

PM_{2.5}-BOUND PAHs IN THE INDOOR AND OUTDOOR AIR OF NURSERY SCHOOLS IN HA NOI, VIET NAM AND HEALTH IMPLICATION

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Abstract. Thirty two daily indoor and outdoor PM_{2.5} samples at sixteen nursery schools in Ha Noi were taken from March to April 2018. 15 individual PAHs were analyzed by GC/MS. Mutagenic potential (MP) and carcinogenic potential (CP) were used to evaluate carcinogenic and mutagenic risk contribution. The average PM_{2.5} concentrations were 38.7 µg/m³ for indoor air and 93.3 µg/m³ for outdoor one, which exceeded WHO Air quality guideline (2004) for ambient air. The mean concentrations of indoor and outdoor Σ15PAHs were 267.1 ng/m³ and 843.4 ng/m³, respectively. There was a good correlation between indoors and outdoors for both PM_{2.5} and PAHs. Indoor MEQ_{Σ7PAHs} was 41.4 ng/m³, whilst outdoor MEQ_{Σ7PAHs} was 137.3 ng/m³. The mean level of indoor TEQ_{Σ15PAH} was 64.6 ng/m³, whereas the outdoor TEQ_{Σ15PAH} was 208.8 ng/m³. Dibenz(a,h)anthracene (DahA) and benzo(a)pyrene (BaP) were the most contributors among indoor and outdoor PAHs to the carcinogenicity and mutagenicity.

Keywords: indoor air quality, PAH, PM_{2.5}, health risk assessment, nursery school, Ha Noi.

Classification numbers: 3.6.2, 3.4.5.

1. INTRODUCTION

The indoor air pollution has been a major threat of global public health, because people spend much more time in enclosed spaces than in outdoor ones, especially in urban areas. It is reported that roughly 80 - 90 % of our time is spent in indoor spaces [1]. Air quality in Ha Noi has gotten worse with the high concentration of fine particles. It was referred that there were

more than 60000 deaths from heart disease, stroke, lung cancer, chronic obstructive pulmonary diseases and pneumonia in Viet Nam in 2016 due to air pollution (WHO, 2018). $PM_{2.5}$ may penetrate deep into the lungs and cardiovascular system. Besides, the adverse health effects of $PM_{2.5}$ strongly depend on contaminants associated with particles, such as polycyclic aromatic hydrocarbons (PAHs), trace elements [2]. It has been demonstrated that approximate 1.6 % of lung cancer cases in China may attribute to the inhalation of PAHs from polluted air [1]. Some PAH derivatives are carcinogenic and mutagenic substances. Benzo[a]pyrene (BaP) and PAH mixtures are classified group 1 as carcinogenic to humans (IARC 2016). PAHs attached to fine particulate matters are confirmed to harm the public health, especially for sensitive groups such as the kids. Predominant PAHs concentration found in $PM_{2.5}$ was concluded to be penetrated deep into the alveolar regions of the lungs [3]. Potential risk and characterization of a complex mixture of PAHs can be identified using carcinogenic equivalency factors (TEF) and mutagenic equivalent factor (MEF). Toxicological studies on individual PAHs relative to BaP has been previously considered [1, 4]. Currently, the number of studies on carcinogenic and mutagenic properties of PAHs has been increasing [4, 5]. Notwithstanding the documentation of the significant effects of PAHs and fine particles outlined above, there is a few researches to date focusing on either the chemical composition or health risk assessment of $PM_{2.5}$ -bounded PAHs in nursery schools in Ha Noi, Viet Nam. The main objectives of this study are: (1) to determine the concentrations of PAHs associated with $PM_{2.5}$ for indoor and outdoor air samples at sixteen nursey schools within Ha Noi metropolis; (2) to examine the indoor to outdoor air ratio; and (3) to estimate the toxicity of PAHs using toxic equivalency factors.

