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Health risk assessment of heavy metal exposure from indoor dust: A case study of residential buildings in Isfahan, Iran

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Abstract
Background: Heavy metals (HMs) are toxic pollutants whose concentrations in confined spaces might cause severe health impacts. This study aimed to determine the concentration and health risk of As, Cd, Co, Pb, Mn, Ni, and V in indoor household dust in Isfahan during 2022-2023.
Methods: Ninety dust samples were collected from 30 sampling homes. After preparation and acid digestion of the samples in the laboratory, the concentrations of the elements were determined using the ICP-OES method and analyzed statistically.
Results: Except for Pb, the HMs' mean concentrations were significantly lower than the permissible limit ($P<0.050$). The maximum daily exposure through ingestion, inhalation, and dermal contact for children and adults were 66.1 and 79.1 mg/kg/d, respectively, with Pb as the relevant element in both groups. Furthermore, the maximum lifetime daily exposure doses of 8-10×26.1 mg/kg/d belonged to Pb. The maximum non-carcinogenic and carcinogenic risk values through direct ingestion, inhalation, and dermal contact were 4.83×1-10 and 1.40×8-10 for children and 5.23×2-10 and 7.91×9-10 for adults, which were associated with Pb in both groups.
Conclusion: The results showed that the HMs content in indoor household dust in Isfahan followed a decreasing trend of Pb > Mn > Ni > V > As > Co > Cd. Moreover, direct ingestion followed by dermal contact and inhalation were the most important exposure pathways to the HMs-contaminated dust for both children and adults.
Keywords: Health risk assessment, Heavy metals, Exposure, Dust
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Introduction

Heavy metals (HMs) are elements with long half-lives that are non-biodegradable and widespread in the environment (1). They have the potential to bioaccumulate and even small amounts can cause physiological effects (2). While some elements like iron (Fe), zinc (Zn), copper (Cu), and nickel (Ni) are essential for biological systems, elements such as arsenic (As), mercury (Hg), lead (Pb), cadmium (Cd), and chromium (Cr) are considered non-essential and can be toxic and hazardous to organisms in small amounts (3,4).

Exposure to elevated levels of the metalloid As has been linked to hepatic and renal damage, as well as an increased susceptibility to bladder, lung, and skin cancers (5,6). Cd exposure can result in hypertension, anemia, osteoporosis, diabetes, and cardiovascular diseases (7,8). Pb exposure is associated with gastrointestinal, neurological, and hematological disorders, cognitive impairment in children, learning disabilities, fainting, and in severe cases, death (9-11). On the other hand, exposure to excessive levels of Co poses detrimental effects on the heart and skin (12). Additionally, the uptake of the above-permissible limit of Mn is associated with the onset of Alzheimer's and Parkinson's diseases (13). Ni exposure above permissible limits can lead to genetic mutations, teratogenicity, neurological and cardiac disorders, and lung cancer (14,15). Vanadium, as a toxic metal, can bind to blood proteins and cause serious complications when individuals are exposed to its upper permissible limit (16).

Given that individuals spend approximately 90% of their time in enclosed spaces such as residential, commercial, administrative, and educational buildings, it is crucial to assess indoor environmental quality for the preservation of citizens' health and safety (17-19). This issue is particularly important for children and adolescents, who have a higher relative inhalation rate per unit of body weight (20,21). Although outdoor air quality significantly impacts indoor air quality, indoor pollutant concentrations can be higher due to sources such as gas combustion, building materials, cleaning agents, and reduced air infiltration caused by energy-saving insulations (22,23).

Particulate matter (PM) is composed of various pollutant materials with diverse particle sizes and complex chemical compositions. PMs smaller than 10 µm can penetrate deep into the lungs and cause health problems (24-26). HMs derived from mobile and stationary sources in urban environments, such as industries, traffic emissions, atmospheric deposition, and geochemical processes, pose a public health concern. After being exposed to particles, these HMs can pose significant health risks and contribute to morbidity and mortality (13,27).

Health risk assessment is a conceptual approach that provides a framework for examining and estimating information related to health or biological effects (28). It involves the identification, collection, and integration of information on health hazards resulting from human exposure to chemical substances through ingestion, inhalation, and dermal contact. Understanding the relationships between exposure, concentration, and adverse effects is crucial in health risk assessment (29).

