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Comprehensive Health Risk Assessment of Trace Metals in PM 2.5 : An Urban Perspective

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Abstract

Air pollution is one of the most significant problems faced by all countries, including Malaysia. Traffic emissions are a significant contributor to air pollutant emissions, particularly in urban areas, which can have adverse health impacts on vulnerable populations. The study conducted at Jalan Raja Muda Abdul Aziz, Kuala Lumpur, was to determine the concentration of selected trace metals and their health risks to humans through inhalation exposure during the Southwest Monsoon (SW) and Monsoon Transition (IM). The study's results showed that the concentration of PM2.5 was higher during the SW monsoon compared to the IM, at 19.95 μg/m^3 and 13.13 μg/m^3, respectively. The ∑ ELCR trend for trace metals varies by age group, with adults having the highest levels, followed by children, toddlers, and infants. In addition, Cr for the adult group had the highest CR values during the SW monsoon compared to the IM monsoon with readings of 9.21×10^{-7} and 7.67×10^{-7}, respectively. The lung exposure assessment showed that the 60-year-old group of men had the highest amount of $PM_{2.5}$-bound trace metal inhalation. The Multiple-Path Dosimetry Model (MPPD) revealed that the head region (46%) had the highest deposition amount, followed by the pulmonary (45%) and tracheobronchial regions (9%). Deposition in the lobar area is highest in the left lower (LL) (35%) and lowest in the right middle (RM) (7%). The deposition mass (D_m) was highest for the adult (1.53×10^{-5} μg) compared to the infant (9.00×10^{-6} μg). However, the mass of deposition per unit area (D_{ma}) indicates that the toddler (0.0467 μg/m^2) has a higher value than the adult (0.0116 μg/m^2).In conclusion, exposure to trace metals bound- $PM_{2.5}$ in the study area has the potential to have health effects on the surrounding community.

1. Introduction

Air pollution is a significant environmental and public health concern worldwide, particularly in urban areas where industrial activities, vehicular emissions, and other anthropogenic sources contribute to the deterioration of air quality. Kuala Lumpur is a rapidly developing urban area with various developments including industry, infrastructure and residential areas and it also contributes to the growth of the Malaysian economy (Wahab et al., 2020). The demand for the transportation system is influenced by several factors, including economic development, increased of activities that can provide income and various social and recreational activities. The number of private vehicles in the city has increased, which has resulted in deteriorating air quality and raising concerns around the city centre (Shafie et al., 2020; Saboori et al.; 2014; Shahbaz et al., 2014).

Among the various air pollutants, particulate matter with an aerodynamic diameter of less than 2.5 micrometers ($PM_{2.5}$) has gained significant attention due to its potential to penetrate deep into the respiratory system and even enter the bloodstream, posing severe health risks (Brook et al. 2010). This is due to the particle size, which will easily enter and penetrate deeply into the lungs and blood (Han et al., 2021; Singh et al., 2021). Exposure to fine particulate matter has been linked to several negative health impacts, such as cardiovascular disease, lung function decline, and even premature infant death (Guaita et al., 2011). Trace metals bound $PM_{2.5}$ are non-volatile, less susceptible to chemical changes and remain in the form when they are released, although they tend to disperse further into the atmosphere (Morawska et al., 2002). The size of particulate matter plays a crucial role in the accumulation of trace metals and their effects, as finer particles have a larger surface area. Therefore, the accumulation of trace metals is more effective than coarser particulate matter (PM) (Zhang et al., 2013).

The presence of trace metals in $PM_{2.5}$ further exacerbates these risks, as metals such as lead (Pb), cadmium (Cd), arsenic (As), and chromium (Cr) are known for their toxicity and carcinogenic properties (Manisalidis et al. 2020). These trace metals can originate from industrial emissions, the combustion of fossil fuels, and the resuspension of road dust, significantly contributing to human exposure and adverse health effects.

Despite increasing awareness of air pollution hazards, comprehensive assessments of trace metal-bound $PM_{2.5}$ in urban areas remain limited, particularly concerning their health risk implications (Liu et al. 2019). Traditional air quality monitoring often focuses on overall PM2.5 concentrations without detailed chemical characterization, thereby overlooking the specific contributions of hazardous trace metals. This knowledge gap poses a challenge in formulating effective air quality management strategies and public health interventions. Furthermore, existing studies predominantly focus on short-term exposure. In contrast, lung exposure assessments are crucial for a more accurate evaluation of potential health outcomes (Gao et al., 2018).

This study aims to assess and evaluate the health risks associated with trace metals bound to PM2.5 in an urban environment. The primary objectives of this research are: (1) to quantify the concentration of trace metals in $PM_{2.5}$ collected from different locations within the study area, (2) to evaluate the carcinogenic and non-carcinogenic health risks associated with human exposure to trace metal-bound $PM_{2.5}$ through inhalation pathways (3) to identify the lung exposure to trace metals-bound $PM_{2.5}$, and (4) to determine the deposition in the human respiratory tract.

By providing a comprehensive evaluation of the chemical composition and health risks associated with $PM_{2.5}$ in an urban setting, this research will offer valuable insights into pollution sources and their impacts on public health. The findings of this study are expected to support policymakers in implementing targeted air pollution control measures and developing strategies to mitigate the adverse health effects of $PM_{2.5}$ exposure, ultimately contributing to improved air quality and public health outcomes.

2. Materials and Methods

2.1 Study area

The sampling location is on the rooftop of Block B of the Faculty of Health Sciences, Universiti Kebangsaan Malaysia (UKM), Kuala Lumpur Campus (3°10'06.3"N, 101°42'03.5"E) at Jalan Raja Muda Abdul Aziz, Kuala Lumpur, and the height is approximately 20 m above ground level. This area was surrounded by various institutions and human activities such as higher education institutions, hospitals, shop buildings, residential areas, wet markets, recreational areas and others. In addition, this faculty is in the center of Kuala Lumpur city, and is located 10, 30 and 40 m from three busy roads in the city of

Kuala Lumpur namely; Jalan Chow Kit, Jalan Raja Muda Abd Aziz and Jalan Pahang respectively. Each road is characterized by congested traffic, featuring vehicles such as public buses, cars, and motorcycles. This area has many major roads and highways that are the cause of traffic jams that are often the focus of the city's citizen.

2.2 $PM_{2.5}$ Sampling

This study was a cross-sectional study conducted from 13 June to 31 October 2022. A total of 30 samples have been collected during the Southwest Monsoon (SW) and Intermonsoon (IM) seasons. The sampling guideline was based on the Compendium Method IO-2.1 by USEPA (Winberry, 1999). Quartz microfiber filter papers (20.3 cm × 25.4 cm, Whatman, UK) were placed into aluminum foil before being burned at 400°C for 5 hours using a furnace (Thermolyne 48000 Furnace). Then, the filter paper will be placed inside the desiccator for 24 hours to achieve equilibrium in humidity and temperature. A high-volume sampler (HVS, Tisch Environmental, USA) was operated for 24-hour sampling at a flow rate of 39.0 CFM to 45.0 CFM per sample.

2.3 Acid Digestion

The acid digestion method was according to the Compendium Method IO-3.1 by USEPA (Mainey & Winberry, 1999). The small cut filter papers (1 cm × 1 cm) were digested using 3.0 mL 65% solution of suprapur nitric acid and 4.0 mL 30% hydrogen peroxide solution. Digestion of trace metals was conducted using microwave-assisted (PerkinElmer Titan MPS Microwave Sample Preparation System). Next, the digested sample was filtered using a 10 mL syringe and Acrodisc filter into 10 mL centrifuge tube. This microwave works in three steps (Perkin Elmer 2013): 1) 10 minutes at a temperature of 160°C with a ramping time of 2 minutes; 2) 25 minutes at a temperature of 180°C with a ramping time of 2 minutes; and 3) 10 minutes at a temperature of 50°C with a ramping time of 1 minute. The total time required for sample digestion is 50 minutes. After that, the digested sample will be left to cool for 30 minutes before being removed from the wave digester. Next, the sample was extracted from the digestion vessel and put into a 50 mL centrifuge tube. The digester vessel needs to be rinsed thoroughly using 25 mL of type II deionized water and then poured into the centrifuge to ensure that all the extracts from the vessel are not left behind. Then, using a 15 mL plastic syringe and a 0.45 µm PTFE filter, the sample extract was filtered and put into a 50 mL centrifuge tube. Finally, 10 mL of the sample extract was poured into a labeled 15 mL centrifuge tube and ready for extraction of trace metal using ICPMS. The filtered stocks were kept in the refrigerator at 4°C before further analysis.

2.4 Backward trajectories and fire hotspots

Wind travel paths of back trajectories and biomass hotspots for each season were plotted. Ensemble averages of backward trajectories were generated using the Lagrangian Hybrid Single Particle Integrated Trajectory Model (HYSPLIT 4.9). Backward trajectories were performed at the sampling location during the sampling day with the highest concentration which is on 6 July 2022 (SW monsoon) and on 27 October 2022 which represents the Intermonsoon. Release heights of approximately 100m, 500m and 1000m for the 120-hour back trajectory and the trajectory start time were chosen at 0000 (UTC). Average trajectories are represented by numerical values and colors (100m – red, 500m – green, 1000m – blue).

2.5 Meteorological Data

Meteorological data such as rain distribution and temperature were obtained from the Official website of Malaysian Meteorological Department (https://www.met.gov.my/en/)

2.6 Multiple-Path Particle Dosimetry

In this study, MPPD model software version 3.04 was used to obtain the deposition of particles in human lungs. There are 3 main steps to carry out this model which involves the selection of the morphometry of the respiratory tract, the particle properties and the exposure scenario. Each step has parameters to fill, according to the specifics of the research to be conducted. Various pathways for this model indicate that estimates are possible for all lung airways where a specific deposition of lobar can be obtained through this method. Therefore, a model of 5 age-specific lobes is adopted in this research. The PM size used for this study is 2.5 µm for the morphometry of the human airways. In addition, the density, aspect ratio, PM diameter should be identified because the properties of the PM play an important role in carrying out this model. There are 3 options to choose for the particle diameter which are the median diameter of the count (CMD), the median diameter of the mass (MMD) and the median aerodynamic diameter of the mass (MMAD). Next on the move involves a disclosure scenario that involves continuous exposure and variable exposure to the PM. Constant exposure is the average PM concentration throughout the research, while variable exposures involving all of the above-mentioned parameters which can be changed as hourly patterns or activities. In addition, there are 9 body orientations available in this model. However, in this study, an upright position has been selected. All particles are assumed to enter the respiratory tract through the nose. Finally, the deposition and discharge dose can be evaluated after the completion of determining all the input data in the model and the deposition of particles in the lungs can be obtained. In this study, the required input data only uses the default data found in the MPPD model software (Manojkumar et al., 2019). The age group found in this software for the age-specific 5 lobe model is only 3 months, 28 months, 3 years, 8 years, 9 years, 14 years, 18 years and 21 years. However, in this study, the 9-year age group was excluded because the results were the same as the 8-year age group. The input data used in this study are as shown in Supplementary 1.

