the values were 7.6 $m^3\ day^{-1}$ for a child and 20 $m^3\ day^{-1}$ for an adult. EF is the exposure frequency (215 school days $year^{-1}$); ED is the exposure duration (6 years); BW is body weight which was 15 kg for a child and 70 kg for an adult; AT is the averaging time ($ED \times 365$ days for non-carcinogenic and 70×365 days for carcinogenic trace metals). PEF is the particle emission factor with a value of $1.36 \times 10^9\ m^3\ kg^{-1}$; SA is the exposed skin area (2800 cm^2); SL is the skin adherence factor (0.2 $mg\ cm^{-2}\ day^{-1}$); and ABS is the dermal absorption factor (0.001). The 95% upper confidence limit (UCL) was used for the concentration C to yield the estimation of the reasonable maximum exposure for non-carcinogenic and carcinogenic elements (Ferreira-Baptista and De Miguel, 2005; Zheng et al., 2010). The hazard quotient (HQ) was then calculated for each element and pathway by dividing ADD with the corresponding reference dose (RFD). The Hazard Index (HI) was then acquired with the sum of HQ, where HI and HQ values of < 1 are considered no significant risk while HI and HQ values of > 1 show the possibility of non-carcinogenic effects. The cancer risk (CR) was calculated as ADD for carcinogenic elements multiplied with the corresponding slope factor where the acceptable value is in the range of 1.0E-06 to 1.0E-04 (Ferreira-Baptista and De Miguel, 2005; Han et al., 2017; Shao et al., 2018).

2.8. Statistical analysis

All the data obtained in this study was checked for normality using statistical P-P plots and Q-Q plots. Pearson's correlation coefficients for ion and trace metal concentrations in dust (indoor and outdoor) and soil dust samples were analysed using Statistical Package for the Social Sciences (SPSS 20). Pearson's correlation is usually used to determine the strength of a linear relationship between two quantitative variables (Chen et al., 2014).

Principle Component Analysis (PCA) was carried out to identify the contribution of various factors to the concentrations of metals in indoor and outdoor dust as well as soil dust samples using XLSTAT 2018 software. In this study, PCA analysis was carried out using the varimax rotation method and an Eigenvalue of > 1 was chosen to obtain the final factor (Wahid et al., 2013). The factor loading with a value of > 0.7 was used to determine the contributions of the ions and trace metals to each factor. Factor loading evaluation is important to estimate the chemical source responsible for each factor where factor loadings were strong (> 0.75), moderate (0.50–0.75), and weak (< 0.05) (Wahid et al., 2013). Multiple Linear Regression (MLR) was then performed using the PM_{2.5} concentrations and score matrices from the PCA analysis. In this study, each score matrix for each source factor was regressed against PM_{2.5} concentration to determine the coefficient of determination value (R²) as described by Zhong et al. (2014).

3. Results and discussion

3.1. PM_{2.5} concentration in indoor and outdoor classrooms

Fine particulate matter concentrations (PM_{2.5}) were measured in indoor and outdoor classrooms during weekdays (WD) and the weekend (WE) (Table 1). The 24 h average concentration during WD was 11.2 ± 0.45 µg m⁻³ (indoor) and 11.4 ± 0.44 µg m⁻³ (outdoor). Overall, both indoor and outdoor classroom PM_{2.5} concentrations were lower than the World Health Organization (WHO) guideline (25 µg m⁻³), the National Ambient Air Quality Standard proposed by USEPA (35 µg m⁻³), and the New Malaysia Ambient Air Quality Standard for 2018 Interim Target-2 (50 µg m⁻³). For 8 h measurements, the PM_{2.5} concentrations ranged from 3.2 to 28 µg m⁻³ for the indoor and from 3.2 to 19 µg m⁻³ for the outdoor classroom environment. Maximum concentrations of up to 28 µg m⁻³ in the indoor classroom during school hours suggest school children's activities contribute to PM_{2.5} concentrations compared to the 8 h outdoor situation, which recorded much lower maximum concentrations. Fig. 1 shows the 8 h average concentration trend of PM_{2.5} during the school term, covering both WD and WE. Overall, the PM_{2.5} concentration trend in the indoor and outdoor classrooms was quite similar. The PM_{2.5} concentration suddenly increased at 10.00 a.m. during recess time when the school

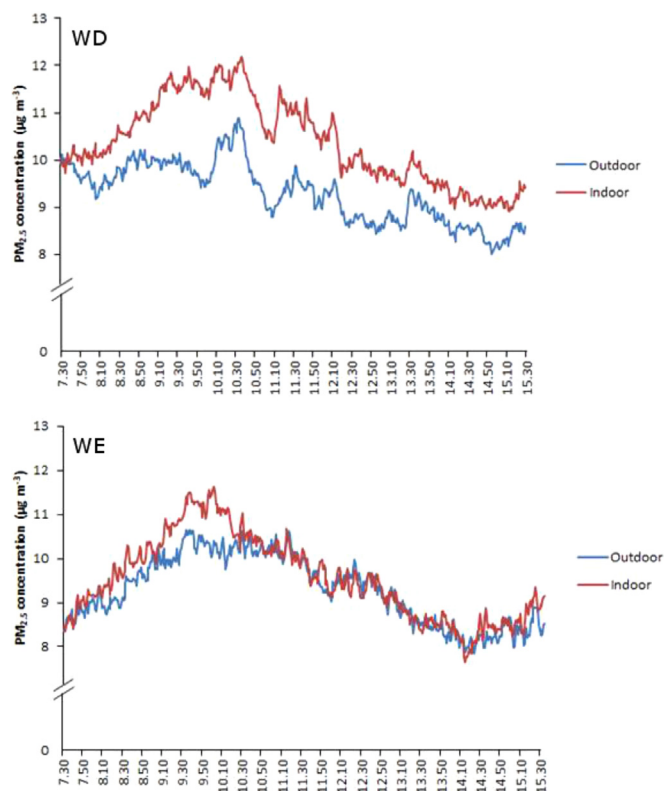


Fig. 1. Average indoor and outdoor 8 h PM_{2.5} concentration during weekdays (WD) and weekend (WE).