Several studies have been conducted to determine the levels of potentially toxic elements in indoor household dust in Iran and other countries. In Khorramabad, Iran, Sabzevari and Sobhanardakani (20) discovered high levels of Pb, Cd, and Ni in dust deposited inside houses among the study cases. They concluded that indoor dust in the city could lead to adverse effects on human health. In another study, conducted by Hashemi et al (30), it was demonstrated that factors such as traffic sources, old building materials, and building paint colors resulted in the excessive accumulation of HMs in the surface dust of indoor environments in houses in Busher City, Southwest Iran. Studies such as the studies by Cheng et al (31) in Chengdu, China and Yaghi and Abdul-Wahab (32) in Muscat, Oman, indicate that the accumulation of HMs in the indoor environment of urban houses can significantly surpass permitted limits, posing substantial health threats to city dwellers. To date, however, no study has been conducted to assess the content and health risks of potentially toxic elements in indoor household dust in the major city of Isfahan, which has a population of nearly 3 000 000 inhabitants. Therefore, this study aimed to determine the content and health risk assessment of As, Pb, Cd, Co, Mn, Ni, and V in indoor household dust in the city of Isfahan in 2022, providing valuable insights for the first time.

Materials and Methods

Study area

Isfahan metropolis, with an approximate area of 300 km^2, is located at the geographical latitude of 38°32' N and longitude of 39°51' E. Evidence indicates that, on average, the residents of Isfahan are exposed to air pollution for approximately 150 days a year. Moreover, the presence of various industries has led to numerous environmental problems, particularly air pollution, in most areas of this metropolis (33,34).

Dust sampling

In this descriptive study, owing to constraints in financial resources and obtaining permissions for utilizing houses as case samples, a total of 30 residential buildings were chosen. Within each house, three dust samples were collected, resulting in a total of 90 samples. The collection process was carried out by a designated individual in all houses to mitigate sampling bias. Dust was collected during the winter of 2023, utilizing a polyethylene brush to gather the maximum amount of dust. We thoroughly assessed all surfaces and conducted sampling in cases where dust was detected. In certain situations, we encountered challenges with collecting sufficient material on the ground and in corners. Often, the collected material consisted mostly of carpet fibers and clothing lint. As a result, we redirected our focus toward higher surfaces such as the upper surfaces of kitchen cabinets, surfaces of bedroom cabinets, and wall cabinets. The samples were then transferred to the laboratory in 50 mL Falcon tubes and stored in a freezer until further analysis (6,35). The geographical coordinates of the sampling locations were recorded using a Garmin ETREX 32X GPS device, and their positions are shown in Figure 1.

Preparation of dust samples and HMs analysis

For instrumental analysis, high-purity chemicals and reagents were procured from Sigma-Aldrich Spain. In the laboratory, the collected samples from each location were combined to obtain a homogeneous sample. The samples were then sieved, and one gram of each sample was weighed using a digital balance model And-hr-200 with an accuracy of 0.0001 g. Next, each sample was thoroughly mixed with 10 mL of nitric acid in a 1:1 volumetric ratio, and the resulting solution was covered with a watch glass. The solution was heated to 90 °C and refluxed for 10 minutes. After cooling the solution to room temperature (25 °C), 5 mL of concentrated nitric acid was added, and the solution was covered again. The refluxing process was repeated for 30 minutes at 90 °C. The solution was then reduced to a volume of approximately 5 mL without boiling and allowed to cool for 5 minutes at room temperature. Subsequently, 2 mL of double-distilled water and 3 mL of 30% hydrogen peroxide were added to the solution, and the mixture was covered with

