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Integrating source apportionment, health risk assessment, and biomonitoring of PTEs in household vacuum dust from a mining and industrial hot spot

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ABSTRACT

Indoor dust is an important exposure pathway for potentially toxic elements (PTEs) in industrial hotspots such as the Meža Valley (Slovenia), impacted by centuries of Pb–Zn mining/smelting and ongoing metallurgical activities. This study is the first to examine household dust from homes of children participating in biomonitoring, providing a unique opportunity to directly assess the relationship between indoor contamination and biomonitoring outcomes. Household vacuum dust from 27 homes (Upper Meža Valley, n = 13; Lower Meža Valley, n = 14) was analysed for 12 metal(loid)s. Statistical analyses identified three dominant source-related groups: (1) Pb–Zn–Cd (legacy mining/smelting), (2) Cu–Sn (Pb recycling), and (3) Cr–Ni–Mo (steel production), indicating a mixed anthropogenic fingerprint across the valley. Health risk assessment of PTEs indicates that ingestion is the primary exposure pathway, with children facing 7–12 times higher non-carcinogenic risks than adults. Children's blood lead levels (BLLs) were generally low (75th percentile = 31 $\mu g\ L^{-1}$), although 6 of 27 exceeded the CDC blood lead reference value of 35 $\mu g\ L^{-1}$. Lead concentrations in household dust significantly predicted BLL variability (adjusted $R^2 \approx 0.23$), highlighting indoor dust as a relevant contributor to internal lead exposure. Cumulative cancer risks remained within acceptable regulatory limits under both worst-case (100% Cr(VI), 9.03×10^{-5}) and more realistic (5% Cr(VI), 1.97×10^{-5}) scenarios. Persistent indoor contamination following the 2023 flooding events, despite ongoing remediation efforts, underscores the need for area-wide mitigation strategies and continued environmental and health monitoring to limit PTE transfer into indoor environments.

1. Introduction

Humans spend up to 90% of their time indoors, making indoor environments a critical determinant of exposure to environmental contaminants (Schweizer et al., 2007; Isley et al., 2022). Research examining indoor environments and their implications for public health has gained significant attention in the last two decades (Buonanno and Kumar, 2025). Indoor dust often contains elevated concentrations of metal(loid)s, such as Pb, Zn, Cu, and As, compared to outdoor dust, even in urban areas with low impact from industrial sources (Rasmussen et al., 2008). A recent global review of urban dust across 59 countries revealed that more than one-third of samples were heavily contaminated with As and 60.3% with Cd, primarily due to industrial activities and traffic emissions closely linked to economic structure (Chen et al., 2024). Residential and commercial areas mainly show contamination from in-house sources and vehicle emissions (Roy et al., 2024). In contrast, indoor dust in industrial zones and mining areas often exhibits significantly higher concentrations of potentially toxic elements (PTEs) and is mostly influenced by anthropogenic activities such as manufacturing processes, e-waste recycling, and mining (Cao et al., 2020, 2022; Roy et al., 2024). Some studies in mining regions have reported extreme Cd enrichment in indoor dust, followed by Zn, Cu, and Pb, and As in areas where arsenic-bearing minerals are processed (Shi and Wang, 2021; Li et al., 2019; Al-Swadi et al., 2022; Chu et al., 2023; Roy et al., 2024).

The relationship between indoor and outdoor dust is dynamic and complex. Outdoor particles enter indoor spaces through ventilation systems, open windows and doors, and are further transported indoors by human activities such as shoes, clothing, and pets (US EPA, 2022). Limited air exchange and continuous deposition can lead to the accumulation of higher concentrations of certain metals indoors, particularly

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in poorly ventilated environments (Rivas et al., 2019; US EPA, 2022). Consequently, house dust is a complex mixture of organic and inorganic particles originating from both external and internal sources, including outdoor soil and background values of PTEs, degree of industrialisation, building materials, consumer products, and human activities (Ibanez et al., 2010; Barrio-Parra et al., 2017). PTEs such as Pb, Cr, Ni, Cd, and Cu are of particular concern due to their toxicity, persistence, and potential for bioaccumulation (Tan et al., 2016; Lim et al., 2024).

Global-scale geochemical analyses of vacuum dust from 2265 households in 35 countries revealed that metal enrichments are predominantly influenced by indoor sources rather than soil tracked indoors (Isley et al., 2022). Pb-Zn-As are mainly associated with legacy Pb sources (leaded petrol and paint), Zn-Cu with building materials (galvanised surfaces, wood preservatives, paints, metal coatings), while Mn typically originates from natural soil sources (Isley et al., 2022; Aguilera et al., 2024). However, site-specific vacuum dust studies at industrial sites clearly demonstrate a significant influence of outdoor industrial emissions to vacuum dust enriched metal concentrations, such as Fe, Co, Pb, Cd, Cr, Mn, Ni, and Zn related with ironworking and other mining activities (Žibret and Rokavec, 2010; Teran, 2020; Fry et al., 2021; Cao et al., 2022), and Hg related with historical Hg mining sites and possibly cement production (Bavec et al., 2017; Zupančič et al., 2021; Gosar and Gaberšek, 2025).

Research on metal contamination in household dust is essential because dust acts as a reservoir for pollutants that can enter the human body via ingestion, inhalation, or dermal contact (Hou et al., 2025; Somsunun et al., 2023). Ingestion is the dominant exposure pathway, accounting for over 50% of total exposure in adults and up to 98% in children (Hou et al., 2025). Children are particularly vulnerable due to frequent hand-to-mouth activity, and higher absorption rates associated with ongoing development (US EPA, 2011; Wang et al., 2022). Elevated concentrations of Cr and Ni in indoor dust have been reported near industrial zones, while Pb remains a major concern in older homes due to legacy lead-based paints (Tan et al., 2016; Al-Momani and Ali, 2024). Chronic exposure to these metals can lead to neurodevelopmental deficits in children, kidney dysfunction, and increased cancer risk in adults (Lim et al., 2024; Hou et al., 2025).

However, studies directly linking indoor dust composition with biomonitoring outcomes remain limited, particularly in mining-impacted regions. This study addresses a gap in current knowledge by investigating indoor metal pollution in the highly contaminated Meža Valley, a region historically affected by intensive mining and smelting activities. Previous studies on household dust in the Meža Valley are over 20 years old (Šajn et al., 2000; Šajn, 2002) or included only a limited number of samples and focused solely on the Upper Meža Valley (Miler and Gosar, 2013). This research provides new insight into the current indoor environment in both the Upper and Lower Meža Valley, following the implementation of the national Programme of Measures to Improve the Quality of the Environment (Official Gazette No. 119/07 and 44/22—ZVO-2; ZVO is an abbreviation for the national Environmental Protection Act). This programme, carried out between 2007 and 2022, included a wide range of remediation actions aimed at reducing environmental pollution in the region. In early 2025, the Decree on the Designation of Degraded Environment and the Programme of Measures for Improving Environmental Quality in Kindergartens, Schools, and Public Playgrounds in the Municipalities of Črna na Koroškem, Mežica, Prevalje, Ravne na Koroškem, and Dravograd was adopted (Official Gazette No. 4/2025, 2025). The decree is of critical importance in the aftermath of the severe floods in 2023, which redistributed contaminated sediments across residential and agricultural areas, thereby increasing environmental and public health risks. This regulatory framework establishes mandatory interventions to reduce lead and cadmium exposure, particularly among children.

Modelling with the IEUBK (Integrated Exposure Uptake Biokinetic) model indicates that soil and dust account for 24–62% of total Pb intake in Mežica and up to 85% in Žerjav, highlighting dust ingestion as one of a dominant pathway for young children (Bavec et al., 2025). Understanding the distribution, sources, and health risks of PTEs in indoor dust is crucial for developing effective mitigation strategies. Therefore, the main objectives of this study were as follows: 1) to quantify the concentrations of PTEs in household vacuum dust across the Meža Valley; 2) to discriminate between contamination patterns and source signatures across the Upper Meža Valley (historical mining and smelting activities and ongoing Pb waste recycling) and the Lower Meža Valley (ongoing iron and steel industry) and; 3) to assess and compare non-carcinogenic and carcinogenic health risks for adults and children, identifying the most critical exposure pathways and elements of concern; 4) to evaluate the predictive relationship between lead (Pb) concentrations in household dust and blood lead levels (BLLs).