children moved around the classroom and left for recess. The highest PM_{2.5} concentration was recorded at 10.30 a.m. when the children returned to the classroom after recess. A sudden decrement in PM_{2.5} concentration was observed after recess time when teaching resumed. Then, the PM_{2.5} concentration slowly decreased until 1.30 p.m. where school time ended but other activities were performed such as classroom cleaning and extra classes (just a few days in a week). During the weekend (WE), indoor and outdoor classroom PM_{2.5} concentrations recorded much higher concentrations for the 24 h sampling duration—around 1.2 times higher compared to WD. Higher WE concentrations may originate from high numbers of vehicles during the WE, as the sampling location is near to a tourist attraction area and close to a main road and busy junction. This result was supported by the recorded higher numbers of vehicles during WE compared to WD for certain times of day in the city centre (DBKL, 2018).

This study also measured PM_{2.5} concentrations during the school

Table 1

Average concentration of PM_{2.5} (µg m⁻³) and range concentration in bracket, with *Finf*, *Cig* (µg m⁻³) and R² value for indoor and outdoor classroom during weekdays (WD) and weekend (WE).

Measurement Duration	WD (n = 83)						WE (n = 32)					
	Indoor	Outdoor	I/O	<i>Finf</i>	<i>Cig</i>	R ²	Indoor	Outdoor	I/O	<i>Finf</i>	<i>Cig</i>	R ²
School Term (occupied)												
24 h	11.2 ± 0.45 (4.5-26)	11.4 ± 0.44 (4.5-23)	1.01 ± 0.03 (0.4-2.1)	0.82	2.08	0.55	13.4 ± 1.05 (5.0-24)	13.3 ± 1.08 (5.5-27)	1.02 ± 0.04 (0.4-1.6)	0.82	1.89	0.76
8 h	10.2 ± 0.45 (3.2-28)	9.61 ± 0.41 (3.2-19)	1.07 ± 0.01 (0.8-1.8)	0.61	2.95	0.79	9.47 ± 0.60 (4.8-18)	9.23 ± 0.61 (4.3-18)	1.04 ± 0.02 (0.7-1.2)	0.76	2.00	0.86
School Holidays (unoccupied)		(n = 26)						(n = 10)				
24 h	10.6 ± 0.73 (3.0-17)	9.98 ± 0.95 (2.5-18)	1.20 ± 0.08 (0.6-2.5)	0.57	4.95	0.54	10.9 ± 1.30 (4.0-17)	9.50 ± 1.11 (3.5-14)	1.20 ± 0.11 (0.6-1.7)	0.73	3.90	0.39
8 h	8.56 ± 0.82 (2.7-20)	8.35 ± 0.85 (3.0-20)	1.04 ± 0.02 (0.8-1.3)	0.94	0.70	0.94	9.76 ± 1.31 (3.9-17)	9.93 ± 1.46 (2.2-17)	1.06 ± 0.08 (0.7-1.7)	0.74	2.95	0.72

holidays (i.e. when the building was unoccupied) when, overall, slightly higher indoor concentrations were observed for 24 h and 8 h sampling durations compared to outdoor concentrations during WD. For WE, higher indoor compared to outdoor concentration was measured for 24 h sampling duration while not much difference between indoor and outdoor concentration for 8 h sampling duration. Higher WE concentrations for 8 h sampling compared to WD are consistent with $PM_{2.5}$ concentrations measured during the school term, and it is suggested these concentrations are influenced by higher numbers of vehicles during WE compared to WD. Higher $PM_{2.5}$ concentrations during 8 h sampling in the WD of the school term compared to the school holidays (inoccupancy) in the school classroom significantly denoted that occupancy of the classroom influenced $PM_{2.5}$ levels in the classroom.

The indoor to outdoor ratio (I/O) is used to determine the contribution of outdoor pollution to indoor areas; in this study, for 24 h and 8 h $PM_{2.5}$ concentrations, the ratio was > 1 . The maximum I/O value of 2.5 was determined during a period when the classroom was unoccupied, indicating that $PM_{2.5}$ concentrations were enhanced indoors. Even though the I/O ratio value was slightly above 1 during both the school term (occupied) and school holiday (unoccupied) periods, this result indicates that there are outdoor source contributions to the classroom—these infiltrate indoors through openings and cracks in the building and windows. Moreover, I/O values of > 1 in this study suggest a contribution from the resuspension of dust due to school children's activities. As reported by Goyal and Khare (2009), factors which affect I/O were ventilation rates, where higher I/O ratios were due to windows being opened and fans being run, while temperature, relative humidity and wind speed also played important roles for I/O values, increasing the infiltration from outdoors. A study by Rivas et al. (2015), had a value of > 1 for I/O for studies in an urban school which indicated an outdoor source contribution, while a study by Zwoździak et al. (2013) had I/O values that exceeded 1 and concluded that there were high contributions of indoor-generated particles and resuspension of dust caused by children's activities. van der Zee et al. (2016) also reported an I/O value of > 1 for PM with diameters of more than $10 \mu m$ (PM_{10}) and $PM_{2.5}$ during teaching hours due to large contributions of indoor-generated particles in school classrooms located near a busy highway.