2. MATERIALS AND METHODOLOGY

2.1. Sampling sites

Outdoor and indoor samples of $PM_{2.5}$ were taken synchronously at sixteen nursery schools within Ha Noi metropolis. Total 32 $PM_{2.5}$ samples were collected daily during teaching hours with the windows/ doors closed due to air conditioning from March to April 2018. The $PM_{2.5}$ sampling inlets were placed around 0.8 m above the floor (to simulate the kid's breathing zone) in the middle of the classroom and minimally 1 m from the walls and/or windows. Simultaneous outdoor sampling was conducted at schoolyard and the inlet was at least 1m away from any obstacle and 1.5 m above ground. $PM_{2.5}$ samples were collected by a cyclone with a filter holder (URG-2000-30EH, University Research Glassware Co., Chapel Hill, NC, USA) and a MiniVol sampler (Arhmetrics, USA), respectively. All samplers were calibrated before sampling to obtain the recommended flowrate (16.7 min/L for indoors and 5.0 L/min for outdoors). $PM_{2.5}$ samples were collected on quartz filters (QMA, 47 mm in diameters), which were prebaked at 900 °C for four hours to remove all possible contaminants [5]. Each sampled quartz filter was placed in a Petri dish, wrapped up in the aluminum foil and stored in a freezer at -4 °C till the PAH analysis.

2.2. Analytical method

2.2.1. Mass analysis

The filters were weighted on an electronic microbalance with an accuracy of 10^{-6} g (AX26 DeltaRange microbalance, Mettler Toledo company, Switzerland). Prior to be weighted, the filters were equilibrated in a balance room for at least 48 hours. Relative humidity of the

balance room was kept at the range of 30 - 40 % and temperature was remained from 20 °C to 25 °C. The electrostatic charge of the filters was eliminated by a constant ionizing air blower (Model YIB01-ODR, Germany) before weighting.

2.2.2. PAHs analysis

Quartz filters were cut into small pieces and put in 15 mL centrifugal tube to analyze PAH in particle phase. Ten µL of Internal Standards Mix (20 ppm, Code: DRE-YA08273300TO) were spiked in all samples and let to reach equilibrium for 20 minutes. Five ml mixture (n-hexane and acetone, 1:1) were added in each facol and ultrasonicated for five minutes by ultrasonic horn. The extraction was repeated 3 times and then concentrated to less than 1 ml by carefully blowing a gentle stream of dry nitrogen. The extracts were then purified by Bond Elute-Si column. The colume was activated and washed by methanol, mixture (n-hexane & acetone), and n-hexane to remove any lipid. The extracts were eluted with 15 mL of a mixture (n-hexane and dichloromethane (1:1)) and then turbo-vaporized to 1.0 mL under a gentle stream of nitrogen prior to GC/MS analysis. The analysis was performed using gas chromatography (Agilent Technologies 6890 N) coupled with mass spectrometry (Agilent Technologies 5973) in the selective ion-monitoring (SIM) mode, transfer lines with MS interface column nut, P/N 05988-20066, 10/pk. A fused silica capillary column (DB-5 30 m × 0.25 mm × 0.25 µm) was used for separation.

Fifteen PAH species are naphthalene (Nap), acenaphthylene (Acy), acenaphthene (Ace), fluoranthene (Flu), phenanthrene (Phe), anthracene (Ant), fluoranthene (Flt), pyrene (Pyr), benzo (a) anthracene (BaA), chrysene (Chr), benzo (b) flouranthene (BbF), benzo (a)Pyrene (BaP) and dibenzo (a,h) anthrancene (DBA), indeno (1,2,3-cd) pyrene (IcP) and benzo (ghi) perylene (BghiP). They are classified into low molecular weight (LMW, 2-3 rings), and high molecular weight (HMW, 4-6 rings).