2. Materials and methods

2.1. Study area

The study area is located in Slovenia, Europe (Fig. 1). The Meža Valley (MV) is divided into the Upper Meža Valley (UMV; Črna na Koroškem, Mežica) and the Lower Meža Valley (LMV; Prevalje, Ravne na Koroškem). Geologically, the UMV mainly consists of Triassic carbonate rocks, with minor occurrences of low-grade metamorphic rocks (Mioč and Žnidarčič, 1983). The area contains Pb–Zn ore bodies that were exploited for approximately 500 years, producing about 1 Mt of Pb and 0.5 Mt of Zn, with minor Mo extraction. The typical mineral paragenesis includes Pb, Zn, and Mo, with Cd as the only trace element present in notably elevated concentrations (Drovenik et al., 1980). The LMV is mainly underlain by various Palaeozoic metamorphic rocks (Mioč and Žnidarčič, 1983). Long-term mining, smelting, and industrial activities have largely overprinted the natural geochemical background, resulting in a heavily burdened environment in the UMV (Pb, Zn, Cd, and locally also Mo and As) (Šajn, 2006; Miler and Gosar, 2012; Finžgar et al., 2014; Gošar et al., 2015; Miler et al., 2022; Goltnik et al., 2022). Current activities include Pb waste recycling, starter battery production, and mining waste processing, all located in Črna na Koroškem. Previous dust analyses from the historical smelting area indicated that the primary source of pollution in the past was primary Pb smelting, whereas mining waste processing is now the main source. Active Pb recycling has a negligible effect (Miler and Gosar, 2019). In snow solid particles surrounding the lead waste recycling plant, anthropogenic phases including Pb, Sb and Sn were found (Miler and Gosar, 2013). In the LMV, pollution from mining and smelting activities has decreased but remains elevated, particularly in river sediments (Miler et al., 2022). The surroundings of the LMV are also influenced by a long-standing (approximately 400 years) iron and steel industry in Ravne na Koroškem. Around the Ravne steel plant, increased levels of As, Cd, Cr, Mn, Mo, Ni, U, V, W and Zn were found in street dust (Teran et al., 2020), and elevated concentrations of Cr, Mo, Ni and W were detected in attic dust and topsoil (Šajn, 2002, 2006). Overall, the area is characterised by complex, overlapping historical and ongoing sources of potentially toxic elements, which contribute to persistent indoor dust contamination.

2.2. Sampling

Household vacuum dust sampling was conducted in spring and summer 2023. In total, 27 samples were collected from households where children participating in the biomonitoring programme resided. Thirteen children lived in the UMV, while fourteen lived in the LMV. Participants provided dust collected during routine vacuum cleaning. In some cases, multiple bags were collected to ensure sufficient material for analysis.

Whole blood samples from children living in the studied households were collected in May 2023 as part of a long-term monitoring programme. A questionnaire assessing lead exposure risk was completed for each child participating in BLL monitoring. The case management

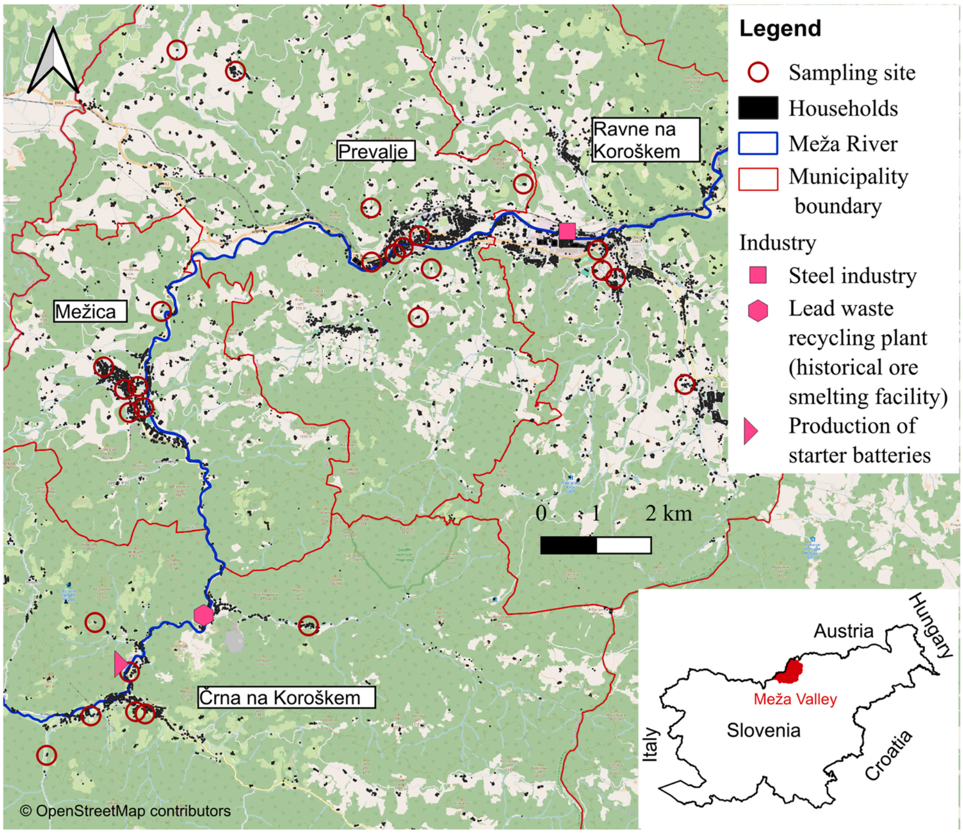

Fig. 1. Study area (Meža Valley, Slovenia) and sampling areas. Exact sampling locations are not shown for privacy and data protection reasons; instead, the approximate sampling area for each household is indicated by a red circle.

protocol for children with BLLs $\geq 100\ \mu g\ L^{-1}$ includes venous BLL measurement, a home visit, and specialist consultation. Access to blood Pb data was approved following an ethics assessment by the National Institute of Public Health (NIJZ).

2.3. Analytical methods

2.3.1. Chemical analysis and quality control

Pulps for analysis were prepared at the Geological Survey of Slovenia. Samples were manually sieved using a column of stainless screen mesh (500, 250, and 125 μm) to obtain the analytical fraction (<125 μm). Pulps were shipped to Bureau Veritas Commodities Canada Ltd., where laboratory tests were performed Total organic carbon (TOC) was determined in 18 out of 27 samples using a LECO analyser, as only these samples yielded sufficient material (5 g) after sieving. At Bureau Veritas Minerals, Mining Laboratory Services Canada (accredited under ISO/IEC 17025), concentrations of 53 elements (CODE AQ250-EXT, BUREAU VERITAS 2024) were measured in all samples ($n = 27$) following a modified aqua regia digestion (1:1:1 HNO_3:HCl:H_2O) of 0.5 g of sample, using inductively coupled plasma mass spectrometry (ICP-MS) and inductively coupled plasma emission spectrometry (ICP-ES). Using a modified aqua regia digestion, a partial digest provides valuable information regarding mobile and easily soluble species (such as sulphides), which is more relevant for environmental assessment and exposure analysis. This study focused on potentially harmful elements (As, Cd, Co, Cr, Cu, Hg, Mn, Mo, Ni, Pb, Sn, Zn), while Al concentrations were used for normalisation in statistical analysis. The precision, accuracy, and bias of the chemical analyses were assessed by duplicate analysis (6 replicates) of certified reference materials (CRMs), with three repetitions each of OREAS45d, OREAS45f, and OREAS262. Accuracy was poor (>20%) for Cd and Mo in two CRMs, and for As and Sn in one CRM. Therefore, it was assumed that the results for these elements are less reliable quantitatively, although their overall distribution and spatial patterns remained important for the research and discussion. On this basis, and considering the multi-CRM validation strategy adopted in the study, these elements were retained in the analysis with explicit acknowledgement of their higher associated uncertainty. The quantitative results for these elements must be used with caution. For the remaining elements, accuracy was good or satisfactory. Precision was good and satisfactory for all elements, except for Sn in four replicates and Hg in two replicates, where it was poor (>20%).

Analyses of lead in whole blood (BLLs) were carried out at the Clinical Institute of Clinical Chemistry and Biochemistry, University Medical Centre Ljubljana, using inductively coupled plasma mass spectrometry (ICP-MS, Agilent 7700, Japan). The laboratory has successfully participated in the INSTAND External Quality Assessment Scheme with validated ICP-MS methods for over eight years.

2.3.2. Data processing

Data processing included raw data preparation and the use of descriptive, bivariate, and multivariate statistical analysis and visualisation using Microsoft Excel and open-source R version 4.3.1 and RStudio software. Spearman correlation and distance matrices were calculated, and hierarchical clustering with Ward's method was performed (Wickham, 2016; Kassambara, 2019; Wei and Simko, 2021), considering all samples ($n = 27$) to observe patterns in element concentrations across the dust samples and to speculate about their potential common sources. The same statistical approach was applied to observe patterns between blood lead levels, elements, and organic matter, as well as between elements and BLLs, to investigate potential relationships between environmental exposure and health impacts. It should be noted that the observed patterns in organic matter content were based on 15 samples, as sufficient material was available only for these. Ordinary Least Squares (OLS) regression was used to assess the linear relationship between house dust Pb concentrations (predictor) and the blood lead level of a child living in the household (response).