Furthermore, as the school building is not a new building, the risk of infiltration of fine particles is very high. The relationship between indoor and outdoor PM concentrations were investigated using the infiltration factor (*Finf*) as shown in Table 1. A higher *Finf* value was determined during the 24 h sampling duration (0.82) for both WD and WE compared to the 8 h sampling duration for WD (0.61) and WE (0.76). This result indicates that $PM_{2.5}$ infiltration occurred during unoccupied classroom perhaps due to wall openings and small gaps between door and wall. During the school holidays (unoccupied), a higher *Finf* value was observed for 8 h sampling duration compared to 24 h sampling duration for both WD and WE. Highest *Finf* was recorded for school holidays (unoccupied) for 8 h sampling duration where factor such as indoor and outdoor pressure difference and effect of wind speed may also affected *Finf* value (Chen and Zhao, 2011). For indoor-generated $PM_{2.5}$ (Cig), a higher concentration was recorded during unoccupied classroom periods during WD with concentration of $4.95 \mu g m^{-3}$ for the 24 h sampling measurement and $0.7 \mu g m^{-3}$ for the 8 h sampling measurement. For the school term (occupied), Cig for the 24 h sampling duration was $2.08 \mu g m^{-3}$ and the 8 h sampling duration was $2.95 \mu g m^{-3}$, both during WD. These results suggest indoor particles were generated indoors, particularly from classroom furniture and natural dust accumulation. It was thought that about 28% of the average 24 h indoor $PM_{2.5}$ concentration was generated indoors. During the school holidays (unoccupied), the Cig value was observed to be 43% of 24 h indoor $PM_{2.5}$ concentration and 8% of indoor 8 h $PM_{2.5}$ concentration. As reported by Gao et al. (2016), the Cig value can be affected by the resuspension of deposited particles by the movement of

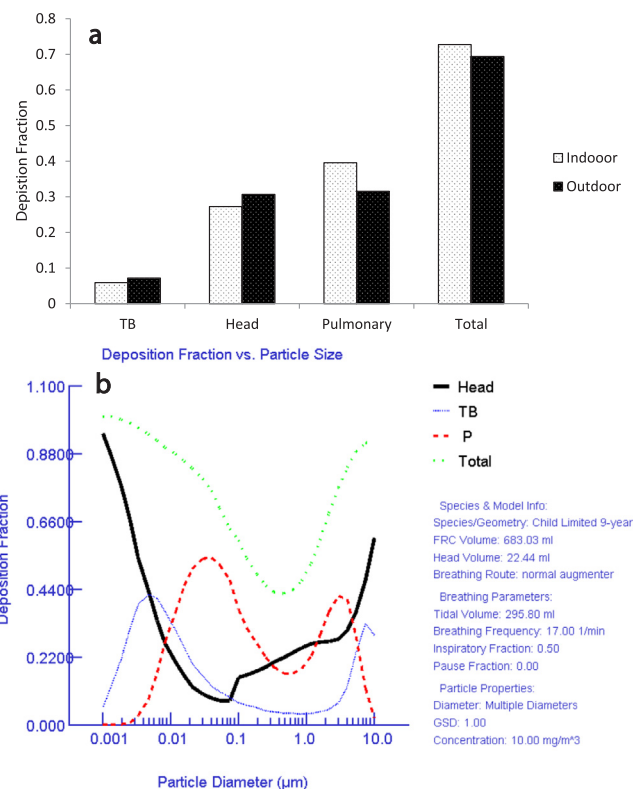


Fig. 2. Deposition fraction of particles for indoor and outdoor scenarios (a) and deposition fraction vs. different particle size (b) in school children respiratory track.

the occupant.

3.2. Deposition fraction of indoor and outdoor particle exposure in school children

Particle deposition in the respiratory tract of school children in indoor and outdoor classroom scenarios was determined using the MPPD model. As reported in Fig. 2a, a higher total deposition fraction was determined for the indoor scenario compared to the outdoor scenario, with values of 0.727 and 0.694 respectively. For the indoor scenario, the highest deposition fraction was in the lungs (0.395), followed by the head (0.277) and the trachea and bronchi (TB) (0.059). The same sequence of deposition fraction was observed for the outdoor scenario, with values of 0.315, 0.306 and 0.071 for the lungs, the head and the TB respectively. In this study, head deposition includes nasopharyngeal and laryngeal deposition (Riebeling et al., 2016). The higher deposition fractions for the indoor scenario compared the outdoor were due to the higher aerosol concentrations and also body orientation, which was selected to be leaning forward to reflect the conditions during learning and teaching in classroom. As reported by Gangwal et al. (2011), aerosol concentrations and breathing patterns were the most important MPPD model inputs, where these input parameters increase alveolar retention by $> 10\%$. Moreover, other parameters such as lung capacity and human respiratory tract structure and morphology also played important roles in the MPPD model (Sarigiannis et al., 2015). Even though a higher breathing frequency was modelled for the outdoor scenario, where school children were assumed to be performing heavier activities, lower total deposition fractions were determined for the outdoor scenario compared to the indoor scenario. Deposition fractions of about 0.03 higher for the head and 0.01 higher for TB, for the outdoor scenario compared to the indoor scenario, reflected that these two body parts accumulate particles during heavy activities.

The deposition of different size fractions of particles from

Table 2
Ions and trace metals concentration ($\mu\text{g g}^{-1}$) in indoor and outdoor dust; and soil dust.