2.7 Lung Exposure Assessment

The average lung exposure to the amount of trace metal bound- PM2.5 for 24 hours of exposure is estimated using this method. There are 3 sub-populations selected for this study in this study which is age groups of 13–19 years (teens), 20 years, 30 years, 40 years, 50 years, 60 years and 70 years old for males and females. The respiratory flow rate and the amount of trace metal bound-PM2.5 inhaled for 8 hours of exposure was used can be obtained by using the equations below (Eq. 3 and Eq. 4) (Quintana-Belmares et al., 2018).

$RFR = TV \times BR \times 60\ min$ (3)

$Itm / 8h = RFR \times \sum Ctm \times 24\ h$ (4)

where RFR is the respiratory flow rate (L inhaled / h), TV is the tidal volume (L / normal inhalation), BR is the respiratory rate (breath / min), Itm / 24h is the amount of trace metal bound-PM2.5 inhaled for 24 hours of exposure, and ∑ Ctm is the sum of the concentration of trace metal. The value of TV and BR is derived from previously reported data under rest conditions (Quintana-Belmares et al., 2018). Then, the RFR results for each age and gender are multiplied by the sum of the concentration of the selected trace metal to obtain the exposure value of the trace metal in the lungs. The estimated amount of trace metal inhalation is based on 24 hours of exposure. Physiological changes with each increase in age mean the limits for younger adult do not always apply. This causes each individual to have a different breathing rate (Barret et al., 2008). The tidal volume and respiratory rates for males and females of particular ages from the previous study as shown in Supplementary 2 (Quintana-Belmares et al., 2018).

2.8 Health Risk Assessment

This model is used to estimate non-carcinogenic risk and carcinogenic risk of exposure to selected trace metals in $PM_{2.5}$ based on USEPA. In this study, we only assessed respiratory pathways because breathing is the main route of exposure to fine suspended particulate matter. Health Risk Assessment (HRA) model involves two types of risk, namely non-carcinogenic risk and carcinogenic risk. The human Health Risk Assessment model involves two risk assessments, namely non-carcinogenic risk and carcinogenic risk involving exposure through inhalation. This study included four specific age groups, namely infant (0–1 year), toddler (2–5 years), children (6–12 years) and adult (19–75 years) (Nadzir et al., 2021). This is due to each population group having a different susceptibility to pollutants. However, the group of 13–18 years was excluded from this study because no parameters could be identified through previous studies. For non-carcinogenic risk, average hourly dose (AHD) for acute exposure and average daily dose (ADD) for chronic exposure were obtained using the below equations respectively (Eq. 5 and Eq. 6) (Morakinyo et al., 2017; USEPA, 2005).

AHD = C × (InhR / BW) (5)

ADD = C × InhR × EF × ED / BW × AT noncarc (6)

Then, the non-carcinogenic risk was obtained through the calculation of outcomes for Hazard Quotient (HQ) and Hazard Index (HI) (Eq. 7 and Eq. 8) (USEPA, 2005).

HQ = ADD / RfD (7)

HI = ∑ HQ (8)

For $PM_{2.5}$, no reference dose (RfD) can be identified through previous studies. Thus, the RfD for $PM_{2.5}$ was derived from the RfC that obtained from the previous studies, where the RfC for $PM_{2.5}$ was 5 µg/m3 (Oliveira et al., 2012) (Eq. 9) (Chalvatzaki et al., 2019).

RfD = RfC × (InhR / BW) (9)

The lifetime average daily dose (LADD) through inhalation was obtained using Eq. (10) which is then multiplied by the slope factor of cancer (CSF) to obtain a cancer risk value (CR) using Eq. (11). Next, an excess lifetime cancer value (ELCR) is obtained using Eq. (12) (USEPA, 2005).

LADD = C × InhR × EF × ED / BW × ATcarc (10)

CR = LADD × CSF (11)

ELCR = ∑ CR (12)

In addition, the CSF value of each carcinogenic metal is based on the IUR derivative as found in Eq. (13) (Chalvatzaki et al., 2019).

CSF = IUR × (BW / InhR) (13)

where, C is the concentration of pollutants in ambient air, InhR is the inhalation rate (m3/day), ED is the exposure duration (days), BW is the body weight of the exposed group (kg), AT is the average time for non-carcinogenic (noncar) and carcinogenic (carc) pollutants (days), EF is the frequency of exposure (day / year), ADD is the average daily dose, RfD is the reference dose for the chemical compounds used, LADD is the average daily dose of a lifetime, IUR is the risk of inhalation units and CSF is a cancer slope factor. The parameter values for the calculation of risk assessment can be referred to the Supplementary 3 (Nadzir et al., 2021). HQ and HI are the last element to be obtained to estimate the non-carcinogenic risk from exposure through inhalation by using the RfD as per Table 1. HQ < 1 and ∑ HI < 1 indicate that no non-carcinogenic risk can occur when exposed to the particulate matter. In addition, the classification of cancer risk for exposure to PM2.5 and trace metals of carcinogen elements can be referred to the Supplementary 4 (Latif et al., 2019).

Table 1
Reference dose (RfD) and inhalation risk unit (IUR) values for the parameters involved

Parameter	Non-carcinogenic reference value		Non-carcinogenic reference value	
	RfD (μg/kg-day)	Reference	IUR $(\mu g/kg\text{-}day)^{-1}$	Reference
$PM_{2.5}$	-	-	8.0×10^{-3}	(Greene & Morris, 2006)
Mg	7.38×10^{2}	(Buranatrevedh, 2013)	-	-
Fe	8.25×10^{3}	(Buranatrevedh, 2013)	-	-
Cr	2.86×10^{-2}	(Zheng et al., 2015)	1.20×10^{-2}	(IRIS, 1998)
Ni	2.06×10^{1}	(Zheng et al., 2015)	2.40×10^{-4}	(IRIS, 1998)
Co	5.71×10^{-3}	(Zheng et al., 2015)	9.00×10^{-3}	(Pavilonis et al., 2017)
As	3.00×10^{-1}	(Kamunda et al., 2016)	4.30×10^{-3}	(IRIS, 1983)
Cd	1.00	(Zheng et al., 2015)	1.80×10^{-3}	(IRIS, 1985)
Pb	3.50	(Zheng et al., 2015)	1.20×10^{-5}	(OEHHA, 2009)
Mn	4.6×10^{1}	(Morakinyo et al., 2021)	-	-
Zn	3.0×10^{2}	(Qu et al., 2012)	-	-
Cu	4.0×10^{1}	(Qu et al., 2012)	-	-
V	7.00	(Zheng et al., 2015)	-	-

2.9 Quality Assurance and Quality Control

Several actions have been taken to ensure the quality of the samples is assured and effective. The wearing of gloves and plastic tweezers is very important throughout the study to avoid any contamination that could interfere with the accuracy of the sample. In addition, each apparatus used has been sanitised in advance to reduce surface exposure to objects. During the cutting process, the part of the surface filled with the sample is at the top. When performing the sample preparation procedure, vinyl gloves should be used to preserve the purity of the sample. All tools and utensils are thoroughly washed with acid and rinsed with water. Calibration has been performed for each instrument used to ensure measurement accuracy and reliability. Next, blank samples for each group of filter paper and field samples are prepared to identify any contamination present. The same analytical weigher is also used throughout the study to ensure weighing accuracy. In addition, the temperature and humidity of the room during weighing should be similar to the temperature and humidity during the moisture balance process. Therefore, analytical scales and decimators are placed in the same room. Upon completion of each process, the filter paper is placed in the refrigerator at a temperature of 4°C. For trace metal analysis, the concentration of trace elements was corrected for reagent and filter blanks samples, which were treated with a method similar to the filters exposed. Urban Particulate Matter SRM1648A corresponds as references material was used to calculate the recovery percentage of trace elements.

3 Results and Discussion

3.1 $PM_{2.5}$ concentrations

A total of 30 samples of $PM_{2.5}$ have been collected during the sampling period. The average concentration of $PM_{2.5}$ is 51% higher during the SW monsoon compared to the IM monsoon at 19.95 $\mu g/m^3$ and 13.13 $\mu g/m^3$ respectively. The difference between the maximum reading of SW monsoon (6th of July) and IM monsoon (27th of October) was as much as 20% where the reading clearly exceeds the Malaysian Ambient Air Quality Guidelines (MAAQS) and the National Ambient Air Quality Standards (NAAQS) from the United States Environmental Protection Agency (USEPA) (DOE, 2020; USEPA, 2022). On the other hand, all of the reading of $PM_{2.5}$ concentration were exceeding the WHO guideline which is 5 $\mu g/m^3$ (WHO, 2021a). Figure 2 shows the daily concentration of $PM_{2.5}$ collected at the sampling site with $PM_{2.5}$ standards value from MAAQS, NAAQS and WHO guideline. Meteorological data from the Malaysian Meteorological Department (METMalaysia) shows an increase of 48% in the average monthly rainfall during the SW and IM monsoon by 155.9 mm and 230.0 mm respectively. Meanwhile, the average temperature is also slightly higher during the SW monsoon compared to the IM monsoon which is 27.5°C and 27°C respectively, as shown in Table 2.

Table 2
Summary of $PM_{2.5}$ concentration and meteorological data during the sampling period.

Element	Overall	Southwest Monsoon (SW)	Intermonsoon (IM)
	13 June – 31 Oct 2022 (n = 30)	13 June – 28 Aug 2022 (n = 17)	13 Oct – 31 Oct 2022 (n = 13)
Average (μg/m^3)	17.00 ± 9.7	19.95 ± 9.34	13.1 3 ± 8.65
Minimum (μg/m^3)	6.15	11.07	6.15
Maximum (μg/m^3)	50.29	50.29	39.98
Monthly average rainfall (mm)	188.0 ± 36.91	155.9 ± 4.92	230.0 ± 0
Daily temperature average (°C)	29.5 ± 1.48	29.5 ± 1.61	29.6 ± 1.27

Overall, the SW monsoon has a higher concentration of $PM_{2.5}$ compared to IM monsoon. The results of this study are similar to the previous studies which stated that $PM_{2.5}$ concentrations are higher during the dry season in Malaysia due to its small rainfall distribution of around 100–150 mm in Peninsular Malaysia (Othman et al., 2022; METMalaysia, 2018). This phenomenon will affect the re-circulation, collection and chemical reaction of airborne contaminants due to stable atmospheric conditions. Open burning practices by farmers and higher incidences of natural forest fires during the dry season, including the low rainfall and high air pressure have led to the accumulation of air pollutants. According to Chantara (2021), the SW monsoon in Peninsular Malaysia was different from the IM monsoon where at this time, there were often thunderstorms that produce the maximum average monthly rainfall. Othman (2021) argues that the diversity of $PM_{2.5}$ was influenced by factors such as meteorological parameters, seasonal effects, and biomass combustion sources. The highest PM emissions in Kuala Lumpur were released by private cars and motorcycles and contribute to air pollution with 214,427 kg released in 2014 which is caused by traffic congestion, fuel burning, and morphological aspects of the background of the city itself (Shafie & Mahmoud, 2020). The increase in $PM_{2.5}$ concentrations was also closely related to emission sources either from local or transboundary sources.