Spatial analysis and maps preparation were performed using the open-source QGIS software, version 3.34.12.

2.3.3. Pollution assessment

To evaluate the degree of element enrichment in vacuum dust, two complementary enrichment factors were calculated: a dust-based enrichment factor (Dust-EF) and a soil-based enrichment factor (Soil-EF). The Dust-EF for each element was calculated as the ratio between the median element concentration in vacuum dust measured in this study and the corresponding background concentration for rural areas of Slovenia (Teran, 2020). This approach highlights deviations from typical indoor dust composition and is particularly sensitive to contemporary anthropogenic inputs, such as emissions from the active steel industry. Dust-EF values were calculated for the Meža Valley (MV; all samples), Upper Meža Valley (UMV; n = 13 samples), and Lower Meža Valley (LMV; n = 14 samples).

To distinguish geogenic (crustal) from anthropogenic (non-crustal) contributions to elemental concentrations in dust, a two-component Al-normalised mass balance model was applied. Aluminium (Al) was used as the conservative lithogenic reference element because it is abundant in aluminosilicate minerals and is not substantially enriched by Pb–Zn mining and smelting activities.

In contrast to approaches based on average upper continental crust compositions, area-specific local soil background concentrations were used to account for regional lithological variability, following recommendations that local geochemical backgrounds provide more appropriate reference values than global crustal averages in heterogeneous geological settings (Reimann and de Caritat, 2000). Local background concentrations for both Al and the investigated elements were obtained from previously published baseline soil data for the LMV and UMV sub-areas (Šajn, 2002; see Table 1). The crustal contribution of element i in sample j was estimated as:

$$C_{crustal,\ ij} = [Al]j \times \left(\frac{Ci}{Al}\right)_{local} \tag{1}$$

where [Al]j is the measured Al concentration in dust sample j, and (Ci/Al) local is the ratio between the local background concentration of element i and the corresponding local background Al concentration for the respective sub-area.

The non-crustal fraction was calculated as:

$$C_{non-crustal,\ ij} = \max\ (0,\ C_{measured,\ ij} - C_{crustal,\ ij}) \tag{2}$$

where negative values were set to zero, indicating that measured concentrations were fully explained by crustal inputs. The percentage non-crustal contribution was calculated as:

$$f_{non-crustal,\ ij}(\%) = \frac{C_{non-crustal,\ ij}}{C_{measured,\ ij}} \times 100 \tag{3}$$

Higher non-crustal percentages indicate increasing anthropogenic enrichment relative to the local geogenic background composition. All calculations and visualisations were performed in R version 4.3.1 using the packages readxl (Wickham and Bryan, 2023), dplyr and tidyr (Wickham et al., 2023a, 2023b), and ggplot2 (Wickham, 2016).

2.4. Health risk assessment

The human health risk assessment followed the US EPA exposure and risk assessment framework (US EPA, 2001, 2002), which is widely used in studies to estimate exposure to PTEs found in indoor and outdoor dust (Barrio-Parra et al., 2017; Zhang et al., 2018; Doyi et al., 2019; Zhao et al., 2021; Asvad et al., 2023; Wang et al., 2023).

Exposure was evaluated for adults and children via ingestion, inhalation, and dermal contact pathways. Non-carcinogenic risk was assessed for As, Cd, Cr, Cu, Mn, Mo, Ni, Pb, and Zn. As Cr speciation in household dust was unavailable, Cr-related risk estimates are presented as a range, with Cr(III) representing the most optimistic scenario and Cr(VI) the worst-case scenario (Zhang et al., 2018).

Input parameters for non-carcinogenic and carcinogenic risks are summarised in Tables S1 and S2. Two soil/dust ingestion scenarios were evaluated: a Reasonable Maximum Exposure (RME) scenario, typically applied in highly contaminated environments, assuming ingestion rates of 200 $mg\ day^{-1}$ for children and 100 $mg\ day^{-1}$ for adults (US EPA, 2017; Somsunun et al., 2023; Hou et al., 2025), and a lower-intake scenario assuming ingestion rates of 100 $mg\ day^{-1}$ for children and 50 $mg\ day^{-1}$ for adults to assess the influence of ingestion rate on HRA outcomes. Dermal absorption factors (ABSd) are not available for all metals,

Table 1

Descriptive statistics (mean, standard deviation - SD, coefficient of variation - CV, median - md, minimum - min, maximum - max, 10th, 25th, 75th, and 90th percentiles) of PTE concentrations ($mg\ kg^{-1}$) in the vacuum dust of the Meža Valley. Median element values were also calculated for the Upper Meža Valley (md UMV) and the Lower Meža Valley (md LMV). Data were compared with median vacuum dust levels from Slovenian urban and rural areas, Slovenian industrial towns (Celje, Idrija and Anhovo), the urban area of Maribor, and Upper and Lower Meža Valley topsoil baseline mean values (0–5 cm).

Meža Valley = MV, (n = 27)	As	Cd	Co	Cr	Cu	Hg	Mn	Mo	Ni	Pb	Sn	Zn
Mean	4.7	2.8	5.2	174.1	116.9	0.27	288.7	13.3	91.6	164.1	57.5	821.2
SD	3.2	1.8	4.1	358.8	86.5	0.84	183.7	14.2	190.4	102.9	137.6	339.3
CV	0.7	0.7	0.8	2.1	0.7	0.00	0.6	1.1	2.1	0.6	2.4	0.4
Md	**4.4**	**2.6**	**4.1**	**87.9**	**82.7**	**0.09**	**294.5**	**10.0**	**42.1**	**156.8**	**13.7**	**848.2**
Md UMV (n = 14)	3.3	2.5	3.0	47.8	70.9	0.07	156.0	4.4	37.5	198.1	14.8	878.1
Md LMV (n = 13)	5.2	2.7	5.4	102.8	124.4	0.10	353.0	13.5	60.8	141.9	12.7	827.5
Min	0.9	0.5	0.7	7.5	15.6	0.01	31.0	1.4	6.0	28.5	1.4	141.8
Max	16.4	8.4	18.6	1914.5	327.3	4.45	795.0	64.9	1012.9	475.8	690.0	1466.1
P10	1.2	0.7	1.8	32.7	41.6	0.05	85.0	2.9	26.1	40.0	4.4	397.1
P25	2.8	1.9	2.9	45.6	59.7	0.06	152.0	3.9	36.8	99.9	9.4	637.1
P75	5.7	3.3	6.0	147.1	138.4	0.14	393.0	15.3	66.5	202.0	28.3	946.5
P90	7.0	4.7	8.6	268.7	279.5	0.21	528.2	26.4	100.1	285.2	101.4	1306.6
Slovenian rural area dust baseline[1] (n = 101)	4.0	1	5.9	64	110	0.24	360	2.9	39	54	20	540
Slovenian urban area dust baseline[1] (n = 137)	3.6	0.98	5.6	70	130	0.36	280	3.1	42	61	19	650
Celje[2] (Slovenia) n = 12	5. 95	3.35	7.98	144	190	n.d.	354	4	48	123	n.d.	736
Idrija[3] (Slovenia) n = 16	5.5	1.7	4.1	61.4	159.6	20.9	225	2.7	37.3	69.1	27.2	656.3
Anhovo[4] (Slovenia) n = 16	2.8	0.95	5.8	50	183	1.479	432	1.94	47	67	n.d.	556
Maribor[5] (Slovenia) n = 27	4.1	1.12	6.2	65	140	0.316	306	2.9	38	69	23	716
Upper Meža valley topsoil baseline[6]	16	2.6	11	70	32	0.146	908	3.1	31	408	4.8	402
Lower Meža Valley topsoil baseline[6]	16	0.96	17	113	46	0.14	1069	2.2	46.6	135	5.6	257

[1]Teran (2020); [2]Žibret and Rokavec (2010), Žibret (2012); [3]Bavec et al., 2017; [4]Gosar and Gaberšek, 2025; [5]Gaberšek and Gosar, 2021; [6]Šajn, 2002.
n.d. = no data.

