



Assessment of house dust trace elements and human exposure in Ankara, Turkey

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Abstract

One of the impacts of the COVID-19 pandemic is leading people remain at homes longer than ever. Considering the elongation of the time people spend indoors, the potential health risks caused by contaminants including heavy metals in indoor environments have become even more critical. The purpose of this study was to evaluate the levels and sources of heavy metals in indoor dust, to assess the exposure to heavy metals via indoor dust, and to estimate the associated health risk. The highest median value was measured for Zn ($263 \mu\text{g g}^{-1}$), while the lowest median concentration value was observed for Cd ($0.348 \mu\text{g g}^{-1}$). The levels of elements measured in the current study were found to be within the ranges reported in the other parts of the world, mostly close to the lower end of the range. House characteristics such as proximity to the main street, presence of pets, number of occupants, and age of the building were the house characteristics influencing the observed higher concentrations of certain heavy metals in houses. Enrichment factor values range between 1.79 (Cr) and 20.4 (Zn) with an average EF value of 8.80 ± 6.80 representing that the targeted elements are enriched ($\text{EF} > 2$) in indoor dust in Ankara. Positive matrix factorization results showed that the heavy metals in the house dust in the study area are mainly contributed from sources namely outdoor dust, carpets/furniture, solders, wall paint/coal combustion, and cigarette smoke. Carcinogenic and non-carcinogenic risk values from heavy metals did not exceed the safe limits recommended by EPA. The highest carcinogenic risk level was caused by Cr. The risk through ingestion was higher than inhalation, and the risk levels were higher for children than for adults.

Keywords Carcinogenic risk · Heavy metals · House dust · Exposure · Risk assessment · Turkey

Introduction

One of the impacts of the COVID-19 pandemic is making people remain at homes longer than ever (Nwanaji-Enwerem et al. 2020). As lockdowns and isolations were applied as a primary strategy in isolation of people to control the spread

of the virus, most people get used to home-office work style with a new understanding of the benefits. Some forms of home-office are estimated to persist even long after the control of the pandemic (Sneider and Singhal 2021). Therefore, the air quality in indoor environments, where we spend most of our daily life, became more important when considered its effect on our health. Nowadays, people spend most of their time (approximately 90% of the day) indoors (González-Martín et al. 2020; Marques et al. 2018; Nwanaji-Enwerem et al. 2020). However, indoor air contains various contaminants such as carbon monoxide, sulfur dioxide, ozone and radon, particle-sized pollutants and biological particles (pollen, fungi, and bacteria), and volatile organic compounds (VOCs) (Cincinelli and Martellini 2017). Dust is an important indoor contaminant as it can serve as a host medium for various organic and inorganic pollutants including trace metals (Barrio-Parra et al. 2018). Any particle with a size less than $300 \mu\text{m}$ is called as dust and is defined as fine powdery

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particulate matter that is precipitated or suspended in the air in indoor environments (Rashed 2008; Turner 2011). Pollutants in dust may originate from internal sources such as fuel combustion, decorative materials, cooking, smoking, and external sources such as soil, mining, smelting, industrial activities, and vehicular emissions (Yadav et al. 2019). Heavy metals may have adverse effects on humans (Darus et al. 2012; Kamran et al. 2013) due to their non-degradable and highly toxic nature. Heavy metals in indoor dust can enter the human body through inhalation, ingestion, and dermal contact (Cao et al. 2015; Hashemi et al. 2020; Kang et al. 2011; Popoola et al. 2012; Rout et al. 2013). Children (1–6 years) are more sensitive to heavy metals in indoor dust due to reasons such as crawling, hands-to-mouth habits, and rapid growth rates (Becker et al. 2022a; Ceballos et al. 2022; Jones and Burstyn 2018). Scientific reports state that dust ingestion is the primary heavy metal exposure route for children, as children tend to play on the floor and swallow dust indirectly (Olujimi et al. 2015). Moreover, dust may be inadvertently swallowed and easily adhere to children's skin due to their proximity to the floor and their tendency to touch objects in addition to put their hands and other objects into their mouths (Latif et al. 2014; Olujimi et al. 2015). Studies on heavy metal pollution in house dust show that features such as the type of heating of houses (Battsengel et al. 2021; Zhao et al. 2020), the type of floor covering, traffic emissions (Capozzi et al. 2019; Duan et al. 2020; Hashemi et al. 2020; Yap et al. 2011), proximity to industrial areas (Yap et al. 2011), wall paint type and colour (Hashemi et al. 2020; Hassan 2012; Ogilo et al. 2017), age of the building (Hassan 2012; Popoola et al. 2012), and smoking habits (Hashemi et al. 2020; Hassan 2012; Zhou et al. 2020) may affect the concentrations of indoor heavy metal levels. Rasmussen et al. (2001) found that indoor dust in electrically heated homes in Canada tends to have both mercury (Hg) and Pb contents higher than those heated by oil or gas. Zhao et al. (2020) reported that house dust shows higher Cd concentration levels in urban areas and during heating periods in China. Yap et al. (2011) reported that proximity to the main street and the industrial regions caused high levels of Cd, Pb, and Zn in dust samples. Another study reported that the primary source of Cd, Cr, Cu, and Zn is the motor vehicle emissions in China, whereas Pb comes mainly from the construction source, and Ni has mixed sources, including nature and traffic (Duan et al. 2020). Similarly, Capozzi et al. (2019) showed that traffic affects indoor pollution in Italy's urban areas. Tong and Lam (2000) observed the effect of wall paint colours on the metal levels in indoor dust, and they found a positive relationship between the gray dye and Mn; the yellow dye and Cu, Cd, Zn, and Pb; the purple dye and Zn and Pb; and green dye and Cu concentrations, and that the Mn and Zn concentrations of the houses using wallpaper were much lower. Hassan (2012) found that wall paint

containing lead causes high Pb concentrations in indoor dust in Egypt. Hashemi et al. (2020) showed that the heavy metals in indoor mainly originate from tobacco and smoking, old building materials, traffic sources, and building paint colours in Iran. It has been reported that the heavy metal concentration in indoor dust is negatively correlated with the ventilation rate and positively correlated with smoking inside the building (Hashemi et al. 2020). Zhou et al. (2020) found that cooking and smoking habits are the main sources of indoor heavy metal pollution in China and that there is a negative correlation between the number of people living in the house and heavy metal concentrations, and this is because people act as a sink of pollutants. Rasmussen et al. (2001) found that higher Pb concentrations occur in dust samples of old houses in Canada. Although urban lifestyles may have similarities among different countries, they also have many differences, such as city planning applications, building material preferences, exhaust controls over vehicles, behaviour in home (wearing shoes at home or not), and cooking habits. All these cultural and geographical differences may affect the quantity and composition of indoor dust. Therefore, studies on different geographical regions are valuable to evaluate the health risks associated with indoor dust pollution.

Although a considerable amount of research has been reported on health risks associated to heavy metals contents of the indoor dust around the world (Bashir et al. 2020; Sobhanardakani 2019; Tan et al. 2016; Zhang et al. 2019), studies reporting on the occurrence and health risks caused by heavy metals in indoor dust are scarce in Turkey. To our knowledge, only three publications exist in this particular field in Turkey (Babaei et al. 2015; Kurt-Karakus 2012; Zararsız and Oztürk 2020). In this manner, the main goals of this study were (1) to determine levels of heavy metals (As, Cd, Co, Cr, Cu, Mn, Mo, Ni, Pb, Sb, Se, Sn, Sr, V, and Zn) in house dust samples collected in urban centres in Ankara (the capital city in Turkey); (2) to determine metal uptake rates for children and adults via ingestion, inhalation, or dermal contact to the dust particles in homes; (3) to estimate the health risk posed by heavy metal exposure through indoor dust at homes; (4) to assess the relationship between the metal levels and house characteristics.

Material and methods

Study area and sampling locations

The province of Ankara is located in the northwest of the Central Anatolia Region of Turkey, with nearly a population of 5.5 M people. It is the second-most populous city in Turkey (TUIK 2020). Sampling was conducted in the most populous districts of the city namely Cankaya

(949,265), Kecioren (942,884), Yenimahalle (703,809), Mamak (682,420), Etimesgut (606,472), Sincan (561,411), Altindag (407,675), Golbasi (142,961), and Elmadag (44,236), respectively. **Government** is the prominent employer in Ankara, and yet Ankara is also an important city located at the centre of Turkey's road and railway networks. Several enterprises have headquarters in Ankara which contribute eventually to the traffic originated pollution rather than industrial pollution. Particularly in Cankaya district, service facilities are available almost in everywhere and shopping malls are spread in every part of the city in addition to the government institutions such as hospitals, schools, and universities. Therefore, the

district has a heavy traffic load. Ivedik and Ostim are the two main organized industrial zones in Ankara, which host the majority of small and medium-sized businesses.