Kuala Lumpur is an urban area with an important traffic network that often experiences traffic congestion, especially in the morning and evening (Sulong et al., 2017; Khan et al., 2016a). According to Azhari (2020), the source of traffic is a major contributor to $PM_{2.5}$ emissions in urban areas associated with traffic jams during peak hours. Research conducted at the sampling station in another nearby urban area in Batu Muda, Kuala Lumpur in 2019 also found that the average concentration of $PM_{2.5}$ during the months of July to September was higher at 61.78 μg/m^3 compared to October-December which is 20.16 μg/m^3 (Zainal et al., 2021). In the same study, the average concentration of $PM_{2.5}$ in another city which is Cheras, Kuala Lumpur found that the concentration of $PM_{2.5}$ during the SW monsoon was 59.54 μg/m^3 and 20.96 μg/m^3 during the Intermonsoon-Southeast Monsoon (IM-NE monsoon). The higher $PM_{2.5}$ concentrations during this period are mostly due to drier weather conditions, a stable atmosphere, effects of local activities, and transboundary movement of air pollution from biomass burning in neighbouring countries (Koplitz et al., 2016; Kusumaningtyas & Aldrian, 2016; Asif et al., 2018). Other than that, the southwest wind that dominates this area is also responsible for transporting pollutants such as $PM_{2.5}$ to Malaysia, especially from Sumatera Island, Indonesia, which is located in the southwest quadrant of the Southeast Asia (SEA) region (Khan et al., 2015; Lee et al., 2017).

The comparison of $PM_{2.5}$ concentration with other previous studies shows the dry season (11.40 μg/m^3) in Singapore has a lower concentration of $PM_{2.5}$ than the wet season (32.80 μg/m^3) which is different from the results of this study. This is because, the air mass in the dry season in Singapore occurs when the NE monsoon was said to originate from the Southern Continent of China (Wang et al. 2022). According to Zhang et al. (2022), the highest secondary $PM_{2.5}$ pollutant emissions in South China occur in winters spread from northern China in the presence of monsoons. Winter season which is also known as the dry season has a stable atmospheric condition that contribute to the accumulation of $PM_{2.5}$. This is likely to lead to an increase in $PM_{2.5}$ in Singapore due to long-distance transportation. However, higher $PM_{2.5}$ concentrations in the wet season in Singapore are said to originate from the northwest coast of Indonesia across South Sumatra province, Lampung province, Banten province, Sawu Sea and partly stem from the Pacific Ocean (Wang et al., 2022). In addition, the concentration of $PM_{2.5}$ in this study was lower than in other countries due to differences in activity in each of these cities (Enotoriuwa et al., 2016) that led to anthropogenic sources. Other studies conducted in Hanoi, Vietnam (Bui et al., 2022) and Colombo, Sri Lanka (Shelton et al., 2022) showed that the study areas carried out construction and industrial activities while Kuala Lumpur, Malaysia was a city that involving the services and construction sector (DOSM, 2020). Therefore, the resulting $PM_{2.5}$ concentrations vary according to the source of emissions and long-distance transportation in addition to different geographical positions and meteorological conditions (Sippula et al., 2013; Valavanidis et al., 2008).

3.2 Trace Metal Concentration

There are 14 trace metals that have been chosen throughout the sampling period as shown in **Fig. 5**. The results show that the selected trace metals are divided into 2 categories which are major elements and minor elements. The major elements including Na, Ca, Mg and Fe while the minor elements including Cr, Ni, Co, As, Cd, Pb, Mn, Zn, Cu & V. Major elements dominate approximately 96% and 97% of the total amount of trace, meanwhile other trace metals that were also analyzed accounted for less than 1% of the total trace metals. Comparatively, the concentration of trace metals is higher during the SW monsoon for all trace metals, except for Na, Ca, Zn & Pb which show high concentrations during the IT Monsoon. The results are in line with some other research such as in Bangi, Malaysia (Khan et al., 2016a), Penang, Malaysia (Saseetheran et al., 2021), Skudai, Malaysia (Dahari et al., 2022), Brazil (Paulino et al., 2014), East Java, Indonesia (Humairoh et al., 2020), Riau, Indonesia (Siregar et al., 2022), Kolakata, India (Das et al., 2015) Central Liaoning, China (Guo et al., 2021) and Harbin, China (Luo et al., 2021) which shows crustal elements contribute to the majority of trace metal concentrations in $PM_{2.5}$. Even so, Na and Ca were found to be higher during the IM monsoon compared to SW monsoon. According to a study in Delhi, the concentration of Na was also found to be higher

during the post-monsoon season, which is a cooler and more humid season than during the monsoon season (hotter and higher wind speed) (Bangar et al., 2021). While in the Czech Republic, Na was found in a higher concentration during winter than in summer (Seibert et al., 2020).

Furthermore, according to Dahari et al. (2022), the concentration of trace metals increases during the SW monsoon when compared to the intermonsoon for several reasons, including local anthropogenic activities and transboundary movement of pollutants as well as possible effects of wind flow patterns & speeds. During the inter-monsoon, the rainy season will begin with lesser amounts of rain than during the Northeast (NE) monsoon. The onset of rain during IM monsoon can cause the concentration of trace metals in the atmosphere to decrease due to the wet deposition process. Conversely, trace metal concentrations are higher during the SW monsoon due to the slower dispersion of pollutants during this time. Furthermore, for pollutants that bind to $PM_{2.5}$, studies show that meteorological conditions such as temperature, relative humidity, wind speed and wind direction are associated with the formation and elimination of these substances (Hua et al., 2016).

An independent t-test was conducted to compare the means trace metals and monsoons. The normality test conducted has shown that the data was in normal distribution. The findings of the study show that there is no significant difference ($p > 0.05$) between SW monsoon and IM monsoon for the concentration of Na, ($p = 0.368$), Mg ($p = 0.642$), Ca ($p = 0.349$), Co ($p = 0.157$), As ($p = 0.385$), Cd ($p = 0.191$), Pb($p = 0.448$), Zn ($p = 0.539$) & Cu ($p = 0.181$, Meanwhile, there is a significant difference between the SW monsoon and IM monsoon for the metal concentrations of Fe ($p = 0.02$), Cr ($p = 0.00$), Ni ($p = 0.001$), Mn ($p = 0.002$), V ($p = 0.029$). The metal concentration of Na, Mg and Ca were associated with the crustal elements or natural elements (Pipal et al., 2014; Rai et al., 2020).

In addition, according to Wanget al. (2020), building materials in cement widely used in construction sites contain a high content of Ca, Mg and Na (Wang et al., 2020). In addition, Na and Ca are also identified as one of the marker substances for pollution sources due to biomass burning and the release of vehicle smoke (Siregar et al., 2022). Additionally, the high Na concentration during IM monsoon can be related to the wind direction during IM monsoon seen from the Andaman Sea because Na is one of the components or tracers of marine aerosols (Bove et al., 2016; Manousakas et al., 2015). In addition, Ca and Mg minerals are also classified as sources of mineral dust or natural components of natural dust (Fugitive dust) which originates from roads, construction, and others (Dall'Osto et al., 2013; Moreno et al., 2013). In most studies state that Zn results from vehicle emission sources (Shi & Lu, 2018) and road dust, mainly from tires, motor oil and even the use of motor vehicle brakes (Alias et al., 2020). This coincides with the study that was conducted in urban areas with dense traffic networks and a growing number of vehicle ownership due to urbanization (Azhari et al., 2021).

A study in Harbin, China recorded the highest average Zn mass which involved the emission of pollutants from traffic density and heat supply industries resulting from burning coal and pollutants formed from atmospheric physics/chemical ion reactions (Wang et al., 2020). He observed during a period without heat supply, there is a higher humidity than during the heat supply period which can cause PM's precipitation. This is likely because high humidity will cause the particles to be too heavy to remain in the air. Thus, dry deposition occurs where particles fall to the ground and result in a decrease in the number and concentration of $PM_{2.5}$. It is different when there is a constant supply of heat in winter which contributes to a more significant release of particles into the atmosphere.

Kim (2018) stated that As, Zn and Pb are the result of the release of the smelting industry. In addition, As is also said to be one of the results of emissions from the charcoal-burning industry (Duan & Tan, 2013). Pb, Cd, Mn and Cu are also associated with the release of traffic that is usually very abundant in urban areas (Sakunko et al., 2022). The metals are usually produced from debris of tire wear and old car combustion systems. According to Amarloei et al. (2020), the metals identified in the atmosphere are primarily caused by traffic emissions, fuel oil combustion, or the re-suspension of particles. However, natural resources such as dust storm events are no exception to being a cause of PM emissions into the atmosphere. The source of emissions for Mn and V is likely due to local industrial emissions. This is because Mn and V are said to be caused by anthropogenic sources of fuel combustion (Thangavel et al., 2022) i.e. emissions from the industry. Additionally, other studies have found that Mn is a marker element for the metal smelting industry (Dai et al., 2015). Wu (2019) and Sakunko (2022) in their study reported that dust in particular soil dust and construction also contributed to an increase in the concentration of Mn in the atmosphere. In addition, in a study in Malamocco, Italy, it was said that V is likely to result from the release of ship traffic due to the burning of heavy oil (Contini et al., 2012). According to Sulong et al. (2017), traffic is usually the main source of ambient $PM_{2.5}$ pollution in the city. This corresponds to a study conducted by Rahman et al. (2011) which linked vehicle emissions which accounted for 30% of $PM_{2.5}$ in the Klang Valley. Meanwhile, Ni and Co are associated with the release of smoke from heavy vehicles using diesel (Das et al., 2020; Lin et al., 2015; Wang & Hopke. 2013) and Fe is associated with vehicle emissions and road dust (Khanna et al., 2015; Pant et al., 2015). Additionally, Cr and Fe are associated with tire impact sensors and brake lining wear on vehicles (Böhlandt et al., 2012; Lin et al., 2015).

