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Assessment of the Personal Dose Received by School Children due to PM_{10} Air Pollution in Lisbon

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ABSTRACT

Investigation of the personal dose caused by air pollution in children is important due to their vulnerability. Exposure to PM_{10} and its components, particularly certain metals, may pose significant health risks therefore many studies have focused on measuring the ambient indoor/outdoor PM_{10} concentrations in school environments. However, little research has aimed at assessing the resultant personal dose. Hence, this study applied a dosimetry model (ExDoM2) to predict the personal dose received by students in five primary schools in Lisbon, Portugal. The calculations were performed for PM_{10} and PM_{10}-bound metals, and the exposed subjects were assumed to be 10-year-old nose breathers. A realistic exposure scenario involving three different settings (the indoor home, indoor school and outdoor school microenvironments) was implemented for an exposure period of one week (Monday–Sunday). Although the students spent only 24% of their total time inside a school (vs. 73% at home), this environment contributed 44% on average to the weekly deposited dose of PM_{10}, providing further evidence that indoor exposure at schools is a major contributor to the total dose. The modeling results showed that the cumulative deposited doses in the respiratory tract (RT) reached as high as 2,004 μg, 0.16 μg, 0.65 μg, 0.58 μg and 0.06 μg for PM_{10}, Cr, Mn, Pb and Ni, respectively, after one week.

Keywords: Particulate matter; Metals; Child; School; Dosimetry model.

INTRODUCTION

Exposure to airborne pollutants involves primarily the study of inhalation of particulate matter (PM) from humans and its health implications. The latter is directly associated with adverse health effects such as cardiovascular and respiratory diseases even in increased mortality and morbidity (Dockery *et al.*, 1993; Pope *et al.*, 2002; Pope and Dockery, 2006; Kelly and Fussell, 2015; Shiraiwa *et al.*, 2017). The World Health Organization (WHO) reports that 4.2 million deaths are due to exposure to outdoor air pollution. Additionally, the International Agency for Research on Cancer (IARC) has recently recognized outdoor air and PM as carcinogenic to humans (Group I) with approximately 15% of deaths from lung cancer attributed to outdoor air pollution (IARC, 2016).

Health impacts vary substantially because of the different contribution that arises from particle source, composition, exposure routine, region or even season (West *et al.*, 2016). The latter also insightfully underline the inconsistency in published research on the dominant factors that provoke the most harmful effects as a direct result of the great variability of exposure characteristics and PM toxicity. Another important factor that yields targeted research for exposure to ambient air pollution is age-grouped health impacts. Buonanno *et al.* (2012) asserted that children represent one of the most vulnerable populations among the total population and are more physically active than adults. Likewise, Mazaheri *et al.* (2014) pointed out that children are generally more susceptible to air pollution than adults. Elderly are also considered to be vulnerable to air pollution among the total population (Almeida-Silva *et al.*, 2015).

Currently, many researchers focus on school environment and its characteristics in order to assess indoor air quality and

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the major sources (Raysoni *et al.*, 2016; Mohammadyan *et al.*, 2017; Bennett *et al.*, 2019; Branco *et al.*, 2019; Majd *et al.*, 2019; Slezakova *et al.*, 2019). Accordingly, higher indoor particle mass concentrations than outdoors were observed in several schools (Raysoni *et al.*, 2016; Mohammadyan *et al.*, 2017; Bennett *et al.*, 2019), a finding that is linked with indoor sources such as the use of chalk and soil/dust resuspension. On the contrary, other studies report higher concentration outdoors (Rovelli *et al.*, 2014; Carrion-Matta *et al.*, 2019; Slezakova *et al.*, 2019) than inside the classrooms. Braniš *et al.* (2009) found high PM concentrations in a naturally ventilated elementary school gym in Prague where the indoor $PM_{2.5}$ level exceeded the WHO limit in 50% of the measured days. Ventilation, building characteristics and indoor sources have strong impact on the measured concentrations; however, the observations depend strongly on the metrics under investigation. Notably, Morawska *et al.* (2017) note that the outdoor air is a primary source for particle number concentration (especially ultrafine particles) whereas increased PM_{10} and $PM_{2.5}$ are linked with indoor sources from the school environments.

In addition, studies can be found in literature that deal with exposure of the students to PM concentrations and other hazardous components (i.e., metals, polycyclic aromatic hydrocarbons [PAHs]) in schools (Pacitto *et al.*, 2018; Carrion-Matta *et al.*, 2019; Oliveira *et al.*, 2019; Rojas *et al.*, 2019). As such, Oliveira *et al.* (2019) found that children attending school in polluted urban areas are exposed to higher levels of PM than children in rural areas. Rojas *et al.* (2019) studied nanoparticle emissions and their species in school areas and found that road traffic is the major source of nanoparticles. Additionally, the authors point out that heavy metals such as Ni and Zn are related to exhaust whereas other trace metals such as Ca, Fe, Cu originated from road dust resuspension and brake wear (Fe, Cu, Mn).

Lastly, studies that investigate the personal dose received by school students are still very scarce and mainly focus on ultrafine particles. Mazaheri *et al.* (2014) found in their study that the highest daily alveolar dose corresponds to home with 55.3% among all studied microenvironments and that for the investigated schools the mean indoor dose was never higher than the outdoor one. Another study that examined particle deposition in the respiratory tract for school children suggests that the home environment accounts for 77% of the total daily dose intake and that younger children showed higher particle deposition than older children (Patterson *et al.*, 2014). Mazaheri *et al.* (2014) consider monitoring of the personal ultrafine particle exposure important at all microenvironments (e.g., school, home).

In this work, the personal dose of PM_{10} and PM_{10}-bound metals for 10-year-old school children is estimated using a dosimetry model. The main objectives are to obtain the cumulative deposited doses for PM_{10} and PM_{10}-bound metals given the exposure scenario (concentration, daily activity profile) and to estimate the internal dose of metals in the human body. Overall, this study aims to recognize the contribution of the school environment to the personal dose received by the students in a typical weekly period.

MATERIALS AND METHODS

Study Area and Field Measurements

PM_{10} concentration and size distribution field measurements were conducted at the metropolitan area of Lisbon, Portugal. Lisbon is located on the Atlantic Ocean coast at the right side of the Tagus River. The climate is Mediterranean. The air quality in the city is significantly influenced by road traffic (Almeida *et al.*, 2009), marine aerosol (Cruz *et al.*, 2015) and Saharan dust events (Almeida *et al.*, 2008).

