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Heavy Metal Contamination in Industrial Surface Dust: Sources, Levels, and Health Risks in Düzce City, Türkiye

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Abstract This study analyzed the surface dust composition to assess environmental pollution in the industrial areas of Düzce. Metal contamination in road dust is concerning because of its toxicity, persistence, and non-biodegradability. Dust samples from 14 industrial zones and one forest control site were compared with reference standards. This study employed a comprehensive methodological approach, including statistical analyses (PCA and HCA) and health risk assessment calculations. The objective was to evaluate the speciation of metallic trace elements and classify the zones according to the dominance of industrial factors. In the study area, the average concentration of potentially toxic elements (PTEs) in decreasing order was Zn (47.59 ± 24.71 mg/kg) > Ni (33.54 ± 12.41 mg/kg) > Pb (32.43 ± 10.54 mg/kg) > Cu (29.14 ± 18.77 mg/kg) > Co (16.40 ± 7.87 mg/kg) > Cd (3.92 ± 1.27 mg/kg). PCA and HCA indicated that vehicle emissions, industrial processes, ferroalloy industries, and coal-fired power plants were the probable sources of these PTEs. Health risk assessment revealed that ingestion was the primary exposure pathway, followed by inhalation and dermal exposure. The non-carcinogenic hazard index (HI) for children and adults followed the order: Cu > Ni > Cd > Zn > Pb. The

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carcinogenic risk for adults ranged from 1.36E-07 to 6.02E-05, whereas for children, it ranged from 1.17E-08 to 4.08E-05. This study concluded that children living in industrial cities, such as Düzce, are particularly exposed to health risks associated with heavy metal pollution in dust. This underscores the imperative need to strengthen air pollution control to protect public health, particularly that of vulnerable populations such as children.

Keywords Anthropogenic activities · Life on land · Road dust · Spatial variability · Sustainable cities and communities

1 Introduction

Industrial sector emissions contribute significantly to environmental pollution, necessitating effective monitoring and mitigation strategies (Liang et al., 2025a). Reducing pollutants released by the industrial and automotive transportation sectors is a crucial target for industrialized nations, requiring both sectors to adopt strategies to decrease atmospheric emissions and mitigate hazardous air pollution (Yang et al., 2024a). Industrial areas in cities are disproportionately exposed to higher levels of pollutants. The resuspension of particulate matter from roadways and adjacent soil is a tangible environmental concern in and around industrial zones. Road dust (RD) indicates atmospheric deposition and accumulation from various sources (Hao et al., 2023). Well-documented health risks (HRs) associated with air pollution include exposure to hazardous pollutants in RD, which impact the immune system, affect breathing and cardiovascular diseases, and promote conditions such as asthma, chronic obstructive pulmonary disease, and lung carcinoma (Ahmad et al., 2021; Bezberdaya et al., 2022). Inorganic and organic contaminants in RD can modulate health risks and depend on the bioavailability and duration of exposure to traffic-related pollutants.

The accumulation of potential toxic elements (PTEs) in the body can lead to organ damage or failure and an increased risk of developing several types of cancer (Faisal et al., 2021). Exposure to RD through the hand-to-mouth pathway and chronic inhalation has been linked to increased morbidity and mortality from various diseases, organ disturbances, cancers, and other chronic ailments, necessitating immediate action to control pollution resulting from industrial activities that degrade the environment (Bahinskyi et al., 2025; Chakraborty et al., 2024). PTEs are considered to be among the most harmful environmental pollutants. Even at minimal concentrations, some PTEs can exhibit toxic, carcinogenic, and lethal effects on human and environmental health (Aguilera et al., 2021; Sharma & Sarkar, 2023). Furthermore, although some PTEs are essential in trace amounts for organisms' nutritional needs, it has been suggested that high concentrations of these substances can be harmful (Heidari et al., 2021). Therefore, appropriate monitoring of changes in PTE contamination and the implementation of measures to mitigate their effects are crucial. PTEs, while injurious to human health and other organisms, have a much greater impact when absorbed through respiration (Faraji et al., 2023). PTEs are released into the environment primarily through mining (Ou et al., 2022), traffic (Mahmoud et al., 2023), and industrial activities (Zhu et al., 2021). Among these pollutant sources, the most diverse releases are due to industrial activities, and studies have shown that pollution levels are higher around industrial facilities (Jeong et al., 2020). After PTEs leave their source, they mix with air, cling to particulate matter, and can be carried away by wind (Rajfur et al., 2025). However, they accumulate more on upper soil surfaces around industrial facilities because of gravity. Therefore, particulate matter known as RD, specifically the layer on the upper surface of the soil, is routinely used as an indicator to assess variations in PTE contamination in the atmosphere (Öncü et al., 2025; Xiao et al., 2020).