2.3. Quality assurance and quality control

The methodology for conditioning, weighing, storing, and transporting samples as well as blank samples complied with the QA/QC procedures defined in the reference method for gravimetric measurements. US EPA Method 610 (US. EPA, 1984) was followed to analyze PAHs in particle phase. PAHs recovery was evaluated by spiking the internal standard mixture containing 15 PAHs. The recoveries of 15PAHs ranged from 80 to 130 %, except for flourence with 60 %. Limit of detection (LOD) were determined based on the Signal/Noise. LOD of Nap, Acy, Ace, Flu, Ant, Phe, Flt, Pyr, BaA, Chr, BbF, BaP, DBA, IcP and BghiP were 0.07 ng/m³, 0.1 ng/m³, 0.05 ng/m³, 0.09 ng/m³, 0.04 ng/m³, 0.09 ng/m³, 0.08 ng/m³, 0.11 ng/m³, 0.21 ng/m³, 0.93 ng/m³, 1.94 ng/m³, 0.95 ng/m³, 4.28 ng/m³ and 0.43 ng/ m³, respectively. SPSS was used for statistical analysis.

2.4. Health risk assessment

The toxic equivalent factor (TEF) and mutagenic equivalent factor (MEF) relating the carcinogenic and mutagenic potency of individual PAH_i to BaP have been used to evaluate the toxic and carcinogenic potency of PAHs [1,3]. Calculations of the carcinogenic equivalent (TEQ) and mutagenic equivalent (MEQ) for the individual PAHs were presented in equations (1) and (2).

$$MEQ = \sum_{i=1}^N (PAH)_i \times MEF_i \quad (1)$$

$$TEQ = \sum_{i=1}^N (PAH)_i \times TEF_i \quad (2)$$

where, MEF_i and TEF_i are the mutagenic and toxic equivalent factor for individual PAH. The contribution of total carcinogenicity and mutagenicity of PAHs was calculated by carcinogenic potential and mutagenic potential. Total 15 PAHs or 8 PAHs were employed to calculate carcinogenic potential (CP) or mutagenic potential (MP) using the following equations (3-4) [6].

$$CP_i = \frac{\frac{(PAH)_i}{(BaP)} \times TEF_i}{\sum_{i=1}^N \frac{PAH_i}{BaP} \times TEF_i} \times 100\% \quad (3)$$

$$MP_i = \frac{\frac{(PAH)_i}{(BaP)} \times MEF_i}{\sum_{i=1}^N \frac{PAH_i}{BaP} \times MEF_i} \times 100\% \quad (4)$$

3. RESULTS AND DISCUSSIONS

3.1 Occurrence of $PM_{2.5}$

The mass concentrations of indoor and outdoor $PM_{2.5}$ in sixteen nursery schools are presented in Fig. 1. The mean concentrations of indoor and outdoor $PM_{2.5}$ were 38.7 and $93.3 \mu\text{g}/\text{m}^3$ respectively, which exceed WHO recommended 24-h limit for $PM_{2.5}$ ($25 \mu\text{g}/\text{m}^3$) approximately 1.6 and 3.7 times. The indoor $PM_{2.5}$ concentrations exceed targeted value of WHO in 50 % of observed nursery schools, whilst an excess of outdoor $PM_{2.5}$ concentrations can be seen in 70 % of measured nursery schools.

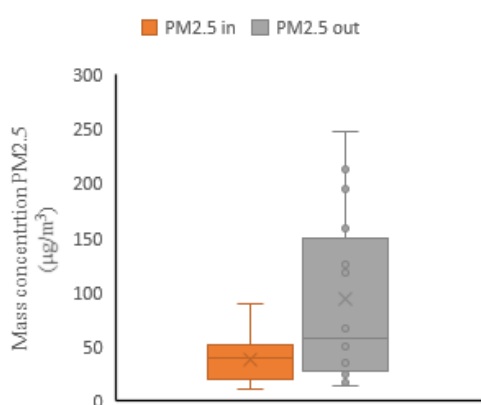


Figure 1. Mass concentrations of $PM_{2.5}$ in Ha Noi.