Field measurements were performed from September 2017 to July 2018. In total, five primary schools (SA, SB, SC, SD and SE) and 34 houses were selected to perform indoor/outdoor sampling. The location of each sampling point is shown in Fig. 1. Furthermore, the particle-bound metal concentrations were derived from four schools (SA, SB, SC and SD) and 23 houses. School SA is located in a coastal area approximately 8 km north of the city center and 3 km east of the Lisbon airport. Schools SB and SC are located north of the city center (4–5 km) and south of the airport whilst School SD is located southwest from the airport nearby two highways very congested (the 2nd Circular and the North–South Axis, at roughly 200 and 60 m away, respectively) and approximately 6 km away from the city center. Finally, School SE is also located close to the coast 1 km east of the city center.

All schools and houses were monitored for five days during the occupied period. In schools sampling was conducted during teaching hours (approximately 8 h day^{-1}), while in homes the sampling period was 15 h on weekdays (4 days) and 24 h on weekend (1 day), considered as the normal occupied period. In schools and houses indoor sampling was performed in the classroom and living room, respectively, whereas, outdoor sampling in schools was conducted in the playground. Further details for the applied methodology can be found in Faria *et al.* (2020).

Fig. 1. Locations of the schools and houses (Google Maps).

A Leckel sampler (MVS6; Sven Leckel, Germany) was used to measure PM_{10} concentration at a constant flow rate of 2.3 m^3 h^{-1}. The Leckel sampler is certified by the European Committee for Standardization as a reference sampler for PM_{10} measurements according to CEN EN 12341. The PM samples were collected onto filters that were weighed before and after sampling on a microbalance (Sartorius R160P; Greifensee, Switzerland), after being equilibrated for at least 24 h in a conditioned room (20℃ and 50% relative humidity). After weighing, filters were temporarily stored in a freezer until analysis. For purposes of quality assurance and quality control (QA/QC) and in order to prove the consistent operation of the Leckel samplers, field blanks were also collected at both indoor and outdoor microenvironments.

Field measurements for the size distribution were performed together with a Sioutas Personal Cascade Impactor Sampler (PCIS; SKC Inc.) and a Leckel MVS6 sampler. The Sioutas PCIS consists of five stages (< 0.25 μm, 0.25–0.5 μm, 0.5–1 μm, 1–2.5 μm and > 2.5 μm). Size distribution data from the Sioutas PCIS correspond to the fine fraction (< 2.5 μm) whereas for coarse particles ($PM_{2.5-10}$) the Leckel data was used. Summarizing, a detailed description of the sampling methodology is given by Martins *et al.* (2020).

The collected samples were analyzed by X-ray fluorescence (XRF) for the determination of the concentration of metals (Manousakas *et al.*, 2018). Sampled filters and corresponding blanks and field blanks were analyzed following the same analytical procedures. The final ambient concentrations were calculated after the subtraction of analytical blank values from the corresponding sample concentrations.

Dose Assessment

The present study aims to evaluate the impact of the different school environments to the personal dose received by the students. Thus, all simulations were conducted using the same average PM_{10} concentration obtained from all houses while modeling a house environment, whilst the data received from each school were implemented in each simulation for each school (SA, SB, SC, SD, SE). In addition, the size distributions (PM_{10} and metals) used in the simulations were chosen from a selected school and a selected house of the campaign as typical size distributions.

The exposed person was considered to be a 10-year-old child, thus, spends his time between the school and the house on weekdays but stays in the house during the weekends. Subsequently, Table 1 presents the daily activity profile that was adopted in all simulations. The type of the environment was derived from a questionnaire that was distributed in the participating schools and as it is shown differs for weekdays and weekends. Nonetheless, the same activity level (sleep, sitting, light exercise) was used for both weekdays and weekends. In practice, Table 1 suggests that the main difference in the daily activity profile between weekdays and weekends lies in the time period 09:00–18:00 where the children are assumed to have the same activity but in different environments (school vs. house). Table 1 also indicates that children spend approximately 96% of their time indoors during weekdays.

Dosimetry Model

The dose received by inhalation due to exposure to PM_{10} and the corresponding bound metals was estimated by ExDoM2 (Chalvatzaki and Lazaridis, 2015). The latter is a revised version of ExDoM (Aleksandropoulou and Lazaridis, 2013) and is based on the International Commission on Radiological Protection (ICRP, 1994, 2015). In the present work, particle size for each stage was considered monodisperse (σ_g = 1) and hence the geometric midpoint (square root of lower cut-off size × upper cut-off size) was used for the calculations. The deposition fraction is given by ICRP (1994):

$$DE_j = n_j \phi_j \prod_{jj=0}^{j-1} \left(1 - n_{jj}\right) \quad (1)$$

where n_j is the deposition efficiency of the j filter, ϕ_j is the fraction of tidal air that reaches the j filter and n_0 is the prefiltration efficiency. Total number of filters was nine which correspond to two filters for the anterior nose (ET1) region; two filters for the posterior nasal passages, pharynx and larynx (ET2) region; two filters for the bronchial (BB) region; two filters for bronchiolar (bb) region and one filter for the alveolar-interstitial (AI) region.

The deposition fraction in the ET region was obtained from the sum of the deposition fraction in ET1 and ET2 regions and then re-partitioned 65% to ET1 region and 35% to ET2 region (Chalvatzaki and Lazaridis, 2015; ICRP, 2015). The physiologically based pharmacokinetic (PBPK) module of ExDoM2 was used for the estimation of the internal dose of metals (As, Pb, Mn, Cd, Cr) in the human body (e.g., kidney) and is based on several studies (Kjellstrom and Nordberg, 1978; O'Flaherty *et al.*, 2001; Sharma *et al.*, 2005; Liao *et al.*, 2008; Chou *et al.*, 2009; Liao *et al.*, 2009; Schroeter *et al.*, 2011). The dose of Pb and Mn in each organ

Table 1. Daily activity profile during weekdays and weekends.