A total of 85 house dust samples were collected in 2019 from Ankara. The houses, where dust samples were collected, were in Cankaya, Altindag, Etimesgut, Elmadag, Kecioren, Golbasi, Sincan, Mamak, and Yenimahalle districts that are known as the central districts of Ankara (Fig. 1). The physical characteristics of houses such as proximity to the main street, number of occupants, and wall paint colour, and the habits of the residents such as smoking were recorded by using questionnaires (Table 1).

Fig. 1 Sampling locations (the sampling areas marked in red; 1: Sincan, 2: Etimesgut; 3: Yenimahalle; 4: Kecioren; 5: Altindag; 6: Cankaya; 7: Elmadag; 8: Mamak; 9: Golbasi)

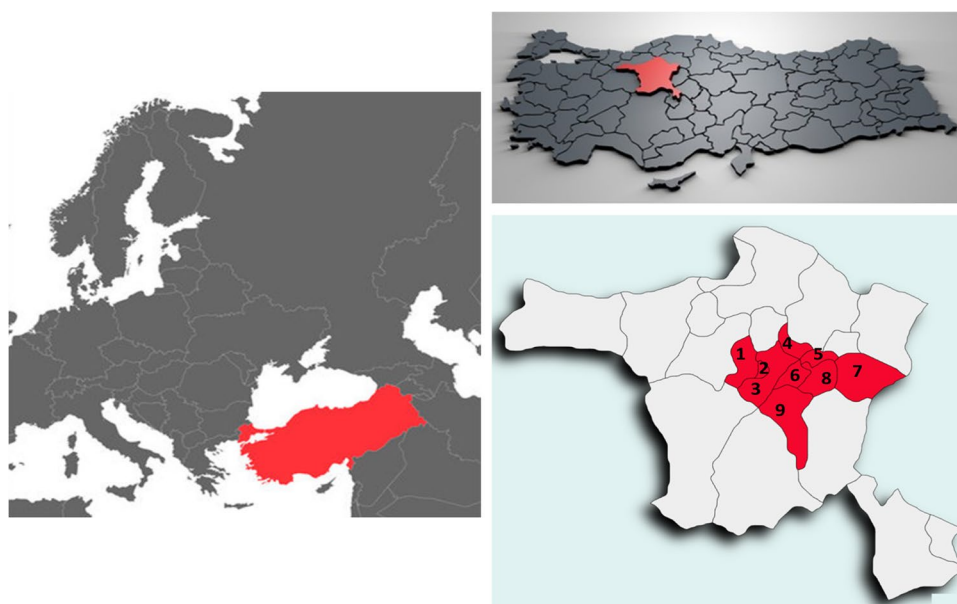


Table 1 Characteristics of sampling locations

Parameter	Values (number of samples in each category)
Age of building	1–5 years (13), 6–10 years (22), 11–20 years (22), >20 years (28)
Close to the main street?	Yes (61), No (24)
Floor cover	Laminate and carpet (74), laminate and hand-made carpet (5), hardwood and carpet (4), mosaic and machine carpet (1), wall-to-wall carpet (1)
Smoking allowed?	Yes (43), No (42)
Number of occupants	2 (1), 3 (15), 4 (53), 5–6 (16)
Floor level	≤0 (11), 1 (23), 2 (18), 3 (9), 4 (7), 5 (6), >5 (11)
Vacuuming frequency	Once day a week (27), 2–3 times in a week (28), 4–7 times in a week (30)
Air conditioning?	Yes (6), No (79)
Windows opened	Once or twice a week (5), three or four times a week (7), every day (73)
Did wall paint in the last one year?	Yes (19), No (66)
Wall paint colour	White (27), cream (31), grey (3), pink (7), yellow (4), beige (4), earth tone (1), green (1), wall-paper (7)
Pets?	Yes (17), No (68)
Asthmatic children?	Yes (38), No (47)

Sample collection

Dust samples were taken from vacuum cleaner bags, which are regularly used for house cleaning to facilitate comparison with studies in the literature. Previous studies suggest that dust samples collected in vacuum cleaner bags are representative in terms of showing the accumulation of pesticides, polybrominated diphenyl ethers (PBDEs), polyaromatic hydrocarbons (PAHs), and heavy metals in indoor environments (Allen et al. 2008; Colt 1998; Colt et al. 2008). Three sampling campaigns were conducted at each of the 85 houses in different seasons for a year; then, the samples were combined and evaluated as a single sample. The house dust samples were placed in sealed bags, labelled, and brought to the laboratory where kept in the freezer until the analysis. Before the analysis, the samples were cleaned from components such as hair, light rubbish, and debris and sieved through a 100 mesh (150- μm) sieve.

Chemicals and reagents

ICP multi-element standard solution with a concentration value of 100 ppm for each targeted element was from CPA-chem (Bulgaria). Nitric acid (HNO_3 , 65%) and sulfuric acid (H_2SO_4 , 96%) were obtained from Sigma Aldrich. Certified standard material, NIST-SRM 2584 (National Institute of Standards and Technology-Trace Elements in Indoor Dust), was also from Sigma Aldrich and provided as a courtesy by Dr. Mahiba Shoeib of Environment Canada (Toronto, Canada). 0.45 μm poresize polytetrafluoroethylene (PTFE) syringe filter tip was obtained from ISOLAB, and ultrapure water was obtained from an PURELAB Flex 3-4 (Elga LabWater, UK).

Sample preparation

The acid digestion process of house dust samples was carried out using a hotplate and aluminium block digestion system. A modification of the HPR-EN-36 (Urban Dust) method specified in the Application Notes of the Milestone microwave device was used. And the digestion process was carried out using 8 mL of 65% HNO_3 and 3 mL of 96% H_2SO_4 to approximately 0.20 g of house dust sample (Milestone 2010). The digestion process was continued until the formed brown smoke disappeared. This process varied between about 5–6 h. The samples were cooled and filtered through a 0.45- μm PTFE syringe filter. The volume of the filtrate was made to 50 mL with ultrapure water and stored in plastic tubes in the refrigerator until instrumental analysis (max. 24 h).

Instrumental analysis

Instrumental analysis was performed using Agilent 7700 model Inductively Coupled Plasma–Mass Spectrometer (ICP-MS). Before the analysis of the samples, an 11-level external calibration with a concentration range of 0.5–1000 ppb was performed for each targeted trace element. The operating conditions of the instrument were as follows: integration time 0.1 s, acquisition time 22.76 s, sampling period 0.31 s, RF power 1550 W, RF matching 1.78 V, makeup gas 0.1 L min^{-1} , carrier gas 0.9 L min^{-1} , He gas flow 4.5 mL min^{-1} , and nebulizer pump 0.1 rps. Readings were undertaken as three replicates.

Quality assurance and quality control

A blank sample was analysed on every ten samples. The blank samples were prepared and analysed in the same manner of dust samples, and only a mixture of acids which were used for sample digestion was used in preparation of blank samples. Results were blank corrected. As a quality control test, the reliability of the device was checked by reading a QC standard (25 ppb) at every 10 samples. Relative standard deviation (RSD) (Becker et al. 2022b) values through the instrumental analysis were 1.09, 1.85, 1.26, 1.58, 0.778, 1.85, 1.21, 1.13, 0.531, 0.959, 1.45, 0.915, 0.466, 0.937, and 4.80 for V, Cr, Mn, Co, Ni, Cu, Zn, As, Se, Sr, Mo, Cd, Sn, Sb, and Pb, respectively.

Method spikes ($n = 10$) were prepared by adding known amounts of heavy metals (20 ppb each) to the acid mixture of digestion and treated in the same manner as the samples.

Recovery ratio (Becker et al. 2022b) of spiked heavy metals were 99.1 ± 4.50 (94.5–107%), 105 ± 5.29 (93.3–109%), 98.0 ± 5.79 (90.7–109%), 102 ± 4.79 (94.7–108%), 99.4 ± 5.14 (90.8–107%), 100 ± 5.05 (92.1–109%), 102 ± 4.87 (95.9–109%), 104 ± 5.38 (91.3–110%), 102 ± 5.94 (95.4–110%), 98.5 ± 6.22 (90.6–109%), 102 ± 7.19 (93.0–112%), 99.3 ± 5.61 (91.6–106%), 100 ± 5.52 (91.5–107%), 100 ± 6.16 (91.8–107%), and 102 ± 5.51 (94.3–110%) for V, Cr, Mn, Co, Ni, Cu, Zn, As, Se, Sr, Mo, Cd, Sn, Sb, and Pb, respectively.