3.3 Lung Exposure Assessment

The total amount of inhaled trace metals bound-$PM_{2.5}$ for major and minor elements have been obtained for both monsoons as shown in **Fig. 6**. The results showed that the amount of major elements inhaled was higher during the IM monsoon compared to the SW monsoon due to the high concentration of major elements. In contrast, the amount of minor elements inhaled was higher during the SW monsoon compared to the IM monsoon. In general, inhalation exposure for both major and minor elements was increasing as age increases. The trend of the inhaled amount of major and minor elements increased from the teenage group until the age of 40 and undergo fluctuation until the age of 70 with the sharp rising to the maximum level at age 60, before decreasing when it reached the age of 70. Although the trend is decreased during the age of 50 years for men, the respiratory rate (L/min inhaled) was increased with age, resulting in an increase in the respiratory flow rate (RFR) for the elderly compared to those in their 30s. Therefore, the maximum RFR is reached at the age of 60 for both sexes, after which it starts to decrease. Therefore, older men and women in their 70s have a higher exposure to trace metals than the 30s group. In contrast, women show an even increase until it reaches the maximum level when reaching the age of 60. However, exposure readings begin to decline after reaching the age of 60. Age and gender play an important role in determining inhalation exposure to trace metals bound-$PM_{2.5}$. The tidal volume used, as a result of previous studies, showed a decrease after the age of 50 for both men and women. In addition, the respiratory rate usually does not

change according to adult age, except when it reaches the age of 60 and above. This is because, as age increases, lung function will begin to decrease (ALA, 2018).

The higher rate of respiratory flow by men is higher than women causing the total inhaled trace metals to be higher in men compared to women group. Lung size and growth, fine particle deposition, gas absorption (blood-gas barrier permeability, airway hyperresponsiveness, vascular response and inflammation are gender-specific (Sharma & Goodwin, 2010). The lungs undergo a phase of growth and maturation during the first two decades of life and reach maximum lung function around the age of 20 years in women and 25 years in men. Lung function remains stable with very minimal changes from age 20 to 35 and begins to decline thereafter. The decline in lung function tests depends on the peak lung function achieved during adulthood, the duration of the plateau phase, and the rate of lung function decline (Clougherty, 2010). Differences in exposure levels for men and women are associated with differences in physiology. According to Schaeffer (2021), the respiratory system reaches maturity in about 20–25 years, after which there is a progressive decrease in pulmonary function.

In addition, the respiratory rate (L/min inhaled) shows an increase when it reaches 60 years. This causes the rate of respiratory flow to increase for the elderly compared to those aged 30 (Quintana-Belmares et al., 2018). Thus, the maximum RFR is achieved at the age of 60 for both sexes before it begins to decline. The age 70 had higher internal exposure through inhalation to trace metals than those aged 30. According to Quintana-Belmares (2018), differences in the physiology of lung function for mens and women contribute to differences in inhalation exposure. This can be attributed to normative aging due to significant structural changes in the lungs, airways, chest walls, and respiratory muscles that ultimately lead to a progressive decrease in lung function (Molgat-Seon et al., 2018). The shrinking size and shape of their maximum expiratory flow-volume curves result in lower ventilation capacity in healthy individuals. According to LoMauro (2021), there are two pathways in respiratory physiology by gender which are hormonal and anatomical. These differences may lead to the respiratory pathway between gender and age thus, result in the different inhaled amounts of trace metals by individuals. Therefore, the most susceptible person in this study is 60s for the men and female group.

3.4 Multiple-Path Particle Dosimetry

3.4.1. Deposition fraction of $PM_{2.5}$ in the human respiratory tract

The deposition fraction covers deposition by area in human respiratory tract which includes the regional and lobar of the human respiratory tract. For the regional deposition fraction, it was found that the total deposition of $PM_{2.5}$ was highest in the head (46%), compared to pulmonary (45%) and TB (9%). In addition, it was found that the highest deposition was in the 21-year group (16%) while the lowest was in the 28-month (12%) group. The downward trend for total $PM_{2.5}$ depositions according to a certain age group is 21 years > 18 years > 8 years > 3 months & 14 years > 3 years > 28 months. However, the 18-year-old group had the highest $PM_{2.5}$ deposition in the head compared to the 3-month group. Besides, the deposition of $PM_{2.5}$ in the pulmonary had the highest value in the 8-year group and the lowest in the 28-month group. In contrast to $PM_{2.5}$ deposition in TB where the 3-month group had the highest value compared to the 21-year group. The results of the study for deposition by area for $PM_{2.5}$ in the human respiratory pathway can be seen in **Fig. 7 (a)**. In this study, $PM_{2.5}$ deposition was highest in the head area (46%) followed by pulmonary and TB parts which are consistent with the Khan et al. (2020) and Manojkumar et al. (2019) studies which found that $PM_{2.5}$ deposition is maximum in the head, Pulmonary and TB. The variety of $PM_{2.5}$ deposition fractions in each area is dependent on the geometry of the airways, the size of the PM and the deposition mechanism. An important mechanism that affects PM deposition in the airways are Brownian diffusion, inertia impaction and gravitational deposition (Darquenne 2020). On the other hand, PMs with smaller diameters and low flow velocities experience gravitational deposition. Meanwhile, the diffusion mechanism is more effective for PMs with an aerodynamic diameter of less than 0.5 µm. In addition, PM deposition in the respiratory tract is maximum in adults compared to children. The findings of this study are in line with the study in India in 2018–2019 (Khan et al., 2020). This is because the deposition will vary due to differences based on gender, pulmonary morphology, respiratory cycle, respiratory scenario and body position (Salma et al., 2002). The deposition of $PM_{2.5}$ in 3 anatomical areas of the respiratory tract was different for each age group. Deposition in high pulmonary areas is generally available for lower age groups (3 months, 8 years and 14 years) which may increase cardiovascular risk (Lv et al., 2021; Manojkumar et al., 2019). The deposition of $PM_{2.5}$ in the head in the adult group directly increases the risk of pharyngitis and rhinitis to them. In addition, high $PM_{2.5}$ deposition in TB for the infant group increased the risk of asthma for the group (Rajput et al., 2019; Sánchez-Soberón et al., 2015).

The $PM_{2.5}$ deposition of lobar in **Fig. 7 (b)** showed that the amount of deposition was highest in LL at 35% and the lowest at RM at 7%. Deposition in LL was 2 and 5 times higher than LU and RM, respectively. When compared between the top and bottom of the lobe, RU and LU showed high highway deposition compared to RL and LL. Comparison of deposition patterns between the left and right lungs was rated higher in the left lung lobe than the right. Overall, the average deposition on the LL side, followed by RL, LU, RU and finally RM parts. The downward trend for total $PM_{2.5}$ depositions according to a certain age group is 8 years > 3 months > 14 years > 3 years > 18 years > 28 months. Based on a specific age group, LU and RU were highest in the 8-year group and lowest in the 28-month group. In contrast to the RU which had the highest deposition for the 3-month versus 18-year group. Besides, RM and RL had a maximum deposition in the 3-month group compared to 28 months. Meanwhile, the left and right lobar comparisons showed higher deposition in the right lobar compared to the left lobar of 51% and 49% respectively. This can be attributed to the volume factor of lobar. The lower lobe has a larger volume and causes a higher PM deposition at the bottom, while the middle lobe has the smallest volume resulting in the least deposition in the middle of the lobar (Asgharian et al., 2006(a); Asgharian et al., 2006 (b)). In addition, according to Asgharian (2006), the diversity of deposition in lobar is also influenced by the effect of gravity among all lobars. Studies have found that primary small cell lung cancer tends to be found in the right lobar compared to the left lobar (Sahmoun et al., 2005). Moreover, as many as 35% of the 978 patients treated with carcinoma were detected in the right upper lobe (RU) (Sung et al., 2020). In addition, according to several studies, lung cancer with idiopathic pulmonary fibrosis is more often found in the lower lobes (Liu et al., 2018a; Xie et al.,

2022). In addition, the anatomical structure, flow rate and greater inertia of PM play an important role in its deposition across the right and left lungs (Islam et al., 2017). Therefore, deposition evaluation is important for predicting and showing various effects due to PM.

3.4.2. Deposition Mass of $PM_{2.5}$ in the human respiratory tract

The deposition mass includes deposition mass (D_m) and deposition mass per unit area (D_{ma}) in the tracheobronchial (TB). The geometry of the airways showed that children had a smaller size and diamater than the adult group. This leads to the difference in deposition mass (D_m) and deposition mass per unit area (D_{ma}) in the TB area. Overall, the 21-year-old group had the highest D_m compared to the 3-month group with readings of 1.53×10^{-5} µg and 1.2×10^{-6} µg respectively. The trend for D_m is 21 years > 14 years > 18 years > 28 months > 3 years > 8 years > 3 months. In contrast, D_{ma} had the highest value in the 3-year group compared to 21 years with readings, 0.0467 µg/m^2 and 0.0116 µg/m^2, respectively. The trend for D_{ma} is 3 years > 28 months > 3 months > 8 years > 14 years > 18 years > 21 years. The overall visuals of D_m and D_{ma} for the deposition of $PM_{2.5}$ on human airways can be referred to **Fig. 8**. Other than that, the D_m of $PM_{2.5}$ deposited in TB was the maximum in adults compared to children. Studies by Khan et al. (2020) also show that D_m increases with age. This is because adult exposure to $PM_{2.5}$ is higher than that of children. Next. the difference in the degree of deposition is due to the difference in PM concentration and the morphology of the parameters of the lung parameters for different age groups. Next, the mass of $PM_{2.5}$ deposited per unit area in TB (D_{ma}) was higher in the group of children than in adults. This is because the high deposition per unit area is due to a greater ratio of surface area to volume when compared to adults (Poorbahrami et al., 2021). Furthermore, children also tend to breathe through the nose (Brown, 2015) and this affects the deposition of higher particles in the child's lower respiratory tract. Children also breathe at higher ventilation minutes compared to their lung volume, resulting in higher particle deposition on the surface of children's lungs (Foos et al., 2008). Although Dm values were lower in infants than in adults, the D_{ma} value was greater in infants due to the larger surface area of TB per unit volume. Differences in deposition levels (D_m and D_{ma}) between different age groups in our study can be attributed to differences in the value of deposition fractions, pulmonary morphological parameters and airway geometry. In addition, breathing also significantly affects the deposition of inhaled PM particles. According to previous studies, it was observed that nasal breathing is associated with deposition in the lower pulmonary and tracheobronchial areas, while breathing through the oronasal causes a smaller amount of deposition (Jia et al., 2020).