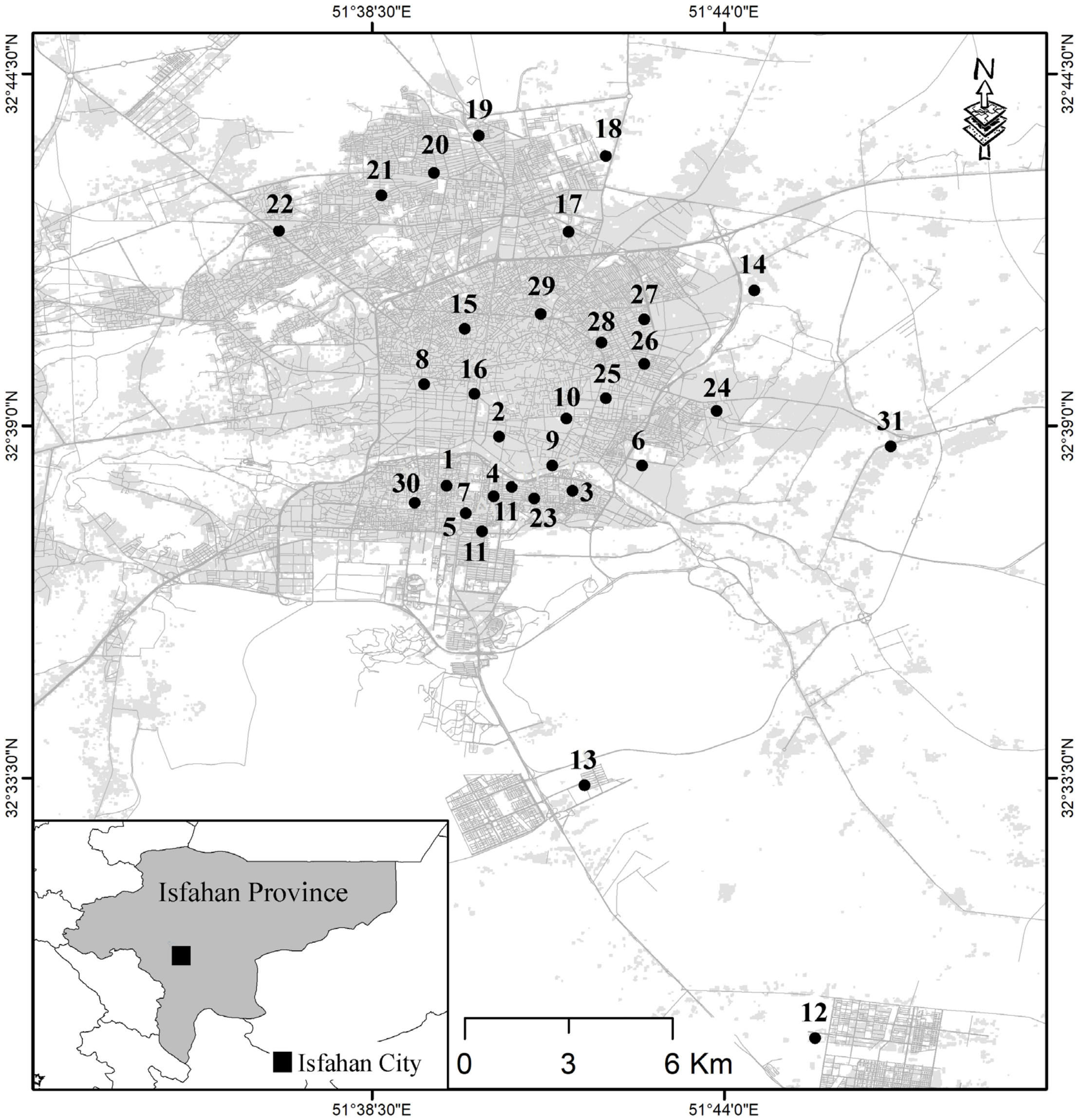

Figure 1. Sampling stations in Isfahan

a watch glass to initiate the hydrogen peroxide reaction. After the completion of the reaction, the solution was again covered with a watch glass and heated to 90 °C until it reached a volume of 5 mL. To reflux the solution for an additional 15 minutes at a temperature of 90 °C, 10 mL of concentrated hydrochloric acid was added to the solution, which was covered and placed on a hot plate. After the solution cooled down, it was filtered through a filter paper (Whatman No. 42) and brought to a volume of 100 mL in a volumetric flask using double-distilled water (13,36). Finally, after preparing standard salt solutions for the studied elements and calibrating the atomic absorption spectrophotometer device, the concentrations of As, Pb, Cd, Co, Mn, Ni, and V were measured.

To ensure quality assurance and quality control, the Lu et al method involving the standard reference materials (SQC-001) was employed (37). The results showed that the average recovery rates for As, Pb, Cd, Co, Mn, Ni, and V were variable, ranging from 91% to 100%, 89% to 101%, 94% to 102%, 90% to 99%, 93% to 101%, 88% to 99%, and 96% to 103%, respectively.

Health risk assessment model

In this study, equations 1 to 3 were used to assess the health risk of exposure to dust particles contaminated with potentially toxic elements through direct ingestion, inhalation, and dermal contact for children and adults (38-40):

$$Ding = C \times \frac{IngR \times EF \times ED}{BW \times AT} \times 10^{-6} \quad (1)$$

In equation 1, the health risk of exposure to dust particles contaminated with potentially toxic elements through direct ingestion (mg/kg/d) is represented by *Ding*, and *C* is the average concentration of the elements in the dust sample (mg/kg). *IngR* shows the ingestion rate of contaminated dust particles, which is equal to 200 mg/d for children and 100 mg/d for adults. *EF* and *ED* represent the frequency of exposure to contaminated dust particles (180 days/year for children and adults) and the duration of exposure (6 years for children and 24 years for adults), respectively. In addition, *BW* is the average body weight (15 kg for children and 70 kg for adults), and *AT* is the average exposure time to the pollutant mixture over a lifetime, which is 365 days/year multiplied by ED for non-carcinogenic effects and 365 days/year multiplied by 72 for carcinogenic effects (38,41,42).