In the present study, instrument detection limit (Kurt-Karakus et al.) and method detection limit (MDL) values were determined according to the method given in EPA Method 200.8 Section 3.5 (USEPA 1994). IDL value for analysed heavy metals were 0.0125 $\mu\text{g g}^{-1}$ except for Cr (0.01 $\mu\text{g g}^{-1}$), Zn (0.0025 $\mu\text{g g}^{-1}$), Sr (0.0063 $\mu\text{g g}^{-1}$), and Sn (0.005 $\mu\text{g g}^{-1}$). MDL values of target metals ($\mu\text{g g}^{-1}$) were 0.018, 0.012, 0.024, 0.018, 0.060, 0.034, 0.003, 0.031, 0.039, 0.009, 0.095, 0.024, 0.007, 0.037, and 0.027 for V, Cr, Mn, Co, Ni, Cu, Zn, As, Se, Sr, Mo, Cd, Sn, Sb, and Pb, respectively. Relative standard

deviation (% RSD) for triplicate analysis of dust samples ($n = 3$) from three houses ranged between 1.36% (Co) and 20.3% (Cd) with an average value of $7.20 \pm 5.50\%$.

The National Institute of Standards and Technology (NIST) standard reference material (SRM 2584) was used to perform the method evaluation and to determine the precision of the analysis. The relative standard deviation (RSD, %) of Cr, As, Cd, and Pb from the certified concentrations was 2.96%, 17.2%, 7.0%, and 2.6%, respectively.

Risk assessment

The methods specified by USEPA were used to assess the risk resulting from exposure to heavy metals via indoor dust (EPA 1996; USEPA 1989). In this regard, the daily chemical intake (CDI, $\mu\text{g g}^{-1}$ body weight day^{-1}) through ingestion, inhalation, and dermal exposure pathways was calculated using the Eqs. (1), (2), and (3) (Hashemi et al. 2020; Kurt-Karakus 2012; Zararsız and Oztürk 2020). The parameters used in these equations and their corresponding values are provided in Supporting Information Table S1.

Calculation of exposure through ingestion:

$$CDI_{\text{ing}} = C_{UCL} \times \frac{R_{\text{ing}} \times F_{\text{exp}} \times T_{\text{exp}}}{ABW \times T_{\text{avg}}} \times 10^{-6} \quad (1)$$

Calculation of exposure through inhalation:

$$CDI_{\text{inh}} = C_{UCL} \times \frac{R_{\text{inh}} \times F_{\text{exp}} \times T_{\text{exp}}}{PEF \times ABW \times T_{\text{avg}}} \quad (2)$$

Calculation of exposure through dermal contact:

$$CDI_{\text{dermal}} = C_{UCL} \times \frac{SAF \times A_{\text{skin}} \times DAF \times F_{\text{exp}} \times T_{\text{exp}}}{ABW \times T_{\text{avg}}} \times 10^{-6} \quad (3)$$

In the equations, C_{UCL} (exposure-point upper confident limit content, $\mu\text{g g}^{-1}$) is considered to give an estimate of “reasonable maximum exposure,” which is the upper limit of the 95% confidence interval for the mean (Hu et al. 2011; USEPA 1989; Zheng et al. 2010b; Zheng et al. 2010a). In this study, C_{UCL} was calculated using the central limit theorem (CLT), since the concentrations of most elements have a non-normal distribution (Eq. 4) (Hashemi et al. 2020; Kurt-Karakus 2012; USEPA 2002; Zararsız and Oztürk 2020).

$$UCL_{1-\alpha} = \bar{X} + \left(z_{\alpha} + \frac{\beta}{6\sqrt{n}} (1 + 2z_{\alpha}^2) \right) S / \sqrt{n} \quad (4)$$

whereas

\bar{X} : arithmetic mean; S : standard deviation; β : skewness; n : number of samples; α is the possibility of making Type I error (false positive); Z_{α} : $(1-\alpha)$ th quantile of the standard normal distribution. For the 95% confidence level, $Z_{\alpha} = 1.645$.

Carcinogenic and non-carcinogenic risks for different exposure routes are calculated using Eqs. (5) and (6) (Hu et al. 2011; Kurt-Karakus 2012; USEPA 2007; Zararsız and Oztürk 2020). The slope factor (SLF), oral reference dose ($R_f D_o$), and bioavailability fraction (BAF) for different metals were taken as shown in Supporting Information Table S2.

$$\text{Carcinogenic Risk} = CDI_{\text{ing/inh/dermal}} \times \text{BAF} \times \text{SLF} \quad (5)$$

$$\text{Hazard Qoutient (HQ)} = (CDI \times \text{BAF}) / R_f D_o \quad (6)$$

The total risk is expressed as the sum of the cancer risks of each exposure route. Calculated total carcinogenic risk values express the probability of cancer in the event that an individual is exposed to any dangerous substance with cancer-causing potential for a lifetime. In this context, acceptable risk values are reported between 1×10^{-6} and 1×10^{-4} (USEPA 2001). Cumulative non-carcinogenic hazard is defined as the hazard index and represents the sum of the hazard ratios of each exposure route. The calculated total hazard index (HI) values express the probability of cancer in the event that an individual is exposed to any hazardous substance that has no cancer-causing potential for a lifetime. In this context, it is reported that acceptable risk values should be below 1 (USEPA 2001).

Enrichment factor

A popular statistic for assessing how much human activity has boosted an element's presence in a sampling medium relative to its average natural abundance is the enrichment factor (EF). Both a background composition and a reference element must be chosen in order to calculate an EF as these decisions have a significant impact on the calculation's outcome. In the current study, in order to assess the influence of anthropogenic activities on indoor dust metal concentrations, EF was calculated for As, Cr, Cu, Ni, Pb, and Zn in each district of Ankara city using Eq. (7) (Barbieri 2016). EF was calculated by normalizing dust trace metal ratios to ratios from global crustal values reported in a previous study (Isley et al. 2021).

$$EF = \frac{\left(\frac{TE}{RE} \right)_{\text{dust}}}{\left(\frac{TE}{RE} \right)_{\text{crust}}} \quad (7)$$

where TE is the element concentration determined in dust and RE is the reference element. EF values are categorized

as follows: no enrichment to minimal enrichment ($EF < 2$), moderate enrichment ($2 < EF < 5$), significant enrichment ($5 < EF < 20$), very significant enrichment ($20 < EF < 40$), and extremely significant enrichment ($EF > 40$) (Gope et al. 2018). Fe (Khademi et al. 2019; Monged et al. 2020), Al (Adimalla et al. 2020; Relić et al. 2019), Ti (Jiang et al. 2020), and Mn (Yadav et al. 2019) are commonly used reference elements in EF calculation.

Positive matrix factorization

In the study, positive matrix factorization (PMF) was employed using Software from the US EPA (version 5.0.14) for source apportionment. In PMF, the base run was selected based on the sample concentration values, and source identification was developed according to representative elements of base run factors. In order to determine the component profiles and source contributions of the pollution sources under non-negative limitations, the model employs the concentration matrix X of the elements and uncertainty information, and the PMF may be written as follows (Eq. (8)):

$$X_{ij} = \sum_{k=1}^p (gikf_{kj} + e_{ij}) \quad (8)$$

where X_{ij} is the concentration of a species, e_{ij} is residual matrix, f_{kj} is factor profile, gik is factor contributions, p is factor number, i is sample number, and j is specie.

By repeatedly breaking down the heavy metal concentration matrix X to produce the ideal matrices G and F , PMF minimizes the objective function Q . The function has the following definition (Eq. (9)).

$$Q = \sum_{i=1}^n \sum_{j=1}^m \left\{ \frac{e_{ij}}{u_{ij}} \right\}^2 \quad (9)$$

Additionally, the element concentration and associated uncertainty (Zhou et al.) must be loaded into PMF. The uncertainty (Zhou et al.) is calculated using a fixed fraction of the MDL according to EPA PMF 5.0 user guide (Eqs. (10) and (11)):

$$\text{When concentration} < \text{the method detection limit (MDL),} \\ u_{ij} = 5/6 \times MDL \quad (10)$$

$$\text{When concentration} > \text{the method detection limit (MDL),} \\ u_{ij} = \sqrt{(\text{error fraction} \times c) + MDL} \quad (11)$$

The minimal value of Q can be determined by any number of F (factor profile) and G (factor contribution) matrices. The term “rotational ambiguity” describes this. PMF has the FPEAK tool to check for rotational ambiguity. For the present dataset, PMF could do five FPEAK runs.

Statistical analysis

SPSS Statistics 26 (IBM) and XLSTAT 2014 were used for statistical analyses. Firstly, normality tests were conducted to determine whether the heavy metal contents showed normal distribution. According to the Shapiro-Wilk normality test, most of the heavy metal levels showed non-normal distribution ($p < 0.001$).

Skewness is a measure of the lack of symmetry of the data distribution around its mean; kurtosis is a measure of the tails of the data distribution (Kurt-Karakus et al. 2008). For these distributions, the skewness (S) and kurtosis (K), standard error of skewness (SES), and standard error of kurtosis (SEK) were calculated using SPSS Statistics 26 (IBM) (Supporting Information Table S3). Data showed significant kurtosis ($K/SEK > 2$) and significant skewness ($S/SES > 2$) for all heavy metals except for vanadium ($K/SEK < 2$ and $S/SES < 2$). As the data did not show normal distribution, the Kruskal-Wallis test was used to compare the median values of data sets with three or more groups/conditions. The Kruskal-Wallis test is the non-parametric equivalent of a one-way analysis of variance. A p -value of < 0.05 shows significant differences between the median of the compared groups. In the presence of only two groups, the Mann-Whitney test, the non-parametric equivalent of a two-sample t -test, was used to compare the medians.