therefore a default value of 0.001 was adopted for most elements, following the Health Risk Assessment Guidance for Metals (ICMM, 2007). The exception was As, for which a value of 0.03 was applied due to its higher dermal uptake potential from soil and dust particles (US EPA, 2002; ICMM, 2007). For the non-carcinogenic risk assessment, an exposure duration of 30 years was applied for adults, whereas for carcinogenic risk assessment, a lifetime of 70 years was used to calculate the lifetime average daily dose (US EPA, 2005, 2011). The average daily exposure dose of PTEs in household dust was calculated separately for three exposure pathways by following Eqs. (4)–(6) (US EPA, 2002) for further non-carcinogenic risk calculations:

$$\mathrm{ADDing} = C \times \frac{EF \times ED \times IngR}{AT \times BW} \times 10^{-6} \tag{4}$$

where C is the concentration of PTEs in household dust (mg kg^{-1}) in this study; EF is the exposure frequency (days year^{-1}); ED is the exposure duration (years); $IngR$ is the ingestion rate (mg day^{-1}); AT is the average lifetime (days); BW is body weight (kg); and 1×10^{-6} is the conversion factor (mg/kg).

$$\mathrm{ADDinh} = C \times \frac{EF \times ED \times InhR}{PEF \times AT \times BW} \tag{5}$$

Where $InhR$ is the inhalation rate (m^3 day^{-1}) and PEF is the particulate emission factor (m^3 kg^{-1}).

$$\mathrm{ADDderm} = C \times \frac{EF \times ED \times SA \times AF \times ABSd}{AT \times BW} \times 10^{-6} \tag{6}$$

Where SA is the exposed skin area (cm^2); AF is the skin adherence factor (mg cm^2 day^{-1}); and ABSd is the dermal absorption factor (unitless). The hazard quotient (HQ) is used to assess the non-carcinogenic risks of PTEs in dust (US EPA, 2002, 2011). It is calculated as the ratio of the average daily dose (ADD) to the reference dose (RfD) according to (Eq. (7)):

$$\mathrm{HQ} = \frac{ADD}{RfD} \tag{7}$$

The Reference Dose (RfD) is the maximum daily amount of PTEs that can be safely consumed without causing non-carcinogenic risks over a person's lifetime. There are three distinct RfDs for different exposure pathways: $RfDing$ (mg kg^{-1} day^{-1}) for ingestion, $RfDinh$ (mg m^{-3}) for inhalation, and $RfDder$ (mg kg^{-1} day^{-1}) for dermal contact.

The cumulative risk from multiple exposures to specific chemicals is represented by the Hazard Index (HI) (Eq. (8)). The total risk of PTEs in dust through various exposure pathways was calculated as the sum of HQ using the following equation (US EPA, 2007):

$$\mathrm{HI} = \sum \mathrm{HQi} \tag{8}$$

where i represents different exposure pathways. HI values below 1 indicate negligible risk, whereas an HI value greater than 1 suggests that non-carcinogenic risk may occur (US EPA, 2001).

To estimate **carcinogenic risks**, the lifetime average daily doses (LADDs) were calculated (Eqs. (9)–(11)) by summing exposure during childhood (labelled c) and adulthood (labelled a). These values were multiplied by the corresponding slope factor (SF; Table S2) to calculate the level of cancer risk (CR) (Eq. (12)):

$$LADDing = \frac{C \times EF}{AT} \times \left(\frac{IngRc \times EDc}{BWc} + \frac{IngRa \times EDa}{BWa}\right) \times 10^{-6} \tag{9}$$

$$LADDinh = \frac{C \times EF}{AT \times PEF} \times \left(\frac{InhRc \times EDc}{BWc} + \frac{InhRa \times EDa}{BWa}\right) \tag{10}$$

$$LADDderm = \frac{C \times EF \times ABSd}{AT} \times \left(\frac{SAc \times AFc \times EDc}{BWc} + \frac{SAa \times AFa \times EDa}{BWa}\right) \times 10^{-6} \tag{11}$$

$$CR = LADD \times SF \tag{12}$$

A CR value below 1×10^{-6} signifies a negligible carcinogenic risk, while a CR value above 1×10^{-4} indicates a significant carcinogenic risk to humans (US EPA, 2001).

3. Results and discussion

3.1. Statistical analysis

3.1.1. PTEs concentrations in household dust

The complete dataset, including element concentrations, BLLs, and organic matter (OM) content, is provided in Table S3, while the corresponding descriptive statistics for PTEs across the 27 sampling sites are summarised in Table 1. Organic matter content ranged from 4.3% to 34.5% with a median of 18.7%. High OM content is typical for this type of dust (Rasmussen et al., 2008; Bavec et al., 2017; Gaberšek and Gosar, 2021) and may act as a binding agent or "sink" for certain elements, facilitating the accumulation and persistence indoors. The median concentrations follow the ascending order: Hg < Cd < As < Co < Mo < Sn < Ni < Cr < Cu < Pb < Mn < Zn. When medians were calculated separately for the Lower Meža Valley (LMV) and Upper Meža Valley (UMV), the values were somewhat higher in the LMV for Cr, Cu, Mn, Mo, and Ni, already suggesting enrichment of these elements in this area.

The dust-based median enrichment factor (Fig. 3) showed moderate enrichment for Cd and Pb in all areas, and for Mo in LMV (and consequently in MV as well). Minimal enrichment of As, Cr, Cu, and Ni was observed in LMV, Zn in all areas, and Mo in UMV. The enrichment of Cd, Pb, and Zn (and partly Mo) clearly reflects the impact of past mining activity and the associated environmental burden on household dust throughout the Meža Valley (Upper and Lower). In LMV, the enrichment of Cr and Ni indicates a possible influence from the steel industry, as previously shown in attic dust (Šajn, 2002) and street dust (Teran et al., 2020).

A comparison of median dust levels in MV, UMV, and LMV with median dust values from other industrial areas (Celje, Idrija, Anhovo) and an urban area (Maribor) across the country (Table 1), based on studies using comparable sampling and analytical methodologies, shows that the dominant industrial activities in the studied towns strongly influence elevated element levels in household dust. Dust from Celje is well known for pronounced industrial contamination linked to ironworks, smelting, and brownfield redevelopment (Žibret and Rokavec, 2010; Žibret, 2012). MV is at least as impacted as Celje in terms of Pb and Zn, but Celje exhibits a distinct Sb anomaly and somewhat higher Cd. MV's Mo levels, however, are more pronounced than in Celje, highlighting a particular alloy/steel-related signature in LMV. Household dust in Idrija is dominated by Hg derived from centuries of mercury mining and smelting (Bavec et al., 2017). For most non-Hg metals, MV is similarly or more contaminated; however, Idrija remains unique due to its extreme Hg levels and elevated Cu and Sn. Around the Anhovo cement plant, dust composition reflects a mixture of geogenic and cement-related sources, including contributions from co-incineration and quarrying (Gosar and Gaberšek, 2025). Anhovo therefore exhibits a cement-plant/industrial fingerprint characterised by high Cu and Mn, whereas MV exceeds Anhovo in Pb, Zn, Cd, and in Cr and Ni within LMV. Maribor represents a typical Slovenian urban environment influenced by both traffic and industry (Gaberšek et al., 2021). Even compared to this major urban centre, MV household dust exhibits stronger Pb–Zn and steel-related signatures, while Maribor displays a more “classic urban” Cu-rich pattern associated with traffic (e.g., brake wear).

The mass balance, referenced to local soil background values, confirms that the anthropogenic (non-crustal) fraction dominates indoor dust across the entire study area for all primary smelter-associated elements (Table S4). Using local rather than global crustal reference values yields more conservative and geochemically credible estimates, yet the results still demonstrate an overwhelming non-crustal signal for the key pollutants.

The strongest anthropogenic signatures were found for Mo, Cu, Zn, Sn, and Cd, with median non-crustal fractions exceeding 94% in both the LMV and UMV (Fig. 2, Table S4). Cadmium showed median non-crustal fractions of 97.0% (LMV; range: 95.3–99.6%) and 94.3% (UMV; range: 83.9–95.6%), consistent with its well-documented co-mobilisation during Pb–Zn smelting. Zinc and Mo followed a similar pattern (Zn: LMV 97.7%, UMV 96.8%; Mo: LMV 98.8%, UMV 96.1%).

Lead, despite an elevated local background (408 mg kg^{-1}) in the UMV, exhibited a predominantly non-crustal signature (LMV median: 92.3%, range: 87.3–97.4%; UMV median: 85.9%, range: 62.5–96.6%). The lower UMV values for Pb reflect the higher local geogenic Pb background in that sub-area, associated with natural ore-bearing geology, and thus provide a more nuanced picture than a global reference would provide. Even when accounting for this elevated natural background, the majority of Pb in UMV indoor dust remains of non-crustal origin.