Elements	Indoor	Outdoor	I/O	Soil
Al	457.6 \pm 22.9	1869.3 \pm 81.9	0.24 \pm 0.04	3066.9 \pm 200.1
Pb	10.60 \pm 4.57	18.2 \pm 9.68	0.70 \pm 0.45	39.6 \pm 2.91
Cr	9.33 \pm 3.88	3.84 \pm 0.99	2.66 \pm 1.63	26.4 \pm 9.05
Fe	219.9 \pm 45.7	652.9 \pm 29.2	0.42 \pm 0.29	2695.2 \pm 135.1
Cd	0.38 \pm 0.13	0.48 \pm 0.43	1.40 \pm 1.01	0.16 \pm 0.02
Co	0.61 \pm 0.45	1.09 \pm 0.72	0.81 \pm 0.51	1.69 \pm 0.06
Zn	140.7 \pm 8.29	386.4 \pm 10.1	0.52 \pm 0.48	66.5 \pm 9.14
V	13.33 \pm 0.70	16.6 \pm 2.05	0.81 \pm 0.09	25.3 \pm 1.12
Ni	3.90 \pm 3.11	1.89 \pm 0.90	2.57 \pm 2.39	4.57 \pm 0.74
As	5.23 \pm 1.05	10.5 \pm 4.91	0.56 \pm 0.25	10.8 \pm 0.72
Cu	10.3 \pm 6.97	6.94 \pm 2.98	1.41 \pm 0.41	17.9 \pm 0.92
Mn	8.03 \pm 0.69	15.6 \pm 7.27	0.58 \pm 0.20	302.1 \pm 25.5
Na ⁺	4.22 \pm 2.56	0.93 \pm 0.36	5.55 \pm 4.95	171.2 \pm 78.7
K ⁺	2.66 \pm 1.13	2.58 \pm 0.64	1.11 \pm 0.69	236.4 \pm 54.1
NH ₄ ⁺	0.68 \pm 0.05	2.34 \pm 1.45	0.47 \pm 0.38	45.3 \pm 10.1
Ca ²⁺	38.5 \pm 35.0	25.1 \pm 9.66	1.37 \pm 0.92	233.3 \pm 65.3
Mg ²⁺	1.67 \pm 1.53	1.27 \pm 0.35	1.36 \pm 1.25	63.2 \pm 22.1
F ⁻	0.03 \pm 0.01	0.18 \pm 0.13	0.21 \pm 0.17	19.2 \pm 8.21
Cl ⁻	4.01 \pm 3.02	1.61 \pm 0.31	2.84 \pm 2.11	101.2 \pm 78.2
NO ₃ ⁻	0.91 \pm 1.43	7.10 \pm 2.24	0.15 \pm 0.10	762.2 \pm 101.2
SO ₄ ²⁻	3.77 \pm 1.86	30.6 \pm 9.37	0.14 \pm 0.11	432.1 \pm 101.3

0.001–10 μm diameter were highly deposited in the head and TB, while particles with sizes between 0.01 and 0.1 μm , and particles with sizes between 1.0 and 5 μm , were highly deposited in the lungs (Fig. 2b). Moreover, coarse particles with sizes 5–10 μm were deposited mainly in the head and TB. A study by Megido et al. (2016) found that coarse particles tend to be deposited in extrathoracic regions (which includes nasal, oral and larynx areas) while fine particles tend to accumulate in the alveolar region.

3.3. Chemical composition of indoor and outdoor dust

The ion and trace metal concentrations in indoor and outdoor dust and soil dust collected in the school environment are shown in Table 2. Overall, an abundance of the earth's crust elements, Al and Fe, was observed with concentrations of 457.6 \pm 22.9 $\mu\text{g g}^{-1}$ (indoor); 1869.3 \pm 81.9 $\mu\text{g g}^{-1}$ (outdoor); and 3066.9 \pm 200.17 $\mu\text{g g}^{-1}$ (soil dust) for Al and 219.9 \pm 45.7 $\mu\text{g g}^{-1}$ (indoor); 652.9 \pm 29.2 $\mu\text{g g}^{-1}$ (outdoor); and 2695.2 \pm 135.1 $\mu\text{g g}^{-1}$ (soil dust) for Fe. For indoor dust, other dominant elements and ions observed were Zn, V, Cu and Ca²⁺ while for outdoor dominant elements and ions were Zn, SO₄²⁻, Ca²⁺ and V. The highest I/O value was recorded for Na⁺ with an average value of 5.55 \pm 4.95 suggesting the influence of indoor sources on this ion. An indoor source was also contributing to Ni, Cu, K⁺, Ca²⁺, Mg²⁺, Cd and Cr concentrations, indicated by I/O values of > 1. For soil dust, NO₃⁻ was observed as the most dominant ion with an average concentration of 762 \pm 101 $\mu\text{g g}^{-1}$. The result from a one-way ANOVA test showed no significant difference ($p > 0.05$) for indoor and outdoor dust, or soil dust composition sampled in different months.

The higher concentrations of crustal elements in dust and soil dust are due to enrichment with these earth elements. Other enriched elements such as Zn, Cu and Ca²⁺ are thought to be influenced by anthropogenic activities such as industrial and motor vehicles as this study was performed in a city/urban area. As reported by Hassan (2012), Al and Fe originate from a natural source, while anthropogenic metals were Pb, Ni, Cd, Zn, Cu and Cr. The highest ion concentration in both indoor and outdoor dust was Ca²⁺; the contribution from cement in the school environment is considered to be the main source. Zhao et al. (2006), Kong et al. (2011) and Zhong et al. (2014) also determined high concentrations of Ca in indoor and road dust where Ca was assumed to be influenced by cement, while it was suggested that Zn originated from tyre wear, galvanized materials, indoor products and

the use of Zn in rubber production. High SO₄²⁻ concentrations in outdoor dust were expected due to the secondary aerosol influence from industrial and vehicle emissions. As reported by Zhao et al. (2006), SO₄²⁻ has a much higher abundance in urban dust due to the contribution of secondary fine aerosol from gas-to-particle conversion of SO₂ during the burning of fossil fuel. A consistent result was observed by Kong et al. (2011), where the major ion determined was SO₄²⁻ for fugitive dust in a coal mine area. For the soil dust, enriched elements and ions were NO₃⁻, SO₄²⁻, Mn, K⁺ and Ca²⁺ along with the crustal elements (Al and Fe). High concentrations of secondary aerosol in soil dust were due to the contribution of industrial and traffic emissions where NO₃⁻ was suggested to originate from the emission of NO_x. Mn was also recorded at high concentrations with a mean concentration of 302.1 \pm 25.5 $\mu\text{g g}^{-1}$; this element was also thought to originate from a crustal source. Slightly higher concentrations of Mn were observed by Qu et al. (2018) with a mean concentration of 640 $\mu\text{g g}^{-1}$. The source of Mn here was suggested to be soil parent materials. A study by Guney et al. (2010) observed that the highest metal concentrations in road dust and soil were Zn and Pb with average concentrations of 245 $\mu\text{g g}^{-1}$ (road dust) and 255 $\mu\text{g g}^{-1}$ (soil) for Zn; and 177 $\mu\text{g g}^{-1}$ (road dust) and 191 $\mu\text{g g}^{-1}$ (surface soil) for Pb.