Results and discussions

Heavy metal concentrations in indoor dust

Mean and median concentrations of heavy metals detected in house dust from Ankara are shown in Fig. 2 while the descriptive statistics of metal concentrations are given in Supporting Information Table S3. Zn showed the highest concentration with a median concentration of $264 \pm 159 \mu\text{g g}^{-1}$, whereas the lowest concentration was measured for Cd ($0.400 \pm 0.600 \mu\text{g g}^{-1}$). The median concentrations of the metals were found in the order Zn ($264 \pm 159 \mu\text{g g}^{-1}$) > Cu ($65.8 \pm 183 \mu\text{g g}^{-1}$) > Sr ($73.4 \pm 37.5 \mu\text{g g}^{-1}$) > Mn ($65.9 \pm 48.3 \mu\text{g g}^{-1}$) > Ni ($32.3 \pm 40.7 \mu\text{g g}^{-1}$) > Pb ($27.5 \pm 22.8 \mu\text{g g}^{-1}$) > V ($26.4 \pm 9.60 \mu\text{g g}^{-1}$) > Cr ($23.8 \pm 15.0 \mu\text{g g}^{-1}$) > Sn ($14.3 \pm 16.3 \mu\text{g g}^{-1}$) > As ($4.40 \pm 4.20 \mu\text{g g}^{-1}$) > Co ($2.30 \pm 2.50 \mu\text{g g}^{-1}$) > Sb ($2.10 \pm 1.70 \mu\text{g g}^{-1}$) > Mo ($1.10 \pm 10.5 \mu\text{g g}^{-1}$) > Se ($0.600 \pm 1.40 \mu\text{g g}^{-1}$) > Cd ($0.400 \pm 0.600 \mu\text{g g}^{-1}$) (Fig. 2). A considerable variation in the concentrations of heavy metals, especially Zn, Cu, Mn, Ni, Pb, Sn, and Mo, found in various dust samples can be inferred from the standard deviations in Fig. 2. Certain heavy metals such as Cu, Zn, and Pb are reported to be also derived from industrial sources (Yongming et al. 2006). However, to our knowledge, there are not known industrial sources nearby

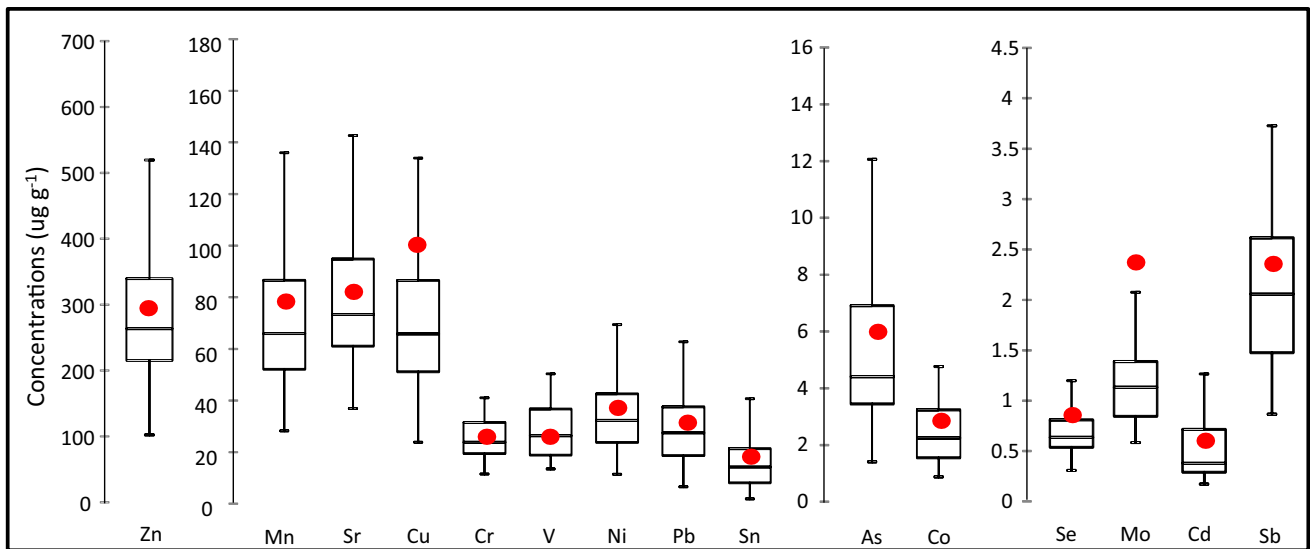


Fig. 2 Heavy metal concentrations ($\mu\text{g g}^{-1}$) detected in analysed samples (the top and bottom ends of the box represents the 75th and 25th percentiles of the data set, respectively. The extensions (“whiskers”)

at either end of the box indicate the maximum and minimum values. The red dots represent mean concentrations, while the median concentrations are indicated by the horizontal line in the boxes)

these houses and yet most of the houses studied in the current research (72%) are located close to main streets (≤ 50 m) (Table 1). Therefore, it is hypothesized that the traffic source is the main contributor to the detected metal levels in this study. On the other hand, Zn (Kelepertzis et al. 2019), Cu (Al-Madanat et al. 2017; Darus et al. 2012), Mn (Iwegbue et al. 2017), Ni (Dehghani et al. 2017), and Pb (Nawazish et al. 2017) are known to be mainly released from vehicle emission. In the current research, we investigated statistical differences in median values of heavy metals in dust from houses categorized based on proximity to the main street. Median values of studied elements were not statistically different ($p > 0.05$) except for Pb. Although the correlation was weak ($p = 0.049$), this statistically significant result shows that the proximity of the houses to the main streets is important in the release of Pb, which is a traffic emission related contaminant, into indoor environments.

The median concentrations of the heavy metals measured in this study are compared with those found in the other locations in Turkey and around the world in Table 2. In published studies, Zn, Cu, Sr, Mn, Ni, Pb, Cr, V, Sn, As, Co, Sb, Mo, Se, and Cd concentrations range between 163 and $1380 \mu\text{g g}^{-1}$, 23.0 and $305 \mu\text{g g}^{-1}$, Sr 57.0 and $170 \mu\text{g g}^{-1}$, 78.8 and $388 \mu\text{g g}^{-1}$, 7.21 and $263 \mu\text{g g}^{-1}$, 28.0 and $209 \mu\text{g g}^{-1}$, 29.9, $152 \mu\text{g g}^{-1}$, 12.0 and $40.7 \mu\text{g g}^{-1}$, 6.02 and $65.0 \mu\text{g g}^{-1}$, 1.57 and $27.0 \mu\text{g g}^{-1}$, 3.66 and $30.2 \mu\text{g g}^{-1}$, 0.140 and $8.85 \mu\text{g g}^{-1}$, 0.760 and $2.03 \mu\text{g g}^{-1}$, and 1.00 and $475 \mu\text{g g}^{-1}$, respectively. The levels measured in our study were found to

be within the ranges reported in the literature, mostly close to the lower end of the range. Wang et al. (2003) reported that concentrations of Cr, Cu, Co, Mn, Mo, Pb, Ni, Sb, Sr, V, and Zn were emitted from diesel vehicle exhausts. Besides, Zn, Mn, As, Cr, and Cu metals may also be caused by industrial activities (Marcazzan et al. 2001; Nriagu and Davidson 1986). Median concentrations of Cr, Co, and Cd in the literature vary between 29.9 and $152 \mu\text{g g}^{-1}$, 5.00 and $30.2 \mu\text{g g}^{-1}$, and 1.00 and $5.00 \mu\text{g g}^{-1}$, respectively. In this study, the median concentrations of Zn, Cu, Mn, Pb, Cr, Co, and Cd were found to be lower than the values reported in the literature. Concentrations of Zn, Cu, Mn, Ni, Pb, Cr, Co, and Cd were lower than the levels reported for house dust in other parts of Turkey (Babaei et al. 2015; Kurt-Karakus 2012; Zararsız and Oztürk 2020). Regarding the findings by Babaei et al. (2015) who also conducted a study in Ankara in 2015, Zn, Cu, Mn, Pb, V, Cr, and Cd levels were found to be approximately 1.67, 1.10, 1.19, 2.51, 1.54, 1.26, and 5.89 times higher, respectively, than those detected in the current study. Such a difference may originate from increasing number of vehicles due to population growth in the city. Median Ni concentrations ($32.3 \mu\text{g g}^{-1}$) in the current study were found to be higher than those reported by the study conducted in Ankara ($31.6 \mu\text{g g}^{-1}$) (Babaei et al. 2015; Zararsız and Oztürk 2020), but lower than the levels reported for a study conducted in Istanbul ($263 \mu\text{g g}^{-1}$) (Kurt-Karakus 2012). This may be due to the fact that Istanbul, a megacity, contains more intertwined residential and industrial areas.