Time	Activity	Environment	
		Weekdays	Weekends
00:00–08:00	Sleep	House (indoor)	House (indoor)
08:00–09:00	Light exercise	House (indoor)	House (indoor)
09:00–13:00	Sitting	School (indoor)	House (indoor)
13:00–14:00	Light exercise	School (outdoor)	House (indoor)
14:00–18:00	Sitting	School (indoor)	House (indoor)
18:00–22:00	Light exercise	House (indoor)	House (indoor)
22:00–00:00	Sleep	House (indoor)	House (indoor)

or tissue group *i* of the human body is given by (Sharma *et al.*, 2005; Liao *et al.*, 2008; Chou *et al.*, 2009; Chalvatzaki and Lazaridis, 2015):

$$\frac{dA_{i,m}}{dt} = Q_i \times \left(C_{a,m} - C_{v,m}\right) - Metabolism_{i.m} - Elimination_{i,m} \quad (2)$$

where $A_{i,m}$ is the dose of chemical *m* in tissue group *i* (μg), Q_i is the blood flow rate to tissue group *i* (L h^{-1}), $C_{a,m}$ is the arterial concentration (μg L^{-1}) of chemical *m* and $C_{v,m}$ is the venous concentration of chemical *m* (μg L^{-1}). The metabolism parameter ($Metabolism_{i,m}$) is zero for Pb and Mn.

The blood flow during no physical activity (sleep/sitting/rest) for children was based on Edginton *et al.* (2006) whilst the mass of organs was based on ICRP (2002). Due to differences between the blood flow and the mass of (some) organs between a 10-year-old male and female, the exposed subject was chosen to be a 10-year-old male child.

The blood flow during light exercise for a 10-year-old male child was estimated as:

$$Q_{i,10\ year\ old\ male,\ Light\ exercise} = \frac{Q_{i,10\ year\ old\ male,\ Rest} \times Q_{i,\ Adult\ male,\ Light\ exercise}}{Q_{i,\ Adult\ male,\ Rest}} \quad (3)$$

On the other hand, the blood flow during no physical activity for adult males was based on ICRP (2002), whereas during light activity the values of ICRP were modified based on Lenz (2010) and Plowman and Smith (2011). A detailed description of the methodology for the modification of the values of ICRP for the calculation of blood flow during light exercise can be found in Chalvatzaki *et al.* (2018).

The dose of Cr(III) in each organ or tissue group i of the human body is given as (O'Flaherty *et al.*, 2001; Chalvatzaki and Lazaridis, 2015):

$$\frac{dA_i}{dt} = KIN \times C_{plasma} - KOUT \times C_i - Elimination_i \quad (4)$$

where A_i is the dose of Cr(III) in tissue group *i* (μg), *KIN* is the clearance of Cr(III) from plasma into tissue (L h^{-1}), *KOUT* is the clearance of Cr(III) from tissue into plasma (L h^{-1}), C_{plasma} is the concentration of Cr(III) in plasma (μg L^{-1}) and C_i is the concentration of Cr(III) in tissue group *i* (μg L^{-1}).

RESULTS AND DISCUSSION

PM_{10} Concentration and Size Distribution

Table 2 presents the indoor/outdoor PM_{10} concentration measured at each school. The highest indoor daily concentrations among the measured schools were obtained in SD and SC with daily PM_{10} reaching 161.3 μg m^{-3} and 119.2 μg m^{-3} respectively, whereas indoor PM_{10} concentration in the rest of the three schools was not higher than 62.4 μg m^{-3}. High indoor PM_{10} concentrations (> 50 μg m^{-3}) are also reported in other studies conducted in schools (Almeida *et al.*, 2011; Tran *et al.*, 2012; Morawska *et al.*, 2017; Paccito *et al.*, 2018; Sánchez-Soberón *et al.*, 2019). Furthermore, Table 2 demonstrates that in most cases the indoor concentration was higher than the outdoor which implies the presence of indoor sources (such as chalk emission during writing and wiping activities). Goel *et al.* (2017) found that gypsum chalks produce more dust than the calcium carbonate chalks. Mohammadyan *et al.* (2017) attribute higher indoor $PM_{2.5}$ to the use of chalk on blackboards and the lack of air conditioning in the investigated schools. In addition, Rovelli *et al.* (2014) asserted that the movement and the activities by students cause resuspension of settled particles in school classrooms. Similar observation, where the indoor PM_{10} levels were higher than the outdoor levels, was found in several schools in Barcelona by Paccito *et al.* (2018) and also in French classrooms during the occupation period by Tran *et al.* (2012).

Additionally, the indoor and outdoor concentrations of the PM_{10}-bound metals at the four schools (no data for School SE) are listed in Table 3. The latter indicates that Cr, Mn and Pb are the dominant among the measured metals in three schools (SA, SC, SD) whilst Cr and Mn were higher by one order of magnitude compared to the rest of the metals in SB. Metal elements are commonly found inside schools within urban areas (Almeida *et al.*, 2011; Tran *et al.*, 2012; Rojas *et al.*, 2019; Sánchez-Soberón *et al.*, 2019). Metals

Table 2. PM_{10} concentration (μg m^{-3}) measured indoors and outdoors for each school.

School	Monday	Tuesday	Wednesday	Thursday	Friday
Indoor					
SA	37.4	30.9	35.0	34.1	52.6
SB	62.4	52.0	39.6	55.3	48.5
SC	91.5	89.9	58.2	119.2	89.7
SD	161.3	79.9	95.9	125.1	82.9
SE	31.5	30.5	37.5	41.9	22.2
Outdoor					
SA	34.9	33.8	35.2	37.0	44.4
SB	26.3	19.0	9.9	14.7	33.8
SC	30.9	44.6	48.8	55.7	45.0
SD	41.6	15.7	23.6	39.0	21.6
SE	47.3	25.7	21.8	18.7	12.6

Table 3. Indoor/outdoor concentration (ng m^{-3}) of PM_{10}-bound metals at the four schools (no data for SE).

	Mon	Tue	Wed	Thu	Fri	Mon	Tue	Wed	Thu	Fri
	Indoor					Outdoor				
SA										
Cr	6.85	3.97	5.52	4.78	6.2	3.33	1.16	0.206	1.42	2.51
Mn	14.16	5.73	0.617	17.71	16.2	19.71	8.48	6.97	6.52	12.43
Ni	0.411	0.411	4.43	0.411	1.11	0.411	5.34	1.93	0.411	0.411
Pb	23.49	8.2	13.7	33.62	23.98	24.97	11.7	0.617	13.3	15.6
SB								SB		
Cr	4.4	0.832	4.79	1.38	4.11	1.11	1.36	0.206	0.772	1.27
Mn	6.33	6.51	0.617	6.51	8.7	3.49	0.617	0.617	2.98	8.34
Ni	0.411	0.411	0.411	0.411	0.411	0.411	0.411	0.411	5.56	2.21
Pb	0.617	0.617	0.617	0.617	0.617	0.617	0.617	0.617	0.617	0.617
SC								SC		
Cr	8.31	7.52	1.94	7.25	6.25	0.939	0.206	0.939	3.82	1.48
Mn	9.67	11.35	8.17	20.96	12.54	6.16	4.49	5.99	12.43	7.27
Ni	1.88	1.54	0.411	0.361	1.86	0.411	0.411	1.21	0.411	0.610
Pb	18.37	31.3	15.5	16.7	20.5	10.8	39.5	18.0	16.9	21.3
SD										
Cr	6.76	6.33	9.26	5.91	13.03	5.89	5.45	1.23	3.41	1.86
Mn	28.79	37.06	35.84	32.75	30.06	20.88	16.07	7.09	16.96	0.617
Ni	1.52	1.71	2.84	1.71	2.47	0.411	0.411	0.411	0.411	0.411
Pb	26.82	22.51	0.617	0.617	0.617	23.33	0.617	0.617	29.29	0.617

