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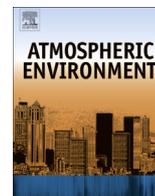


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## Indoor/outdoor relationships of PM<sub>10</sub>, PM<sub>2.5</sub>, and PM<sub>1</sub> mass concentrations and their water-soluble ions in a retirement home and a school dormitory



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### HIGHLIGHTS

- Higher correlations were found between indoor and outdoor for PM<sub>2.5</sub> and PM<sub>1</sub>.
- Higher percentages of secondary inorganic aerosols were found for PM<sub>2.5</sub> and PM<sub>1</sub>.
- For all the ions, the I/O was lower than the unity, except for calcium in PM<sub>10</sub>.
- We had significant indoor sources for PM<sub>10</sub>, but not for PM<sub>2.5</sub> and PM<sub>1</sub>.
- Mobile sources were the most important source of PM in Tehran.

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### ABSTRACT

Indoor/outdoor particulate matter (PM<sub>10</sub>, PM<sub>2.5</sub>, and PM<sub>1</sub>) and their water-soluble ions were measured in a retirement home and a school dormitory in Tehran, from May 2012 to January 2013. Hourly indoor/outdoor PM concentrations were measured using GRIMM dust monitors and 24-h aerosol samples were collected by low-volume air samplers. Water-soluble ions were determined using an ion chromatography (IC) instrument. Although the mean outdoor PM concentrations in both sampling sites were almost equal, the mean indoor PM<sub>10</sub> in the school dormitory was approximately 1.35 times higher than that in the retirement home. During a Middle Eastern dust storm, the 24-h average PM<sub>10</sub>, PM<sub>2.5</sub>, and PM<sub>1</sub> concentrations were respectively 3.4, 2.9, and 1.9 times as high as those in normal days outdoors and 3.4, 2.8, and 1.6 times indoors. The results indicated that secondary inorganic aerosols were the dominant water-soluble ions of indoor and outdoor PM. We found that the smaller the particle, the higher the percentage of secondary inorganic aerosols. Except for PM<sub>10</sub> in the school dormitory, strong correlations were found between indoor and outdoor PM. We estimated that nearly 45% of PM<sub>10</sub>, 67% of PM<sub>2.5</sub>, and 79% of PM<sub>1</sub> in the retirement home, and 32% of PM<sub>10</sub>, 76% of PM<sub>2.5</sub>, and 83% of PM<sub>1</sub> in the school dormitory originated from outdoor environment.

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### 1. Introduction

Numerous studies have shown that exposure to atmospheric particulate matter (PM) is associated with adverse health impacts

such as heart and respiratory diseases (Pope and Dockery, 2006). The percentage of particle deposition in different regions of the respiratory system directly depends upon particle size (Arhami et al., 2010). Coarse particles (PM<sub>10</sub>) have been associated with respiratory hospital admissions (Brunekreef and Forsberg, 2005). Fine PM (PM<sub>2.5</sub>) has been more strongly correlated with cardiovascular and respiratory effects (Bell, 2012). Ultrafine particles (PM<sub>0.1</sub>) can penetrate deep into the alveolar regions of the lungs (Sioutas et al., 2005). Along with physical properties (e.g. size and mass), chemical composition (e.g. organic carbon, elemental carbon, sulfate and nitrate salts, metals) also affects the toxicity of PM (Bell, 2012).

Indoor air pollution originating from outdoor air pollution is an important global risk factor requiring increasing efforts in the field of research and policy-making (Bruce et al., 2000). People spend more than 80–90% of their time in indoor environments (Simoni et al., 2003); therefore, the existence of indoor PM, even at low levels, may have considerable adverse health effects, mainly because of long exposure times. Understanding of the source, behavior, and properties of indoor PM is of essential importance for exposure estimation and emission control (Polidori et al., 2007). Overall, indoor PM comprises PM originating from outdoor environment that has infiltrated into the indoor environment, PM emitted directly from indoor sources (primary PM), and PM formed indoors (secondary PM) from precursors emitted from indoor and/or outdoor sources (Meng et al., 2004; Weschler, 2004).

High concentrations of indoor PM in schools have been reported because of inadequate ventilation, irregular housekeeping, disproportionate number of pupils with respect to the area and the volume of the room, and extreme physical activity leading to the resuspension of PM (Fromme et al., 2008; Polednik, 2012). Since the emission sources of indoor PM are different from those of outdoor PM, its composition and health effects may also vary (Schwarze et al., 2006). Therefore, the emission sources, size distribution, and chemical characteristics of indoor PM need to be explored.

Although there are a number of studies investigating indoor/outdoor PM mass concentrations and the ionic composition of different PM size fractions in schools (Braniš et al., 2005; Buonanno et al., 2013; Fromme et al., 2008; Pegas et al., 2012) and a minor but significant number of studies in retirement homes (Devlin et al., 2003; Polidori et al., 2007; Rodes et al., 2001), none of them explored simultaneously the PM mass concentration and the ionic composition of three different PM sizes (i.e. PM<sub>10</sub>, PM<sub>2.5</sub>, and PM<sub>1</sub>).

The present paper was carried out as part of the Health Effects of Air Pollution Panel Study (HEAPPS) to investigate the impacts of indoor and outdoor exposures to PM on the level of biomarkers (inflammation and coagulation markers) in two groups of healthy young (15–18 years) and elderly (>65 years) subjects. The main objectives of present paper were: (a) to evaluate indoor/outdoor relationships and trends of PM<sub>10</sub>, PM<sub>2.5</sub>, and PM<sub>1</sub> mass concentrations; (b) to characterize water-soluble ions of indoor/outdoor PM; and (c) characterize indoor/outdoor sources of indoor PM<sub>10</sub>, PM<sub>2.5</sub>, and PM<sub>1</sub> in a retirement home and a school dormitory in Tehran, the capital of Iran. The results demonstrated in this paper will be used by HEAPPS researcher to investigate the associations between the levels of indoor and outdoor PM<sub>10</sub>, PM<sub>2.5</sub>, and PM<sub>1</sub> and inflammation and coagulation markers in the elderly and healthy young subjects.