Table 2 Heavy metal concentrations in indoor dust and comparison to the published data ($\mu\text{g g}^{-1}$)

Country	City	Zn	Cu	Sr	Mn	Ni	Pb	V	Cr	Sn	As	Co	Sb	Mo	Se	Cd	References
Turkey	Ankara ^a	263	65.7	73.3	65.9	32.3	27.5	26.4	23.8	14.3	4.41	2.25	2.06	1.10	0.604	0.348	This study
Turkey	Ankara ^a	440	72.5	nd	78.8	31.6	69.2	40.7	29.9	6.02	1.57	nd	0.140	nd	nd	2.05	Babaei et al. (2015)
Turkey	Bolu ^a	551	94.0	170	241	63.0	41.0	nd	152	65.0	27.0	30.2	nd	nd	nd	nd	Zararsız and Öztürk (2020)
Turkey	İstanbul ^a	832	156	nd	156	263	28.0	nd	55.0	nd	nd	5.00	nd	nd	nd	1.00	Kurt-Karakus (2012)
Austria	Wels ^a	395	190	57.0	234	49.0	29.0	12.0	31.0	nd	nd	9.00	nd	nd	nd	nd	Lanzerstorfer (2017)
Iran	Bushehr ^a	567	186	nd	nd	57.0	209	nd	143	nd	nd	nd	nd	nd	nd	5.00	Hashemi et al. (2020)
Australia	Across the country ^a	1260	nc	186	246	30	126	nc	86	nc	19	nc	nc	nc	nc	nc	Isley et al. (2021)
China	Across the country ^a	849	nc	204	218	52	112	nc	108	nc	10	nc	nc	nc	nc	nc	Isley et al. (2021)
Nigeria	Across the country ^a	163	23.0	nc	339	20.2	43.7	nc	77.0	nc	3.3	nc	nc	nc	nc	nc	Isley et al. (2021)
Greece	Across the country ^a	664	158	nc	230	52.3	57.0	nc	97.9	nc	6.9	nc	nc	nc	nc	nc	Isley et al. (2021)
UK	Across the country ^a	448	97.4	nc	229	25.9	82.1	nc	63.6	nc	4.5	nc	nc	nc	nc	nc	Isley et al. (2021)
USA	Across the country ^a	1380	305	nc	335	99.0	40.0	nc	77.0	nc	10.8	nc	nc	nc	nc	nc	Isley et al. (2021)
Australia	Sydney ^a	1258	222	nd	189	36	112	nd	71	nd	13.5	nd	nd	nd	nd	nd	Doyi et al. (2019).
Nigeria	Ogun State ^a	373	55.4	62.5	388	7.21	49.7	22.4	39.2	0.57	2.41	3.66	nd	0.76	1.31	475	Olujimi et al. (2015)
China	Chengdu ^b	675	161	nd	nd	nd	123	nd	82.7	nd	nd	nd	nd	nd	nd	2.37	Cheng et al. (2018)
Japan	Tokyo and Hiroshima ^a	896	266	69.2	224	56.5	49.1	24.6	66.3	17.3	nd	4.43	8.85	2.03	nd	1.04	Yoshinaga et al. (2014)
Turkish-SQG ^c		23,464	3129	nc	nc	1564	400	548	235	46,929	0.400	23.0	nc	391	391	70.0	Official-Gazette (2010)
USEPA-SCS ^d		23,000	nc	nc	nc	1600	400	nc	100,000*	nc	22	3100	nc	nc	nc	37	Provoost et al. (2006)
CCME-SQG ^e		200	63	nc	nc	50	140	130	64	nc	12	nc	nc	nc	nc	10	CCME (2014)

nd not detected, nc not included/reported

^aMedian concentrations

^bMean concentrations

^cLimit value for ingestion or dermal contact published in Turkish Soil Quality Guidelines (Turkish-SQG)

^dUSEPA-SCS: The US Environmental Protection Agency Soil Clean-up Standard

^eCCME-SQG: Canadian Council of Ministers of the Environment Soil Quality Guidelines for Parks and Residential Areas

*Chromium III

The concentrations of Sr and V in the current study were 1.28 and 2.20 times higher, respectively, in comparison to those reported in house dust in Wels (Austria) (Lanzerstorfer 2017). Lower levels of Wels study than this study may be associated with the non-existence of a large-scale industry with heavy metal emissions in or near Wels (Lanzerstorfer 2017). In the current study, the V and Sr levels were measured to be higher in Cankaya and Yenimahalle districts than in the other districts. The houses sampled in Cankaya and Yenimahalle districts were close to the main street and industrial areas. In addition, the highest Pb (6.60–135 $\mu\text{g g}^{-1}$), Cr (12.5–130 $\mu\text{g g}^{-1}$), Sn (3.95–81.7 $\mu\text{g g}^{-1}$), Mn (46.3–399 $\mu\text{g g}^{-1}$), Ni (16.6–386 $\mu\text{g g}^{-1}$), Cu (32.4–1463 $\mu\text{g g}^{-1}$), and As (1.77–22.6 $\mu\text{g g}^{-1}$) concentrations were obtained in Cankaya district. Cankaya is the most populated district of Ankara and has a high traffic load. Se (0.450–12.3 $\mu\text{g g}^{-1}$), Mo (0.570–97.9 $\mu\text{g g}^{-1}$), and Sb (1.38–11.5 $\mu\text{g g}^{-1}$) were present at the highest levels in the samples from Yenimahalle district, which is also influenced by the traffic and industry. The houses where dust samples were collected in the Mamak district were also close to the main street and showed higher Zn (102–1395 $\mu\text{g g}^{-1}$), Cd (0.140–4.38 $\mu\text{g g}^{-1}$), and Pb (8.34–62.8 $\mu\text{g g}^{-1}$) levels. Much higher indoor heavy metal levels (Zn, Cu, Ni, Pb, Cr, and Cd) than found in the current study were reported by Hashemi et al. (2020) for Bushehr (Böhlandt et al. 2012), and researchers stated that this can be associated with the industrial load, an increasing number of vehicles due to the economic growth, and a large number of construction sites in the city centre.

To our best knowledge, there are no dust guidelines for heavy metals in Turkey or in another part of the world. Therefore, the heavy metal concentration in dust was compared with soil guidelines. Turkish Soil Quality Guidelines (Turkish-SQG) (Official-Gazette 2010), The United States Environmental Protection Agency Soil Clean-up Standard (USEPA-SCS) (Provoost et al. 2006) and Canadian Council of Ministers of the Environment Soil Quality Guidelines for Parks and residential areas (CCME-SQG) (CCME 2014) were used for comparison purpose in the current study (Table 2).

Sources and house characteristics influencing metal concentrations

Correlation between metal concentrations and house characteristics

The heavy metal concentrations measured in the dust samples exhibited significant differences (Kruskal Wallis and Mann-Whitney U test, $p < 0.05$) depending on several factors, such as pet ownership, air-conditioner use, proximity to the main street, number of occupants in the house, age of the building, wall paint colour, and smoking. Box and whisker plots of heavy metal concentrations are shown in Supporting Information Fig. S1–S7.

Figure S1 shows the heavy metal levels measured in the dust samples collected from the houses with and without air conditioner. The concentration difference between the status of air conditioner use and heavy metal levels (except for Sb) was not statistically significant ($p > 0.05$). Only the median concentration value of Sb ($p = 0.013$) in the group with an air conditioner (4.94 $\mu\text{g g}^{-1}$) was found to be slightly higher than the group without an air conditioner (2.01 $\mu\text{g g}^{-1}$) (Supporting Information Figure S1). There is no study in the literature showing that there is a correlation between Sb and air conditioning. Al-Harbi et al. (2021) found relatively low concentration of As and Co, while levels of Fe, Al, Zn, and Mn in dust collected from air filters of home air conditioners were higher. The median concentration value of Pb ($p < 0.05$) in the dust samples from the houses close to the main street (29.3 $\mu\text{g g}^{-1}$) was found to be significantly higher than the other group (22.9 $\mu\text{g g}^{-1}$) (Supporting Information Figure S2). The sources of Pb in indoor dust vary from Pb-based paints to automobile-related petroleum Pb deposition (Doyi et al. 2019). Pb is considered as one of the indicators of vehicular traffic pollution which agrees with the findings of the current study (Iodice et al. 2016; Lin et al. 2005; Singh et al. 2002). Yap et al. (2011) also reported a significant relationship between proximity to the main street and Pb concentration, along with Cd and Zn. The median concentration values of Ni (37.2 $\mu\text{g g}^{-1}$, $p < 0.05$), Se (0.740 $\mu\text{g g}^{-1}$, $p < 0.05$), and Mo (1.29 $\mu\text{g g}^{-1}$, $p < 0.05$) of the group with pet ownership were found to be significantly higher than the group without pet ownership (concentrations were 30.0 $\mu\text{g g}^{-1}$, 0.590 $\mu\text{g g}^{-1}$, and 1.03 $\mu\text{g g}^{-1}$, respectively) (Supporting Information Figure S3). Berny et al. (1994) have reported a significant relationship between Pb levels in children and the presence of domestic pets. Tong (1998) has also reported higher Pb and Cu concentration levels when there is a pet in the house. This may be partly because of pets often bring dust from outdoor. Pets usually stay close to the floor and spend most of their time playing with soil or dust which may be contaminated with heavy metals (Tong 1998). Therefore, this result may explain the higher metal concentration obtained in houses of pet owners. Concentrations of heavy metals in with different wall paint colours can be seen in Figure S4. A similar order of heavy metal abundancy can be noticed, and no significant relationship was found between the heavy metal levels and wall paint colour of the houses (Kruskal Wallis test ($p > 0.05$)). However, there are studies in which a positive relationship was found between yellow dye and Cd, Cu, Pb, and Zn levels; purple dye, and Zn and Pb levels; green dye and Cu levels (Chatopadhyay et al. 2003; Tong and Lam 2000). However, the current study did not reveal a statistically significant