The calculations of Igeo, pollution index and EF were performed for indoor dust, outdoor dust and soil dust (Fig. S2). The Igeo values recorded by the trace metal elements were mostly < 0 which indicated an unpolluted environment. Only As, Zn and Pb recorded values of > 0. The highest Igeo value was determined to be for Zn in outdoor dust (1.86) which indicates that outdoor dust was moderately polluted by Zn. A study by Iwegbue et al. (2018) determined a value of Igeo of < 0 for trace elements in indoor office dust while Fe, Cr and Ni had positive values for dust samples in an electronic workshop, suggesting this type of indoor environment was moderately polluted. The pollution index values for all elements were < 1 except for As, Zn and Pb (Fig. S2). The highest pollution index was observed for Zn in outdoor dust which indicated a high level of pollution of this element. A study by Kong et al. (2011) found that Cu, Zn, Pb, Cd and Hg recorded pollution index values of > 3, indicating high levels of pollution for the dust samples from coal-based city. Iwegbue et al. (2018) found high pollution indexes of > 1 for Cd, Cr, Cu, Ni and Zn in dust sampled in an electronic workshop, a cybercafé and offices, where slight to severe pollution in indoor environments was identified. The integrated pollution index (IPI) was also calculated to determine the average pollution index for dust and soil dust. The IPI value for indoor dust, outdoor dust and soil dust were 0.31, 0.60 and 0.50 respectively. The IPI value classification is: IPI \leq 1 shows low levels of pollution; 1 < IPI \leq 2 shows moderate levels of pollution; 2 < IPI \leq 5 shows high levels of pollution. Higher IPI values for outdoor dust compared to indoor dust and soil dust were found, suggesting higher levels of pollution of outdoor dust.

EF were calculated for indoor and outdoor dust as shown in Fig. S2. Overall, the highest EF value was observed for indoor compared outdoor dust, with the highest EF value for Zn, indicating an anthropogenic source of Zn. Other trace metals recorded EF values of > 10 for indoor dust except for Al, Fe, Cd and Mn, while for outdoor dust Al, Cr, Fe, Cd, Co, V, Ni and Mn had EF values of < 10. These results revealed higher EF values for indoor dust compared to outdoor dust and the highest I/O for EF was Cr (9.91) followed by Cu (6.11). An EF value of > 10 shows that the degree of enrichment arose from anthropogenic origins (Iwegbue et al., 2018). As reported by Luo et al. (2012), an EF value of > 1 can be considered an indicator of the influence of anthropogenic sources due to the enhancement of the metals compared to local background values. High EF values were observed for Cd in street dust in Kuala Lumpur City Centre by Han et al. (2013) where particles with sizes 123 < d < 250 μm had higher EF values compared to particles with sizes d < 63 μm . High EF values for Zn in dust (EF > 100) were recorded by Joshi et al. (2009) where these elements were designated as having a high anthropogenic influence. Moreover, soil dust had the highest EF value for As (83.2) followed by Pb (52.0) where both

elements were classified as extremely enriched with high influences of anthropogenic sources.

Pearson correlation matrix analysis was performed with correlation coefficient values for indoor and outdoor dust element compositions as shown in Table S4. Significant correlations ($p < 0.01$) were observed between indoor and outdoor elements such as Co, Zn, V, As, Cu, Mn, Pb and Cr in outdoor dust with Na^+ , Ca^{2+} , Cl^- and Cu in indoor dust. Other significant correlations were observed between outdoor NH_4^+ and indoor NO_3^- ; outdoor Na^+ and V/Zn; outdoor F^- and Na^+ and; outdoor NO_3^- and indoor Co/ Zn. The relationships of ions and trace metals in soil dust were determined as shown in Table S5. There were a few significant correlations observed for soil dust such as Al with Pb, Al with Fe, Al with V, Pb with V, Cr with Mn and Cd with As ($p < 0.01$). The relationship between outdoor trace metals with indoor Na^+ , Ca^{2+} and Cl^- levels indicates a mutual relationship between elements/ions that originates from natural sources, while the relationship between outdoor NH_4^+ and indoor NO_3^- shows a traffic and industrial source. These relationships between elements/ions in soil dust suggest common sources of these elements/ions.

3.4. Sources of indoor and outdoor dust

Five source factors for indoor dust and three for outdoor dust were identified (Table 3) and source contributions of indoor dust, outdoor dust and soil dust are shown in Fig. 3.

3.4.1. Sources of indoor dust

Sources of indoor dust were: road dust (69%); vehicle and cigarette sources (20%); oil combustion (6%); marine and secondary sources (3%); and coal combustion and traffic contributions (2%). The crustal factor is usually identified by the presence of elements such as Al and Fe (Amato et al., 2014), although Xie and Zhao (2018) identified a metal manufacturing dust factor from high factor loadings of both Al and Fe. As reported by Han et al. (2017), the source of Co is mainly from the machinery industries while Zn and As originate from human activities such as steel plants, metallurgy industries and commercial areas. Thus, road dust was identified by the high factor loading of Al, Fe, Co, Zn and

As in this study, where road dust is suggested to be transported to indoor classrooms via the movement of the school children. There are also high pollution levels from various industrial activities around the school area. Zhong et al. (2014) identified the main sources of indoor dust in a university building as from the earth's crust where strong factor loading was characterized by Cd, Mg^{2+} , and Ca^{2+} .