$$Ddermal = C \times \frac{SA \times SL \times ABS \times EF \times ED}{BW \times AT} \times 10^{-6} \quad (2)$$

In equation 2, the health risk of exposure to dust particles contaminated with potentially toxic elements through inhalation (mg/kg/d) is shown by *Dinh*. *InhR* and *PEF* are the respiratory rate (6.7 m^3/day for children and 20.0 m^3/day for adults) and the soil dispersion factor (1.36 multiplied by 10^9 m3/kg), respectively (27,43).

$$Ddermal = C \times \frac{SA \times SL \times ABS \times EF \times ED}{BW \times AT} \times 10^{-6} \quad (3)$$

where Ddermal is the health risk of exposure to dust particles contaminated with potentially toxic elements through dermal contact (mg/kg/d). *SA*, *SL*, and *ABS* are the body surface area exposed to elements (2800 cm^2 for children and 5700 cm^2 for adults), skin adherence factor (0.200 mg/cm^2/day for children and 0.070 mg/cm^2/day for adults), and dermal absorption coefficient equal to 0.001, respectively (44).
The carcinogenic risk resulting from exposure to dust particles contaminated with As, Pb, Cd, Co, and Ni through inhalation was assessed using Eq. 4 (43):

$$LADD = \frac{C \times EF}{PEF \times AT} \times \left[\frac{InhRchild \times EDchild}{BWchild} + \frac{InhRadult \times EDadult}{BWadult} \right] \quad (4)$$

where *LADD* is the average daily exposure dose to dust particles contaminated throughout the lifetime (mg/kg/d).

The potential non-carcinogenic and carcinogenic risks associated with exposure to dust particles contaminated with the studied elements were calculated using Eqs. 5 to 7:

$$HQ = \frac{D}{RfD} \quad (5)$$

$$HI = \sum HQi \quad (6)$$

where *HQ* is the non-carcinogenic toxicity risk, *D* is the average daily absorbed concentration of elements resulting from exposure to dust particles contaminated through each route of absorption (mg/kg/d), and *RfD* (mg/kg) is the reference concentration of the studied elements (Table 1). Values of HI where $HI \leq 1$ indicate no risk from exposure to dust particles contaminated and $HI > 1$ indicates potential effects or risks from exposure to dust particles contaminated (45).

$$CR = D \times SF \quad (7)$$

In equation 7, *CR* and *SF* are the carcinogenic risk and slope factor for carcinogenicity through inhalation of the contaminated dust particles, respectively (42). The SF values are provided in Table 1.

Statistical analysis

The data were statistically processed using SPSS version 19. To examine the normality of the data distribution, the Kolmogorov-Smirnov test was used, and the homogeneity of variances was assessed using Levene's test. Moreover, the mean concentrations of the HMs were assessed using the one-sample t-test based on the guidelines of the World Health Organization (WHO). Descriptive statistics, including minimum, maximum, mean, standard deviation, and coefficient of variation were calculated for the HMs in the dust samples.

Results

Descriptive statistics of the HMs from indoor dust samples are presented in Figure 2. Accordingly, the concentration of As, Pb, Cd, Co, Mn, Ni, and V (mg/kg) varied from 0.20 to 4.20, 0.29 to 3762, 0.100 to 10.3, 0.901 to 50.7, 0.59 to 615, 0.11 to 0.77, and 0.009 to 0.52, respectively. Except for Pb, the significance level (*P* value) of the normality test was greater than 0.050, confirming that the values of As, Cd, Co, Mn, Ni, and V follow a normal distribution in the dust samples.

The results of the one-sample t-test comparing the mean values with the WHO reference values showed that, except for Pb, which did not have a statistically significant difference from the reference value ($P > 0.050$), the mean values of As, Cd, Co, Mn, Ni, and V had a statistically

Table 1. Reference concentrations (RfD) and carcinogenic slope factor (SF) of the studied elements (41,43,46-48)

	Element						
	As	Pb	Cd	Co	Mn	Ni	V
RfDing	$3.00*4^{-10}$	$3.50*3^{-10}$	$1.00*3^{-10}$	$2.00*2^{-10}$	$4.60*2^{-10}$	$2.00*2^{-10}$	$7.00*3^{-10}$
RfDinh	$43.0*1^{-10}$	$3.52*3^{-10}$	$1.00*3^{-10}$	$5.71*6^{-10}$	$1.43*5^{-10}$	$2.06*2^{-10}$	$7.00*3^{-10}$
RfDdermal	$1.23*4^{-10}$	$5.25*4^{-10}$	$1.00*5^{-10}$	$1.60*2^{-10}$	$1.84*3^{-10}$	$5.40*3^{-10}$	$7.00*5^{-10}$
Inhal. SF	15.1	0.009	6.30	9.80	-	0.840	-

significant difference from the reference value ($P < 0.050$) and were below the permissible limit.