3.5 Health Risk Assessment

3.5.1. AHD, ADD and Non-carcinogenic risk of $PM_{2.5}$

The infant group recorded the highest ADD values of $PM_{2.5}$ exposure for the SW monsoon, followed by the IM monsoon. The value of ADD was higher than AHD for both SW and IM monsoons. The AHD and ADD trends for $PM_{2.5}$ during SW and IM monsoons are the same: infant > toddler > children > adult which were recorded in the Supplementary 5. However, the HQ value of $PM_{2.5}$ shows no difference for all age groups during the SW and IM monsoons as shown in Table 3. The acute HQ value of the SW monsoon indicates no non-carcinogenic risk was identified (HQ < 1). Toddler and children showed similar HQ readings of 1.67×10^{-1}, followed by adult 1.66×10^{-1} and infant 1.65×10^{-1}. This suggests that the effects of short-term exposure are more likely on children while long-term exposure has the same risk effect on all age groups. Similar to IM monsoon which showed no non-carcinogenic risk for acute exposure of $PM_{2.5}$ (HQ < 1) where toddler and children showed the same HQ value of 1.10 × 10 − 1, followed by adult at 1.09×10^{-1} and infant at 1.08×10^{-1}. However, the chronic HQ value for all age groups is above the safe level (HQ > 1). The reading of the chronic HQ value of the SW monsoon was higher than the IM monsoon where it was 3.83 and 3.52, respectively. This is due to the higher concentration of $PM_{2.5}$ during the SW monsoon. A high content of pollutants can increase the risk to humans. HQ ratings in Beijing, China in 2015 showed the maximum HQ value was in winter, compared to summer, autumn and spring (Zhang et al., 2020a). This is due to the activity of the locals for space heating as well as a decrease in the height of the mixture (mixing height) during winter causing pollutants to accumulate in the atmosphere. (Li et al., 2015). A study conducted by Opara (2021) during the dry season at the waste disposal site in Metropolis Owerri, Nigeria showed that the $PM_{2.5}$ HQ of all age groups to was below the safe level (HQ < 1). The study showed that infants had the highest chronic HQ readings of $PM_{2.5}$ compared to children and adults. However, the results of both studies still show that children and infants have a higher risk than the adult groups. In addition, the similarity that can be noted in both studies is that the chronic HQ value of $PM_{2.5}$ was higher than the acute HQ value of $PM_{2.5}$ for all age groups. The exposure of air pollutants to infants, toddlers and children is particularly at high risk because the development of children's organs, such as the lungs, is not fully developed compared to adults. Children also physiologically have a higher respiratory rate. Therefore, they breathe faster than adults, exposing them to higher concentrations of pollutants (Zhou et al., 2019).

Table 3
Hazard quotient for acute and chronic exposure of $PM_{2.5}$ for different age groups.

Group	SW monsoon (n = 17)		IM monsoon (n = 13)	
	Acute	Chronic	Acute	Chronic
	HQ	HQ	HQ	HQ
Infant	1.65×10^{-1}	3.83	1.08×10^{-1}	2.52
Toddler	1.67×10^{-1}	3.83	1.10×10^{-1}	2.52
Children	1.67×10^{-1}	3.83	1.10×10^{-1}	2.52
Adult	1.66×10^{-1}	3.83	1.09×10^{-1}	2.52

In addition, Edlund et al. (2021) also stated that children have high respiratory rates which are closely associated with increased levels of pollutant inhalation. Exposure to $PM_{2.5}$ can harm newborns and children as it is at risk of causing brain developmental damage (Egondi et al., 2018). Long-term exposure to $PM_{2.5}$ is associated with a 5% increase in all causes of death, in addition to cardiovascular disease (CVD) and respiratory diseases (Wang et al. 2020). Additionally, the effect of long-term exposure to ambient $PM_{2.5}$ on cardiovascular events estimated by 12 cohort studies has found that the association of cardiovascular disease increases with long-term exposure to atmospheric $PM_{2.5}$. For example, stroke is the most studied cardiovascular disease, and its risk increases consistently with higher $PM_{2.5}$ exposure. In addition, cohort studies have been conducted for endocrine and metabolic diseases, the results prove that the long-term effects of $PM_{2.5}$ increase the risk of endocrine &; metabolic diseases. Furthermore, it is also associated with increased mortality, disease, type 2 diabetes mellitus, kidney disease and chronic obstructive pulmonary disease (Yang et al., 2022). For respiratory diseases, the Children's Health Study conducted in the United States has proven a link between long-term exposure to $PM_{2.5}$ and a decrease in children's lung growth (Gauderman et al., 2004). In addition, studies in Taiwan show that long-term exposure to $PM_{2.5}$ and $PM_{2.5–10}$ in residential areas was also associated with a decrease in lung function among seniors (aged ≥ 65 years) (Chen et al., 2019). In contrast, short-term exposure to $PM_{2.5}$ is associated with the risk of several causes of hospitalisation due to non-hypertensive congestive heart failure, obstructive pulmonary disease and chronic bronchiectasis, acute myocardial infarction and others. In addition, it was also found that short-term exposure to $PM_{2.5}$ was associated with an increase in hospitalisation rates due to septicemia, fluid and electrolyte disorders and acute kidney failure (Wei et al., 2019

3.5.1. AHD, ADD and non-carcinogenic risk of trace metal bound-$PM_{2.5}$

ADD Fe was the highest recorded for infant in the SW monsoon followed by the IM monsoon where it was 7.17×10^{-2} µg/kg-day and 3.34×10^{-2} µg/kg-day, respectively. The trend for AHD and ADD values for selected trace metals bound-$PM_{2.5}$ during the SW monsoon: Fe > Mg > Zn > Cu > Pb > Mn > Cr > Ni > V > As > Cd > Co. It is also different for the IM monsoon where the trend for AHD and ADD values: Fe > Mg > Zn > Cu > Pb > Mn > Cr > Ni > As > V > Cd > Co, where the successive changes are in As and V. The AHD and ADD values for selected trace metals bound-$PM_{2.5}$ can be referred from the Supplementary 7. The highest HQ values were recorded for chronic exposure compared to acute exposure which suggests that long-term non-carcinogenic effects are higher than short-term effects. Moreover, HI (acute and chronic) values were low for the adult compared to infant for both monsoons. Overall, the acute and chronic trends for the age groups during the SW and IM monsoons are: Infant > toddler > children > adult. Besides, all trace metals show no non-carcinogenic risk as the HQ value for each parameter does not exceed the safe levels (HQ < 1) as shown in Table 5. Furthermore, the HI value for each age group was higher during the SW monsoon compared to the IM monsoon. The chronic HQ value of Cr was more dominant during the SW monsoon recorded for infant where the readings were 7.20×10^{-2}. However, the chronic HQ value of Mn was more dominant during the IM monsoons recorded for infant where the readings were 2.32×10^{-2}. The trends of the trace metals for both acute and chronic exposure during SW and IM monsoons are as: Cr > Mn > Co > Pb > As > Cu > Cd > Zn > V > Ni > Mg > Fe and Mn > Cr > Co > Pb > As > Cu > Cd > Zn > Mg > V > Ni > Fe respectively, as shown in Table 5. The HQ values of Cr, Mn and Co are higher compared to other trace metals. The findings of a study in Chinese cities found that the HQ for Co was the highest followed by Cd and Cr (Liu et al. 2021). A study in Rayong, Thailand found the highest risk of non-carcinogenic was from Mn, followed by Ni, Cd and Cr (Kawichai et al., 2022). The South Africa study found that the highest HQ of all study populations was from Cu, followed by Ni and As (Morakinyo et al., 2021). HI studies of Cu, Cd, Ni, Pb, Zn and Cr exposure in India to child and adult groups were less than 1 and the group of children showed a higher risk compared to adults (Kumari et al., 2021). In addition, the study in Rayong, Thailand also obtained similar results from this study where HI values of less than 1 for all age groups (infants, children & adults) (Kawichai et al., 2022). In addition, a study conducted during the haze season in Chiang Rai, Thailand also found that HI values are less than 1 in the order of Cd > Cr > Mn > Ni (Niampradit et al., 2022). On the other hand, a study in Indonesia during a haze incident involving forest burning in Riau found HI values > 1 for trace metals Co, Ni, Cr and some other trace metals (Siregar et al., 2022).

The main exposure of the Cr trace metal is through the inhalation road site. Chronic exposure of Cr-based workers was found to be at greater risk of developing pneumonia, bronchitis and other respiratory problems, including occupational asthma. In addition, according to some case studies, very high exposure to Cr (VI) compounds may cause acute respiratory symptoms such as shortness of breath, coughing and wheezing. Inhalation of Cr compounds (VI) can result in respiratory problems such as hyperplasia, inflammation of the lungs and fibrosis. In the same study, 4 workers exposed to Cr had occupational asthma after a latency period of 3 months–9 years. Two workers also reportedly had rhinitis and another worker had a skin rash (Hessel et al., 2021). Prolonged exposure to Ni results in a variety of acute and chronic health effects (Begum et al., 2022). Acute effects, such as irritation of the upper respiratory tract, pneumonitis and even death have occurred in workers exposed to fine particulate matter Ni (< 0.5 µm in diameter) at very high concentrations (≥ 0.5 mg/m^3) (Phillips et al. 2010). Chronic rhinitis, sinusitis, and nasal septum perforation were all reported among nickel industry workers. For workers exposed to dust or nickel smoke, there is also an incidence of lung fibrosis and decreased lung capacity. There is evidence that workers who have been exposed to nickel dust over a long period of time have radiological signs of nickel pneumoconiosis (Begum et al., 2022). The main route of exposure to Co is through inhalation and the highest levels of Co in the workplace are found in hard metal manufacturing, Co salt production, and metallurgical related industries (Scarselli et al., 2020b). The liver is the first site for cobalt accumulation in the body, and the accumulation of Co in the liver causes glutathione levels to decrease. In a study of Co particle exposure to 582 patients, 56% of heavy metal workers had mild to moderate elevated liver enzymes, while others had hepatomegaly (Cheung et al., 2016). In addition, a study in Baoding, China showed a high HQ value for children of HI = 4.15 (HI > 1) which is significantly higher than the adult HQ of HQ = 6.72×10^{-1} (HQ < 1). This is because studies conducted during the heating period led to an increase in the concentration of $PM_{2.5}$, thus increasing the health risks of the surrounding population, especially children. The highest concentration was recorded by Pb with HQ = 3.68 (HQ > 1). In addition, a study conducted in Western Pretoria, South Africa (Morakinyo et al., 2021) showed that toddlers had the highest ∑ HI compared to infants, followed by children and adults.