such as Cr, Mn, Ni and Pb are reported to be associated with traffic exhaust, heavy-fuel combustion in industries, heating emissions and tire/brake wear (Tran *et al.*, 2012; Rojas *et al.*, 2019). Moreover, a comparison of the indoor concentration with the outdoor one suggests important differences between the measured concentrations. Specifically, Cr was found in higher concentrations indoors than outdoors in all cases except in SB on Tuesday. Other cases where an indoor metal concentration was higher than the outdoor can be observed for Mn in SB, SC and SD as well as for Ni in SC and SD. Regarding Pb high indoor concentrations were obtained for SA, SC and SD.

The average indoor PM_{10} and bound metal concentrations obtained from the measurements in the houses are presented in Table 4. Accordingly, averaged daily PM_{10} concentration ranged from 17.3 to 26.9 μg m^{-3} whilst higher metal concentrations correspond to Pb and Mn.

The size distributions (PM_{10} and metals) used in the simulations were from a selected school (SC) and a selected house and are presented in Figs. 2 and 3, respectively. Fig. 2(a) shows that the mass fraction of coarse particles (> 2.5 μm) was higher than fine particles at School SC (indoor) with the corresponding values being 0.55, 0.73, 1.00, 1.00 and 1.00 for PM, Cr, Mn, Ni and Pb respectively. Regarding the outdoor school environment (Fig. 2(b)), it is observed that the mass fraction of coarse particles was equal to 0.43, 0.64, 1.00, 1.00 and 1.00 for PM, Cr, Mn, Ni and Pb, respectively. These results imply that the coarse fraction dominated over the fine particles in School SC and that the studied metals were bound to the coarse particles (both indoors/outdoors). Likely, higher indoor metal concentrations are due to dust/soil resuspension that was transferred from outdoors and therefore carried these elements indoors. Tran *et al.* (2012) found increased percentages of trace metal elements in the fine fraction during unoccupied periods. The authors linked this observation with the decrease of coarse particles during unoccupied periods (absence of human presence) in the schools.

The opposite characteristic is found for houses (Fig. 3) where the fine fraction was dominating over the coarse fraction for all cases shown, i.e., the mass fraction of fine particles was equal to 0.78, 0.71, 0.73, 1 and 0.82 for PM, Cr, Mn, Ni and Pb, respectively.

Deposited Dose for PM_{10}

The cumulative deposited dose of PM_{10} in the respiratory tract and in the regions of ET and lungs are shown in Fig. 4 for each school. Higher weekly deposited dose was received by students in Schools SD and SC while the rest of the three schools were characterized by lower deposition. The higher deposited dose obtained for these two schools is directly linked with the higher PM_{10} concentrations measured inside both schools compared to the other three schools. Therefore, a student in School SD (higher dose) received 2,004 μg in the respiratory tract, 1,486 μg in the ET region and 518 μg in lungs for 1 week exposure. The corresponding values for a student in School SE (lower dose) was 1,156 μg, 826 μg and 330 μg respectively. Moreover, the results indicate that higher deposited dose was obtained in the ET region compared to the lungs, a finding that is associated with particle size. Zwozdziak *et al.* (2017) found that coarse particles are primarily deposited in the ET region. The nose has the ability to filter particles and hence to protect the lungs (Foos *et al.*, 2007); therefore, the knowledge of the regional deposition is important because the incidence of lung diseases depends on the deposited dose in the lungs or specific region of lungs (Patterson *et al.*, 2014). Other studies (Ferguson *et al.*, 2013; Zwozdziak *et al.*, 2017) asserted that coarse particles can cause inflammatory response.

Table 4. Average daily indoor PM_{10} concentration (µg m^{-3}) and average daily PM_{10}-bound metal concentration (ng m^{-3}) obtained from houses.

	Monday	Tuesday	Wednesday	Thursday	Friday	Saturday	Sunday
PM_{10}	18.6	19.4	18.7	17.3	26.9	17.8	19.6
Cr	0.760	0.986	0.696	0.704	0.450	1.46	0.272
Mn	2.74	2.67	1.94	3.17	2.26	3.18	1.22
Ni	0.549	0.846	0.972	0.822	0.411	0.642	0.411
Pb	4.06	4.15	3.72	4.43	4.68	2.9	1.99