The daily exposure values to HMs-contaminated indoor dust, separately for children and adults, through direct ingestion, inhalation, and dermal contact, as well as the average lifetime daily dose of exposure to particle-contaminated dust are presented in Table 2. Moreover, the values of non-carcinogenic and carcinogenic risk indices resulting from exposure to HMs-contaminated indoor dust are provided in Table 3 separately for children and adults.

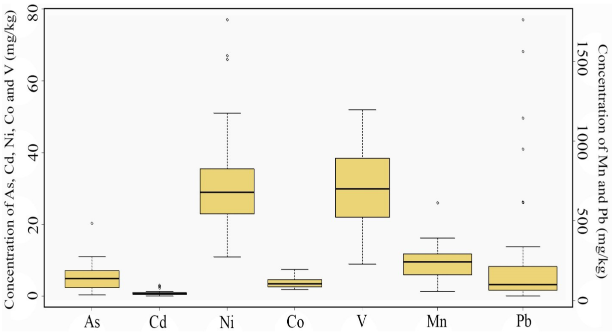

Figure 2. Descriptive statistics of the HMs concentration (mg/kg) in indoor dust samples from residential homes

Based on the results, the maximum HM exposure values through direct ingestion, inhalation, and dermal contact with contaminated dust were 1.66×3^{-10} mg/kg/d for children and 1.97×4^{-10} 79.1 mg/kg/d for adults, and were associated with Pb in both groups. The maximum lifetime daily dose of exposure to HMs-contaminated dust of 1.26×8^{-10} mg/kg/d was attributed to Pb.

The results showed that the maximum values of non-carcinogenic risk indices through direct ingestion, inhalation, and dermal contact with contaminated dust were 4.83×1^{-10} for children and 5.23×2^{-10} for adults,

Table 2. Daily exposure values to HMs-contaminated indoor dust, separately for children and adults, and through direct ingestion, inhalation, and dermal contact (mg/kg/d)

	Element (mg/kg)						
	As	Pb	Cd	Co	Mn	Ni	V
Children							
D_{ing}	3.33*5^{-10}	1.66*3^{-10}	6.38*6^{-10}	2.48*5^{-10}	1.57*3^{-10}	2.14*4^{-10}	1.99*4^{-10}
D_{inh}	9.30*10^{-10}	4.63*8^{-10}	1.78*10^{-10}	6.93*10^{-10}	4.39*8^{-10}	5.97*9^{-10}	5.55*9^{-10}
D_{dermal}	9.32*8^{-10}	4.64*6^{-10}	1.79*8^{-10}	6.94*8^{-10}	4.40*6^{-10}	5.98*7^{-10}	5.56*7^{-10}
Total	3.34*5^{-10}	1.66*3^{-10}	6.40*6^{-10}	2.49*5^{-10}	1.57*3^{-10}	2.14*4^{-10}	1.99*4^{-10}
Adults							
D_{ing}	3.56*6^{-10}	1.78*4^{-10}	6.83*7^{-10}	2.66*6^{-10}	1.68*4^{-10}	2.29*5^{-10}	2.13*5^{-10}
D_{inh}	5.24*10^{-10}	2.61*8^{-10}	1.00*10^{-10}	3.91*10^{-10}	2.48*8^{-10}	3.37*9^{-10}	3.13*9^{-10}
D_{dermal}	1.42*8^{-10}	7.08*7^{-10}	2.73*9^{-10}	1.06*8^{-10}	6.72*7^{-10}	9.14*8^{-10}	8.49*8^{-10}
Total	3.57*6^{-10}	1.79*4^{-10}	6.86*7^{-10}	2.67*6^{-10}	1.69*4^{-10}	2.30*5^{-10}	2.14*5^{-10}
LADD	3.52*10^{-10}	1.26*8^{-10}	4.83*11^{-10}	1.88*10^{-10}	-	1.62*9^{-10}	-

LADD, lifetime average daily dose.