Düzce city has a mixed land-use pattern of industrial, residential, and agricultural areas (Koç et al., 2025). Major industrial facilities, including automotive factories and wood-processing facilities, are located in Düzce. The rapid increase in industrialization and urbanization contributes to increased traffic and atmospheric deposition, leading to elevated levels of PTEs and other pollutants in surface soils (Yang et al., 2022). The majority of these factories belong to the automotive and wood industries, as well as foundries and metal-cutting. These industries can emit PTEs such as As, Cd, Cr, Cu, Ni, Pb, and Zn, which can accumulate in the surrounding environment and pose risks to human health and the environment. Although industrial entities are the predominant sources of PTE pollution, distinct industrial operations release various PTEs at different concentrations, leading to heterogeneous PTE contamination (Özlü, 2024; Lu et al., 2025). Most studies have focused on the characterization of RD, and few have incorporated source identification and apportionment (Chen et al., 2022; Roy et al., 2024). Surface soil dust may serve as a significant reservoir for PTE distribution, which can enter the human body through direct ingestion, inhalation of dust, or consumption of contaminated plants grown near roads with heavy traffic or in polluted industrial zones (Abdulaziz et al., 2022; Respondek et al., 2024). Similar studies have been conducted in Türkiye to assess the origins and potential health hazards associated with PTEs in surface soil dust (Birgül, 2024; Ulutaş, 2022). In addition, the long-term effects of industrial operations on surface soil dust composition have not been sufficiently clarified. This study seeks to fill existing gaps by examining the PTEs of RD within Düzce's industrial area. This study focuses on (1) identifying the factors that affect RD development, including various metal interactions; (2) categorizing zones impacted by human activities through principal component analyses of RD chemistry; and (3) using the spatial distribution of PTEs to assess potential ecological and human risks for both adults and children, considering different exposure pathways.