relationship between wall colour and heavy metal levels in the dust samples. A significant relationship between smoking and heavy metals (Pb and Zn) was reported in the literature (Cheng et al. 2018). Zhou et al. (2020) also reported higher Cd levels in the dust samples from smokers' rooms than those from non-smokers' rooms. However, the current study did not show any statistically significant correlation between the heavy metal levels and smoking habit (Mann-Whitney U test ($p > 0.05$)) (Supporting Information Figure S5). Kruskal-Wallis test was applied to examine if the number of residents influences the heavy metal levels in the house. No significant relationship was found ($p > 0.05$) for all metal concentrations except for Sr ($p = 0.022$) (Supporting Information Figure S6). Mann-Whitney U test was conducted to find out the origin of Sr difference. The significance level for all effects was accepted as $0.05/5 = 0.01$ by applying the Bonferroni correction. The groups that made the difference for Sr were determined as the houses with three occupants ($105 \mu\text{g g}^{-1}$, $p < 0.05$) and four occupants ($69.2 \mu\text{g g}^{-1}$, $p < 0.05$). Zhou et al. (2020) found a negative correlation between the number of occupants and Cr, Cd, Ni, Mn, Sb, V, Pb, and Zn, stating that people act as a sink of pollutants. However, such a relationship was not observed in the current study. Statistically significant relationships were found between Ni and Cd levels and the age of the building ($p < 0.05$) (Supporting Information Figure S7). Mann-Whitney U tests were conducted to find out the groups causing the difference for these trace elements. The significance level for all effects was accepted as $0.05/4 = 0.0125$ by applying the Bonferroni correction. The groups that made the difference for Ni ($43.3 \mu\text{g g}^{-1}$, $p < 0.05$) were determined as buildings with ages older than 20 years and buildings with ages younger than 5 years ($28.3 \mu\text{g g}^{-1}$, $p < 0.05$). The groups that made the difference for Cd were determined as the building ages of 1 to 5 years ($0.280 \mu\text{g g}^{-1}$, $p < 0.05$) and 6 to 10 years ($0.585 \mu\text{g g}^{-1}$, $p < 0.05$). Rasmussen et al. (2013) reported that there is a significant increase in heavy metal load for Cr, Cu, Ni, and As by building age as there is more dust in older homes, and determined that Pb, Cd, and Zn concentrations increase with house age. It was concluded that the reason for the increase in the concentration is not only the presence of the metal source but also the dustiness that increases due to the age of the house (Rasmussen et al. 2013). The heavy metal concentrations measured in the dust samples exhibited no significant differences (Kruskal Wallis and Mann-Whitney U test, $p > 0.05$) depending on several factors, such as house cleaning frequency, ventilation, floor level, and floor cover of houses.

The correlations between the concentrations of heavy metals in dust samples were analyzed using the Spearman

correlation model, and the results are reported in Supporting Information Table S4. The following positive correlations were observed at a 99% confidence interval: V vs Mn, Co, As, Se; Cr vs Ni, Mn, Co, Cu, As, Se and Sr; Mn vs Co, Ni, Cu, As, Sr and Mo; Co vs Ni, As, Sr, Mo; Ni vs Cu, Zn, As and Sr; Cu vs Zn, As, Se, Sr, Mo, Sn and Pb; Zn vs As, Pb, Sn; As vs Se, Sr and Pb; Se vs Sr and Pb; Sn vs Sb and Pb, while positive correlation was observed between V vs Ni, Cu, Sr; Cr vs Zn and Mo; Mn vs Zn and Se; Co vs Cu, Zn, Cd; Ni vs Mo; Cu vs Sb; Zn vs Se and Sb; As vs Mo and Sn; Se vs Mo and Sn; Sr vs Cd; Sb vs Pb at a 95% confidence level. No statistically significant correlation was observed for the rest of the elements.

Enrichment factor

As stated earlier, EF is commonly used to determine the extent of heavy metal enrichment in environmental compartments. Isley et al. (2021) reported EF values of selected heavy metals for indoor dust samples collected from 35 countries. In order to compare the EF values reported by Isley et al. (2021) and EF values determined in the current study, we calculated the EF values for the same suite of elements (As, Cr, Cu, Ni, Pb, Zn) and the same reference element (Mn) reported by these researchers. Additionally, in order to develop a comparative enrichment measure for each district as well as for the provincial data set, a cumulative enrichment factor ($cEF = \sum EF$) was calculated for the selected heavy metals (Supporting Information Table S5).

In general, $EF < 2$ indicates natural conditions while $EF > 2$ suggest anthropogenic influence (Barbieri 2016). For the whole data set of Ankara province, EF values range between 1.79 (Cr) and 20.4 (Zn) with an average EF value of 8.80 ± 6.80 representing that the targeted elements are enriched ($EF > 2$) in indoor dust in Ankara. Over all enrichment of Zn was the greatest (20.4) followed by $\text{Cu} > \text{Pb} > \text{As} > \text{Ni} > \text{Cr}$. This order was similar to that reported by Isley et al. (2021) for EF values calculated for 35 countries. District basis variations were present with the lowest enrichment of Cr ($EF = 1.14$) in Elmadag, while the greatest enrichment was for Zn ($EF = 26.3$) in Mamak. Such unique differences in EF reflect local anthropogenic activities in surrounding areas of the sampling sites. Obesity, hypertension and type-2 diabetes are associated to the cumulative trace metal exposure (Wang et al. 2018). It is also suggested the consideration of cumulative risks in assessments of children's trace metal exposures instead of risks associated to each trace metal individually (Cao et al. 2016). For this purpose, the cumulative enrichment factors (cEF) (Supporting Information Table S5) were compared detected in the current study for Ankara province to the values reported for other countries (Isley et al. 2021). cEF value of Ankara was greater compared to cEF value reported for Nigeria (10.5), Mexico

(23.9), UK (31.7), Ghana (36.8), Greece (38.1), New Caledonia (40.8), and Croatia (44.5), while Ankara showed lower cEF values compared to USA (545.8), New Zealand (56), China (61.4), and Australia (64.7). Among the districts of Ankara, cEF value was the greatest in Sincan (73.7) and followed by Mamak (59.9) > Altindag (59.0) > Cankaya (54.7) > Etimesgut (53.3) > Kecioren (46.3) > Yenimahalle (45.4) > Golbasi (39.6).

Positive matrix factorization

In this study, the PMF model was used to analyze the sources of heavy metals in house dust. The results showed that the heavy metals in the house dust in the study area mainly contributed from five sources. The factor fingerprints plot (Fig. 3) illustrates the distribution of metal species in various sources and provides the percentage contribution of metal species in various sources. Factor profiles are shown in Fig. 4. Factor 1 explains 41.4% of all data, factor 2 (4.9%), factor 3 (5.5%), factor 4 (21.1%), and factor 5 (27.4%).

Factor 1 (outdoor dust) had high concentration of V, Sb, Se, Mo, Cd, As, Cu, Zn, Ni, and Pb. For these elements, sources are designated as traffic emissions and roadside dust. The possible sources of these elements, which are found in high amounts in house dust, maybe the contaminated dust carried from the outdoor. It is noteworthy that the houses with high scores for this factor are close to the main street and on lower floors. The fact that the houses are ventilated every day also has a significant effect on the transport of polluted outdoor dust to the house. It has been reported in some studies that the main source of Pb, Zn, Cd, Cr, Cu, and Ni is traffic emissions (Al-Rajhi et al. 1996; Hassan 2012).

Factor 2 (carpets/furniture) represents a large incidence of the explained variation of Sb, Cr, and Mo. Hand-woven carpets are used in homes with a high score for this factor.