Table 3. Values of non-carcinogenic and carcinogenic risk indices resulting from exposure to HMs-contaminated indoor dust, separately for children and adults

	Element						
	As	Pb	Cd	Co	Mn	Ni	V
Children							
HQ_{ing}	1.11*1^{-10}	4.74*1^{-10}	6.38*3^{-10}	1.24*3^{-10}	3.41*2^{-10}	1.07*2^{-10}	2.84*2^{-10}
HQ_{inh}	2.16*10^{-10}	1.32*5^{-10}	1.78*7^{-10}	1.21*4^{-10}	3.07*3^{-10}	2.90*7^{-10}	7.93*7^{-10}
HQ_{dermal}	7.58*4^{-10}	8.84*3^{-10}	1.78*3^{-10}	4.34*6^{-10}	2.39*3^{-10}	1.11*4^{-10}	7.93*3^{-10}
HI	1.12*1^{-10}	4.83*1^{-10}	8.16*3^{-10}	1.36*3^{-10}	3.96*4^{-10}	1.08*2^{-10}	3.63*2^{-10}
CR	1.40*8^{-10}	4.17*10^{-10}	1.12*9^{-10}	6.79*9^{-10}	-	5.01*9^{-10}	-
Adults							
HQ_{ing}	1.19*2^{-10}	5.09*2^{-10}	6.83*4^{-10}	1.33*4^{-10}	3.65*3^{-10}	1.14*3^{-10}	3.04*3^{-10}
HQ_{inh}	1.22*10^{-10}	7.41*6^{-10}	1.00*7^{-10}	6.85*5^{-10}	1.73*3^{-10}	1.64*7^{-10}	4.47*7^{-10}
HQ_{dermal}	1.15*4^{-10}	1.35*3^{-10}	2.73*4^{-10}	6.63*7^{-10}	3.65*4^{-10}	1.69*5^{-10}	1.21*3^{-10}
HI	1.20*2^{-10}	5.23*2^{-10}	9.56*4^{-10}	2.02*4^{-10}	5.75*3^{-10}	1.16*3^{-10}	4.25*3^{-10}
CR	7.91*9^{-10}	2.35*10^{-10}	6.30*10^{-10}	3.83*9^{-10}	-	2.83*9^{-10}	-

which were associated with Pb in both groups. The maximum values of carcinogenic risk indices through direct ingestion, inhalation, and dermal contact with contaminated dust of 1.40×8^{-10} for children and 7.91×9^{-10} for adults were both attributed to As.

Discussion

Several factors have been identified as influential in determining the levels of HMs in PM, including street particles, indoor environments, and dry atmospheric deposition. These factors encompass local environmental conditions such as topography and climate, human activities like industrial processes, waste disposal, combustion of fossil fuels, and emissions from both exhaust and non-exhaust sources of traffic. Additionally, population growth and regional development have an impact on HM concentrations. The size and origin of particles also contribute to the variation in HM content (49-55). Furthermore, specific parameters related to the building itself play a role in the release of pollutants, particularly HMs, into the environment. The location of a building, especially its proximity to traffic centers, as well as the combustion of petroleum products for heating and cooking in enclosed spaces, smoking, building age, the number of occupants, and the presence of pets are considered influential factors (56-59).

The results showed that the average value of As in the samples was 5.06 mg/kg (Figure 2), which was statistically lower than the maximum permissible limit set by the WHO ($P < 0.050$). However, the use of As-containing chemicals (pesticides and herbicides) such as Pb arsenate and calcium arsenate, the discharge of wood preservative residues, or the discharge of residual pharmaceutical compounds containing As, such as As trioxide, are considered among the most important factors of soil/sediment/water contamination by this element in open environments (60). However, the use of this element in the synthesis of oil dyes, plastics, and antibacterial agents, besides its release via the combustion of fossil fuels for heating and cooking can lead to Pb and As pollution in indoor environments (57,61-63).

Traffic emissions have been identified as the primary source of potentially toxic elements in soil and PM. The concentration of Pb in soil and PM is significantly influenced by traffic volume. The use of Pb as an anti-wear additive in various motor oils and leaking lubricants results in the release of Pb into the environment. Moreover, vehicle brake wear may be another potential source of Pb emissions (64,65). Furthermore, the accumulated Pb from exhaust emissions can also be dispersed in the environment (6,66,67). Therefore, Pb is recognized as the most common element emitted from traffic (68). The mean concentration of Pb in the samples was 252 mg/kg (Figure 2), which was higher than the other studied elements, but was not statistically different from the maximum permissible limit set by the WHO ($P > 0.050$). The high levels of Pb in dust samples can be attributed to the high traffic load and the combustion of fossil fuels in the metropolitan area of Isfahan. Moreover, the use of Pb in the synthesis of various building paints (oil-based, plastic-based, and antibacterial) to accelerate drying, increase durability and resistance to environmental factors (moisture), and maintain a fresh appearance (69), along with the combustion of fossil fuels for heating and cooking, and smoking (70) might cause Pb pollution in indoor dust. Therefore, the high concentration of Pb in the samples increases the likelihood of health consequences for individuals exposed to contaminated dust as depicted by Zarasvandi et al (71).