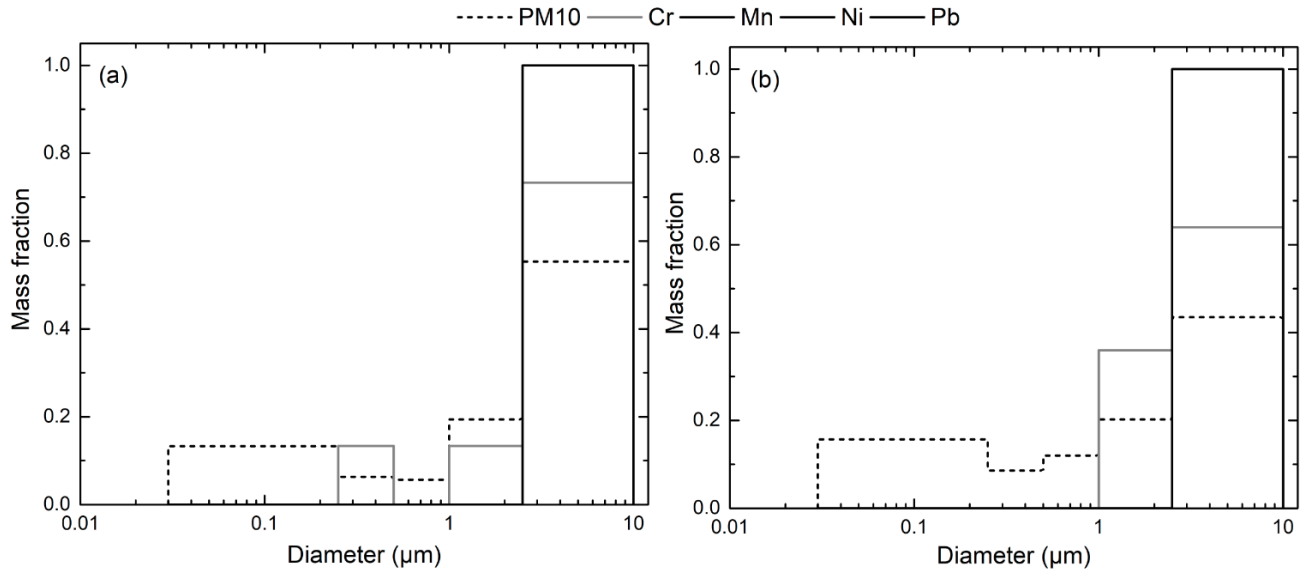

Fig. 2. Size distributions of PM_{10} and PM_{10}-bound metals for (a) indoor and (b) outdoor school (SC) environment. Mn, Ni and Pb had the same mass fraction distribution and are plotted with the same color.

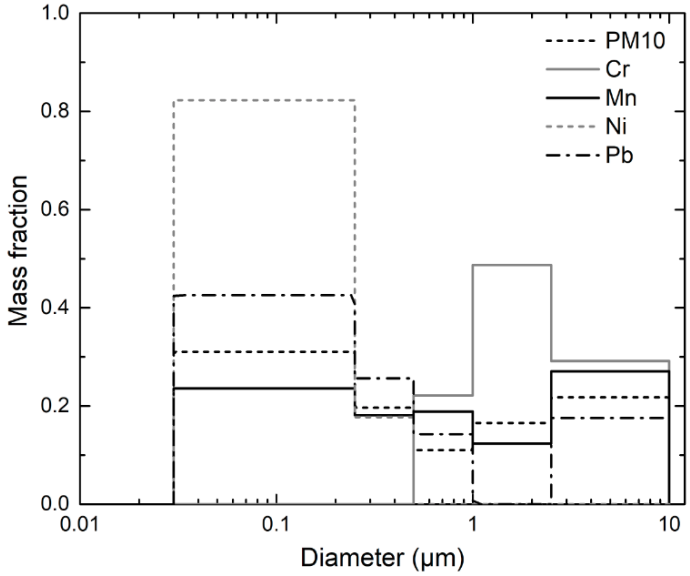

Fig. 3. Size distribution of the mass fraction of PM_{10} and PM_{10}-bound metals in indoor house environment.

Subsequently, Table 5 presents the hourly, daily and weekly deposited dose received by the students in each school and for each environment. Higher estimates of the hourly deposited dose for the school environment were obtained when being outdoors during weekdays (16–36 µg). The indoor school environment follows with values ranging between 9–30 µg whilst the house environment corresponds to the lower hourly deposited dose at 6 µg. This finding alters when looking at the averaged daily doses. Specifically, higher average daily dose (72–239 µg) corresponds to the

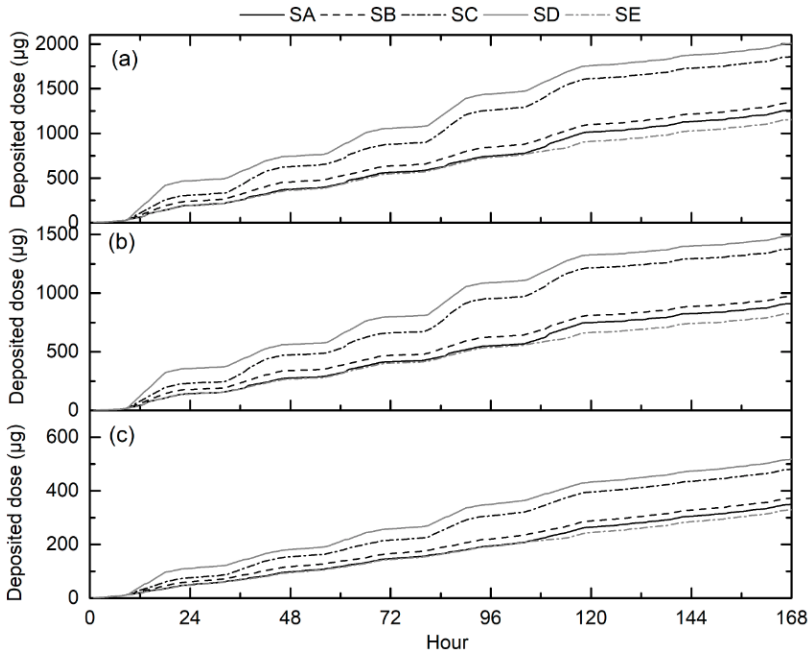

Fig. 4. Cumulative deposited dose (µg) of PM_{10} in the (a) respiratory tract, (b) ET (ET1 + ET2) region and (c) lung (BB + bb + AI) region for each school during the 1 week exposure scenario.

Table 5. Average hourly deposited dose (µg), average daily deposited dose (µg) and cumulative deposited dose (µg) at the end of one week for each school. Hourly and daily doses are shown for weekdays (wo) and weekends (we). The contribution of the type of the environment at the weekly deposited dose is also given.

	Hourly (µg)		Daily (µg)		Weekly (µg)	Contribution (%)
SA	wo	we	wo	we		
House (indoor)	6.0	5.1	90.5	122.6	697.7	55.3
School (indoor)	10.4	0.0	83.3	0.0	416.6	33.0
School (outdoor)	29.3	0.0	29.3	0.0	146.4	11.6
SB						
House (indoor)	6.0	5.1	90.5	122.6	697.7	51.9
School (indoor)	14.1	0.0	113.1	0.0	565.3	42.0
School (outdoor)	16.3	0.0	16.3	0.0	81.7	6.1
SC						
House (indoor)	6.0	5.1	90.5	122.6	697.7	37.5
School (indoor)	24.6	0.0	196.7	0.0	983.3	52.9
School (outdoor)	35.5	0.0	35.5	0.0	177.3	9.5
SD						
House (indoor)	6.0	5.1	90.5	122.6	697.7	34.8
School (indoor)	29.9	0.0	239.0	0.0	1,195.2	59.6
School (outdoor)	22.3	0.0	22.3	0.0	111.3	5.6
SE						
House (indoor)	6.0	5.1	90.5	122.6	697.7	60.4
School (indoor)	9.0	0.0	71.7	0.0	358.7	31.0
School (outdoor)	19.8	0.0	19.8	0.0	99.2	8.6