High factor loadings of NO_3^- (-0.825) and Cd (0.817) were identified as originating from vehicle and cigarette emissions. The oxidation of local gaseous NOx emissions from traffic and industrial plants produces NO_3^- (Amato et al., 2014; Mansha et al., 2012) which can then infiltrate the indoor environment (Viana et al., 2015). The influence of cigarette smoke emissions was identified with the element Cd which is usually used as a cigarette marker in indoor environments (Na et al., 2004; Slezakova et al., 2009). Even though smoking is prohibited throughout the school, the presence of cigarette smoke may influence indoor air quality due to transfer from smokers. As reported by Slezakova et al. (2009) and Hunt et al. (2011) cigarette smoke was found to be a major source of indoor pollution and is usually found in the fine fraction. The oil combustion source factor for indoor dust was identified mainly due to the presence of V. The outdoor source of V is the combustion of crude and residual oil which particularly relates to emissions from heavy machinery. V is enriched in emissions from crude or heavy oil fractions such as those used in power plants, large ships and automobile parts production industries (Han et al., 2017; Mohamad et al., 2016). The marine and secondary sources were identified with high factor loadings (> 0.70) for Na^+ , Cl^- , Ca^{2+} , Cu, Mg^{2+} , SO_4^{2-} and K^+ . As reported by Amato et al. (2014) and Khan et al. (2016), Na^+ and Cl^- are usually used as tracers for fresh sea salt particles while Taner et al. (2013) suggested that a high loading of Cu could be related to a sampling location that is near busy roads. As SO_4^{2-} had a moderate factor loading contribution, this ion is expected to penetrate indoors from outdoors as SO_4^{2-} particles have high penetration factors compared to SO_2 . SO_2 is produced from vehicle and industrial emissions (Patterson and Eatough, 2000). The coal combustion and traffic sources were characterized by F and Pb. F is also considered a marker for coal used both for domestic cooking and in power plants (Ando et al., 1998) while Pb is usually used as a marker for traffic sources (Al-Khashman, 2007).

Table 3
Factor loading after varimax rotation using PCA for indoor and outdoor dust.

Elements	Indoor					Outdoor		
	PC1	PC2	PC3	PC4	PC5	PC1	PC2	PC3
Na^+	0.924	- 0.105	- 0.359	0.054	0.061	- 0.270	0.213	0.936
K^+	0.894	- 0.005	0.050	0.251	- 0.367	- 0.631	0.416	0.548
NH_4^+	- 0.466	- 0.472	- 0.005	- 0.179	0.726	0.272	0.927	0.158
Ca^{2+}	0.945	- 0.001	- 0.167	0.267	0.092	0.642	0.648	0.293
Mg^{2+}	0.781	- 0.201	0.442	0.066	- 0.388	0.025	0.674	0.661
F^-	- 0.010	- 0.151	0.976	- 0.142	0.071	0.937	0.056	- 0.088
Cl^-	0.989	- 0.121	0.043	- 0.052	- 0.045	- 0.773	- 0.072	0.627
NO_3^-	- 0.390	- 0.296	- 0.258	- 0.825	- 0.117	- 0.312	0.315	0.860
SO_4^{2-}	0.899	- 0.123	0.416	0.041	0.048	- 0.736	0.499	0.365
Al	- 0.327	0.833	0.223	0.300	0.244	- 0.378	0.879	0.282
Pb	- 0.203	0.125	0.912	0.295	0.155	0.967	0.177	- 0.129
Cr	- 0.489	0.323	- 0.493	0.625	0.149	0.954	- 0.061	- 0.172
Fe	0.125	- 0.947	- 0.133	0.257	0.068	0.612	0.706	0.290
Cd	0.380	0.296	- 0.071	0.817	- 0.310	- 0.302	- 0.923	0.025
Co	- 0.139	0.973	- 0.083	0.150	- 0.066	0.928	0.090	- 0.339
Zn	- 0.175	0.925	- 0.239	0.237	0.025	0.992	0.023	- 0.084
V	0.125	0.453	0.392	0.115	0.782	0.959	0.198	- 0.033
Ni	- 0.043	0.913	- 0.243	0.319	0.068	0.897	0.073	- 0.350
As	- 0.163	0.832	0.144	0.510	0.014	0.932	0.196	- 0.229
Cu	0.913	- 0.356	- 0.189	0.035	- 0.044	0.954	0.163	- 0.245
Mn	0.295	0.613	0.478	0.550	0.080	0.916	0.306	- 0.228
Eigenvalue	8.05	6.79	3.29	1.51	1.33	12.8	5.98	1.18
% Variance	38.3	32.3	15.6	7.23	6.35	61.1	28.4	5.65
Possible Source	Marine + secondary source	Road dust	Coal combustion + traffic	Vehicle + cigarette emission	Oil combustion	Industrial activity	Soil	Sea salt + fuel combustion

Factor loading > 0.7 in bold.