In contrast to studies in Rayong, Thailand (Kawichai et al. 2022) and Mecca, Saudi Arabia (Abdulaziz et al., 2021) recorded the same HQ for all age groups involved. However, Rayong has ∑ HI below the safe level (∑ HI < 1). In contrast to Mecca which has ∑ HI above the safe level (∑ HI > 1). The main cause of the high health risk in Mecca is the increase in vehicles for transportation purposes for the group performing Hajj or Umrah, as well as dust storms, construction

projects and resuspension dust from cars (Habeebullah et al., 2016; Nayebare et al., 2018). Farahat (2021), on the other hand, does not rule out the possibility of an increase in $PM_{2.5}$ due to industrial long-distance transportation from the surrounding cities. In contrast to the study in Dalian, China (Zhao et al., 2021) showed a higher HQ than the results of this study. Moreover, the adult HQ (HQ = 2.90) is 32% higher than the children's HQ (HQ = 2.20). ∑ HI above safe levels (∑ HI > 1) of both age groups indicates that there was a non-carcinogenic risk when exposed during that period. The study in Dalian was carried out during the spring celebrations where there was a reduction in air pollutant emissions due to factory closures and reduced traffic. However, the high HQ V for this study is due to Dalian's role as a developed shipbuilding industry and a large port company in northeast China (Shi, 2015). Heavy fuel oil is the main fuel used by cargo ships docked at the port, being the main cause for the release of metal V bound $PM_{2.5}$ (Zhao et al., 2021). The presence of metal V as a marker of oil combustion is also supported by Lin (2015). Therefore, the high health effects on V are expected to stem from the burning of cargo ship oil in Dalian, China. In addition, the high HQ Mn in this study are in line with several other studies in Kuala Lumpur, Malaysia (Nadzir et al., 2021), Rayong, Thailand (Kawichai et al., 2022) and Obio-Akpo, Nigeria (Afolabi et al., 2022). There are many studies linking neurotoxic disorders and cognitive deficits to exposure to Mn (Williams et al., 2012; Expósito et al., 2021). According to Yao et al. (2021), Mn inhalation even in low concentrations can cause adverse health effects, especially in the group of children. This is also supported by Rodrigues et al. (2018) who states that high Mn exposure to children is closely related to deficits in certain neuropsychological domains such as cognition. In addition, Pb had the highest HQ for studies in West-Pretoria, South Africa (Morakinyo et al., 2021) and Baoding, China (Liu et al. 2018b). High HQ Pb can be associated with urban areas that have high traffic flow (Lin, 2015; Lough, 2005). However, Pb is also associated with emissions from industrial areas such as in Western Pretoria, South Africa (Morakinyo et al. 2020) and Baoding, China (Liu et al. 2018b). Exposure to Pb is not only associated with the brain, but it can also even affect the central nervous system (Gundacker et al. 2021) so that it can cause coma, seizures and death (WHO, 2022). According to Gundacker et al. (2021) also, adult group exposure to Pb resulted in cardiovascular and nephrotoxin effects. In addition, As has the highest HQ for the study of Apia City, Samoa and Mecca, Saudi Arabia. This coincides with several past studies stating that As's sources of emissions stem from industries related to metals and the burning of charcoal and fossil fuels (Hsu et al., 2016; Sulong et al., 2017; Zhang et al., 2020b; Zhao et al., 2021). Similar to Pb, As is also a pollutant that has high toxicity and is dangerous when exposed. According to Zhang et al. (2020b), cardiovascular disease is one of the diseases that can be suffered when exposed to arsenic through inhalation. Therefore, more special attention should be paid to the risk of non-carcinogenic due to these trace metals that are capable of posing a risk to human health.

Table 5
Hazard Quotient (HQ) and Hazard Index for acute and chronic exposure of trace metal bound-$PM_{2.5}$ for different age groups.

Group	Element	SW monsoon (n = 17)		IM monsoon (n = 13)	
		Acute	**Chronic**	**Acute**	**Chronic**
		HQ	**HQ**	**HQ**	**HQ**
Infant	Mg	1.81×10^{-6}	4.21×10^{-5}	1.63×10^{-6}	3.77×10^{-5}
	Fe	3.74×10^{-7}	8.69×10^{-6}	1.74×10^{-7}	4.04×10^{-6}
	Cr	3.10×10^{-3}	7.20×10^{-2}	7.34×10^{-4}	1.70×10^{-2}
	Ni	2.00×10^{-6}	4.65×10^{-5}	7.06×10^{-7}	1.64×10^{-5}
	Co	5.89×10^{-4}	1.37×10^{-2}	4.12×10^{-4}	9.57×10^{-3}
	As	5.84×10^{-5}	1.36×10^{-3}	4.86×10^{-5}	1.13×10^{-3}
	Cd	9.18×10^{-6}	2.13×10^{-4}	6.83×10^{-6}	1.59×10^{-4}
	Pb	8.31×10^{-5}	1.93×10^{-3}	1.03×10^{-4}	2.38×10^{-3}
	Mn	2.25×10^{-3}	5.23×10^{-2}	1.00×10^{-3}	2.32×10^{-2}
	Zn	4.06×10^{-6}	9.44×10^{-5}	5.11×10^{-6}	1.19×10^{-4}
	Cu	2.93×10^{-5}	6.80×10^{-4}	1.77×10^{-5}	4.11×10^{-4}
	V	3.36×10^{-6}	7.80×10^{-5}	1.49×10^{-6}	3.46×10^{-5}
	∑ HI	$\mathbf{6.13 \times 10^{-3}}$	$\mathbf{1.42 \times 10^{-1}}$	$\mathbf{2.33 \times 10^{-3}}$	$\mathbf{5.41 \times 10^{-2}}$
Toddler	Mg	1.67×10^{-6}	3.83×10^{-5}	1.50×10^{-6}	3.43×10^{-5}
	Fe	3.45×10^{-7}	7.91×10^{-6}	1.60×10^{-7}	3.68×10^{-6}
	Cr	2.86×10^{-3}	6.55×10^{-2}	6.76×10^{-4}	1.55×10^{-2}
	Ni	1.84×10^{-6}	4.23×10^{-5}	6.50×10^{-7}	1.49×10^{-5}
	Co	5.42×10^{-4}	1.24×10^{-2}	3.80×10^{-4}	8.71×10^{-3}
	As	5.38×10^{-5}	1.23×10^{-3}	4.48×10^{-5}	1.03×10^{-3}
	Cd	8.45×10^{-6}	1.94×10^{-4}	6.29×10^{-6}	1.44×10^{-4}
	Pb	7.65×10^{-5}	1.75×10^{-3}	9.46×10^{-5}	2.17×10^{-3}
	Mn	2.07×10^{-3}	4.75×10^{-2}	9.22×10^{-4}	2.11×10^{-2}
	Zn	3.74×10^{-6}	8.59×10^{-5}	4.71×10^{-6}	1.08×10^{-4}
	Cu	2.70×10^{-5}	6.19×10^{-4}	1.63×10^{-5}	3.74×10^{-4}
	V	3.10×10^{-6}	7.10×10^{-5}	1.37×10^{-6}	3.15×10^{-5}
	∑ HI	$\mathbf{5.65 \times 10^{-3}}$	$\mathbf{1.30 \times 10^{-1}}$	$\mathbf{2.15 \times 10^{-3}}$	$\mathbf{4.92 \times 10^{-2}}$
Children	Mg	1.05×10^{-6}	2.40×10^{-5}	9.39×10^{-7}	2.15×10^{-5}
	Fe	2.16×10^{-7}	4.95×10^{-6}	1.01×10^{-7}	2.30×10^{-6}
	Cr	1.79×10^{-3}	4.10×10^{-2}	4.24×10^{-4}	9.71×10^{-3}
	Ni	1.16×10^{-6}	2.65×10^{-5}	4.08×10^{-7}	9.34×10^{-6}
	Co	3.40×10^{-4}	7.79×10^{-3}	2.38×10^{-4}	5.45×10^{-3}
	As	3.37×10^{-5}	7.73×10^{-4}	2.81×10^{-5}	6.43×10^{-4}
	Cd	5.30×10^{-6}	1.21×10^{-4}	3.94×10^{-6}	9.03×10^{-5}

	Pb	4.80×10^{-5}	1.10×10^{-3}	5.93×10^{-5}	1.36×10^{-3}
	Mn	1.30×10^{-3}	2.98×10^{-2}	5.78×10^{-4}	1.32×10^{-2}
	Zn	2.35×10^{-6}	5.38×10^{-5}	2.95×10^{-6}	6.76×10^{-5}
	Cu	1.69×10^{-5}	3.88×10^{-4}	1.02×10^{-5}	2.34×10^{-4}
	V	1.94×10^{-6}	4.45×10^{-5}	8.62×10^{-7}	1.97×10^{-5}
	∑ HI	**3.54×10^{-3}**	**8.11×10^{-2}**	**1.35×10^{-3}**	**3.09×10^{-2}**
Adult	Mg	6.68×10^{-7}	1.54×10^{-5}	5.99×10^{-7}	1.38×10^{-5}
	Fe	1.38×10^{-7}	3.18×10^{-6}	6.42×10^{-8}	1.48×10^{-6}
	Cr	1.14×10^{-3}	2.64×10^{-2}	2.71×10^{-4}	6.24×10^{-3}
	Ni	7.38×10^{-7}	1.70×10^{-5}	2.60×10^{-7}	6.00×10^{-6}
	Co	2.17×10^{-4}	5.01×10^{-3}	1.52×10^{-4}	3.50×10^{-3}
	As	2.15×10^{-5}	4.97×10^{-4}	1.79×10^{-5}	4.13×10^{-4}
	Cd	3.38×10^{-6}	7.80×10^{-5}	2.52×10^{-6}	5.80×10^{-5}
	Pb	3.06×10^{-5}	7.06×10^{-4}	3.79×10^{-5}	8.73×10^{-4}
	Mn	8.30×10^{-4}	1.91×10^{-2}	3.69×10^{-4}	8.50×10^{-3}
	Zn	1.50×10^{-6}	3.45×10^{-5}	1.88×10^{-6}	4.34×10^{-5}
	Cu	1.08×10^{-5}	2.49×10^{-4}	6.52×10^{-6}	1.50×10^{-4}
	V	1.24×10^{-6}	2.86×10^{-5}	5.50×10^{-7}	1.27×10^{-5}
	∑ HI	**2.26×10^{-3}**	**5.21×10^{-2}**	**8.60×10^{-4}**	**1.98×10^{-2}**