The elements with the lowest non-crustal fractions were Co, Mn, and As. Manganese showed the highest geogenic contribution in the UMV (median non-crustal: 65.0%; range: 49.7–78.0%), consistent with the naturally Mn-rich lithology of the Eastern Alps. It has been shown that Mn is predominantly of natural origin (originating from soil tracked indoors) in vacuum dust (Isley et al., 2022). Arsenic also showed notable area-specific variability (LMV median: 79.6%, range: 52.6–90.6%; UMV median: 67.7%, range: 24.7–89.1%), likely reflecting both lithogenic As associated with sulphide mineralisation and diffuse anthropogenic inputs. These elements should be interpreted with appropriate caution, whereas the dominant pollutants (Pb, Zn, Cd, Cu, Mo, Sn) are unambiguously dominated by anthropogenic loading in both sub-areas.

Collectively, these results provide direct quantitative evidence that indoor dust in the Meža Valley reflects long-term industrial emissions more strongly than local geology. For the primary Pb–Zn smelter-associated elements, local soil geochemistry accounts for less than 15% of the measured indoor dust concentration in the LMV and less than 20% in the UMV, even when conservative area-specific background reference values are applied.

This study illustrates how different industrial sectors produce distinct geochemical fingerprints in household dust, and how these signatures can overlap and mix within a valley. Household dust background values provide a more appropriate reference for EF calculations than local soil backgrounds because they better represent the actual indoor depositional environment, avoid the mismatch created by using geologically controlled soil baselines, and fill an important gap in the literature, where dust-specific baselines are largely absent and national-scale soil datasets are often used inappropriately for local conditions.

Dust element patterns in the Meža Valley (MV) were compared with global indoor dust datasets. Shi and Wang (2021) compiled a worldwide synthesis of PTEs in indoor dust from more than 120 studies across five continents, covering rural, urban, and industrially impacted environments. Isley et al. (2022) further expanded this perspective with a harmonised analysis of trace metals (As, Cu, Cr, Mn, Ni, Pb, Zn) in residential dust from 35 countries, using a unified analytical protocol and risk assessment framework. In the international literature, Pb and Zn consistently appear as the most enriched metals in indoor dust, particularly in older urban centres and mining or smelting regions, while Cr, Ni, and Mn typically reflect contributions from traffic, steel production, and other metal-processing activities. The MV dataset, especially when UMV and LMV are evaluated separately, fits well within this global typology but also occupies the upper end of concentration ranges for several elements characteristic of Pb–Zn mining/smelting and steel production. Cd levels in the MV are comparable to or exceed the mid-to-upper ranges reported internationally for Pb–Zn mining towns. Cu concentrations fall within typical global urban values but show local enhancement in LMV, consistent with industrial and traffic influences. The pronounced enrichment of Cr, Ni, Mo, and W in LMV aligns with global observations of steel-related emissions, although Mo and W are less frequently reported in international datasets. Their clear elevation

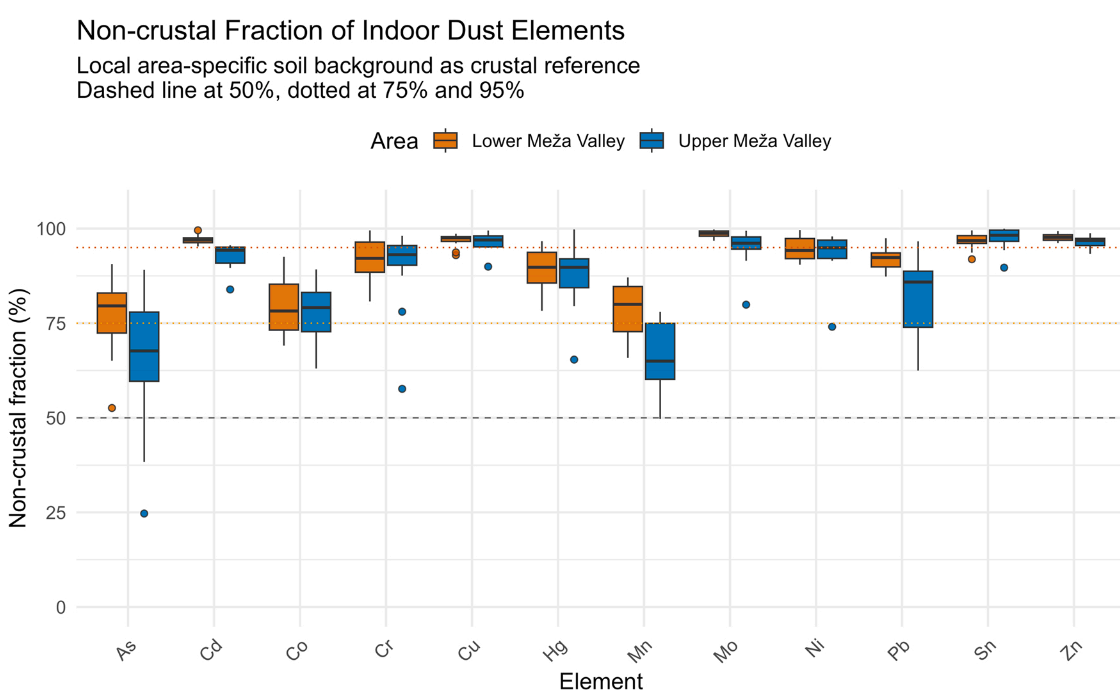

Fig. 2. Non-crustal fractions of potentially toxic elements in indoor dust from the Lower and Upper Meža Valley, calculated using area-specific soil background concentrations as the crustal reference.

in the MV suggests that these elements may serve as useful tracers of alloy and steel production in indoor environments. Compared with large cities represented in global datasets, the MV exhibits a more pronounced metallurgical signature and a relatively weaker dominance of traffic-related elements. This positions the valley closer to the "industrial/mining" archetype described in both global reviews (Shi and Wang, 2021; Isley et al., 2022), rather than a typical traffic-dominated megacity profile.

The correlogram is presented in Fig. S1. Very strong positive correlations (0.8–1.0) were found between As, Mn and Co, and between Mo and Cr. Fairly strong positive correlations (0.6–0.79) were found between Mo and As; Co, Mn and Mo; Ni, Cd and Zn; and Cu and Ni, Sn. Moderate positive correlations (0.4–0.59) were observed between Hg and Co, Mn; Cr and As, Co, Mn; Ni and As, Co, Mn, Mo, Cr; Pb and As; Zn and As, Co, Pb; Cd and Hg, Co, Mn, Pb; Cu and Hg, As, Co; and Cd and Sn. Regarding organic matter (OM) content ($n = 15$), a fairly strong positive correlation was found with Hg, and moderate positive correlations with Mo, Cr, Cu, and Ni. The dendrogram indicates two main clusters based on the hierarchical clustering results.

3.1.2. Source identification

Hierarchical clusters of elements indicate the existence of different sources, which have already been identified to some extent by enrichment ratios. The first cluster includes Mo, Cr, Ni, Hg, As, Co, and Mn (Fig. 4), which most likely represent elements associated with the iron and steel industry and support previous findings (Šajn, 2002; Teran et al., 2020). The second cluster includes Pb, Zn, Cd, Cu, and Sn. The associations of Pb, Zn, and Cd reflects the impact of past mining activities, while the association of Cu and Sn most likely reflects contributions from the Pb waste recycling industry and its influence on the distribution of dust element concentrations in the study area (Fig. 4). Our results support the findings of Miler and Gosar (2013), who demonstrated that anthropogenic sources of PTEs in household and attic dust from Žerjav are primarily Pb smelting (indicated by Pb sulphates with minor concentrations of Cu and Zn and Zn-sulphates with Cu) and secondary Pb-recycling (indicated by Pb-K sulphates, Pb-sulphates with minor concentrations of Sb, Sn, and Cl, spherical Pb-sulphates with minor Cu, and Pb-Sb-Sn-oxides and sulphates). In addition, geogenic-anthropogenic phases (Fe-oxyhydroxide sulphates and oxyhydroxides with minor concentrations of Zn, Cu, and Pb, Pb-sulphates and carbonates/oxides, Ba-sulphates, Zn-sulphides and carbonates/oxides, Fe-sulphides and Pb-phosphates) sourced from mining and mine waste mechanical processing were identified as well as Zn-rich Ba-sulphates sourced from white pigment widely used in interior paints and Fe- oxyhydroxide with minor concentrations of Cu, Mn, and Ti, presumably originating from the oxidation and wear of steel-based household utensils or other industrial processes.