2 Materials and Methods

2.1 Study Area

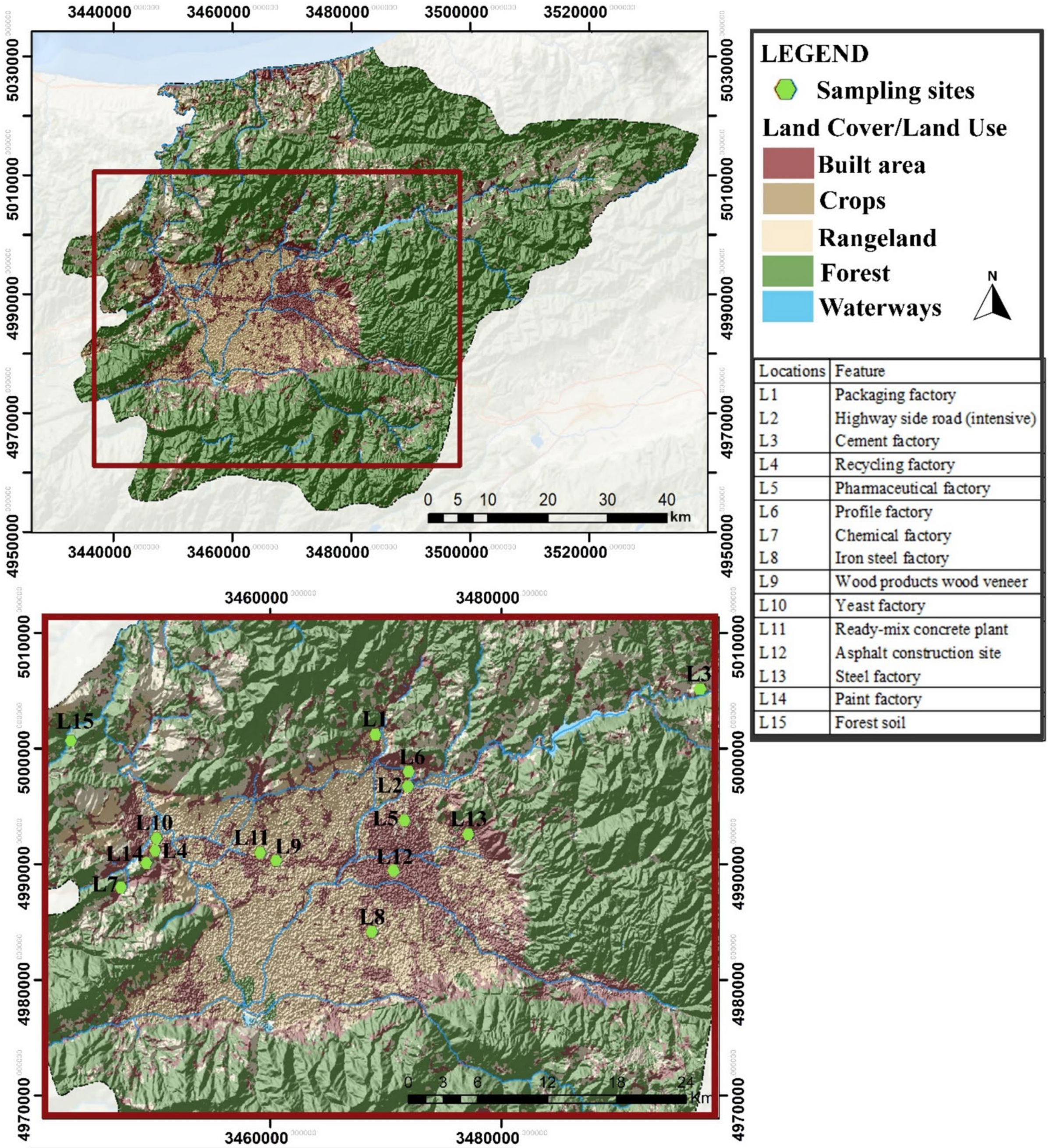
The study was conducted on dust collected from around industrial establishments in and around Düzce city, where they are densely located. It is stated that Düzce province has the 5th most polluted air in Europe. Studies have revealed that PTE pollution is also very high in Düzce (Koc et al., 2024). Samples were collected from dust on the upper soil surface around 14 factories operating in different sectors and locations (Fig. 1). As a reference for the study, samples were also collected from a forest area where no industrial establishments existed. Thus, the study was conducted at 15 locations.

2.2 Data Collection

Soil surface dust was collected using a small brush and plastic shovel, and then stored in clear plastic bags with labels (Jancsek-Turóczi et al., 2013). Three representative sample points were meticulously selected at each designated site. The samples were sieved in the laboratory, placed in appropriately labeled glass Petri dishes, and dried in an oven maintained at 45 °C for 2 weeks. 0.5 g portions of each dried sample were sieved through a 2-mm sieve and preserved in clean containers for subsequent physical and chemical analysis (Han et al., 2011). They were then combined with 6 mL of 65% HNO_3 and 2 mL of 30% H_2O_2 and processed in a specialized microwave oven. The solutions obtained were then poured into flasks and distilled to a final volume of 50 mL using ultrapure water. PTE concentrations were determined by analyzing the prepared samples using inductively coupled plasma-optical emission spectrometry (ICP-OES, SpectroBlue, Spectro, Germany) and applying the appropriate dilution factor. This methodology has recently become common for assessing PTE levels in RD and soil sample (Neisi et al., 2016).