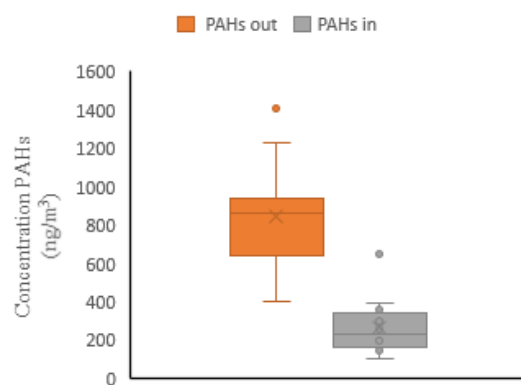


Figure 2. Concentrations of PAH in Ha Noi.

In this study, the mass concentrations of indoor $PM_{2.5}$ are lower than those of outdoor one, which are corresponding to the publications in a school dormitory in Tehran and in Silesian kindergartens in Poland [3, 7]. However, some previous results in a middle school in Xi'an in China and primary schools in Northern Iran are on the contrary. These researches also revealed that increased concentrations of indoor $PM_{2.5}$ were related to the pupil's activities [2, 8]. Besides, the concentrations of both indoor and outdoor $PM_{2.5}$ in this study are higher than those in school dormitory in Tehran and in Silesian kindergartens in Poland as well [3, 7]. It is noticeable that indoor and outdoor $PM_{2.5}$ concentrations are higher at nursery schools located near heavy traffic roads, which was attributed to vehicular exhaust. High indoor $PM_{2.5}$ levels observed in our study could be explained by infiltration from outdoor environment and the

movement of children during school days. It can be supposed that, besides the suspension of deposited particles through kid's activities, outdoor penetration is also a significant contribution to indoor PM_{2.5}.

3.2. Occurrence of PAH

Cumulative concentrations of indoor and outdoor 15 PAHs from sixteen nursery schools in Ha Noi are presented as boxplots in Fig. 2. Their indoor concentrations range from 103.9 ng/m³ to 653.2 ng/m³, while the outdoor ones vary from 406.4 ng/m³ to 1411.9 ng/m³. The mean concentrations of outdoor Σ 15PAH are 3.2 times higher than those in indoors. The higher values of Σ 15PAH, both in outdoors and indoors, are observed at nursery schools locating nearby busy streets, possibly resulting from vehicular emission. The concentrations of indoor and outdoor Σ 15PAH in nursery schools in Northern Viet Nam are higher than those in a middle school in Xi'an in China, school dormitory in Tehran and Silesian kindergartens in Poland [2, 3, 7]. The variations in PAH concentrations are characterized by different sampling sites and typical factors such as emission sources, ventilation systems in buildings, characteristics of pollutants, human activities, etc.

Individual PAHs have their own chemical, physical and toxicological properties, thus, it is important to analyze concentration of individuals. Nap, Phe and Ant are present the most prominent in LMW, whereas BbF, BaP and DahA concentrations appear highest in HMW in indoors and outdoors. The proportion of LMW and HMW in PAHs is insignificantly different between indoor and outdoor air in this study. Similar findings were observed in Silesian kindergartens in Poland and Lithuania primary schools [7, 9]. The studies of PM_{2.5} bounded BaP inside nursery schools are scarce in Viet Nam. However, as shown in this study, the mean concentration of indoor BaP (16.1 ng/m³) in observed nursery schools in Northern Viet Nam is higher than that in middle school in Xi'an, China (2.3 ng/m³), Silesian kindergartens in Poland (3.6 ng/m³), and Lithuania primary schools (3.2 ng/m³) [2, 7, 9]. There is a good correlation between Σ 15PAHs in indoor and outdoor air that may be attributed to the same emission sources. The proportion of LMW is relatively lower than that of HMW in the observed schools. It is likely that the use of air conditioners can influence the transport of pollutants between indoor and outdoor. Consequently, the concentrations of PAHs decline during the air conditioning compared with natural ventilation [10].