Polyester and animal wool used in carpets contain Sb and Cr, respectively. The chromium content of dyed wool acts as a good flame retardant (Ingham et al. 2012). Antimony is commonly found in polyester fibres due to its employment as a catalyst in the manufacturing of polyethylene terephthalate and as a synergistic flame retardant in numerous new and recycled polymers (Biver et al. 2021). On the other hand, while the amount of chromium in steel determines whether it is considered “stainless,” molybdenum increases all stainless steel’s resistance to corrosion. The stainless steel used in household furniture can be a source for Mo and Cr.

Pb and Sn elements have the highest concentration explained by Factor 3 (solders). Although non-Pb solder is widely used today, Pb and Sn were common components of solder. Soldering touches containing Sn and Pb are still used in indoor electronic devices, toys and others made of plastic and cheap jewellery made of metals may be a reason for the coexistence of these two elements in house dust. In a study by Yoshinaga et al. (2014), it was found that there is a high correlation between Pb and Sn measured in house dust.

It has been found that the highest variation Mn, Se, Sr, Cd, As, Co, and Pb elements present in Factor 4 are caused by wall paint and coal combustion. During the combustion of coal, some heavy metals (As, Cd, Co, Cr, Cu, Mn, Ni, Pb, and Zn) can be produced (Rout et al. 2013). As, Cd, Zn, Cu, and Pb elements are used in colour pigments of plastic paints and oil paints, and in antibacterial paints (Chattopadhyay et al. 2003; Tong and Lam 2000). Pb is also added to paint to accelerate drying, increase durability, maintain a fresh appearance, and resist moisture that causes corrosion (Kumar and Gottesfeld 2008). Although some countries limit use of lead in paint due to its toxicity, Turkey is not one of them. The highest variations V, Mn, Se, Mo, Cd, As, Cu, Ni, Zn, and Co elements present in Factor 5 are attributed to cigarette smoke. In this group, the factor

Fig. 3 Percentage contribution of elements to different factors (factor fingerprints)

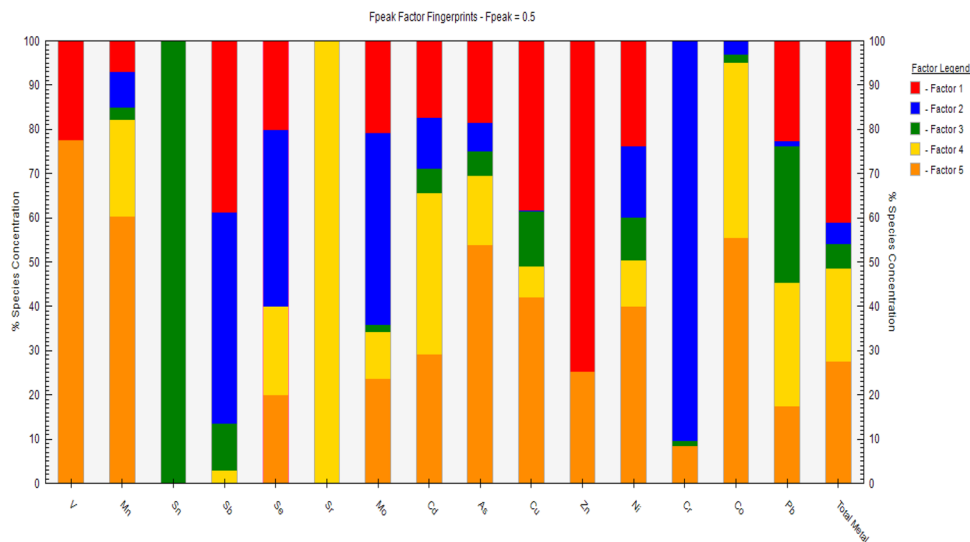


Fig. 4 Factor profiles after rotation



contributions were high at the smoking-allowed houses. Several studies reported that tobacco smoke contains a number of heavy metals, including Cd, Cr, Pb, and Ni (Ashraf 2012; Eržen and Kragelj 2006; Galażyn-Sidorczuk et al. 2008). The source of some elements with high concentration (As, Ni, Cd, Cr, and Co) in this factor may be in the wall dust. According to research, metals including Pb, Zn, Cd, Cu, Mn, Ni, Co, Cr, and Fe are used in wall paints and other building products (Abagale et al. 2013; Mielke et al. 2001).

Estimation of human exposure to heavy metals

Carcinogenic and non-carcinogenic risks were calculated to evaluate the human health risk as a result of exposure to heavy metals in house dust via inhalation, ingestion, and dermal contact routes. Both carcinogenic and non-carcinogenic health risks were calculated for adults and children (1-6 years old) in the current study. The levels calculated for carcinogenic and non-carcinogenic risks are given in Table 3.

Carcinogenic risk

Pb, Cd, and Cr, known as carcinogenic metals, were analyzed, and the carcinogenic risk was defined for routes of ingestion and inhalation of house dust. US Environmental Protection Agency (USEPA) has not established an SLF for dermal contact (USEPA 2011). Environmental Protection Agency (EPA) states that cancer risk lower than 1×10^{-6} can be considered as negligible (EPA 1996), and that some kind of remediation is required for values above 1×10^{-4} , whereas the cancer risk is significant for values higher than 1×10^{-4} (Hu et al. 2011). Higher carcinogenic risk values have been observed for both adults and children in exposure through ingestion compared to those for inhalation (Table 3). The risk of ingestion and inhalation of indoor dust for children was higher than the risk calculated for adults. This result is due to the lower body weight of children as well as higher rates of dust ingestion. As it is shown in Table 3, for adults, the calculated total

Table 3 Carcinogenic and non-carcinogenic risk from heavy metals in the studied indoor dust

Element	C_{UCL} ($\mu\text{g g}^{-1}$)	CDI ($\mu\text{g g}^{-1} \text{day}^{-1}$)		Children		Adults	
		Children	Adults	HQ	Carcinogenic risk	HQ	Carcinogenic risk
$Pb_{\text{carcinogen}}$	37.3	Ingestion	7.35×10^{-5}	2.70×10^{-6}		4.87×10^{-7}	1.79×10^{-8}
		Inhalation	6.85×10^{-9}	1.32×10^{-9}		2.24×10^{-10}	4.34×10^{-11}
		Dermal	2.40×10^{-7}	3.59×10^{-7}		-	-
		Total risk				4.88×10^{-7}	1.79×10^{-8}
$Cd_{\text{carcinogen}}$	0.754	Ingestion	1.49×10^{-6}	5.46×10^{-8}		1.03×10^{-5}	3.77×10^{-7}
		Inhalation	1.39×10^{-10}	2.68×10^{-11}		4.01×10^{-10}	7.76×10^{-11}
		Dermal	4.86×10^{-9}	7.27×10^{-9}		-	-
		Total risk				1.03×10^{-5}	3.77×10^{-7}
$Cr_{\text{carcinogen}}$	30.4	Ingestion	5.99×10^{-5}	2.20×10^{-6}		8.09×10^{-6}	2.97×10^{-7}
		Inhalation	5.58×10^{-9}	1.08×10^{-9}		6.18×10^{-8}	1.19×10^{-8}
		Dermal	1.96×10^{-7}	2.93×10^{-7}		-	-
		Total risk				8.15×10^{-6}	3.09×10^{-7}
$Pb_{\text{non-carcinogen}}$	37.3	Ingestion	7.35×10^{-5}	7.87×10^{-6}	1.64×10^{-2}		1.75×10^{-3}
		Inhalation	6.85×10^{-9}	3.86×10^{-9}	1.53×10^{-6}		8.60×10^{-7}
		Dermal	2.40×10^{-7}	1.05×10^{-6}	5.35×10^{-5}		2.33×10^{-4}
		HI			1.64×10^{-2}		1.99×10^{-3}
$Cd_{\text{non-carcinogen}}$	0.754	Ingestion	1.49×10^{-6}	1.59×10^{-7}	6.84×10^{-4}		7.33×10^{-5}
		Inhalation	1.39×10^{-10}	7.81×10^{-11}	6.37×10^{-8}		3.59×10^{-8}
		Dermal	4.86×10^{-9}	2.12×10^{-8}	2.23×10^{-6}		9.75×10^{-6}
		HI			6.86×10^{-4}		8.31×10^{-5}
$Cr_{\text{non-carcinogen}}$	30.4	Ingestion	5.99×10^{-5}	6.42×10^{-6}	5.39×10^{-3}		5.78×10^{-4}
		Inhalation	5.58×10^{-9}	3.15×10^{-9}	5.02×10^{-7}		2.83×10^{-7}
		Dermal	1.96×10^{-7}	8.54×10^{-7}	1.76×10^{-5}		7.68×10^{-5}
		HI			5.41×10^{-3}		6.55×10^{-4}
Ni	49.3	Ingestion	9.73×10^{-5}	1.04×10^{-5}	2.04×10^{-3}		2.19×10^{-4}
		Inhalation	9.07×10^{-9}	5.11×10^{-9}	1.90×10^{-7}		1.07×10^{-7}
		Dermal	3.18×10^{-7}	1.39×10^{-6}	6.68×10^{-6}		2.91×10^{-5}
		HI			2.05×10^{-3}		2.48×10^{-4}
Cu	149	Ingestion	2.93×10^{-4}	3.14×10^{-5}	2.92×10^{-3}		3.13×10^{-4}
		Inhalation	2.73×10^{-8}	1.54×10^{-8}	2.72×10^{-7}		1.53×10^{-7}
		Dermal	9.57×10^{-7}	4.18×10^{-6}	9.55×10^{-6}		4.16×10^{-5}
		HI			2.93×10^{-3}		3.55×10^{-4}
Zn	335	Ingestion	6.60×10^{-4}	7.07×10^{-5}	1.61×10^{-3}		1.72×10^{-4}
		Inhalation	6.15×10^{-8}	3.47×10^{-8}	1.50×10^{-7}		8.43×10^{-8}
		Dermal	2.16×10^{-6}	9.40×10^{-6}	5.25×10^{-6}		2.29×10^{-5}
		HI			1.61×10^{-3}		1.95×10^{-4}
Mn	89.9	Ingestion	1.77×10^{-4}	1.90×10^{-5}	6.03×10^{-4}		6.46×10^{-5}
		Inhalation	1.65×10^{-8}	9.31×10^{-9}	5.62×10^{-8}		3.17×10^{-8}
		Dermal	5.79×10^{-7}	2.53×10^{-6}	1.97×10^{-6}		8.59×10^{-6}
		HI			6.05×10^{-4}		7.32×10^{-5}