2.3 Statistical Methods

The Statgraphics Centurion 19 statistical package was used to analyze all measurement results. Data evaluation was conducted using a 95% confidence interval. The statistical package SPSS was used, with descriptive statistics, Box & Whisker plots (to visualize the best features in a database by analyzing the distribution of output parameters), Analysis of Variance (ANOVA), correlation, factor analysis, and cluster analysis. Scatter plot of the original monitoring dataset, which includes clusters of isolated outliers, normal points, and clusters of outliers. To verify the principal component analysis (PCA) and hierarchical clustering analysis (HCA), we systematically clustered the eight metals. This procedure created one cluster from the 45 supplied variables. The most favorable outcomes for HCA were achieved using auto-scaled datasets, applying the squared Euclidean distance, and implementing Ward's hierarchical clustering technique. In this analysis, the correlations among the measured elements provided preliminary information. The values of these correlation coefficients ranged from −1 to + 1, indicating the strength of the linear association between the variables. The closer the correlation coefficients are to 1, the more likely it is that the parameters are excreted from the same source or in the same chemical form as the



Locations	Feature
L1	Packaging factory
L2	Highway side road (intensive)
L3	Cement factory
L4	Recycling factory
L5	Pharmaceutical factory
L6	Profile factory
L7	Chemical factory
L8	Iron steel factory
L9	Wood products wood veneer
L10	Yeast factory
L11	Ready-mix concrete plant
L12	Asphalt construction site
L13	Steel factory
L14	Paint factory
L15	Forest soil

Fig. 1 Sampling sites in Düzce city

other parameters. If the correlation coefficient is positive (+), the other parameter also increases as the value of one parameter increases. If the correlation coefficient is negative (-), the other decreases as one value increases. The study results were subjected to analyses to identify the sources of metals and their associations and to determine whether significant differences exist among sites and within the RDS. The Duncan test was employed to assess variables that showed statistically significant differences at a minimum confidence threshold of 95% ($P < 0.05$). The findings were summarized in a tabular format.

2.4 Analytical Quality Control

To ensure the precision and accuracy of our laboratory experiment, we implemented Quality Assurance/Quality Control (QA/QC) measures using 10% (n = 9) certified reference materials (CRM), 2% (n = 2) blank samples, and 10% (n = 9) replicate samples for all analyses. The accuracy of the method was assessed by examining a reference standard material (SRM) from the International Atomic Energy Agency (IAEA) as a quality control measure (soil 7). The SRM analysis indicated that the accuracy was within ± 5%. Sampling was conducted in the wind direction. During the sampling process, the operators wore cotton lab coats, nitrile gloves, and cotton boot covers to shield themselves from plastic fibers, rubber from shoes, and other contaminants. Five standard solutions were used to calibrate the ICP-OES device, resulting in linear calibration curves with r2 > 0.99. Samples were analyzed in duplicate, with relative standard deviations (RSDs) remaining below 20%. Random sample repeats were conducted every 15 samples to verify the accuracy. Reagent blanks showed concentrations below the instrument detection limits, and blank corrections were applied as needed. All analyses were conducted under optimized conditions with a signal-to-noise ratio of 10:1.

2.5 Human Health Risk Assessment (HRA)

In line with the United States Environmental Protection Agency (USEPA) guidelines from 1989, HRA models were employed to assess carcinogenic risk (CR), the lifetime average daily dose (LADD), and non-carcinogenic risk, which encompasses the Hazard Index (HI) and hazard quotient (HQ) for both adults and children. The average daily dose (*AAD*, mg/kg/day) was calculated for HRA following USEPA (1996, 1997) guidelines, taking into account ingestion (AAD_{Ing}, mg/kg/day), inhalation (AAD_{Inh}, mg/kg/day), and dermal absorption (AAD_{der}, mg/kg/day) pathways, as detailed in Table S1 and the subsequent equations:

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

$$AAD_{Inh} : C \times \frac{InhR \times EF \times ED}{PEF \times BW \times AT} \quad (2)$$