## 2. Material and methods

### 2.1. Study area, and sampling sites and schedule

Indoor and outdoor PM<sub>10</sub>, PM<sub>2.5</sub>, and PM<sub>1</sub> mass concentrations and their water-soluble ions were evaluated in a retirement home

(Tohid retirement home) and a school dormitory (Hejrati) in the center of Tehran, Iran (Fig. S1 in Supplementary material). The retirement home was located in a residential area, approximately 25 m away from a major street and 650 m away from a major freeway. The school dormitory was approximately 200 m away from a major freeway, and 40 m away from a major street, and 1.1 km away from retirement home. More details about each sampling location are given in Table 1.

Six-day samplings were performed twice a month at each location; in the retirement home, PM samplings were conducted during the first and the third consecutive 6 days of each month, while those in the school dormitory were carried out during the second and the fourth consecutive 6 days of the month. Sampling schedule was implemented from May 14, 2012 through January 19, 2013 in the school dormitory and from May 21, 2012 to January 26, 2013 in the retirement home. Thus, 108 and 96 PM samples (24-h samples) were collected in the retirement home and in the school dormitory, respectively.

Two identical indoor and outdoor sampling stations were setup at each site. The indoor sampling station at the retirement home was installed in a common room of the building that was used for dining, resting, and watching TV. It is noteworthy that the elderly spent more than 90% of their time there. The indoor sampling site in the school dormitory was located in a multipurpose room that was used by pupils aged 15–18 years mainly for sleeping, resting, studying, and occasionally as a classroom. The room was equipped with 40 beds and one whiteboard. Except for ten days in each season when pupils were on school holidays, the room was occupied all year long. The indoor air samplers were placed 130 cm above the ground in the middle of the rooms at least 2 m away from the windows, doors, and ventilation inlets and more than 1 m away from the walls. Outdoor instruments for ambient air sampling were installed at the rooftop of the buildings at the height of approximately 5 m above the ground.

Temperature and relative humidity both indoors and outdoors were measured by meteorological sensors. Wind speed, wind direction, visibility, and precipitation data were obtained from a local meteorological monitoring station (Mehrabad). The average indoor temperature in both sampling sites was almost equal and indoor was 6 °C warmer than outdoors. The average indoor relative humidity at the retirement home was higher than that for the school dormitory; additionally, outdoor levels were higher than indoor values (Table S1 in Supplementary material).

### 2.2. PM measurement

Simultaneous indoor and outdoor PM (PM<sub>10</sub>, PM<sub>2.5</sub>, and PM<sub>1</sub>) concentrations were measured by portable GRIMM dust monitors-

**Table 1**  
Main characteristics of the retirement home and the school dormitory.

Parameter	Retirement home	School dormitory
Age of the building (year)	14	25
Area	Urban, residential	Urban, residential
Floor area (m <sup>2</sup> )	500	200
Floor type	Tile without carpeting	Tile without carpeting
Mode of ventilation	Natural (window opening), water cooler	Natural (window opening), water cooler
Heating system	Hot water radiators equipped with natural gas heaters	Electric stove
Windows type	Undamaged PVC	Undamaged metal
Number of occupants	73	42
Indoor smoking	NO	NO
Cooking	NO	NO

models 1.107 for outdoor and 1.108 for indoor (GRIMM Aerosol Technik GmbH, Ainring, Germany). The GRIMM dust monitors operate at a flow rate  $1.2 \text{ l min}^{-1}$ . In addition, 24-h PM samples were simultaneously collected both indoors and outdoors on PTFE filter papers ( $47 \mu\text{m}$  dia.,  $0.5 \mu\text{m}$  pore size, from SKC) using two identical low-volume air samplers (FRM OMNI™ Air Sampler, multi-cut inlet; BGI, Inc., USA) operating at a flow rate of  $5 \text{ l min}^{-1}$ . Filters were exposed for 24 h for six consecutive days at each site. All PM filters were conditioned in a desiccator for 48 h before and after the sampling. They were then weighed three times by a microbalance (Mettler-Toledo; weight uncertainty  $\pm 2 \mu\text{g}$ ) and the average weight was recorded. The air inlets of the GRIMM dust monitors were located at least 1 m apart from those of the low-volume air samplers, both indoors and outdoors. The two types of samplers showed statistically comparable results (for all PM size fractions,  $r > 0.9$ ,  $p < 0.0001$ ) (Fig. S2 in Supplementary material). PM data measured by GRIMM dust monitors were corrected by applying related correction factors obtained from the comparison of real-time and gravimetric measurements.

During the study period, a dust storm that had most likely originated from the Middle East entered Tehran on May 25, 2012 (dusty day). In that day, therefore, we performed additional sampling and measured indoor and outdoor PM concentrations and compared the results to a non-dust storm day (i.e. June 1, 2012). Indoor/outdoor PM mass concentrations were measured only in the retirement home, because the school dormitory was closed on that day.

In the present study, dust storm was defined according to the classification presented by Hoffmann et al. (2008).

### 2.3. Chemical analysis

Three-day PM filters were composited to analyze their water-soluble ions (sodium, ammonium, potassium, magnesium, calcium, chloride, nitrite, nitrate, and sulfate). For this purpose, out of 108 and 96 daily PM samples, we analyzed 36 and 32 composited filters for their ionic content in the retirement home and in the school dormitory, respectively. Composited PM filters were put in a glass vial and then 15 ml of ultrapure water (specific resistance  $\geq 18 \Omega \text{ cm}$ ) was added to it. The vial was then shaken for at least 60 min and subsequently ultrasonicated for 30 min to complete extraction. Afterward, the extracted solution was filtered through a microporous membrane (pore size,  $0.45 \mu\text{m}$ ; diameter, 25 mm), and the filtrate was stored in plastic vials at  $4^\circ\text{C}$  until chemical analysis. An ion chromatography instrument (Metrohm 850 Professional IC, Switzerland) operating at a flow rate of  $0.7 \text{ ml min}^{-1}$  was used to analyze the inorganic ions. Field and laboratory blanks and spiked samples analyzed along with the composite PM samples for determining method limit of detection (LoD) and recovery efficiencies of water-soluble ions (Table S2 in Supplementary material).