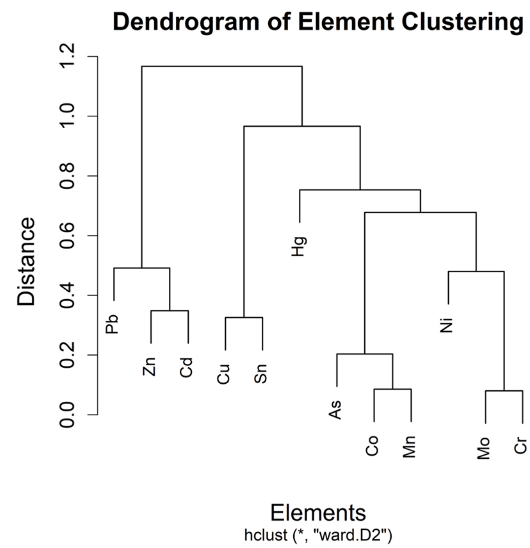

Fig. 4. Dendrogram of 12 metal(loid) elements in indoor dust samples. Clusters (Ward's method, Euclidean distance) indicate common pollution sources.

3.2. Blood lead levels

The distribution of BLL and their relationship with dust Pb concentrations are shown in Fig. 5. Overall, BLLs are generally low, with most samples (BLL Q75 = 31 μg L^{-1}) falling below the Blood Lead Reference Value (BLRV) of 35 μg L^{-1}. This threshold was established by the CDC in 2021 to identify children with exposure levels above the 97.5th percentile of the US population (US CDC (United States Centers for Disease Control and Prevention), 2024). In our study, only six children had BLL above the BLRV. Furthermore, only two children had BLL above 50 μg L^{-1}, a level associated with sufficient evidence for adverse health

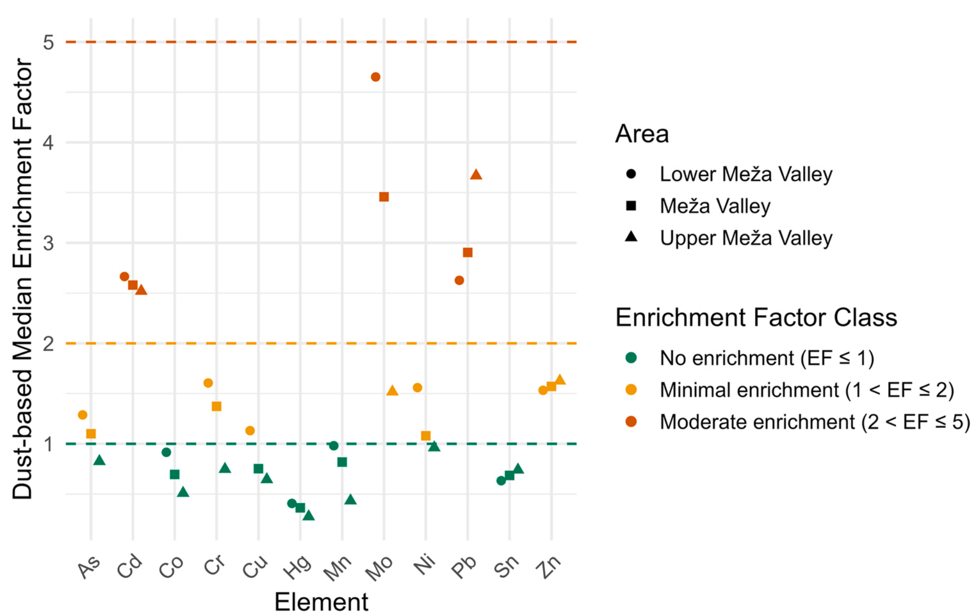

Fig. 3. Median dust-based enrichment factors (EFs) for the Meža Valley (MV), Lower Meža Valley (LMV), and Upper Meža Valley (UMV).

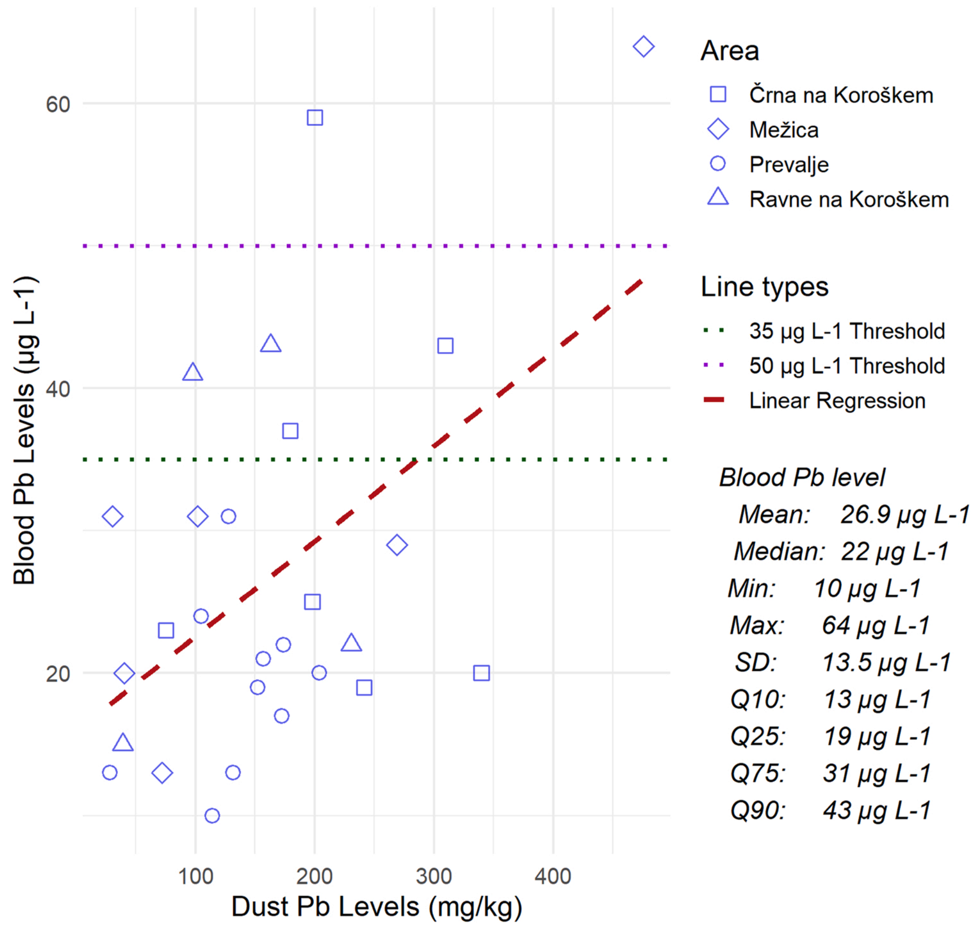

Fig. 5. Correlation between Pb concentrations in household vacuum dust (mg kg^{-1}) and blood lead levels (µg L^{-1}) in children from the Meža Valley.

effects (World Health Organization (WHO), 2021; ATSDR (Agency for Toxic Substances and Disease Registry), 2024). These two children were from Mežica (BLL = 64 µg L^{-1}) and Črna na Koroškem (59 µg L^{-1}), areas severely burdened by historical mining and smelting activities. Ordinary Least Squares (OLS) regression results (Table S5) indicated that BLL is 15.97 µg L^{-1} at 0 mg kg^{-1} dust Pb and increases by 0.066 µg L^{-1} on average per 1 mg kg^{-1} increase in dust Pb. The p-value for the slope was < 0.05, indicating that lead concentration is statistically significant in explaining the variation in BLL, and the t-value was > 2, supporting the conclusion that dust Pb is a meaningful predictor. Only about 23% (strength of relationship adjusted to sample size) of the variability in BLL is explained by lead concentration according to the OLS model. Our previous study (Bavec et al., 2025) showed that the major exposure pathway for children in Žerjav, where smelting ore facilities were previously located, was from soil/dust, ranging between 56.9–84.8%. In Črna na Koroškem and Mežica, soil/dust exposure ranged between 25.4–57.8% and 27.6–61.7%, respectively, indicating that dietary exposure predominated in scenarios, that included local foods. The influence of drinking water and outdoor air was negligible (< 5%).

3.3. Health risk assessment

3.3.1. Non-carcinogenic risk

Calculated average daily exposure doses (ADD) for selected metal

Table 2
Estimated hazard quotients for all three pathways and the Hazard Index (HI) for children and adults.