3.3. Source identification

3.3.1. The relationship of indoor and outdoor air

One of the most important factors of air quality management is the source identification. Indoor-outdoor concentration ratios (I/O) of individual PAHs provide a rough indicator of pollution origins. If the I/O ratio is greater than one then the indoor source is stronger than outdoor one. In addition, on the contrary, if indoor source is weaker then the ratio would be less than 1 [6, 9, 11]. Besides, the correlation coefficients (*r*) between the indoor and outdoor PAHs are applied to identify whether the individual PAH measured in indoors is originated from outdoors [6, 12]. In this study, the observed outdoor PAH and PM_{2.5} are higher than those in indoors. The corresponding I/O values vary from 0.3 to 0.7, with the mean value of 0.4 for PAHs and 0.3 - 1.7 with mean value of 0.7 for PM_{2.5} indicating that indoor PM_{2.5} and PAHs are greatly influenced by outdoor sources. The good correlation of indoor and outdoor PM_{2.5} and PAHs with correlation coefficient (*r* = 0.6 and 0.8) implies that indoor PM_{2.5} and PAHs concentration are dominated by outdoor sources. The similar results also were recorded when

the correlation coefficient between indoors and outdoors are used as indicator of the infiltration factor of $PM_{2.5}$ from outdoors to indoors [6, 12]. These sources may include traffic emissions (gasoline and diesel engines), domestic cooking (LPG, biomass, coal briquette, etc.), industrial activities and construction sites, etc. [1, 11, 12]. Linear regression equations are employed to obtain the correlation of the indoor and outdoor $PM_{2.5}$ and PAHs as following: $C_{in} = C_s + F_{in} C_{Cout}$. (C_{in} , C_{out} for $PM_{2.5}$ indoor and outdoor source, F_{in} : infiltration factor; C_s : indoor concentration contributed in indoor source). The linear regressions of indoor and outdoor relationship of $PM_{2.5}$ and PAHs concentration are drawn in two following equations: $C_{PM_{2.5} in} = 3.4 C_{out} - 34.4$ and $C_{PAH in} = 2.8 C_{out} + 117.4$. It can be seen from two above equations that $PM_{2.5}$ from indoor sources (C_s) as intercept is less than zero, which might be attributed to some species' decomposition in indoor $PM_{2.5}$. Concentration of PAHs from indoor sources is 117.4 ng/m^3 , suggesting that approximately 43 % of indoor PAHs come from indoor sources. This result is consistent with the previous studies in which lighter PAHs could originate from indoor activities (food cooking or evaporation from building materials) [12, 13].

3.3.2. Diagnostic ratios

The diagnostic ratios are used as an indicator of possible PAH sources such as $BaA/(BaA+Chr)$, $Flt/(Flt+Pyr)$ and $IcdP/(IcdP+BghiP)$. Figure 3 (a,b) shows the scatter ratio-ratio plot of $BaA/(BaA+Chr)$ vs $Flt/(Flt+Pyr)$ and $IcdP/(IcdP+BghiP)$ vs $BaA/(BaA+Chr)$ in observed sites. When $Flt/(Flt+Pyr)$ is lower than 0.4, it is defined as the petrogenic source (petroleum); from 0.4 to 0.5, it is as fuel oil source; and above 0.5 it is as coal and biomass combustion. In other cases, if $BaA/(BaA+Chr) < 0.2$ or > 0.35 then it is considered to be petrogenic and combustion sources, respectively. The value of $IcdP/(IcdP+BghiP)$ ranging from 0.2 to 0.5 is regarded as good marker for petroleum source; 0.35 to 0.7 as diesel source. $BaA/(BaA+Chr)$ and $IcdP/(IcdP+BghiP)$ being higher 0.5 are identified as biomass burning [1, 11].