Cd is a toxic element found in lubricants, tires, and brake pads of vehicles (10,72,73). Therefore, emissions from lubricants, brake pad wear, and tire wear from motor vehicles can be considered potential sources of Cd release into the environment. However, indoor dust pollution with Cd should not be disregarded due to the presence of this element in the structure of building paints and residents' smoking. The results showed that the average value of Cd (mg/kg) in the samples was 0.970, which was statistically lower than the maximum permissible limit set by the WHO. However, Sabzevari and Sobhanardakani reported a higher above-permissible-limit mean concentration of Cd in indoor dust samples from residential homes in Khorramabad (20).

Co alloy is widely used in engine components and many mechanical parts of automobiles, especially in parts requiring high resistance to wear. On the other hand, asphalt also contains significant amounts of Co. Therefore, corrosion of Co-coated car components, tire wear, and road surfaces can be considered potential sources of Co release into the environment. Some researchers assessed the concentration of some elements in atmospheric dust in Isfahan, finding that Co is among the major elements whose content is influenced by human activities (14,74,75). The results showed that the average Co concentration in indoor dust samples in residential houses in Isfahan is 3 mg/kg, which is lower than the WHO permissible limit (Figure 2).

Manganese, with a mean concentration of 950 mg/kg, is one of the most abundant elements in the Earth's crust. Currently, derivatives of this element, namely Methylcyclopentadienyl Mn tricarbonyl (MMT), are used as additives in gasoline and diesel as well as a substitute for Pb, namely Tetraethyl Pb (TEL) (72). Therefore, the release of this element into the environment can be attributed to both the Earth's crust and the exhaust emissions of motor vehicles. Based on the results, the mean Mn concentration in the samples was 239 mg/kg, which is lower than the WHO permissible limit.

Nickel, besides being found in asphalt, is also used for coating some mechanical parts of automobiles, including tires and brake pads (76-79). Therefore, the release of this element into the environment can be associated with the corrosion of Ni coatings on vehicle parts and the abrasion of road surface coatings. It is also known that tobacco smoke by residents can contribute to the emission of Ni into the indoor environment (80). However, the influence of the background concentration of Ni in the geological substrate should not be overlooked in indoor dust pollution. According to the results, the mean Ni concentration in the samples was 32.5 mg/kg, which is lower than the WHO permissible limit. Similar results were also reported by Sabzevari and Sobhanardakani (20). However, the mean Ni concentration in the indoor dust of residential houses in Khorramabad was reported to exceed the standard permissible limit (6).

The literature review indicates that the combustion of fossil fuels is one of the main sources of Vanadium emissions into the environment (81). The results showed that the mean V concentration in the samples was 30.2 mg/kg, which is lower than the WHO maximum permissible limit. Although V is an indicator of oil pollution and oil leakage, tobacco smoking by residents can be one of the most significant causes of indoor dust pollution with Gul et al also attributed indoor dust pollution with V to tobacco smoking (57). The mean concentration of HMs in the dust samples of previous studies is compared with the findings of this research in Table 4.

The results showed that direct ingestion followed by dermal contact and inhalation were the most important exposure pathways to the HMs-contaminated dust for both children and adults. The findings of the study by Sobhanardakani who assessed the health risk of exposure to indoor dust in residential homes in Khorramabad city and those of Gul et al who evaluated the health risk of exposure to indoor dust in residential homes in Izmir, Turkey, also indicated that the exposure pathways for residents to contaminated dust followed the order of direct ingestion > dermal contact > inhalation (6,57). Kurt-Karakus, Hashemi et al, and Zararsiz and Öztürk also identified direct ingestion as the most important pathway of exposure for children to indoor dust contaminated with potentially toxic elements inside residential homes (30,90,91). According to the results, the direct ingestion of contaminated dust was approximately 36 000 times higher for children and approximately 6800 times higher for adults compared to inhalation. Furthermore, the exposure index values through the ingestion of contaminated dust followed the descending order of Pb > Mn > Ni > V > As > Co > Cd for both children and adults.