### 2.4. Data analysis

The real-time data collected by the GRIMM dust monitors were logged at 1-min means. Hourly and 24-h averages were calculated from the 1-min data. Simple linear regression was applied to determine the relationship between (i) PM data from GRIMM dust monitors and those from the gravimetric method, (ii) indoor and outdoor PM mass concentrations, and (iii) indoor and outdoor inorganic ions. Differences between groups were tested for significance using the Mann–Whitney  $U$  test.

To apportion indoor PM among indoor and outdoor sources, we used a mass balance approach, which has been previously applied by several studies (Hoek et al., 2008). Overall, indoor PM

concentration originates mainly from PM released from indoor sources and outdoor PM infiltrating into the indoor environment. Therefore, the following equation can be used:

$$C_{\text{in}}^{\text{PM}} = C_{\text{ing}}^{\text{PM}} + (F_{\text{inf}}^{\text{PM}} \times C_{\text{out}}^{\text{PM}}) \quad (1)$$

where,  $C_{\text{in}}^{\text{PM}}$  and  $C_{\text{out}}^{\text{PM}}$  are the total concentration of indoor and outdoor PM ( $\mu\text{g m}^{-3}$ ), respectively;  $C_{\text{ing}}^{\text{PM}}$  is the PM generated indoors ( $\mu\text{g m}^{-3}$ ); and  $F_{\text{inf}}^{\text{PM}}$  is the infiltration factor for PM. A few studies have demonstrated that the infiltration factor for PM can be calculated from the infiltration factor for sulfate and a correction factor (Fromme et al., 2008). Assuming that there is no sulfate source in the indoor, indoor/outdoor concentration ratio of sulfate is equal to the infiltration factor for sulfate. Hence, the following equation can be used to estimate the indoor-generated PM:

$$C_{\text{ing}}^{\text{PM}} = C_{\text{in}}^{\text{PM}} - \left[ \frac{\beta^{\text{PM}}}{\beta^{\text{sulf}}} \times \frac{C_{\text{in}}^{\text{sulf}}}{C_{\text{out}}^{\text{sulf}}} \times C_{\text{out}}^{\text{PM}} \right] \quad (2)$$

where,  $\beta^{\text{PM}}$  and  $\beta^{\text{sulf}}$  are linear relationships of increase of indoor PM per outdoor PM and increase of indoor sulfate per outdoor sulfate, respectively;  $C_{\text{in}}^{\text{sulf}}$  and  $C_{\text{out}}^{\text{sulf}}$  are indoor and outdoor particulate sulfate concentrations. According to the above equation, we calculated the infiltration of outdoor  $\text{PM}_{10}$ ,  $\text{PM}_{2.5}$ , and  $\text{PM}_1$  into the retirement home and the school dormitory using linear regression analysis.

## 3. Results and discussion

### 3.1. Total PM mass concentrations and temporal trends

Summary of statistics for the indoor and outdoor PM mass concentrations in the sampling locations are given in Table 2.

Maximum daily indoor and outdoor  $\text{PM}_{10}$  concentrations in the retirement home reached  $240.1$  and  $360.0 \mu\text{g m}^{-3}$ , respectively. These high levels of  $\text{PM}_{10}$  were measured during a dusty day which occurred on May 25, 2012 that had most likely originated from the Middle East (Givhchi et al., 2013). The highest 24-h outdoor  $\text{PM}_{2.5}$  and  $\text{PM}_1$  concentrations in the school dormitory were  $118.0$  and  $108.3 \mu\text{g m}^{-3}$ , respectively. The maximum daily indoor  $\text{PM}_{2.5}$  concentration in the retirement home was  $72.3 \mu\text{g m}^{-3}$ , while that of  $\text{PM}_1$  was found to be  $42.0 \mu\text{g m}^{-3}$  which was equally observed at both monitoring sites.

High levels of airborne PM in the schools environment have been reported in numerous studies (Daisey et al., 2003; Lee and Chang, 2000). Many authors report that indoor concentrations of PM in schools were higher than those observed outdoors; they have also supported the fact that increased indoor PM concentrations are related to the pupils' activity (Braniš et al., 2005; Fromme et al., 2008; Heudorf et al., 2009). For retirement homes, our findings concerning higher PM concentrations outdoors than those indoors are in good agreement with the literature (Holguín et al., 2003; Polidori et al., 2007).

Time series of daily average indoor and outdoor PM mass concentrations during the sampling period are presented in Fig. 1. During the study period, regardless of the dusty day, steady trends were observed for indoor and outdoor  $\text{PM}_{10}$  concentrations in the retirement home. In addition, outdoor  $\text{PM}_{10}$  concentrations were always higher than those observed indoors. The same pattern was observed for outdoor  $\text{PM}_{10}$  concentrations measured in the school dormitory, but indoor  $\text{PM}_{10}$  concentrations increased during the cold season (in this study, October to January). In the cold season, as presented in Fig. 1, indoor  $\text{PM}_{10}$  levels in the school dormitory were higher than those outdoors. This can most likely be attributed to

**Table 2**  
Descriptive statistics for indoor (in) and outdoor (out) PM<sub>10</sub>, PM<sub>2.5</sub>, and PM<sub>1</sub> mass concentrations (in  $\mu\text{g m}^{-3}$ ) and indoor-to-outdoor ratios (I/O) in the retirement home and the school dormitory from May 2012 to January 2013.