	Children				Adults			
Metal(oid)s	HQing	HQinh	HQder	HI	HQing	HQinh	HQder	HI
As[a]	9.38×10^{-1}	6.89×10^{-6}	3.84×10^{-2}	9.76×10^{-1}	1.00×10^{-1}	2.22×10^{-6}	5.87×10^{-3}	1.06×10^{-1}
As[b]	1.88×10^{-1}	6.89×10^{-6}	3.84×10^{-2}	2.26×10^{-1}	2.01×10^{-2}	2.22×10^{-6}	5.87×10^{-3}	2.60×10^{-2}
Cd	3.32×10^{-2}	1.22×10^{-6}	9.31×10^{-3}	4.26×10^{-2}	3.56×10^{-3}	3.93×10^{-7}	1.42×10^{-3}	4.98×10^{-3}
Cr(VI)	3.75×10^{-1}	1.44×10^{-3}	5.24×10^{-2}	4.29×10^{-1}	4.01×10^{-2}	4.64×10^{-4}	8.01×10^{-3}	4.86×10^{-2}
Cr(III)	7.49×10^{-4}	/	/	7.49×10^{-4}	8.03×10^{-5}	/	/	8.03×10^{-5}
Cu	2.64×10^{-2}	9.67×10^{-7}	2.47×10^{-4}	2.67×10^{-2}	2.83×10^{-3}	3.11×10^{-7}	3.77×10^{-5}	2.87×10^{-3}
Mn	2.69×10^{-2}	2.77×10^{-3}	7.53×10^{-5}	2.97×10^{-2}	2.88×10^{-3}	8.90×10^{-4}	1.15×10^{-5}	3.78×10^{-3}
Mo	2.56×10^{-2}	/	7.16×10^{-3}	3.27×10^{-2}	2.74×10^{-3}	/	1.09×10^{-3}	3.83×10^{-3}
Ni	2.69×10^{-2}	9.61×10^{-7}	2.79×10^{-4}	2.72×10^{-2}	2.88×10^{-3}	3.09×10^{-7}	4.26×10^{-5}	2.93×10^{-3}
Pb	5.73×10^{-1}	2.09×10^{-5}	1.06×10^{-2}	5.83×10^{-1}	6.14×10^{-2}	6.73×10^{-6}	1.62×10^{-3}	6.30×10^{-2}
Zn	3.61×10^{-2}	1.33×10^{-6}	5.06×10^{-4}	3.67×10^{-2}	3.87×10^{-3}	4.27×10^{-7}	7.73×10^{-5}	3.95×10^{-3}

[a] In estimation, a conservative value of 6.0×10^{-5} for RfDing and 0.03 for ABSd was used (USEPA IRIS, 2025).
[b] In estimation, a value of 3.0×10^{-4} for RfDing and 0.03 for ABSd was used (US EPA, 2012).

(loid) elements in household dust are presented in Table S6, while the non-carcinogenic health risk assessment for children and adults is summarised in Table 2. Among the evaluated elements, As and Pb show the highest hazard quotients (HQ) and dominate the overall hazard index (HI) for both age groups. For children, As is the primary risk driver (HI = 0.976), almost entirely due to the ingestion pathway (HQing = 9.38×10^{-1}), which accounts for over 96% of the total HI. It should be noted that for As, the recently updated oral reference dose for ingestion (RfDing) of 6.0×10^{-5} mg kg^{-1} day^{-1} (US EPA, 2025) was used (to our knowledge, this value is not yet widely applied in published studies) instead of the previous value of 3.0×10^{-4} mg kg^{-1} day^{-1} (US EPA, 2012). Therefore, additional scenarios using alternative RfDing values were also evaluated. In addition, a value for ABSd of 0.03 was used (also in Zhao et al., 2021; Hou et al., 2025), while some studies use a value of 0.001 for all metals (Yadav et al., 2019) or 0.03 for children and 0.001 for adults (Doyi et al., 2019). Results obtained using different RfDing values showed that total exposure is strongly influenced by the selected value. For example, the conservative RfDing value (6.0×10^{-5} mg kg^{-1} day^{-1}) resulted in an HQing of 9.38×10^{-1} for children, just below the regulatory threshold of concern, whereas the higher RfDing value (3.0×10^{-4} mg kg^{-1} day^{-1}) resulted in 1.88×10^{-1} (Table 3). However, the use of the conservative RfDing value may lead to an overestimation of risk for children. The enrichment factor (EF) of 1.1 indicates that concentrations of As in the studied area are not elevated and fall within the average range for Slovenian dust; thus, the use of a less conservative RfDing value provides a more realistic estimate for children. For adults, HI values for both scenarios were well below 1, indicating no significant risk.

Lead is one of the major contributors to overall exposure (HQing = 5.73×10^{-1}; HI = 0.583), which reflects the environmental characteristics of the study area. The region has a long history of mining and is still affected by current Pb-recycling operations, both of which contribute to the persistence of lead in the local environment (Šajn, 2002; Miler and Gosar, 2013, 2019). Although the calculated HI values for Pb remain below the regulatory threshold, the well-established neurotoxic effects of lead, particularly its irreversible impacts on cognitive development, behaviour, and the central nervous system, underscore the need for continued vigilance. Even low-level exposure can be harmful, especially for children; therefore, targeted measures to reduce Pb exposure and limit future accumulation in the environment remain necessary.

Results identified oral ingestion as the dominant pathway for all assessed elements (As, Cd, Cr(VI), Cr(III), Cu, Mn, Mo, Ni, Pb), consistent with previous studies (De Miguel et al., 2007; Zhang et al., 2018; Zhao et al., 2021; Kurt-Karakus, 2012). A sensitivity analysis using varied ingestion rates (200/100 mg day^{-1} for children and 100/50 mg day^{-1} for adults) confirmed that ingestion remains the predominant route. For all studied elements, HQing values were several orders of magnitude higher than those for inhalation and dermal contact (Fig. S2), confirming that oral intake is the critical route for human exposure to PTEs in the Meža Valley. The HI values for children and adults, follow the ascending order: Cr(III) < Cu < Ni < Mn < Mo < Zn < Cd < Cr(VI) < Pb < As. These results reflect the distinct geochemical signature of the study area, where concentrations of Pb, Zn, Cd, and Mo are significantly elevated compared to the Slovenian dust baseline due to legacy mining and smelting activities. In the absence of site-specific speciation data, the HRA was modelled for both chromium species: a worst-case scenario assuming 100% Cr(VI) across all pathways and a more realistic scenario assuming Cr(III) dominance via ingestion. This approach likely overestimates the actual risk, as environmental chromium is typically dominated by the less toxic trivalent form. For HQing the range between species spans from 7.49×10^{-4} to 3.75×10^{-1} for children and from 8.03×10^{-5} to 4.01×10^{-2} for adults. Notably, even under the highly conservative assumption that total Cr exists as Cr(VI), all calculated values remain well below the critical threshold of 1.0, indicating that chromium alone does not pose an immediate non-carcinogenic health threat in this indoor environment. Numerous studies have demonstrated that Cr(VI) concentrations typically constitute less than 1–5% of total chromium under natural or moderately contaminated conditions (Choppala et al., 2013). In this context, the Cr(VI) scenario should be interpreted as an upper-bound, worst-case estimate rather than a realistic depiction of environmental exposure. The Cr(III)-based results, supported by literature evidence on typical Cr(VI)/Cr(III) ratios, strongly suggest that actual non-carcinogenic risk from Cr in the study area is likely to be substantially lower than indicated by the Cr(VI)-only model. Furthermore, the high organic matter content (median 18.7%) measured in this study likely facilitates the reduction of Cr(VI) to the less toxic Cr(III) form (Wittbrodt and Palmer, 1995). Consequently, while the assessment based on Cr(VI) is intentionally conservative to ensure public health safety, it likely overestimates the true risk to the local population.

Comparison of exposure profiles reveals that adults consistently have substantially lower HQ and HI values, generally reduced by about one order of magnitude compared to children. Specifically, adult risk values are 8–10 times lower for As, 9–10 times lower for Pb, and approximately 7–12 times lower for most other elements. This disparity reflects lower intake rates per unit of body mass, reduced accidental dust ingestion, and less intensive hand-to-mouth behaviour in adults. Consequently, children face a markedly elevated non-carcinogenic risk profile, with As and Pb emerging as the primary risk drivers. In contrast, adult exposure remains well below established safety thresholds across all pathways, underscoring the pronounced vulnerability of younger populations in environmental health risk assessment.

3.3.2. *Carcinogenic risk*

Carcinogenic risk was estimated for As, Cd, Cr, Ni, and Pb (Table 3) based on the lifetime average daily dose (LADD; Table S6), which accounts for cumulative exposure over a lifespan by integrating childhood and adulthood into a single averaged value (US EPA, 2005; ATSDR (Agency for Toxic Substances and Disease Registry), 2022; Zhang et al., 2018). The cumulative carcinogenic risk (CI) across all assessed metal (loid)s was 9.03×10^{-5} (using the 100% Cr(VI) scenario) and

Table 3
Carcinogenic risks estimated for As, Cd, Cr, Ni, and Pb.