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

$$HI : \sum HQ_i \quad (4)$$

$$HQ : \frac{ADD}{RfD} \quad (5)$$

$$LADD = \frac{C \times \mathrm{EF} \times \mathrm{IngR} \times \mathrm{ED}}{\mathrm{PEF} \times \mathrm{AT} \times \mathrm{BW}} \quad (6)$$

$$\mathrm{CR} : LADD \times \mathrm{SF} \quad (7)$$

where C is the concentration of contaminant in the medium (mg/kg), EF is the exposure frequency (days/year), ED is the exposure duration (years), BW is the body weight (kg), and AT is the averaging time (days). A Hazard Index (HI) exceeding 1 indicates a potential non-carcinogenic effect of toxic metals. At the same time, an HI below 1 implies no HRA and corresponds to the reference dose (RfD) for a specific toxic metal. Regarding Cancer Risk Assessment (CRA) with a one in a million chance, a value less than 10^{-6} is considered an acceptable threshold. The qualitative definitions of ILCR are as follows: very low risk ($\leq 10^{-6}$), low risk (10^{-6}–10^{-4}), moderate risk ($\geq 10^{-4}$–10^{-3}), high risk (10^{-3}–10^{-1}), and very high risk ($\geq 10^{-1}$).

3 Results and Discussion

3.1 Assessment of Metal Concentration in the Road Dust

The variations in trace element concentrations of Mg, Ca, K, Fe, Al, P, S, Mn, Sc, Bi, B, and Na are presented in Table S1, the changes in As, Ba, Be, Sb, Cd, Ni, Cu, Pb, Co, Ga, In, Li, Pt, and Sn concentrations in Table S2, the changes in Se, Ag, Tl, Zn Sr, V, Pd, Au, Ge, Mo, and Nb concentrations in Table S3. determined by ANOVA and Duncan tests. A summary of the descriptive statistics for each metal concentration is given in Table 1. According to the statistical analysis conducted in this study, the elements with the highest averages are the most variable PTEs, followed by Fe (22,782 mg/kg), Mn (12,694 mg/kg), Ca (40,748 mg/kg), K (3632 mg/kg), and Al

(35,847 mg/kg). The elements with the lowest averages are as follows: Be (2.37 mg/kg), Cd (3.92 mg/kg), Ag (9.86 mg/kg), and Bi (10.32 mg/kg).

The present study shows that Fe is abundant in some natural environments, but accumulation with Ni, Cu, Zn, Cr, Cd, and Pb may indicate industrial and traffic sources (Long et al., 2021). Brake linings, tire dust, and components are among the sources related to the transport sector. Similarly, Shahab et al. (2023) synthesized 114 studies from the past two decades, encompassing over 5000 sampling locations across 61 cities in 21 countries, to evaluate the current status and potential risks of PTEs in urban RD. The review summarizes the concentrations, origins, distributions, health implications, sampling protocols, and analytical techniques employed in PTEs research related to urban RD (Wang et al., 2025). Key findings reveal that Cd, Zn, and Cu concentrations in numerous urban RD samples worldwide exceed the Grade II limits set by Chinese regulations for allowable PTE concentrations in soil. Kabir et al. (2022) investigated the contamination, spatial distribution, and potential ecological and human HRs of PTEs (Pb, Co, Cr, Cd, Zn, Ni, and As) in RD samples from the Kushtia-Jhenaidah national highway in Bangladesh, considering traffic load and land-use types. The mean concentrations were: Pb (50.8 ± 15.7 mg/kg), Cu (82.6 ± 17.2 mg/kg), Cr (20.1 ± 5.7 mg/kg), Cd (0.18 ± 0.06 mg/kg), Zn (126.2 ± 40.7 mg/kg), Ni (14.1 ± 4.7 mg/kg), and As (0.12 ± 0.04 mg/kg). Pb, Cu, Cd, and Zn levels were notably elevated compared to background values. Another study by Mostafa et al. (2024) examined environmental quality in Greater Cairo, which is declining due to traffic, overpopulation, and urban planning issues. RD samples from six major roads were collected and analyzed by ICP-OES to assess pollution levels using indices. Multivariate analysis was used to identify metal sources, geochemical associations, and ecological/HRs. The average PTEs concentrations, in descending order, were: Fe (1181.12 mg/kg), Zn (201.36 mg/kg), Pb (66.10 mg/kg), Ni (32.42 mg/kg), Cu (26.07 mg/kg), Cr (26 mg/kg), Co (14.78 mg/kg), As (2.76 mg/kg), and Hg (0.93 mg/kg). An analysis of RD in Delhi by Jangirh et al. (2024) reveals elevated levels of particulate matter and PTEs, particularly in areas with high traffic and industrial activity. The average PTEs concentration follows this decreasing sequence: Cr > Zn > Cu > Co > Pb > Ni > Cd.