Monitoring site	No. of samples <sup>a</sup>		PM <sub>10</sub>		PM <sub>2.5</sub>		PM <sub>1</sub>	
			Mean $\pm$ SD	Min–max	Mean $\pm$ SD	Min–max	Mean $\pm$ SD	Min–max
Retirement home	108	In	39.7 $\pm$ 25	7.5–240.1	16.9 $\pm$ 10	4.4–72.3	10.7 $\pm$ 7	3.1–42.0
		Out	85.6 $\pm$ 37	17.4–360.0	30.1 $\pm$ 16	10.9–92.9	21.6 $\pm$ 15	7.1–82.6
		I/O	0.46 $\pm$ 0.12	0.18–0.80	0.56 $\pm$ 0.12	0.20–0.99	0.50 $\pm$ 0.11	0.16–0.98
School dormitory	96	In	53.7 $\pm$ 35	8.5–145.0	19.0 $\pm$ 11	5.0–52.7	11.0 $\pm$ 8	3.0–42.0
		Out	81.7 $\pm$ 29	25.9–162.0	33.6 $\pm$ 20	12.7–118.0	25.4 $\pm$ 19	7.7–108.3
		I/O	0.66 $\pm$ 0.37	0.16–1.94	0.56 $\pm$ 0.16	0.34–1.00	0.43 $\pm$ 0.11	0.3–0.98

<sup>a</sup> 24-h Samples.

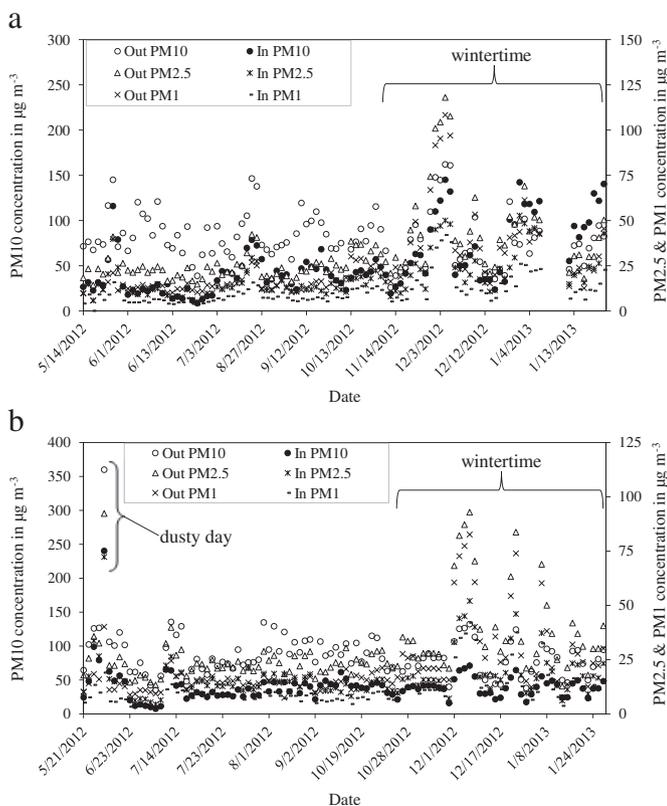
the resuspension of coarse particles due to the indoor activity of pupils who spend more time indoors in cold seasons than in warm seasons (May to September). The levels of finer particulate size fractions (i.e. PM<sub>2.5</sub> and PM<sub>1</sub>), which originate mainly from traffic-related combustion processes, increased in the cold season. The highest daily mass concentrations of PM<sub>2.5</sub> and PM<sub>1</sub> were detected in December. During wintertime, thermal inversion is the dominant phenomenon that increases PM pollution in Tehran. Because of the close proximity of the two sampling locations, similar patterns were observed for PM<sub>2.5</sub> and PM<sub>1</sub> in both monitoring sites. Although the mean outdoor PM<sub>10</sub> mass concentrations in both sampling sites were approximately equal, the mean indoor PM<sub>10</sub> mass concentration in the school dormitory was approximately 1.35 times higher than that in the retirement home. This difference may be due to the higher level of activity in the school dormitory than in the retirement home that may have increased the coarse particles. Our study, thus, adds to the previous studies the fact that

high level of physical activity in indoor increases the level of coarse particles possibly to levels higher than those found outdoors.

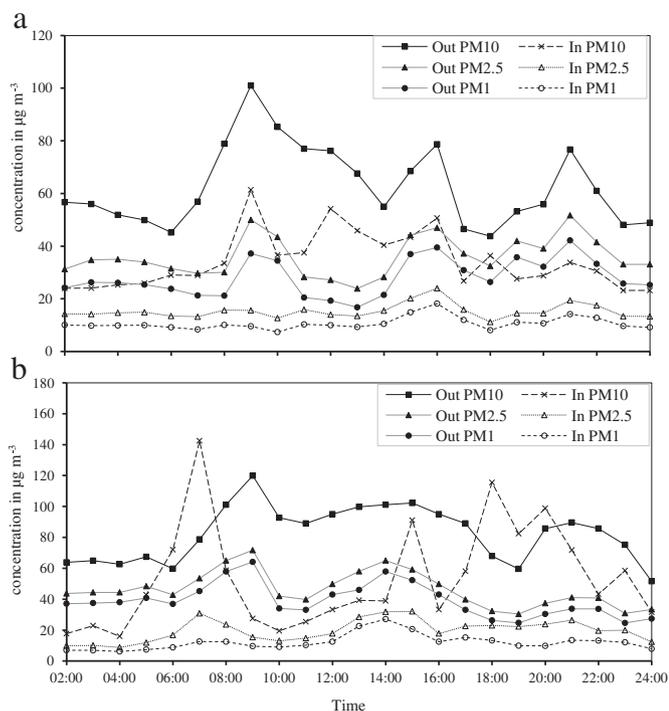
The Iranian national PM standards are the same as WHO guidelines, which are 50 and 25  $\mu\text{g m}^{-3}$  for daily average PM<sub>10</sub> and PM<sub>2.5</sub> concentrations, respectively. Currently, no guidelines or standard levels have been proposed for PM<sub>1</sub> levels. As shown in Fig. 1, in the retirement home, the diurnal mean concentration of outdoor PM<sub>10</sub> and PM<sub>2.5</sub> exceeded WHO guidelines in 90% and 50% of the sampled days, respectively. The 24-h average indoor PM<sub>10</sub> and PM<sub>2.5</sub> concentrations also exceeded in 15% and 11% of the sampled days, respectively. In the school dormitory, daily mean outdoor PM<sub>10</sub> and PM<sub>2.5</sub> concentrations exceeded WHO guidelines in 84% and 54% of the sampled days, respectively, while the corresponding exceedances for indoor concentrations were 38% and 21%.