3.5.2 LADD and carcinogenic risk of trace metal bound-$PM_{2.5}$

The adult group recorded the highest LADD Pb values in the IM monsoon followed by the SW monsoon where the readings were 1.20×10^{-4} µg/kg-day and 9.70×10^{-5} µg/kg-day, respectively. On the other hand, the lowest LADD value was seen in the infant group of each trace metal on both monsoons. The trend for LADD inhalation exposure for SW and IM monsoons is Pb > Ni > Cr > As > Cd > Co and Pb > Cr > Ni > As > Cd > Co respectively. The LADD values for selected trace metals during SW and IM monsoons can be referred to Supplementary 8. This study indicates that all assessed trace metals are at the permissible level. The ∑ ELCR trend for trace metals for each age group is adult > children > toddler > infant. In addition, Cr for the adult group had the highest CR values in the SW monsoon compared to the IM monsoon with readings of 9.21×10^{-7} and 7.67×10^{-7}, respectively. Besides, the trends for CR in this study for SW and IM monsoons as shown as: Cr > As > Co > Cd > Ni > Pb and Cr > As > Co > Cd > Pb > Ni respectively. Table 6 shows CR values for selected trace metals in this study during SW monsoon and IM monsoon. The results showed that trace metals bound $PM_{2.5}$ had high CR values in adults, followed by children, toddler and infants as shown in Table 6. Therefore, the adult age group faces a potential risk of cancer (ELCR 10^{-5} to 10^{-6}), while the other age group is not estimated to have any risk of cancer (ELCR < 10^{-6}). This is in line with several past studies in Kuala Lumpur, Malaysia, and Rayong, Thailand. Next, a study in West Pretoria, South Africa (Morakinyo et al., 2021) showed a definite risk class for adults (ELCR > 10^{-4}), while the possible risk for infants and toddler (ELCR 10^{-4} to 10^{-5}) against exposure in winter. In contrast to studies in Din Daeng, Thailand (Ahmad et al., 2022) and Mecca, Saudi Arabia (Abdulaziz et al., 2022) where the adult group showed a class of possible risks except for children in Din Daeng, Thailand who were at potential risk. However, each of the studies was different from the study in Makassar, Indonesia (Rauf et al., 2021). The study showed that ELCR for children is slightly higher than for adults. Nevertheless, both are in negligible risk classes (ELCR < 10^{-6}). In addition, Pb contributed to the highest CR for the study in Makassar, Indonesia. Although the Pb readings of the Makassar study indicate negligible risks, an increase in Pb in the future may lead to toxicity in the body when exposed over a long period of time. According to Cui et al. (2023), a recent study has found that the blood and human nervous system have a high sensitivity to UN exposure. In addition, the results of this study show that As has the highest ELCR compared to other metals which is similar to other studies as shown in Table 5.6. ELCR As was higher in studies in Kuala Lumpur, Malaysia, Rayong, Thailand, Din Daeng, Thailand, and Mecca, Saudi Arabia. However, CR values in Kuala Lumpur and Rayong are in the negligible risk class (ELCR < 10^{-6}), while Din Daeng and Mecca are in the risk potential class (ELCR 10^{-5} to 10^{-6}). In contrast to CR As for studies in West Pretoria, the risk is definite (CR > 10^{-4}), which requires more specific attention. According to Zhang et al. (2020b), exposure to As over a long period of time either through inhalation or ingestion will lead to diseases such as skin cancer and respiratory cancer. Therefore, more detailed attention and action should be taken to overcome this problem while reducing exposure.

Table 6
Cancer Risk (CR) and Excess Lifetime Cancer Risk (ELCR) for exposure of trace metal bound-$PM_{2.5}$ for different age groups.

Group	Element	SW monsoon (n = 17)	IM monsoon (n = 13)
Infant	Cr	1.51×10^{-7}	1.03×10^{-7}
	Ni	4.03×10^{-9}	1.42×10^{-9}
	Co	1.23×10^{-8}	8.63×10^{-9}
	As	3.07×10^{-8}	2.56×10^{-8}
	Cd	6.73×10^{-9}	5.01×10^{-9}
	Pb	1.43×10^{-9}	1.77×10^{-9}
	∑ ELCR	**2.05×10^{-7}**	**1.45×10^{-7}**
	Risk level	**Negligible**	**Negligible**
Toddler	Cr	2.60×10^{-6}	6.16×10^{-7}
	Ni	2.42×10^{-8}	8.53×10^{-9}
	Co	7.40×10^{-8}	5.18×10^{-8}
	As	1.84×10^{-7}	1.53×10^{-7}
	Cd	4.04×10^{-8}	3.00×10^{-8}
	Pb	8.58×10^{-9}	1.06×10^{-8}
	∑ ELCR	**2.93×10^{-6}**	**8.70×10^{-7}**
	Risk level	**Possible**	**Negligible**
Children	Cr	5.20×10^{-6}	1.23×10^{-6}
	Ni	4.84×10^{-8}	1.71×10^{-8}
	Co	1.48×10^{-7}	1.04×10^{-7}
	As	3.68×10^{-7}	3.07×10^{-7}
	Cd	8.08×10^{-8}	6.01×10^{-8}
	Pb	1.72×10^{-8}	2.12×10^{-8}
	∑ ELCR	**5.87×10^{-6}**	**1.74×10^{-6}**
	Risk level	**Possible**	**Possible**
Adult	Cr	1.30×10^{-5}	3.08×10^{-6}
	Ni	1.21×10^{-7}	4.26×10^{-8}
	Co	3.70×10^{-7}	2.59×10^{-7}
	As	9.21×10^{-7}	7.67×10^{-7}
	Cd	2.02×10^{-7}	1.50×10^{-7}
	Pb	4.29×10^{-8}	5.30×10^{-8}
	∑ ELCR	**1.36×10^{-4}**	**4.35×10^{-6}**
	Risk level	**Definite**	**Possible**

The findings of study in West Pretoria also suggest that exposure to $PM_{2.5}$ can cause a definite carcinogenic risk to all ages. The CR value for the adult population was the highest, while the infant group had the lowest value. In addition, the results of this study are also in line with the study of Sulong et al. (2017) in Kuala Lumpur and a study in Rayong, Thailand (Kawichai et al., 2022) that showed adults have the highest risk compared to infants, children and adolescents. This is because the exposure period (ED) value used for adults in this study was 30 years, different from the ED value for infants and children of only 1 to 12 years. It shows that the exposure of carcinogenic substances to adults is longer, thus causing these carcinogenic substances to accumulate in

their bodies and have cancer effects over a long period of time (Wang et al. 2018). In contrast to the study in Hyedrabad, India, although all ∑ ELCR values in those countries belong to the definite risk, the infant age group has the highest ΣELCR (Kumari et al. 2021). Next, the CR values of carcino-genic metals Cr, Ni and Co are in the safe range (10^{-4} to 10^{-6}) for all age categories and for both seasons except for Cr metals for the adult group with a value of 1.30×10^{-5}. This means that 1 to two people in the 100,000 vulnerable population are potentially cancerous. In addition, the CR of Cr trace metals is seen as the highest in each age category and during both monsoons compared to Ni and Co. This indicates that exposure to Cr is one of the things to note at the study site. Next, the value of ΣELCR is the sum of all CR values of all carcinogenic trace metals for each age group as shown in Table 5.5. Through the study of cancer risk, it was found that the risk of cancer in the population in the study area was relatively low. Despite this, there are still risks that the population around the Faculty of Health Sciences in Kuala Lumpur has to face against other pollutants that could be a threat to human health. The calculation of the amount of ΣELCR in this study is prone to some uncertainty. Among the uncertainties are input data as a result of models and estimates, the impossibility of linking between the effects of pollutants observed with undetected pollutants, there are specific factors that are not evaluated and only emphasize general factors. In addition, the actual amount of pollutants in an area is uncertain. Finally, the exposure scenario is to use an approximate average value, where it is possible that the average weight of the local population is more than that used.

5. Conclusion

In conclusion, the pollution levels of $PM_{2.5}$ at Jalan Raja Muda Abdul Aziz, Kuala Lumpur are at unsafe levels as there are some readings that exceed the guidelines of the Malaysian Ambient Air Quality Guidelines (MAAQS), Ambient Air Quality Standards (NAAQS) from the United States Environmental Protection Agency (USEPA) and WHO guidelines. In addition, it was found that the concentration of $PM_{2.5}$ is higher during the SW Monsoon compared to the IM monsoon because the dry season provides suitable environmental conditions for PM accumulation such as low rainfall distribution and stable and static atmospheric conditions during the SW monsoon. $PM_{2.5}$, which consists of various harmful components, especially trace metals can cause health risks to humans. Next, lung exposure assessments showed that the 60-year-old group had the highest flow rate of inhalation which led to higher inhalation of trace metals. The elder group has a higher exposure to $PM_{2.5}$ because the function of their organs has decreased with age. Even worse when there is no safe threshold for inhalation exposure, trace metals bound-$PM_{2.5}$ against the development of this respiration-related disease.

In addition, regional deposition fraction of $PM_{2.5}$ in the human respiratory tract was found head area had a highest deposited $PM_{2.5}$, followed by pulmonary and tracheobronchial. The lobar deposition fraction of $PM_{2.5}$ had the highest value in the lower left lobar (LL) while the lowest deposition is in the center-right lobar (RM). Next, the sedimentation mass (D_m) was highest for the 21-year group compared to 28 months. However, the mass of deposition per unit area (D_{ma}) indicates that the 3-year group has a higher value than 21 years. Based on the health risk assessment (HRA) analysis, chronic exposure to $PM_{2.5}$ is estimated to provide non-carcinogenic risk (HQ > 1) and carcinogenic risk (CR > 10^{-4}) to all vulnerable age groups. However, exposure to trace metals bound-$PM_{2.5}$ indicates that only the adult age group is at probable and possible risk of developing cancer (∑ ELCR 10^{-4} to 10^{-6}). Metal Cr is a major concern due to its higher risk which can cause severe health effects. This problem is an issue that needs to be emphasized as continuous exposure can have adverse health effects such as cancer that can lead to death. Overall, $PM_{2.5}$ emissions need to be well controlled to avoid any collision of established air quality standards. This control needs to be emphasized as it can have a detrimental effect on the health of the vulnerable group especially children, the elderly, pregnant women and groups with pre-existing diseases. $PM_{2.5}$ can also affect both in the short and long term.

Declarations

Declaration of Competing Interest

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

Author Contribution

Nor Faizah Afandi: Methodology, Sampling, Analysis, Data curation, Writing. Nor Syazwani Azri Khairel Sabri: Methodology, Sampling, Analysis, Data curation, Writing. Sharifah Mazrah Sayed Mohamad Zain: Conceptualization, Methodology, Sampling, Analysis, Data curation. Zurahanim Fasha Annual: Methodology, Analysis. Nuralia Syahida Mohd Sharif: Analysis, Writing. Anas Ahmad Jamhari: Analysis, Writing. Muhammad Ikram A Wahab: Project administration, Conceptualization, Data curation, Review and Editing, Supervision.