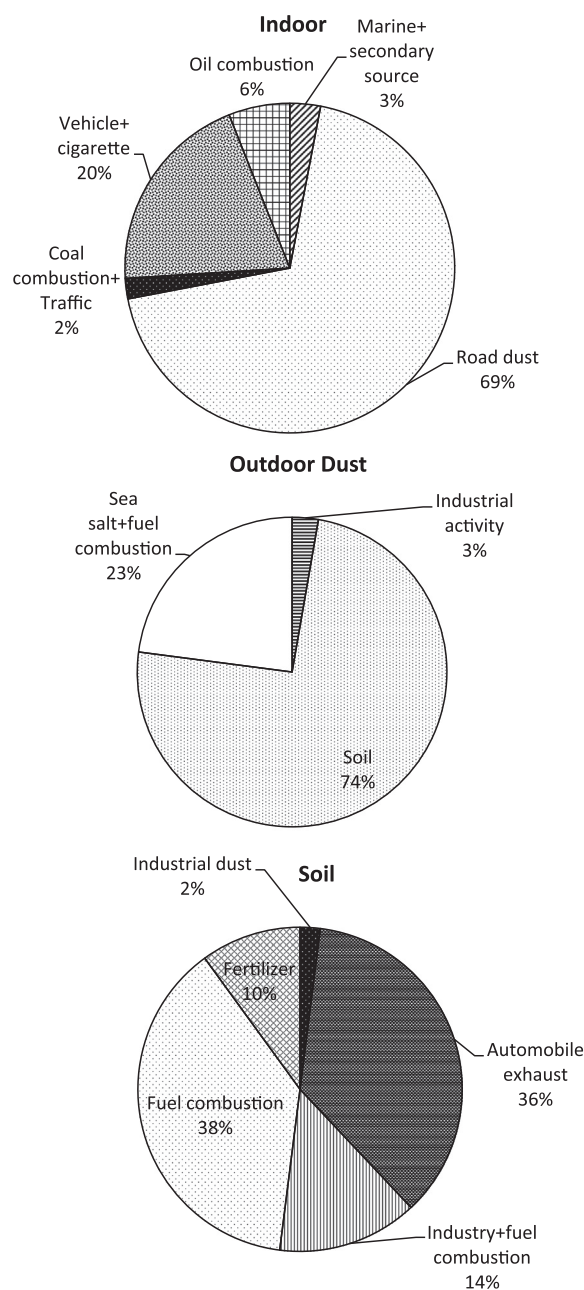


Fig. 3. Source apportionments of indoor and outdoor dust; and soil dust.

3.4.2. Sources of outdoor dust

Soil, sea salt and fuel combustion, and industrial activity recorded contributions of 74%, 23% and 3%, respectively, for the outdoor dust (Fig. 3) (Table 3). The high factor loading for NH_4^+ , Al and Cd showed the contribution of soil—as reported by John et al. (2007) NH_4^+ can come from undisturbed soil and fertilized land while Al originates from soil content. The sea salt and fuel combustion source recorded high factor loadings of Na^+ and NO_3^- of 0.936 and 0.860 respectively. The sea salt source was identified as the main contribution to Na^+ (Marenco et al., 2006) where the sea breeze and transportation of Na^+ particles influence the Na^+ contribution in the outdoor environment. The sea salt source in Kuala Lumpur was also observed by other researchers (Sulong et al., 2017; Wahid et al., 2013) where the Malacca Strait, which is around 40 km from Kuala Lumpur, was considered the probable source (Sulong et al., 2017). Han et al. (2013), who studied the composition of street dust in Kuala Lumpur City Centre, found that the highest contribution was from exhaust emissions, with the elements Pb,

Cr and Ni as indicators, while other source contributions were from soil and brake pad wear. The industrial activity source was identified with high factor loadings for Pb, Cr, Co, Zn, V, Ni, As, Cu and Mn. As reported by John et al. (2007) and Minguillón et al. (2012), high factor loadings of Cd, Cr, As and Ni showed a contribution from industrial sources. High factor loadings of Cu, Pb, Zn and Co were also characterized as emissions from various sources (Kabadayi and Cesur, 2010). A study by Marenco et al. (2006) had high factor loadings of elements such as Ca, Ti, V, Mn, Fe, where contributions from anthropogenic sources were identified for a sampling location at high altitude. A study by Meng et al. (2007) performed source apportionment analysis for pooled indoor and outdoor samples in three cities in the United States where sources of secondary aerosol, mobile/road dust, combustion, sea salt and soil factors were determined—similar source factors of outdoor dust in this study. Moreover, a study by Al-Khashman (2007) in a city area determined the major contribution of industrial activities and traffic to outdoor dust, which consistent with this study.

3.4.3. Source of soil dust

Source apportionment of soil dust revealed five source factors with Eigenvalues of > 1 (Table S6). Fuel combustion; automobile exhaust; industry and fuel combustion; fertilizer; and industrial dust sources contributed to 38%, 36%, 14%, 10% and 2% respectively for soil dust (Fig. 3). K^+ , F and SO_4^{2-} showed strong factor loadings which indicates a fuel combustion source. Biomass burning, usually determined by the presence of K^+ (Schlesinger, 2007), was considered the main combustion source, with contributions from anthropogenic activities such as coal and sulphur fuel combustion that were identified with F and SO_4^{2-} (Mishra et al., 2009; Pereira et al., 2011). The elements Cd and As had strong factor loadings of 0.969 and 0.904 respectively and described the contribution of anthropogenic such automobile exhaust. As reported by Cachada et al. (2012), Cd and As can be potential hazardous and toxic elements. As was also found at high concentrations in urban soil compared to soil from a rural area, suggesting an anthropogenic contribution (Liu et al., 2016) while the Cl⁻ contribution was from vehicle exhaust (Gulson et al., 1995). An industry and fuel combustion source factor was identified from the strong factor loadings for Zn, Cu and SO_4^{2-} . As reported by Yang et al. (2017), high factor loadings of Zn and Cu in soil indicated a high contribution of industrial activity sources such as metal industries and also municipal inputs. Fertilizer factor was indicated with NH_4^+ and suggested the influence of a fertilizer source in soil dust as NH_4^+ is usually found in soil due to fertilizer application (Nixon and Fulweiler, 2009). Previous studies have determined anthropogenic sources such steel production industries, coal combustion, cement factories, vehicle exhaust and burning of fossil fuel as the sources of pollution in soil (Al-Khashman, 2007; Rastmanesh et al., 2017; Yang et al., 2017). Fe, Co, Ni and Mn also had high factor loadings (> 0.70) which indicates sources such as the industrial dust.

3.5. School children health risk assessment

School children exposure to indoor and outdoor dust containing metal contaminants through dermal contact, ingestion and inhalation was determined in this study. The values of HQ and CR are shown in Table 4. The finding of the child health risk assessment for non-carcinogenic elements showed no significant risk ($\text{HQ} < 1$) for all exposure pathways for both indoor and outdoor dust. The highest HI value was recorded for the ingestion pathway with values of 2.54E-01 (indoor dust) and 7.06E-01 (outdoor dust) but this indicates no significant risk. For the dermal pathway, the highest CR value was recorded for Pb (outdoor dust) with a CR value of 1.60E-06, slightly lower than the acceptable value range. A total CR value of 2.23E-06 (lower than acceptable limit) was also determined. For the ingestion pathway, Pb in indoor dust and Ni, Pb and As in outdoor dust recorded higher CR values than the acceptable limit (1.0E-06) with a total CR value of 2.97E-05 (indoor dust) and 1.10E-04 (outdoor dust), but still fall in the

Table 4
Hazard Quotient (HQ) and Cancer Risk (CR) value for indoor and outdoor dust via dermal, ingestion and inhalation exposure pathway.