### 3.2. Diurnal variations of indoor and outdoor PM concentrations

Fig. 2 shows the diurnal variations of indoor and outdoor PM<sub>10</sub>, PM<sub>2.5</sub>, and PM<sub>1</sub> concentrations in both the retirement home and the school dormitory on typical sampling days (the average concentration of all the sampling days excluding the dusty day). As shown in this figure, in the retirement home, diurnal variations of indoor and outdoor PM<sub>10</sub> are fairly similar ( $r = 0.67$ ). It is also shown that the patterns of indoor and outdoor PM<sub>2.5</sub> ( $r = 0.78$ ) and PM<sub>1</sub> ( $r = 0.80$ ) were arguably similar. In the school dormitory, identical patterns were observed for indoor and outdoor PM<sub>2.5</sub> ( $r = 0.70$ ) and PM<sub>1</sub> ( $r = 0.75$ ), which were approximately governed by the pattern of road traffic in Tehran. These results show that outdoor PM sources may have had an important impact on the indoor PM levels. However, different patterns were observed for indoor and outdoor PM<sub>10</sub> concentrations in the school dormitory ( $r = 0.20$ ). Most of the fluctuations took place for indoor PM<sub>10</sub> concentrations in the school dormitory. This could result from varied levels of activity by pupils and the changes in the rate of infiltrated air due to more uncontrolled opening of doors and windows (Polednik, 2012). Therefore, our study suggests that when PM<sub>2.5</sub> and PM<sub>1</sub> are measured outdoors, variations of indoor PM<sub>2.5</sub> and PM<sub>1</sub> follow those of outdoor air. Maximum hourly outdoor PM levels in both the retirement home and the school dormitory were mostly recorded from 8 to 10 in the morning. The highest values of indoor PM<sub>10</sub> concentrations in the school dormitory were observed from 6 a.m. to 8 a.m., when the pupils wake up; interestingly, indoor PM<sub>10</sub> levels during this time period were higher than outdoor PM<sub>10</sub> levels. This is probably due to the peak activity of pupils during this time of the day, including wearing uniforms, cleaning shoes and the floor, shaking clothes and blankets, and doing some exercise. Increased levels of PM at the beginning and the end of classes were also reported by other studies (Polednik, 2012). The results showed fairly similar patterns between the two sampling sites for outdoor PM<sub>10</sub> ( $r = 0.65$ ), PM<sub>2.5</sub> ( $r = 0.84$ ), and PM<sub>1</sub> ( $r = 0.86$ ). Similarity of patterns in both



**Fig. 1.** Time series for diurnal averages of indoor and outdoor PM mass concentrations in the school dormitory (a), and the retirement home (b).



**Fig. 2.** Typical diurnal variations of indoor (in) and outdoor (out) PM<sub>10</sub>, PM<sub>2.5</sub>, and PM<sub>1</sub> concentrations in the retirement home (a) and the school dormitory (b).

sampling sites could be attributed to the close proximity of the two sampling sites. It also implies that they are affected by same major sources. The lower correlation coefficient for PM<sub>10</sub> can be attributed to the fact that in both sampling sites, this size fraction was also affected by local sources such as construction and demolition.

### 3.3. Impact of dust storm on indoor and outdoor PM mass concentrations

Summary statistics for daily mean indoor and outdoor PM mass concentrations during dusty day and normal day obtained from hourly averages are shown in Table 3. During the dusty day, the highest hourly mean outdoor PM<sub>10</sub>, PM<sub>2.5</sub>, and PM<sub>1</sub> concentrations reached 514, 134, and 56  $\mu\text{g m}^{-3}$ , respectively; the maximum hourly indoor values were also elevated to 364, 99, and 26  $\mu\text{g m}^{-3}$  for PM<sub>10</sub>, PM<sub>2.5</sub>, and PM<sub>1</sub>, respectively.

Therefore, our study indicated that daily mean indoor and outdoor PM<sub>10</sub> concentrations during the dust storm were 3.4 times higher than those observed in normal day. Indoor PM<sub>2.5</sub> and PM<sub>1</sub> concentrations increased by 2.8 and 1.6 times, while outdoor PM<sub>2.5</sub> and PM<sub>1</sub> concentrations rose by 2.9 and 1.9 times, respectively.