indoor school µenvironment followed by the house at 91 µg and the lower dose (16–36 µg) corresponds to the outdoor school environment. Daily deposited dose equal to 49.6 µg is reported at Sánchez-Soberón *et al.* (2018) in a school in Spain. The above results highlight the relative contribution of each environment to the received dose by the students. Although the outdoor environment presents the higher hourly rates, time spent in each environment and ambient concentration has significant impact on the received dose for longer exposure. Therefore, the cumulative dose by the end of one week has higher values for house and indoor school environments suggesting that these two environments are the major contributors to the total received dose by the end of the week. In more detail, the house has an average contribution of 48% whilst the indoor school environment has a contribution of 44% to the received dose. According

to the weekly time schedule used in the simulations the indoor school environment corresponds to the 24% of the time spent whereas the house environment corresponds to 73%. This finding highlights the impact of the indoor school environment to the students, i.e., only 24% of their weekly time spent inside the school but this percentage corresponds to almost half of the weekly received dose. Recall that this study focuses on the impact from the different school environment; thus the obtained doses for house is the same for all case studies, although it is found that children attending the same school can receive different dose due to characteristics of their house (Buonanno *et al.*, 2012).

A comparison between the realistic exposure scenario and a scenario where simulations consider the student being all the time outdoors is presented in Fig. 5. The same daily activity profile was used in both scenarios where for the realistic scenario it was implemented as it is given in Table 1, whereas in the outdoor exposure scenario the same activity profile was adopted but all environments were considered to be at school outdoors. The simulations were conducted for School SC only for weekdays. SC was selected due to the higher outdoor concentration compared to the other schools. Outdoor PM_{10} concentration and the size distribution was the same in both scenarios.

Fig. 5 indicates that the outdoor exposure scenario provides higher daily dose compared to the realistic scenario for every weekday except on Monday. The average daily deposited dose was 390 μg and 323 μg for the outdoor and the realistic exposure scenario respectively. This observation is linked with the activity profile used and the corresponding ambient concentrations used to model each environment. For example, using the realistic exposure scenario a student is exposed to higher concentration during indoor school hours compared to outdoors by a factor of 1.2–3.0 but lower concentration when being at the house (outdoor concentration higher by a factor of 1.7–3.2 than indoor home concentration). The higher dose on Monday when using the realistic scenario compared to the outdoor scenario is primarily due to the considerably lower outdoor PM_{10} concentration that corresponds to this day (30.9 μg m^{-3}).

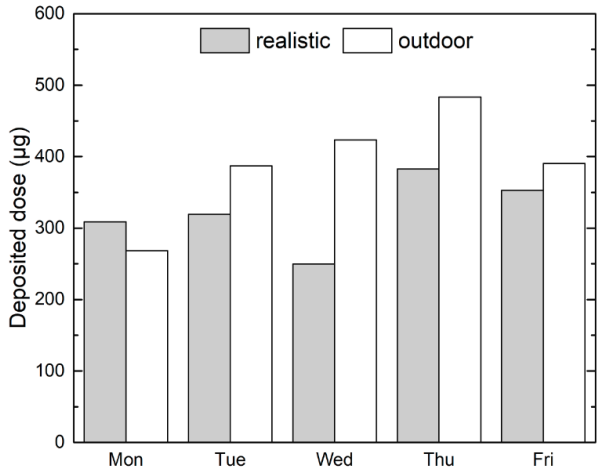

Fig. 5. Daily deposited dose in the respiratory tract for realistic and outdoor exposure scenario.

Finally, the retention of PM_{10} in the respiratory tract (sum of all compartments of the respiratory tract) and the mass transferred to the oesophagus, lymph nodes and blood (absorption into the blood) at the end of one week are shown in Fig. 6. No absorption into the blood takes place in the ET1 region (ICRP, 1994, 2015) whilst the absorption of PM_{10} in blood was assumed to be moderate. Accordingly, the highest dose was obtained to the oesophagus for all schools (764 μg, 817 μg, 1,164 μg, 1,260 μg and 692 μg for students in Schools SA, SB, SC, SD and SE respectively). This finding is associated with higher deposited dose in the ET region whereby particles transferred to the oesophagus by mucociliary action (ICRP, 2015). In particular, the deposited particles into the ET region are transferred more quickly to the oesophagus in comparison with the other regions of the respiratory tract which require more time to reach the ET2 region (ICRP, 2015). Huang *et al.* (2017) found that high PM_{10} concentrations were associated with increased mortality due to oesophageal cancer. In addition, Baccarelli *et al.* (2008) pointed out that exposure on PM_{10} may cause effects on blood clotting, whilst Brook (2008) proposed that PM constituents reaching systemic circulation can interact directly with the cardiovascular system.

Deposited Dose for Metals

Fig. 7 presents the cumulative deposited dose of the four PM_{10}-bound metals (Cr, Mn, Ni and Pb) in the respiratory tract (ET1 + ET2 + BB + bb + AI). In these calculations, the metal oxide densities were used as discussed in Chalvatzaki *et al.* (2018). The highest weekly deposited dose of Cr, Mn and Ni was received by a student in School SD, with the corresponding values being 0.16 μg, 0.65 μg and 0.06 μg respectively. For Pb, a student in School SC received the highest weekly deposited dose (0.58 μg). Overall, the results presented in Fig. 7 indicate that School SB preserved the lower doses, a finding that is associated with the lower metal concentration measured for all four metals indoors. Inhalation of particle-bound metals can cause carcinogenic

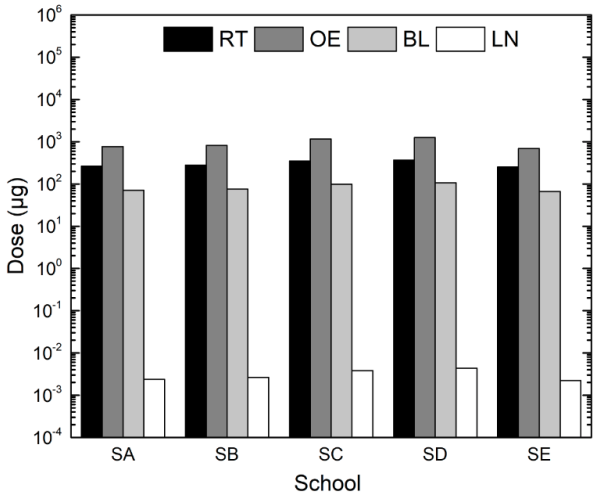

Fig. 6. Retention of PM_{10} in the respiratory tract (RT) and mass transferred to the oesophagus (OE), lymph nodes (LN) and absorbed into the blood (BL) for each school.