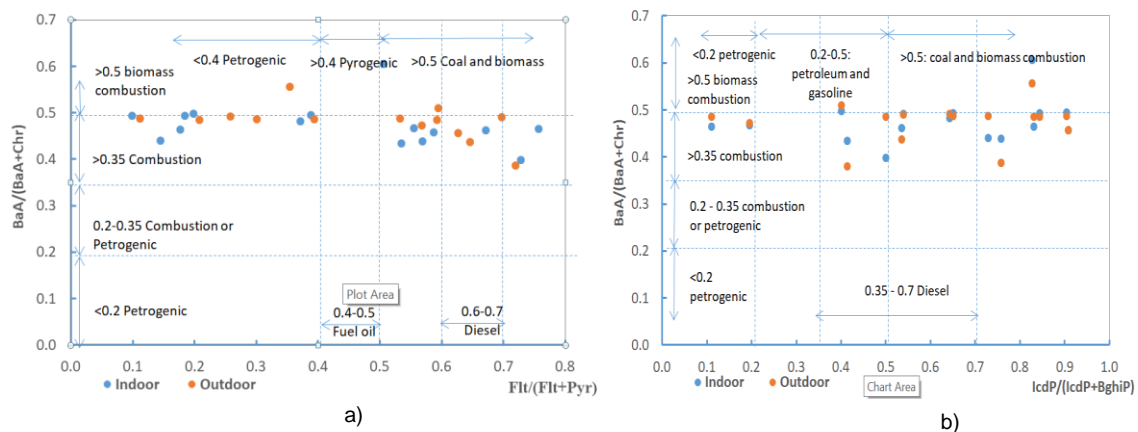


Figure 3. (a) $Flt/(Flt+Pyr)$ vs. $BaA/(BaA+Chr)$ and (b) $IcdP/(IcdP+BghiP)$ vs. $BaA/(BaA+Chr)$.

In Figure 3a, in all samples, $Flt/(Flt+Pyr)$ are either lower 0.4 or above 0.5, while ratio of $BaA/(BaA+Chr)$ is above 0.4 suggesting that coal/biomass combustion and petroleum are primary PAHs sources [1, 11, 12]. In Figure 3b, the diagnostic ratio of $IcdP/(IcdP+BghiP)$ varies from 0.1 to 1, while $BaA/(BaA+Chr)$ is above 0.35, jointly representing that mixed sources including diesel vehicles, gasoline vehicles, coal or biomass combustion may attribute to the primary PAHs sources in investigated nursery schools that are corresponding to potential sources

of PAHs in schools in Beijing, in Xi'an in China and Lithuania [1, 3, 9].