**Table 3**  
Descriptive statistics of daily indoor (in) and outdoor (out) PM mass concentrations during a dusty day and a normal day (in  $\mu\text{g m}^{-3}$ ).<sup>a</sup>

	Dusty day		Normal day		Dusty day/normal day	
	In	Out	In	Out	In	Out
	Mean $\pm$ SD (min–max)	Mean $\pm$ SD (min–max)	Mean $\pm$ SD (min–max)	Mean $\pm$ SD (min–max)	Mean $\pm$ SD (min–max)	Mean $\pm$ SD (min–max)
PM <sub>10</sub>	240 $\pm$ 86.7 (87–364)	360 $\pm$ 82.7 (167–514)	71 $\pm$ 37.8 (23–165)	106 $\pm$ 35.6 (53–170)	3.4 $\pm$ 2.8 (1.9–12.8)	3.4 $\pm$ 1.8 (2.2–7.8)
PM <sub>2.5</sub>	72 $\pm$ 18.5 (34–99)	92 $\pm$ 20.5 (44–134)	26 $\pm$ 14.8 (11–65)	32 $\pm$ 8.2 (20–68)	2.8 $\pm$ 1.73 (0.99–7.8)	2.9 $\pm$ 1.07 (1.6–5.8)
PM <sub>1</sub>	19 $\pm$ 4.5 (10–26)	40 $\pm$ 8.2 (21–56)	12 $\pm$ 3.4 (6–18)	21 $\pm$ 4.6 (13–29)	1.6 $\pm$ 0.64 (0.78–3.3)	1.9 $\pm$ 0.61 (1.2–3.5)

<sup>a</sup> Atmospheric visibility and wind speed on a dusty day (May 25, 2012): 1.9 km and 8.8  $\text{m s}^{-1}$ , respectively; atmospheric visibility and wind speed on a normal day (June 1, 2012): 10 km and 4.7  $\text{m s}^{-1}$ , respectively.

These results revealed that coarse particles were more affected by the dust storm than were finer size fractions, which is consistent with the findings observed by previous studies (Jayaratne et al., 2011).

### 3.4. Ionic composition of PM

Descriptive statistics for water-soluble ions in indoor and outdoor PM at the two sampling sites can be found in Table S3 from Supplementary material. Nitrite levels were below LoD in all PM size fractions at the two sampling sites. Levels of potassium and chloride were below LoD in indoor and outdoor PM<sub>1</sub>; sodium levels were only above the LoD for the outdoor PM<sub>1</sub> in the school dormitory (Table S3). Of all the ions detected, nitrate was the dominant constituent of atmospheric particulate matters which was followed closely by sulfate. Particulate nitrate is formed through the photo-oxidation of nitrogen dioxide emitted from combustion processes (Ho et al., 2003). Sulfate aerosols are formed through the heterogenous or homogenous reactions of sulfur dioxide (Cheng et al., 2000). Contribution of ions to the total water-soluble ions and mass concentrations of indoor and outdoor PM<sub>10</sub>, PM<sub>2.5</sub>, and PM<sub>1</sub> for each sampling site are given in Fig. S3 (from Supplementary material) and Fig. 3, respectively. Regardless from indoor or outdoor and sampling site, secondary inorganic aerosols (sum of sulfate, nitrate, and ammonium) were the main water-soluble ions, and their percentage increased with decreasing PM size. In fact, they accounted for about 60–80% of the mass of water-soluble ions in PM<sub>10</sub> (5–20% of PM<sub>10</sub> mass), 75–85% in PM<sub>2.5</sub> (20–40% of PM<sub>2.5</sub> mass), and more than 90% in PM<sub>1</sub> (25–45% of PM<sub>1</sub> mass). Our study, thus, adds the evidence that the smaller the particle, the higher the percentage of secondary inorganic aerosols. This study also suggests that sea-salt species (sodium and chloride) and crustal elements (potassium, magnesium, and calcium) are mainly found in coarse particles. High levels of secondary inorganic aerosols indicate a high degree of anthropogenic pollution in this area. They are most likely formed secondarily from NO<sub>x</sub> and SO<sub>2</sub> gases. It is noticeable that the high values of sodium, potassium, magnesium, and calcium detected for indoor PM<sub>10</sub> in the school dormitory may indicate resuspension of soil deposited on the floor and other surfaces such as beds, benches, and shoes. Our results with respect to the concentrations of water-soluble ions in outdoor PM<sub>10</sub> nearly concur with those from other studies (Nicolas et al., 2009; Pey et al., 2009). In addition, except for nitrate, similar ionic concentrations were observed for outdoor PM<sub>2.5</sub> in the Western Mediterranean (Pey et al., 2009).

Using Na<sup>+</sup> as a tracer of sea salt, excess sulfate (non-sea-salt sulfates (nss-SO<sub>4</sub><sup>2-</sup>) can be estimated  $\text{nss-SO}_4^{2-} = [\text{SO}_4^{2-}] - [\text{Na}^+] \times 0.2516$  (Ho et al., 2003). According to this estimation, regardless of indoor or outdoor air and sampling site, the non-sea-salt sulfates comprised more than 80%, 92%, and 97% of the total sulfate in PM<sub>10</sub>, PM<sub>2.5</sub> and PM<sub>1</sub>, respectively, indicating a substantial anthropogenic origin.

Previous studies showed that the mass ratio of nitrate/sulfate is a good indicator of the relative importance of stationary and mobile sources of nitrogen and sulfur in the atmosphere (Wang et al., 2005). It is reasonable to use nitrate as an indicator of mobile emission and sulfate as that of stationary emission. Regardless of indoor or outdoor air and sampling site, the mean value of nitrate/sulfate ratio during the study ranged from 1.17 to 1.21 for  $PM_{10}$ , 1.16 to 1.40 for  $PM_{2.5}$ , and 1.23 to 1.45 for  $PM_1$ . The results indicated that mobile sources were the most important source of PM in Tehran.