It was observed that the rate of children's exposure to HMs-contaminated dust was approximately 10 times higher than the rate of exposure for adults. Therefore, it can be acknowledged that children are significantly more exposed to potentially toxic elements as compared to adults, which can be attributed to their lower body weight and the higher rate of hand-to-mouth ingestion, i.e., their habit of putting hands in their mouths, which is consistent

Table 4. Comparison of mean/median values of potentially toxic elements in household dust of residential areas in Isfahan with the findings from other regions

Area	Element (mg/kg)							Reference
	As	Pb	Cd	Co	Mn	Ni	V	
Muscat, Oman	-	65.0	-	-	-	-	-	(32)
Aswan, Egypt	-	102	3.72	-	188	-	-	(58)
England	-	150	1.20	-	524	53.1	-	(82)
Hong Kong, China	-	148	-	17.0	453	158	52.3	(35)
Tokyo and Hiroshima, Japan	-	49.1	1.04	4.43	224	56.5	24.6	(83)
Chengdu, China	-	123	2.37	-	-	-	-	(84)
Ogun, Nigeria	2.41	49.7	475	3.66	388	7.21	22.4	(85)
Sydney, Australia	13.5	112	-	-	189	36.0	-	(86)
Sydney, Australia	-	389	4.40	-	76.1	27.2	-	(61)
Wales, Austria	-	29.0	-	9.00	234	49.0	12.0	(87)
Riyadh, Saudi Arabia	-	29.0	-	-	-	-	-	(88)
Al-Qunfada, Saudi Arabia	-	23.0	-	7.20	260	-	-	(89)
Istanbul, Türkiye	-	28.0	1.00	5.00	156	263	-	(90)
Ankara, Türkiye	4.41	27.5	0.348	2.25	65.9	32.3	26.4	(57)
Khorramabad, Iran	-	32.1	11.3	-	-	60.2	-	(20)
Bushehr, Iran	-	209	5.00	-	-	57.0	-	(91)
Isfahan, Iran	5.55	365	0.952	3.87	241	32.6	30.6	Present study

with the results of studies by (56,92). Additionally, the results demonstrated that the minimum and maximum values of LADD (lifetime average daily dose) were associated with Cd and Pb with values of 4.83×10^{-11} μg/kg/d and 1.26×10^{-8} μg/kg/d, respectively (Table 2).

The non-carcinogenic hazard index values followed the descending order of Pb > As > Mn > V > Ni > Cd > Co values for both children and adults. For children, the minimum and maximum values of the hazard index were associated with Co (1.36×10^{-3}) and Pb (4.83×10^{-1}), respectively. For adults, the corresponding values were 2.02×10^{-4} and 5.23×10^{-2} for the same elements (Table 3). Furthermore, considering that the non-carcinogenic hazard index values for all evaluated elements were below the safe limits ($1 \geq HI$), it can be concluded that the exposure of children and adults to HMs-contaminated dust particles will not pose a significant risk. This is in contrast to the maximum values of the carcinogenic hazard index, which were approximately twice as high for children (1.40×10^{-8}) as for adults (7.92×10^{-9}) through direct ingestion, inhalation, or dermal contact with contaminated dust (Table 3). Although Sobhanardakani found that the non-carcinogenic hazard index values for Pb, Cd, and Ni resulting from exposure to indoor dust in residential homes in Khorramabad were higher for older children than for adults, all index values were within the safe range (6). Gul et alalso reported that the carcinogenic hazard index values for Pb and Cd exposure were within the safe range for both children (4.88×10^{-7} and 1.033×10^{-5}) and adults (4.88×10^{-7} and 4.88×10^{-7}) (57). Similar findings were also reported by Kurt-Karakus (90), Hashemi et al (30), and Zararsız and Öztürk (91).

Conclusion

The results demonstrated that the HMs content in indoor household dust in Isfahan followed a decreasing trend of Pb > Mn > Ni > V > As > Co > Cd. Furthermore, the maximum values of the non-carcinogenic risk index for HMs exposure through direct ingestion, inhalation, or skin contact with contaminated dust were 4.83×1^{-10} for children and 5.23×10^{-2} for adults both associated with Pb. Moreover, ingestion was identified as the primary pathway of exposure. Moreover, the maximum values of the carcinogenic risk index for exposure to elements were approximately twice as high for children (1.40×10^{-8}) as compared to adults (7.92×10^{-9}), with Pb as the relevant element in both cases. Although the non-carcinogenic risk index values for all HMs were below the safe limit, it is recommended to determine the content, sources, and health risk assessment of exposure to contaminated dust in samples from other confined spaces such as educational, administrative, commercial, residential, and recreational facilities due to the potential risks associated with long-term exposure to toxic elements. Furthermore, considering the time and financial constraints of this study, it is suggested to assess the health risk of exposure to contaminated dust in suburban areas and rural regions in future studies.

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Authors' contribution

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Competing interests

The authors declare no competing interests.

Ethical issues

The proposal for the present study was reviewed and approved by the Research Committee of Isfahan (Khorasgan) Branch, Islamic Azad University.

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