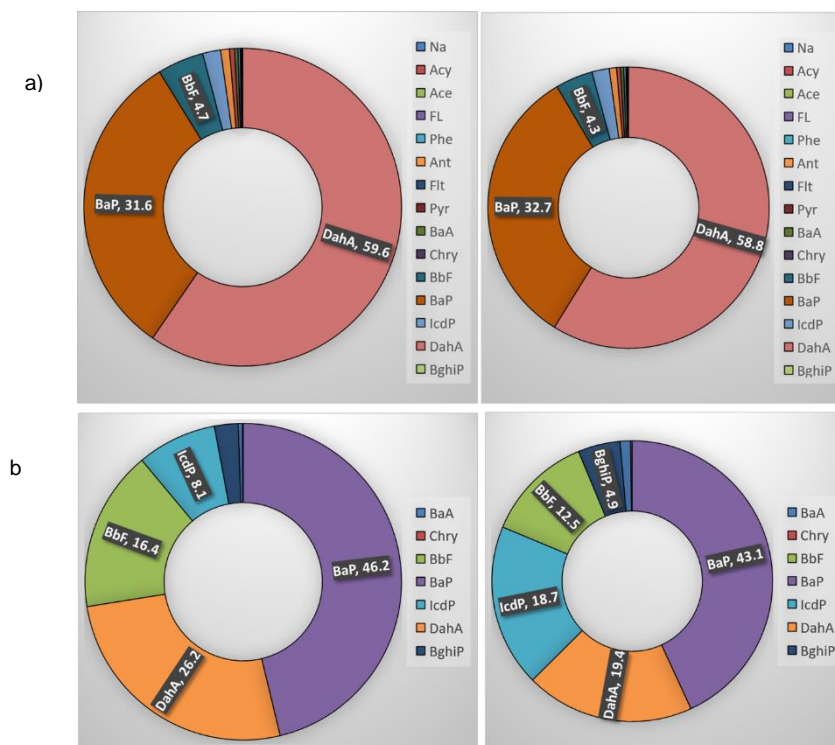


Figure 4. (a) Carcinogenic contribution and (b) mutagenic contribution of PAH in indoor/outdoor air

3.4. Health risk assessment

In this study, comparable levels of indoor $MEQ_{\Sigma 7PAHs}$ are detected in the range of 8.8 ng/m^3 to 229.5 ng/m^3 , whereas those of outdoor $MEQ_{\Sigma 7PAHs}$ vary from 27.9 ng/m^3 to 339.3 ng/m^3 in sixteen nursery schools. The mean concentration of indoor MEQ (44.4 ng/m^3) is lower than that of outdoor MEQ (137.3 ng/m^3), that is consistent with previous researches [3, 7]. The indoor MEQ concentrations are higher than those in some schools in China, Poland, and Lithuania [2, 7, 9]. The mean concentrations of indoor and outdoor $TEQ_{\Sigma 15PAH}$ are 64.6 ng/m^3 and 208.8 ng/m^3 , respectively. The health risk assessment of carcinogenic and mutagenic PAHs takes into account not only individual concentrations of PAH but also carcinogenic and mutagenic potential of each compound. The proportions of carcinogenic and mutagenic contribution for indoors and outdoors are calculated according to equations (1-4) as shown in Fig. 4 (a,b). There is slight difference in the contribution of PAHs individuals related to carcinogenicity and mutagenicity between indoors and outdoors. Particularly, the species with most contribution to the total TEQ are dibenz(a,h)anthracene (DahA) of 59.6 %, followed by benzo(a)pyrene (BaP) of 31.6 % for indoors, and DahA of 58.8 %, followed by BaP of 32.7 % for outdoors. Whilst the most contribution to mutagenicity of PAHs is BaP of 46.2 %, followed by DahA of 26.2 % for indoors and BaP of 43.1 %, followed by 19.4 % for outdoors, respectively. The carcinogenic and mutagenic contribution of PAH bound $PM_{2.5}$ is unity in this study. In both outdoor and indoor samples, the highest contribution to TEQ belongs to DahA, whereas BaP is the highest contribution to MEQ. These findings are not definitely consistent with previous studies, which documented that BaP was the most maker for carcinogenic contribution [6, 7]. However, these

results consider BaP and DahA as makers for the mutagenic and carcinogenic potential of the environmental PAH mixtures. Insufficient ventilation and strong infiltration from outdoors and indoor activities in observed nursery schools may result in poor indoor quality, which is associated with adverse health impact. It is cautious that, MEQ and TEQ values are underestimated because not all of PAHs are analyzed.

4. CONCLUSIONS

The concentrations of indoor and outdoor PM_{2.5} in almost sampling sites are higher than WHO air quality guideline (2004). The concentrations of outdoor PAHs are higher than those of indoor ones. It may be attributed to the emissions of PAHs from potential sources such as vehicular emissions and biomass/coal combustion. Outdoor sources are primary contributors to PAHs in the indoor environment. Nap, Phe and Ant are present the most prominent in LMW, whereas BbF, BaP and DahA appear most in HMW in all samples. Highest concentration of $\Sigma 15$ PAHs and PM_{2.5} are occurred at some nursery schools located nearby congested traffic streets. BaP and DahA are indicated as the most contributors to carcinogenicity and mutagenicity of PAHs samples. Approaches such as mutagenic and carcinogenic equivalents, mutagenic and carcinogenic potential may identify the quantitative contribution related to BaP to health risk assessment.