### 3.5. Relationship between indoor and outdoor PM

As can be seen from Table 2 and Fig. 2, in the retirement home, the I/O ratio was below 1 for all PM size fractions. In the school dormitory, although this ratio was always below 1 for  $PM_{2.5}$  and  $PM_1$ , in some occasions it surpassed 1 for  $PM_{10}$ , which implies the existence of an important indoor  $PM_{10}$  emitting source. Results of regression analysis for indoor and outdoor PM and particulate sulfate on typical sampling days in the two monitoring sites are given in Table 4. Correlations for  $PM_{10}$  were lower than those for  $PM_{2.5}$  and  $PM_1$  in the two sampling sites. This is probably because of higher infiltration rate of smaller size fractions, lower losses during infiltration of smaller size fractions, or more indoor  $PM_{10}$  sources. Poor correlation between indoor and outdoor  $PM_{10}$  in the school dormitory indicated that a major fraction of indoor  $PM_{10}$  had originated from indoor sources. Since no major source of particles existed in the dormitory, pupils' activity was most likely the main factor for elevated indoor  $PM_{10}$  levels. Contrary to coarse particles, in the school dormitory, good correlation were observed between

**Table 4**

Regression analysis of indoor and outdoor PM mass concentrations and particulate sulfate on typical sampling days in the two sampling locations.

	Retirement home				School dormitory			
	$n^a$	$r^b$	$\alpha^c$	$\beta^d$	$n^a$	$r^b$	$\alpha^c$	$\beta^d$
$PM_{10}$	107	0.78	10.23	0.40	96	0.51	15.53	0.61
$PM_{2.5}$	107	0.92	3.71	0.54	96	0.85	3.10	0.50
$PM_1$	107	0.95	1.70	0.48	96	0.95	1.55	0.38
Sulfate- $PM_{10}$	36	0.90	0.09	0.65	32	0.87	0.08	0.53
Sulfate- $PM_{2.5}$	36	0.93	0.06	0.51	32	0.89	0.05	0.31
Sulfate- $PM_1$	36	0.94	0.04	0.42	32	0.92	0.04	0.29

All intercepts were statically significant ( $p < 0.05$ ).

<sup>a</sup> Number of 24-h samples.

<sup>b</sup> Pearson correlation coefficient.

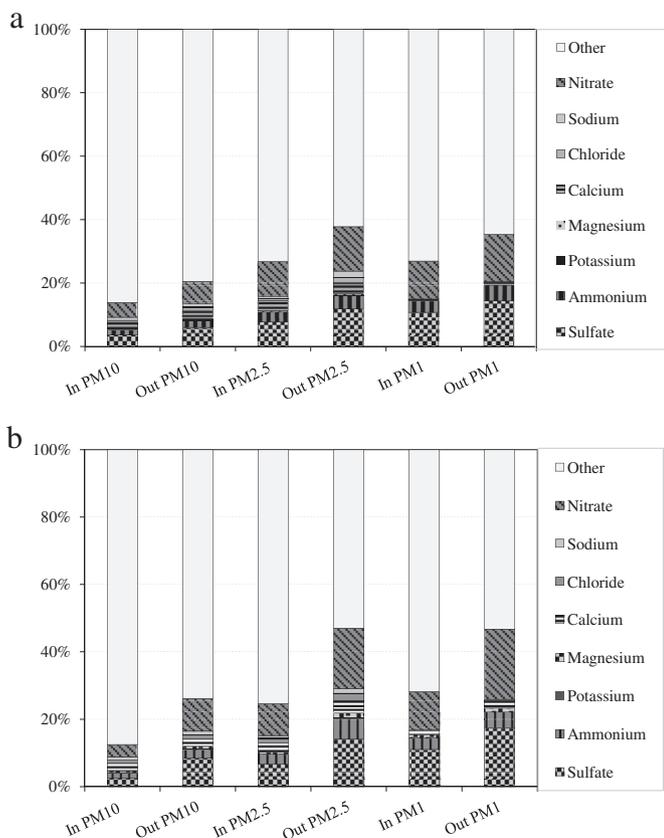
<sup>c</sup> Intercept ( $\mu\text{g m}^{-3}$ ).

<sup>d</sup> Line slope.

indoor and outdoor  $PM_{2.5}$  and  $PM_1$ , which can be explained by the influence of ambient air on indoor  $PM_{2.5}$  and  $PM_1$ . Furthermore, in the retirement home, a fair association was found between indoor and outdoor  $PM_{10}$ ,  $PM_{2.5}$ , and  $PM_1$ , suggesting that the infiltrated outdoor PM was the main source of indoor PM, especially for  $PM_{2.5}$  and  $PM_1$ . Good correlations were also found between indoor and outdoor particulate sulfate ( $r = 0.87$ – $0.94$ ), indicating that the levels of indoor particulate sulfate were mainly affected by outdoor particulate sulfate. It was also found that the concentrations other water-soluble ions had varied indoor/outdoor correlations. For example, a strong correlation was recorded for nitrate in  $PM_1$  ( $r = 0.87$ ,  $p < 0.01$ ), while a weak correlation was observed for potassium in  $PM_{10}$  ( $r = 0.28$ ,  $p < 0.01$ ).

During the dusty day I/O ratios were  $0.67 \pm 0.16$  for  $PM_{10}$ ,  $0.78 \pm 0.09$  for  $PM_{2.5}$ , and  $0.47 \pm 0.05$  for  $PM_1$ . Results indicated that the I/O ratios during the dusty day were virtually equal to those of the normal day (Table 2), which is mainly because a dust storm almost equally affects both indoor and outdoor environments; except  $PM_1$ , for which the outdoor concentrations increased slightly more than the indoor concentrations during the dust storm (1.9 times vs. 1.6 times), which led to a slightly higher I/O ratio during the dust storm. Results of regression analysis between hourly indoor and outdoor PM on the dusty day showed a relatively good correlation for  $PM_{10}$  ( $r = 0.77$ ), and very good ones for  $PM_{2.5}$  ( $r = 0.90$ ) and  $PM_1$  (0.93). Results also indicated that the correlation of indoor/outdoor PM improved as the size of the particle decreased. Good correlation between indoor and outdoor PM during the dusty day indicated that a major fraction of indoor PM had originated from outdoor PM sources. This can be due to the fact that the dust storm occurred in a warm season when windows and doors are often open, which in turn increases the outdoor infiltration rate. Additionally, most Iranians tend to use water cooler, which allows the outdoor PM to easily transport to the indoor environment.