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Figures

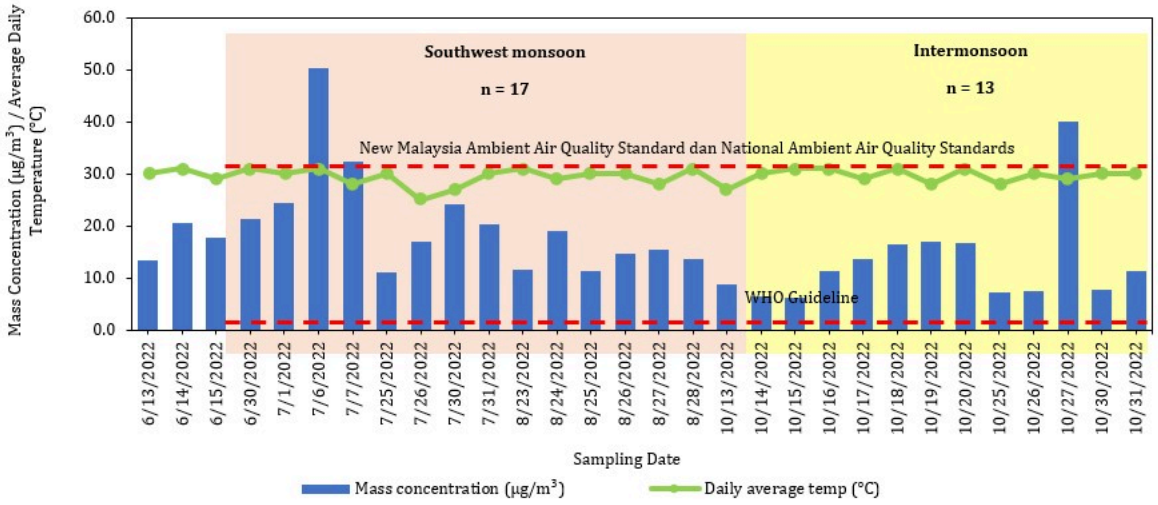


Figure 1

Fig. 2. Daily concentration of $PM_{2.5}$ during the sampling period.

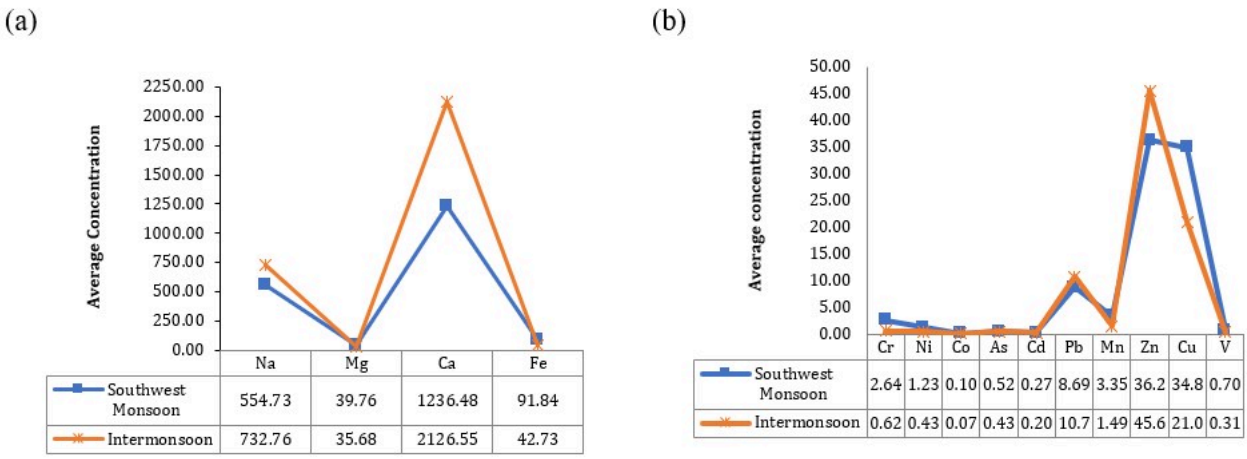


Figure 2

Fig. 5 Concentration of trace metals bound $PM_{2.5}$: (a) Major elements; (b) Minor elements.

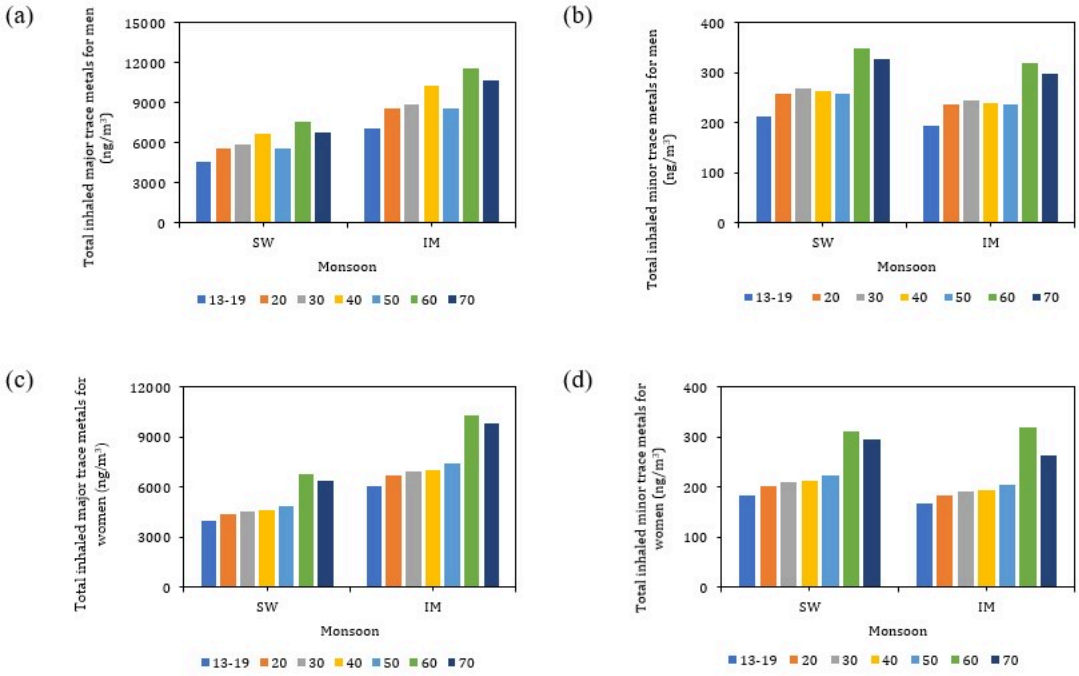


Figure 3

Fig. 6. Total inhaled of major and minor trace metals exposure for men and women

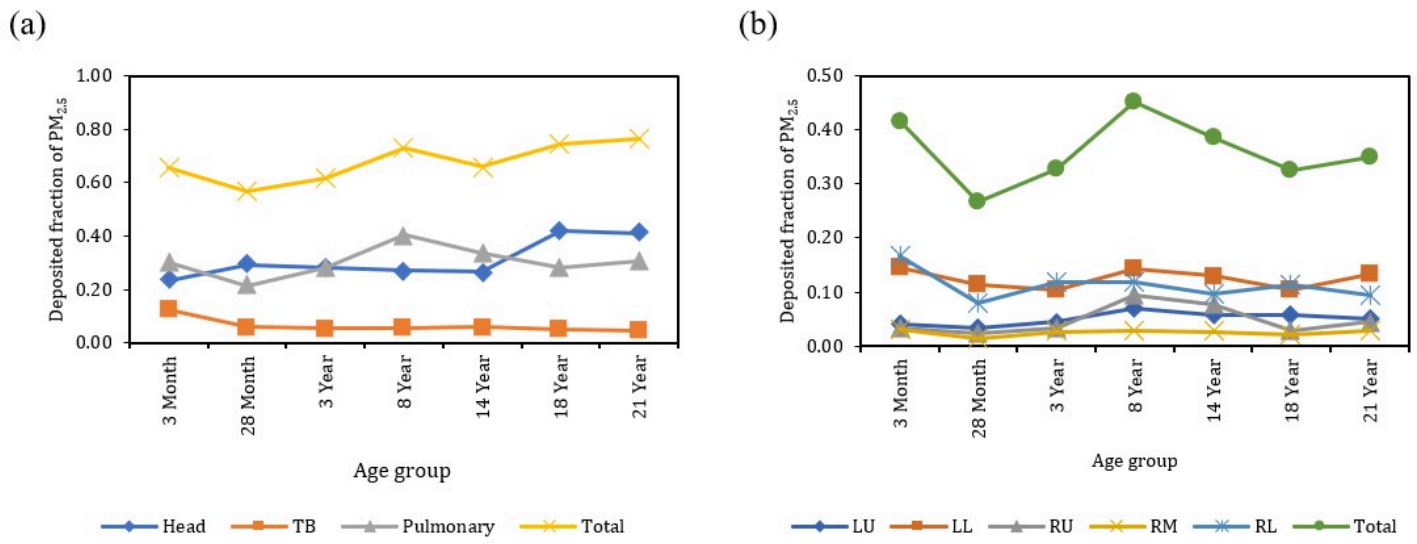


Figure 4

Fig. 7. Deposition fraction of $PM_{2.5}$ in human respiratory tract: (a) Regional deposition fraction; (b) Lobar deposition fraction.

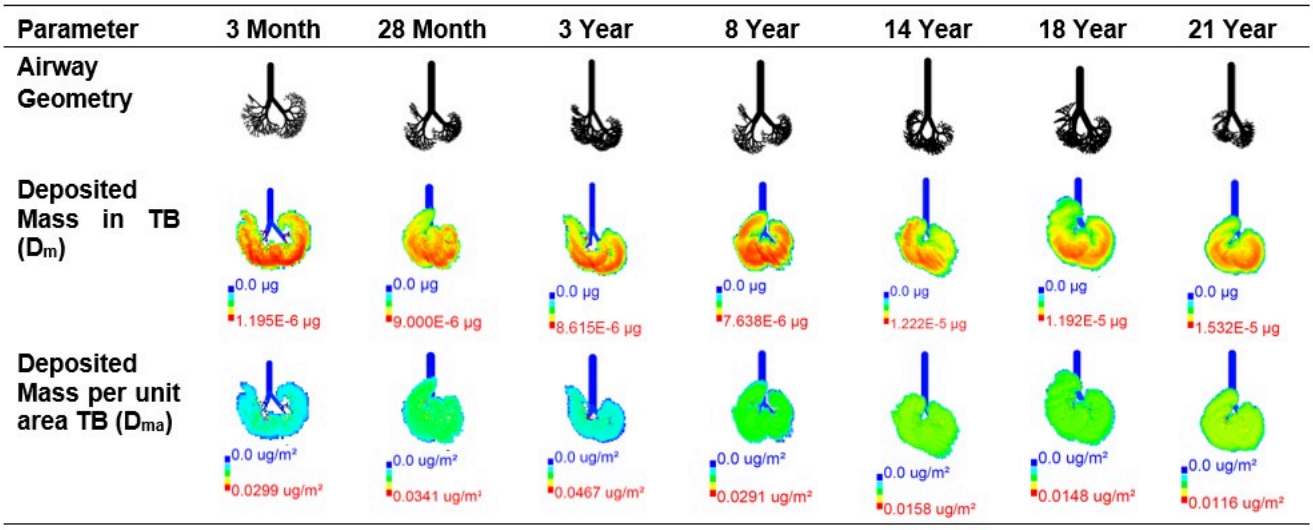


Figure 5

Fig. 8. Deposited mass (D_m) and deposited mass per unit area (D_{ma}) of $PM_{2.5}$ on human airways.

Supplementary Files

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