Metal(loid)s	CRing	CRinh	CRder	CI
As	1.11×10^{-5}	6.93×10^{-9}	2.61×10^{-6}	1.37×10^{-5}
Cd	/	1.72×10^{-9}	/	1.72×10^{-6}
Cr(VI)[a]	7.40×10^{-5}	3.79×10^{-7}	/	7.43×10^{-5}
Cr(VI)[b]	3.70×10^{-6}	1.89×10^{-8}		3.72×10^{-6}
Ni	/	3.71×10^{-9}	/	3.71×10^{-9}
Pb	2.24×10^{-6}	6.92×10^{-10}	/	2.24×10^{-6}
$\sum^{100\%\ \mathrm{Cr\ (VI)}}$				9.03×10^{-5}
$\sum^{5\%\ \mathrm{Cr\ (VI)}}$				1.97×10^{-5}

[a] A conservative (worst-case) scenario, assuming 100% of total Cr is present as Cr(VI).
[b] A realistic scenario, assuming 5% of total Cr is present as Cr(VI).

1.97×10^{-5} (using the 5% Cr(VI) scenario). Both values fall within the risk management range of 1×10^{-6} to 1×10^{-4} defined by the US EPA (2005).

The estimated risk for the individual metal(loid)s ranged from 3.71×10^{-9} for Ni to 7.43×10^{-5} for Cr(VI) (worst-case scenario). In the 100% Cr(VI) scenario, hexavalent chromium was the dominant contributor, accounting for approximately 82.3% of the total risk, predominantly through the ingestion pathway. Arsenic (As) contributed about 13.9%, followed by Pb (about 2.5%) and Cd (about 1.7%), while the contribution of Ni was negligible (<0.01%). It is important to acknowledge that the risk for Cr is likely overestimated in the worst-case scenario due to the absence of speciation data. While total Cr was conservatively assumed to be Cr(VI), literature suggests that Cr(VI) in indoor dust typically represents only 1–2% of total Cr (Rasmussen et al., 2001). Thus, the 5% Cr(VI) scenario provides a more plausible estimate. In this scenario, the hierarchy of contributors shifted significantly. Arsenic (As) became as the dominant contributor, accounting for 69.5% of the total cumulative risk (1.97×10^{-5}), followed by Cr(VI) at 18.9%, Pb at 11.4% and Cd at 0.2%. However, the estimated risk for As is likely overestimated due to the use of highly stringent slope factors and conservative exposure assumptions. Furthermore, considering the regional geochemical context, arsenic does not appear to be a problematic element in this area. Its enrichment factors (EF) show no enrichment relative to the Slovenian dust-based median (Fig. 3), indicating that its presence is of natural geogenic origin rather than due to anthropogenic activities. Consequently, the estimated carcinogenic risk for As is likely overestimated due to the application of highly stringent regulatory slope factors and conservative exposure assumptions. The observed risk values therefore reflect the rigorous nature of the risk assessment guidelines rather than localised environmental contamination or industrial emissions.

Dermal and inhalation pathways contributed minimally to the overall risk compared to ingestion. The predominance of ingestion as the primary exposure pathway underscores the need for interventions to reduce oral intake, such as controlling contamination in food and water sources.

4. Conclusions

Despite extensive remediation efforts and outdoor monitoring, indoor contamination by PTEs in the highly polluted Meža Valley, a historical Pb–Zn mining hotspot, has remained insufficiently studied. This research addresses this gap by providing a crucial scientific basis for assessing PTE levels and their sources in household vacuum dust. Statistical analyses identified three dominant anthropogenic sources in household dust: Pb, Zn, and Cd clearly reflect the long-term legacy of historical Pb–Zn mining and smelting activities; Cu and Sn are most probably associated with ongoing lead-recycling activities; while Cr, Ni, and Mo indicate additional contributions from the contemporary steel industry, particularly in the Lower Meža Valley. These sources were consistently supported by enrichment factors, correlation patterns, and hierarchical clustering. For Cd, Mo, As, and Sn, the results and conclusions are based primarily on relative spatial distributions, co-occurrence patterns, and multi-element associations that are internally consistent across the dataset and supported by known geochemical and anthropogenic signatures. This approach enables meaningful interpretation of source-related patterns despite analytical limitations, while acknowledging that improved analytical accuracy would be required for more robust quantitative assessments and comparative risk calculations in future studies. Most importantly, the results demonstrate that indoor dust contamination in the Meža Valley is shaped by both persistent historical pollution and ongoing industrial emissions, each leaving a distinct geochemical signature in the environment. The health risk assessment identifies ingestion as the primary exposure pathway for both carcinogenic and non-carcinogenic risks, with children facing non-carcinogenic hazards 7–12 times higher than adults. In contrast, dermal and inhalation contributions remain negligible, suggesting that mitigation strategies must prioritise reducing incidental dust ingestion. Although the cumulative carcinogenic risk falls within the acceptable regulatory range, these results are influenced by conservative assumptions; specifically, the risks for Cr and As appear overestimated due to a lack of speciation data and the application of stringent toxicity factors for As, which shows no enrichment relative to the Slovenian regional background values. Consequently, environmental management should remain focused on the neurotoxic threats of lead, as indoor dust concentrations are a significant predictor of blood lead levels in most vulnerable populations.

These results provide a baseline prior to the 2023 catastrophic floods, and offer a vital framework for prioritising remediation under the National Environmental Decree starting in 2025. In the wake of flood-driven sediment redistribution, continued monitoring and targeted interventions, such as hygiene education and dust suppression, are essential. Such actions are indispensable for minimising hand-to-mouth transfer, reducing the PTE body burden, and mitigating the long-term health consequences of multi-source exposure in the Meža Valley.

While we acknowledge that the chemical dataset is not analytically ideal for all elements, we stress that this study represents the first and unique assessment of these materials. Sample collection required extensive coordination with local authorities, and access to such samples is exceptionally limited. Despite these limitations, the results offer valuable baseline information and a robust foundation for future investigations, which are outlined as a priority for follow-up studies using improved analytical protocols.

5. Limitations and further research

Several limitations of the present study should be acknowledged. First, the use of household vacuum bags provided by participants may have introduced variability related to vacuum cleaner type, cleaning frequency, and household-specific collection practices. In addition, the cross-sectional design does not fully capture potential temporal or seasonal fluctuations in PTE concentrations, while sample heterogeneity may also reflect differences in household characteristics, such as building age, ventilation efficiency, and the presence of decorative materials. Furthermore, the relatively limited sample size reflects considerable ethical and logistical constraints, as household dust collection depended on voluntary participation and the study design required paired sampling of household dust and children's blood from the same households. Accordingly, the findings should be interpreted with appropriate caution.

A key limitation is that the human health risk estimates were calculated using total PTE concentrations rather than their bioaccessible fractions. This approach is inherently conservative and may overestime of the actual absorbed dose. Given that lead is a potent neurotoxin with no established safe exposure threshold for children, these findings underscore the urgent need for further research into the bioaccessibility of PTEs in the Meža Valley.

To refine these estimates, future research will incorporate probabilistic modelling using Monte Carlo simulations. This will allow a more robust characterisation of uncertainty and variability in exposure parameters, such as ingestion rates and body weight, providing a more realistic distribution of health risks across the affected population. Additionally, we are currently conducting isotope analyses across multiple environmental media, and SEM/EDS analyses planned for household dust, road dust, and soil samples. These advanced analytical techniques will enable more precise source apportionment and provide a deeper understanding of the transport mechanisms of metallurgical fingerprints.

Comparing HRA results across international studies remains challenging due to the lack of standardised exposure factors and statistical distributions. These differences may arise from regional variations in behavioural patterns, demographic characteristics, environmental conditions, and methodological choices. In some cases, the parameters used are not explicitly reported, making comparisons practically impossible. Miletić et al. (2023) reviewed published studies related to HRA in soils and sediments and emphasised the need for standardisation of input data, which is essential to ensure consistency and comparability among risk assessments. To address these barriers and ensure full traceability, we have meticulously documented all input parameters and exposure scenarios in the manuscript and Supplementary materials. This transparency is intended to facilitate more robust benchmarking for future studies in contaminated mining regions.

CRediT authorship contribution statement

Neda Hudopisk: Writing – review & editing, Resources, Data curation. **Matej Ivartnik:** Writing – review & editing, Resources, Data curation. **Teja Čeru:** Writing – original draft, Visualization, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Špela Bavec:** Writing – original draft, Visualization, Methodology, Investigation, Formal analysis, Data curation, Conceptualization.

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Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.ecoenv.2026.120396.

Data availability

Data will be made available on request.

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