Table 5 presents I/O ratios for water-soluble ions in different PM size fractions measured in the two sampling sites; note that these values are adjusted for PM mass concentrations (i.e. water-soluble ions mass concentration divided by the PM mass concentration) (Fromme et al., 2008). For almost all the ions, the I/O was lower than the unity, except for calcium in  $PM_{10}$  measured in the school dormitory, for which an I/O ratio of 1.1 was found, probably because of indoor calcium sources. Our results regarding I/O ratios for sulfate, sodium, and magnesium in  $PM_{10}$  and  $PM_{2.5}$  as well as that of calcium in  $PM_{10}$  measured in the school dormitory are very similar to findings of the study of Fromme et al. (2007), who measured the ionic composition of different PM size fractions in classrooms in Munich, Germany. High I/O ratios for sulfate and nitrate in school environments have also been reported by other studies (John et al., 2007; Pegas et al., 2012).



**Fig. 3.** Mass balance of indoor and outdoor PM fractions in the retirement home (a), and the school dormitory (b).

**Table 5**  
Mean I/O ratios of water-soluble ions (adjusted for PM mass concentrations).

	Retirement home			School dormitory		
	PM <sub>10</sub>	PM <sub>2.5</sub>	PM <sub>1</sub>	PM <sub>10</sub>	PM <sub>2.5</sub>	PM <sub>1</sub>
Sodium	0.9	0.3		0.9	0.6	0.7
Ammonium	0.7	0.7	0.8	0.6	0.5	0.7
Potassium	0.4			0.5	0.0	
Magnesium	0.7	0.8		0.6	0.4	0.9
Calcium	0.9	1.0	0.9	1.1	0.8	0.5
Chloride	0.6	0.4		0.8	0.6	
Nitrate	0.7	0.8	0.8	0.4	0.5	0.5
Sulfate	0.6	0.7	0.7	0.3	0.5	0.6

### 3.6. Contribution of indoor and outdoor PM sources to the total indoor PM levels

In the absence of particular sources of indoor air pollution, sulfate content of PM can be used to determine the impact of outdoor PM sources on indoor PM levels. Using Eq. (2) and linear regression analysis of PM and particulate sulfate data (Table 4), the fractions of PM<sub>10</sub>, PM<sub>2.5</sub>, and PM<sub>1</sub> generated indoors were calculated. For the retirement home, approximately 79% of PM<sub>1</sub>, 67% of PM<sub>2.5</sub>, and 45% of PM<sub>10</sub> had originated from outdoor air. For the school dormitory, we found that nearly 83% of PM<sub>1</sub>, 76% of PM<sub>2.5</sub>, and 32% of PM<sub>10</sub> in the indoor air had originated from outdoor sources. Results indicated that in both sampling sites a major fraction of PM<sub>10</sub> originated from indoor sources, which can be because of the resuspension of particles due to the personnel and visitor movement in the retirement home and the pupils activities in the school dormitory. Our results confirmed that in the micro-environments with high levels of human activity indoor PM<sub>10</sub> concentrations are mainly affected by the level of activity, whereas finer fractions of indoor PM could be significantly influenced by concentrations of ambient particles, which originate mainly from traffic-related combustion processes. Results also indicated that a small fraction of PM<sub>2.5</sub>, and particularly PM<sub>1</sub>, had been released from indoor sources; this portion may have been produced through the formation of secondary particles (formation of particles from gases) (Arhami et al., 2010), or released from photocopiers, printers, and hard copy instruments (Byeon and Kim, 2012) that existed in both sampling sites. Our study thus adds to previous studies that the PM size affects the infiltration of outdoor PM, and that there are significant indoor sources for PM<sub>10</sub>, but not for PM<sub>2.5</sub> and PM<sub>1</sub>. Regardless of the sampling site, the values obtained from Eq. (2) had a fairly good correlation with the intercept values presented in Table 4 (indicating the level of particles generated indoors) for PM<sub>2.5</sub> and PM<sub>1</sub>, but not for PM<sub>10</sub>. This is mainly because the intercept values for PM<sub>10</sub> were obtained from indoor/outdoor data that did not have a good correlation. In fact, the better the correlation between the indoor and outdoor values, the more consistent the values obtained from Eq. (2) with intercept values.

## 4. Conclusion

Our results on Indoor/outdoor relationships of PM mass concentrations and their water-soluble ions in a retirement home and a school dormitory showed that daily average indoor and outdoor PM concentrations usually exceeded the WHO guidelines and national standard levels, particularly in the school dormitory. Our findings approved that without other sources of PM (such as cooking and smoking), people's activity is a major factor accounting for elevated indoor levels of PM<sub>10</sub>, and that PM<sub>2.5</sub> and PM<sub>1</sub> could be significantly affected by outdoor PM concentrations. In fact, regardless of indoor or outdoor and sampling site, about 32–45% of PM<sub>10</sub>, 67–76% of

PM<sub>2.5</sub>, and 79–83% of PM<sub>1</sub> had originated from the outdoor environment. Higher infiltration rate may be the main reason for observing higher correlations for smaller size fractions of PM.

The results showed that secondary inorganic aerosols (sum of sulfate, nitrate, and ammonium ion) were the main water-soluble ions, and their percentage increased with the decreasing of PM size. We found that the dust storm increased the indoor and outdoor concentrations of all fractions of PM, but coarse particles were more affected by the dust storm than were finer fractions. The indoor air quality is a very specific field of study, but from the local data it is possible to get information that is valid more in general. The results presented in this paper will be applied by HEAPPS researchers to evaluate the associations between indoor and outdoor PM levels and biomarkers (inflammation and coagulation markers) in the elderly and healthy young subjects.

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## Appendix A. Supplementary data

Supplementary data related to this article can be found at <http://dx.doi.org/10.1016/j.atmosenv.2013.10.048>.

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