Element	Indoor		Outdoor	
	HQ	CR	HQ	CR
Dermal exposure				
Mn	1.53E-06		5.29E-06	
Cu	1.52E-05		7.89E-06	
Zn	2.54E-05		7.58E-05	
Ni	2.33E-05	9.33E-07	8.28E-06	3.32E-07
Cr (VI)	1.99E-05	1.02E-07	4.62E-05	2.38E-07
Cd	1.55E-05		3.45E-05	
Al	2.26E-05		8.59E-05	
Pb		8.28E-07		1.60E-06
As	5.76E-04	2.22E-08	1.67E-03	6.44E-08
Fe	1.05E-05		4.34E-05	
Total	7.10E-04	1.89E-06	1.98E-03	2.23E-06
Ingestion exposure				
Mn	5.48E-04		1.89E-03	
Cu	5.44E-03		2.82E-03	
Zn	9.08E-03		2.71E-02	
Ni	8.32E-03	7.84E-06	2.96E-03	1.19E-05
Cr (VI)	7.09E-03	1.82E-06	1.65E-02	5.31E-06
Cd	5.53E-03		1.24E-02	
Al	8.06E-03		3.07E-02	
Pb		1.48E-05		3.56E-05
As	2.06E-01	5.29E-06	5.96E-01	5.75E-05
Fe	3.74E-03		1.55E-02	
Total	2.54E-01	2.97E-05	7.06E-01	1.10E-04
Inhalation exposure				
Mn	8.93E-08		3.08E-07	
Cu	1.52E-07		7.88E-08	
Zn	2.54E-07		7.56E-07	
Ni	2.32E-07	4.20E-10	8.27E-08	1.49E-10
Cr (VI)	1.98E-07	3.32E-10	4.62E-07	7.73E-10
Cd	1.54E-07		3.45E-07	
Al	2.25E-07		8.57E-07	
Pb		2.69E-09		5.19E-09
As	5.75E-06	3.21E-10	1.67E-05	9.30E-10
Fe	1.05E-07		4.33E-07	
Total	7.16E-06	3.76E-09	2.00E-05	7.04E-09

tolerable limit (1.0E-06 to 1.0E-04). For the inhalation pathway, the HQ for all studied elements showed no significant risk, which was similar to the CR values for all studied elements and indicates a low risk of inhalation of both non-carcinogenic and carcinogenic elements. Overall, higher HI and total CR values for outdoor dust compared to indoor dust were observed, indicating a higher health risk from outdoor non-carcinogenic and carcinogenic elements. The sequence of child health risk routes for indoor and outdoor dust was ingestion > dermal > inhalation.

The HQ and CR values for exposure to non-carcinogenic and carcinogenic elements in soil dust for all pathways are listed in Table S7. For the dermal pathway, the highest HQ was recorded for Cr (VI) while the highest CR value was recorded for Pb. The HI value showed no significant risk while the total CR was higher than the acceptable limit but still in the tolerable range. The ingestion pathway exhibited HI values of 4.43E-01 and 8.08E-05 for total CR which indicated a low risk for non-carcinogenic health effects, but there was a risk of carcinogenic health effects. Child health risk via the inhalation pathway was also low with no significant health risks of exposure to both non-carcinogenic and carcinogenic elements. The sequence of pathways for non-carcinogenic and carcinogenic risk in soil dust was ingestion > dermal > inhalation.

Outdoor dust exposure assessment revealed higher health risks compared to indoor dust, mainly due to high ambient pollution from traffic, industry emissions and other anthropogenic activities. The minimal values of HQ and CR for all pathways reflect low concentrations of these elements, especially in dust particles which are in the coarse particle size range. As reported by Othman et al. (2018),

exposure to different sizes of particle can give different values of HQ and CR, where fine particles provide much higher health risks. A study by Hu et al. (2012) also found that fine particles produce higher HQ and CR values compared to coarse particles, which indicates that fine particles significantly affect human health. Moreover, major health risks via exposure through ingestion pathways in this study were also consistent with the study by Neisi et al. (2016), where non-carcinogenic elements in indoor dust via ingestion pathways recorded higher HQ values compared to dermal and inhalation pathways.

4. Conclusions

The concentrations of PM_{2.5} and the compositions (ions and trace metals) of dust in an indoor and outdoor school classroom located in Kuala Lumpur City Centre were investigated. The average 24 h sampling duration indoor PM_{2.5} concentration was 11.2 ± 0.45 µg m⁻³ and the outdoor was 11.4 ± 0.44 µg m⁻³. Both indoor and outdoor 24 h PM_{2.5} average concentrations were below the WHO threshold value (25 µg m⁻³), the National Ambient Air Quality Standards by USEPA (35 µg m⁻³), and the New Malaysia Ambient Air Quality Standard for PM_{2.5} Interim Target (IT-2) for 2018 (50 µg m⁻³). The deposition fraction of particles in the respiratory tract of children was found to be mainly deposited in the lungs, with higher total deposition fractions in indoor compared to outdoor environments. The dominant elements were Al, Fe, Zn, Ca²⁺ in indoor and outdoor dust where road dust and soil were suggested as the major sources respectively. I/O values were recorded at > 1 for Na⁺, Cr, Cd, Ni, Cu, K⁺, Ca²⁺, Mg²⁺ and Cl⁻ where the presence of indoor sources as the main contribution of these elements other than the contribution from outdoor. For the school child health risk assessment, there was no significant risk from either the non-carcinogenic or the carcinogenic elements derived from the dust samples, where the sequence of exposure pathways was ingestion > dermal > inhalation for both the indoor and outdoor school classroom. This study suggests detailed investigations of indoor air quality and the maintenance of school classroom ventilation systems, especially in highly urbanized areas such as Kuala Lumpur, are required.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.ecoenv.2018.12.042.

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