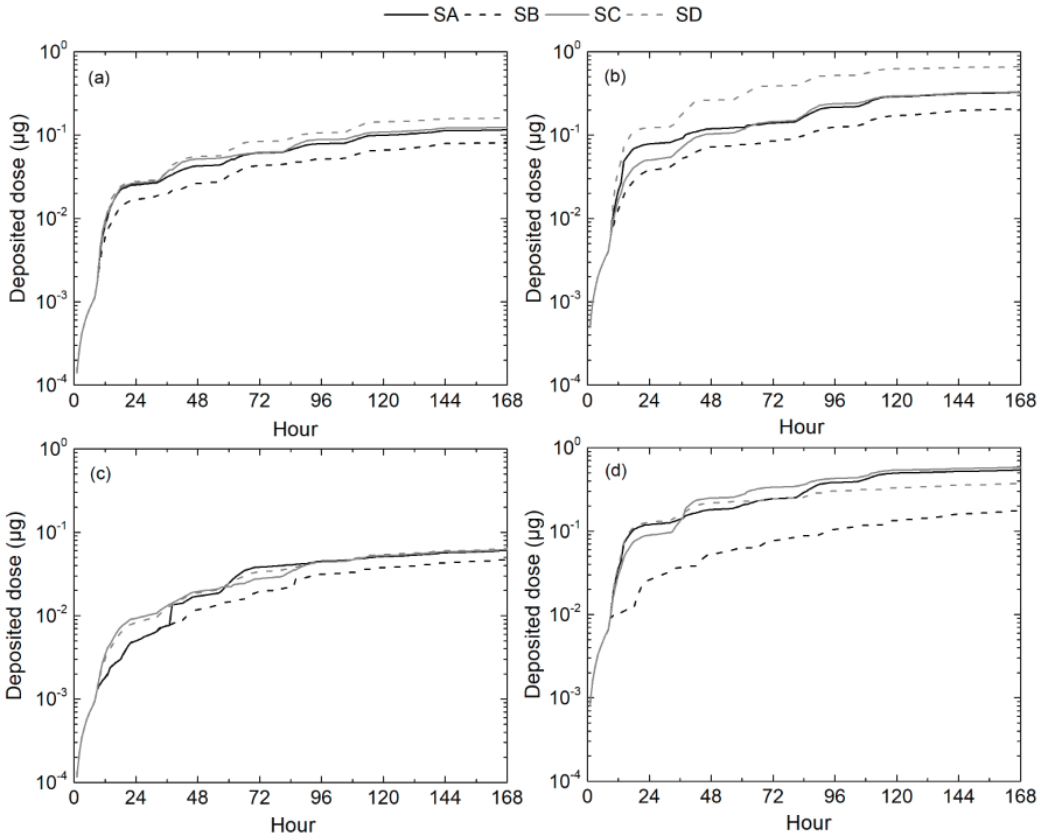

Fig. 7. Cumulative deposited dose in the respiratory tract for PM_{10}-bound metals: (a) Cr, (b) Mn, (c) Ni and (d) Pb in each school (no data for School SE).

and non-carcinogenic effects in adults (Chalvatzaki *et al.*, 2019); thus, evaluation of metal concentrations in airborne particles and its implication on school students' health is necessary.

At the end, all plots in Fig. 7 imply that the lower dose for all four metals was obtained for School SB. The differences in the deposited dose for each metal are indicative of the concentrations measured in each environment and school, with these results highlighting the importance of examining the impact of both particulate matter concentrations along with the concentrations of the hazardous metal components. Furthermore, the contribution of each environment at the weekly deposited dose for each metal is shown in Table 6. The results suggest that the house and indoor school environment have the greatest contribution to the weekly deposited dose, a finding that is in agreement with the results obtained for PM_{10} (Table 5). Specifically, the indoor school environment has the highest contribution at the weekly deposited dose in the respiratory tract for Cr in all schools. The same characteristic is observed for Mn except in School SB where the house environment has the greatest contribution (51%). Regarding Ni, in the houses was found higher contribution than in the schools. Lastly, for Pb a substantial contribution (93%) of the house environment was found in SB as well as Pb had the higher contribution in SD. However, the indoor school environment was the greater contributor to the weekly deposited dose of Pb for Schools SA (56.2%) and SC (51.8%). Overall, the highest contribution to the deposited dose was originated by the indoor school environment for Cr and Mn (57% and 55%, respectively), whereas Ni and Pb (54% and 49%, respectively) had greatest contribution by the house. Again, evaluation of the results should take into account the time spent in each microenvironment (24% for indoor school vs. 73% for house).

Table 6. Contribution (%) of the type of the environment at the weekly deposited dose of each metal at each school.

	Cr	Mn	Ni	Pb
SA				
House (indoor)	33.0	32.5	51.7	30.5
School (indoor)	59.3	49.4	33.0	56.2
School (outdoor)	7.7	18.1	15.3	13.3
SB				
House (indoor)	46.6	50.9	66.4	93.0
School (indoor)	47.5	40.7	12.8	5.1
School (outdoor)	6.0	8.4	20.7	1.9
SC				
House (indoor)	30.7	32.0	50.5	28.3
School (indoor)	63.1	56.0	44.2	51.8
School (outdoor)	6.2	11.9	5.3	19.8
SD				
House (indoor)	23.8	16.1	49.1	44.0
School (indoor)	64.6	73.8	47.4	40.3
School (outdoor)	11.5	10.2	3.5	15.8

Fig. 8 presents the internal dose of Cr, Mn and Pb in each school. Accordingly, the highest dose of Cr was found in the GI tract (2.7–5.4 × 10^{-2} μg) for all schools. Cr is transferred to the GI tract mainly via mucociliary clearance (O'Flaherty *et al.*, 2001). Specifically, particles deposited to the ET region are transferred to the oesophagus faster in comparison with the particles deposited in the lower respiratory tract (ICRP, 1994, 2015). Regarding Mn, at the end of the week the highest dose was obtained in other tissues (1.4–4.9 × 10^{-1} μg) for all schools. WHO (1999) reports that inhalation exposure to Mn particles affects lungs and the nervous system. ATSDR (2012) also reports that inhalation of Mn particles cause inflammatory response in the lung region. Lastly, model simulations indicate that the major accumulation of Pb occurs in the bones (7.3–2.2 × 10^{-1} μg) and blood (2.2–6.7 × 10^{-2} μg). According to other studies, Pb accumulates in the bones while Pb in blood reflects recent exposure (Rabinowitz, 1991; O'Flaherty, 1993; IPCS, 1995). WHO (2019) asserts that high levels of Pb attacks the brain and the central nervous system. Children are known to be more susceptible to heavy metals in comparison with adults and the exposure can prevent their physical growth and development (Zeng *et al.*, 2019). Thus, exposure assessment of students within the school environment constitutes a challenge for modern research.