carcinogenic risks arising from ingestion and inhalation of Pb, Cd, and Cr were 1.79×10^{-8} , 3.77×10^{-7} , and 3.09×10^{-7} , respectively, while it was 4.88×10^{-7} , 1.03×10^{-5} , and 8.15×10^{-6} for Cd, Pb, and Cr, respectively, for children. In summary, the total cancer risk values calculated for both children and adults are within the safe limits of EPA (1×10^{-6} and 1×10^{-4}). Therefore, it can be stated

that cancer risk is acceptable for Pb, Cd, and Cr because of exposure to indoor dust for the samples studied.

Noncarcinogenic risk

In terms of noncarcinogenic risk, the highest risk values for all metals in both adults and children were obtained for the

ingestion, followed by the dermal contact route. Health risk due exposure to indoor dust via inhalation, ingestion, and dermal contact resulted in the order $Mn < Cd < Zn < Ni < Cu < Cr < Pb$ for both adults and children. USEPA (2001) reports that acceptable hazard index (HI) values should be below 1 for noncarcinogenic risk. An overall assessment of ingestion + inhalation + dermal contact exposure routes showed all HI values were lower than 1 showing that HI values were in the safety limits in the current study. Pb, As, and Cd require special attention due to their characteristics such that they can cause neurological and developmental disorders in contact with Pb and As (Masindi and Muedi 2018; Mason et al. 2014), and the kidneys are the main target organ in contact with Cd (Masindi and Muedi 2018). HI values above the EPA's safe limit ($HI < 1$) for Cd and Pb were not observed in the current study.

Kurt-Karakus (2012) reported similar non-carcinogenic and carcinogenic risk results due to exposure to heavy metals through indoor dust in Istanbul, Turkey. Zararsız and Oztürk (2020) have reported non-carcinogenic and carcinogenic risk values within EPA's safe limits and stated that the ingestion is the main exposure route to the heavy metals for both adults and children. Hashemi et al. (2020) found that the non-carcinogenic and carcinogenic risk values resulting from heavy metals had exceeded the safe limit recommended by EPA. Both Hashemi et al. (2020) and Kurt-Karakus (2012) reported that ingestion is the main route of exposure based on the carcinogenic risk assessment of heavy metals in children, while for adults, the main route of exposure was dermal contact.

Relation to presence of asthmatic child in houses

In the current study, 38 samples were collected from houses where asthmatic children lived. Therefore, a possible relation between heavy metal levels and presence of asthmatic syndromes was also investigated since it is known that indoor quality can pose harmful health effects such as respiratory and cardiopulmonary pathologies and asthma, especially in children (Stamatelopoulou et al. 2017; Yang et al. 2009).

As stated earlier, in the presence of only two groups, the Mann-Whitney test, the non-parametric equivalent of a two-sample *t*-test, was used to compare the medians in the current study. In order to compare median values of elements detected in homes with an asthmatic child to the concentration values detected in homes without an asthmatic child, Mann-Whitney non-parametric test was utilized. Statistical test showed the following casual results: $Sn_{asthmatic} (17.3 \mu g g^{-1}) > Sn_{non-asthmatic} (12.2 \mu g g^{-1})$ ($p = 0.047$); $Co_{asthmatic} (1.87 \mu g g^{-1}) < Co_{non-asthmatic} (2.52 \mu g g^{-1})$ ($p = 0.041$); $Pb_{asthmatic} (29.9 \mu g g^{-1}) > Pb_{non-asthmatic} (25.1 \mu g g^{-1})$ ($p = 0.044$) (Supporting Information Figure S8). Recent studies report that the concentrations of Cd (Park et al. 2016), Cr,

and Mn (Zeng et al. 2016) in the blood are significantly associated with asthma. Wang et al. (2017) reported that there is a positive relationship between Pb and asthma. Moghtaderi et al. (2020) found that there is no correlation between heavy metal concentrations in school dust and heavy metals, which agrees with the findings of the current study. However, the size of the samples is too small to generalize the results to the whole community.

Conclusions

Indoor air quality should be recognized as a critical factor in public health policy as the durations of our stays in indoor environments will likely increase not only because of the pandemics' situations but also because of the changing working habits and office hours of the changing world.

This study once more highlights the presence of toxic metals, namely Zn, Cu, Sr, Mn, Ni, Pb, V, Cr, Sn, As, Co, Sb, Mo, Se, and Cd in indoor environments. The metals with the highest levels in house dust were Zn, Cu, Sr, and Mn. The variations in the levels of Zn, Cu, Mn, Ni, Pb, Sn, and Mo were higher than the other metals. The highest Cr, Sn, Mn, Ni, Cu, and As levels were obtained in the Cankaya district; the houses in this district were close to the main street and the industrial areas as well. The measured median levels of Cu, Zn, Ni, Pb, Cr, As, and Co in our study were found to be lower compared to concentrations reported for other countries, while median concentration of Ni was higher compared to level reported for Nigeria, UK, and Australia.

Most of the indoor conditions, such as air conditioner use, wall paint colour, smoking habits of the occupants, and the number of the occupants, could not be statistically associated with the heavy metal levels of the dust. On the other hand, statistically significant relationships were found for Pb levels and proximity to the main street; for Ni, Se, and Mo levels and pet ownership; Ni and Cd levels and the age of the building.

The current study showed that the proximity to the main streets and industrial places would influence the indoor heavy metal levels in a negative way. Therefore, the health effects brought by the traffic and industry as the sources should be seriously considered in the city planning efforts by the relevant authorities. The citizens also need to be informed with the knowledge that the location and several indoor conditions would influence the health safety of their homes.

Zn showed the highest enrichment followed by $Cu > Pb > As > Ni > Cr$. Cumulative enrichment factors value of Ankara was greater compared to value reported for Nigeria, Mexico, UK, Ghana, Greece, New Caledonia and Croatia while Ankara showed lower cumulative enrichment factors value compared to USA, New Zealand, China, and Australia. Positive matrix factorization showed that outdoor dust

influenced the heavy metal composition of houses that are close to the main street and on lower floors. Total cancer risk values calculated for both children and adults are within the safe limits of EPA (1×10^{-6} and 1×10^{-4}). The overall assessment of ingestion + inhalation + dermal contact exposure routes showed all HI values were lower than 1 showing that HI values were in the safety limits in the current study. Comparison of median values of elements detected in homes with an asthmatic child to the median concentration values detected in homes without an asthmatic child showed that certain elements such as Sn, Co, and Pb showed higher concentrations in homes of asthmatic children.

Carcinogenic and non-carcinogenic risk values from heavy metals had not exceeded the safe limit recommended by EPA. The highest carcinogenic risk level was caused by the metal Cr for both children and adults. The risk of exposure through ingestion was higher than the risk of exposure through inhalation for both children and adults. Risk levels were higher in children than in adults but did not exceed the safe level. Although the risk levels in our study were found to be below the safe limits, we also have observed that the heavy metal levels in indoor dust may have a large variability. It may be inferred that an unconsciously selected location of the house may cause an increased exposure to heavy metals followed by increased health risks. However, it should be kept in mind that number of the dust samples in the current study is still considered a small sample size therefore authors would like to emphasize that results of the current study cannot be taken as representative of entire indoor environments in Ankara or in Turkey.

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Declarations

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