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REFERENCES

1. Chen Y., Li X., Zhu T., Han Y., Dong Lv. - PM_{2.5}-bound PAHs in three indoor and one outdoor air in Beijing: Concentration, source and health risk assessment, *Sci. Total Environ.* **586** (2017) 255-264.
2. Xu H., Guinot B., Niu X., Cao J., Ho K., Zhao Z., Hang S., Liu S. - Concentrations, particle-size distributions, and indoor/outdoor differences of polycyclic aromatic hydrocarbons (PAHs) in a middle school classroom in Xi'an, China. *Environ Geochem Health* **37** (2014) 861-873.
3. Hassanvand M., Naddafi K., Faridi S., Nabizadeh R., Sowlat H., Momeniha F., Gho A, Arhami M., Kashani H., Niazi S., Rastkari N., Nazmara S., Ghani M., Yunesiaa M. - Characterization of PAHs and metals in indoor/outdoor PM₁₀, PM_{2.5}, PM₁ in a retirement home and a school dormitory, *Sci. Total Environ.* **527** (2015) 100-110.
4. Vo H., Nguyen H., Minoru Y. - Human health hazard of polycyclic aromatic in road dust in Ha Noi metrolophis, Vietnam *J. Sci. & Technol.* **54** (2016) 27-34.
5. Thuy N.T.T., Dung N.T., Sekiguchi K., Thuy L.B., Hien N.T.T., Yamaguchi R. - Mass concentrations and carbonaceous compositions of PM_{0.1}, PM_{2.5}, and PM₁₀ at urban locations in Ha Noi, Viet Nam. *Aerosol Air Qual Res.* **18** (2018) 1591-1605.
6. Delgado S., Stark C, Harrison R. - Carcinogenic potential, levels and sources of polycyclic aromatic hydrocarbon mixtures in indoor and outdoor environments and their implications for air quality standards, *Environ Int.* **37** (2011) 383-392.
7. Błaszczuk E, Rogula-Kozłowska W, Klejnowski K, Fulara I, Mielżyńska-Švach D. - Polycyclic aromatic hydrocarbons bound to outdoor and indoor airborne particles (PM_{2.5})

- and their mutagenicity and carcinogenicity in Silesian kindergartens, Poland. *Air Qual Atmos Health* **10** (2017) 389-400
8. Mohammadyan M, Alizadeh-Iarimi A, Etemadinejad S. - Particulate Air Pollution at Schools: Indoor-Outdoor Relationship and Determinants of Indoor Concentrations, *Aerosol Air Qual Res.* **17** (2017) 857-864..
 9. Krugly E, Martuzevicius D, Sidaraviciute R. - Characterization of particulate and vapor phase polycyclic aromatic hydrocarbons in indoor and outdoor air of primary schools, *Atmos Environ.* **82** (2014) 298-306.
 10. Lv J., Zhu L. - Effect of central ventilation and air conditioner system on the concentration and health risk from airborne polycyclic aromatic hydrocarbons, *J. Environ Sci.* **23** (2013) 531-536.
 11. Wang W., Yao L., Meng C. - Indoor/outdoor relationships and diurnal/nocturnal variations in water-soluble ion and PAH concentrations in the atmospheric PM_{2.5} of a business office area in Jinan, a heavily polluted city in China, *Atmos Res.* **153** (2014) 276-285.
 12. Sangiorgi G., Ferrero L., Ferrini B. S. - Indoor airborne particle sources and semi-volatile partitioning effect of outdoor fine PM in offices, *Atmos Environ.* **65** (2013) 205-214.
 13. Jia C., Batterman S. A. - Critical Review of Naphthalene Sources and Exposures Relevant to Indoor and Outdoor Air, *Int J. Environ Res. Public Health* **7** (2014) 2903-2039.