CONCLUSIONS

This study used a dosimetry model to estimate the personal doses of PM_{10} and PM_{10}-bound metals (Cr, Mn, Ni and Pb) for 10-year-old school children. Data obtained from five schools and 34 houses were analyzed using a 1 week exposure scenario in which the children were assumed to spend their time in three locations: 1) the indoor home microenvironment, 2) the indoor school microenvironment and 3) the outdoor school microenvironment.

The modeling results showed that the cumulative deposited dose of PM_{10} in the respiratory tract after one week ranged from 1,156 μg to 2,004 μg for the students at the five schools. The variation in the estimated (weekly) deposited dose was directly linked with the indoor concentrations, i.e., higher indoor concentrations led to a higher deposited dose. Furthermore, a comparison between the hourly and daily deposited dose rates revealed the relative contribution of each environment to the total dose. Although a higher hourly dose rate was found for the outdoor school environment, the largest contributor to the daily dose was the home environment (48%), followed by

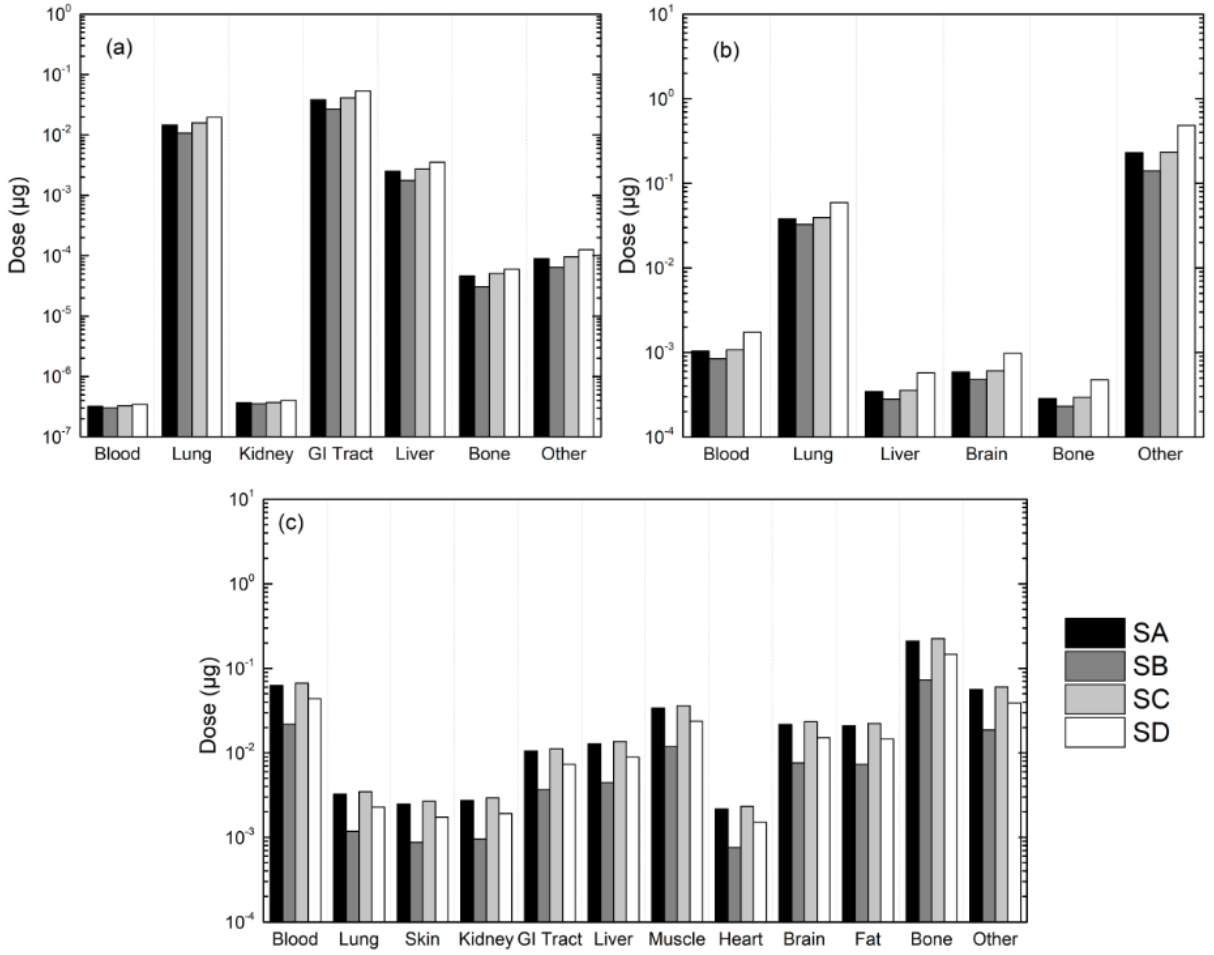

Fig. 8. Internal dose (μg) of metals in the human body for each school for (a) Cr, (b) Mn and (c) Pb at the end of 1 week exposure scenario.

the indoor school environment (44%). The importance of clean indoor air in schools is highlighted by the nearly equal contributions from the home and indoor school environments, despite the fact that the children spent 73% of their weekly time in the former and only 24% in the latter.

Higher doses of PM_{10}-bound metals were also associated with increased indoor concentrations, and a significantly weaker dose was calculated for the school with the lowest metal concentration (SB) than the other schools. Again, the modeling results indicated large contributions from the indoor environments, in particular, Ni (54%) and Pb (49%) from homes and Cr (57%) and Mn (55%) from schools. Ultimately, each metal exhibited unique behavior in terms of being accumulated by the human body.

This study identifies the contributions of different environments (home and school) to the dose of PM_{10} (and bound metals) received by school children via particulate air pollution, which is critical to assessing (and minimizing) the health risks for this vulnerable group.

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