Han et al. (2023) analyzed RD samples (n = 140) from Anyang, China, and they were analyzed for ten PTEs. HRA were quantified using a Monte Carlo simulation and bioaccessibility analysis. Results indicated substantial Mn, Zn, Pb, Cd, and Cu accumulation. Mean concentrations of Mn, Zn, Pb, V, Cr, As, Cd, Ni, Cu, and Co were 1244.1 mg/kg, 332 mg/kg, 88.7 mg/kg, 86.8 mg/kg, 97.6 mg/kg, 18.4 mg/kg, 2.4 mg/kg, 19 mg/kg, 53.2 mg/kg, and 7.4 mg/kg, respectively. Concentrations of most PTEs exceeded regional background levels for Henan Province, China; Zn, Pb, and Cd concentrations were notably elevated. RD samples from five areas in Kunming were analyzed for PTEs levels, distribution, sources, and bioaccessibility by Yang et al. (2024b). The study also investigated the adverse impacts of dust extracts on human corneal cells and their mechanisms. The levels of As, Cd, Cr, Cu, Pb, and Zn in RD were greater than in the soil, with commercial and residential sites showing the most pollution. Suvetha et al. (2022) examined the concentration of six PTEs in RD from Tiruchirappalli, a Tier II city in South India. Five metals were identified, while cadmium was below detection limits in all 22 samples. The mean metal concentration followed this order: Fe (506.39 µg/g) > Zn (47.08 µg/g) > Cu (11.84 µg/g) > Cr (11.47 µg/g) > Pb (0.24 µg/g). Higher metal counts were observed in the outer city than in the inner city, likely due to greater industrial activity and less frequent cleaning.

As presented in Fig. 2, there was a considerable variation in PTE concentrations ($p < 0.05$, one-way ANOVA) between metals. Studies have found that the levels of PTEs in the analyzed dust samples in Baiyin, China were the mean concentrations of Cr, Mn, Ni, Cu, Zn, Cd, Pb, As, and Hg were determined to be 67.59 mg/kg, 414.10 mg/kg, 171.75 mg/kg, 91.36 mg/kg, 369.92 mg/kg, 2.94 mg/kg, 71.27 mg/kg, 6.37 mg/kg, and 1.46 mg/kg, respectively with the exceptions of Mn and As, the concentrations of PTEs were all observed to be greater than the baseline levels in Gansu soil. In particular, Cd and Hg exhibited substantially higher concentrations, exceeding the soil background values by 33.79 and 85.88 times, respectively (Gu et al., 2025). Other studies worldwide have also reported similar findings (Howlader et al., 2025; Simatupang et al., 2025).

Table 1 Statistical evaluation of measured metals (mg/kg)

Elements	Count	Minimum	Maximum	Average ± Standard deviation	Median
Mg	45	3,482.30	28,263.80	12,694.20 ± 6989.08	11,393
Ca	45	2,257.50	147,904	40,748.90 ± 40,324.50	23,290.80
K	45	1,704.80	8,379.30	3,632.09 ± 1567.57	3,248.50
Fe	45	11,273.80	41,849.50	22,782.70 ± 9320.37	24,634
Al	45	13,057	75,296.00	35,847.80 ± 18,579.80	26,387.30
P	45	359.80	1,396.80	658.28 ± 267.24	535.80
S	45	489	5,889	1,569.90 ± 1370.09	1,079.50
Na	45	416.00	2,942.80	1,225.58 ± 851.04	930.80
Mn	45	375.30	5,190.30	1,135.57 ± 1162.16	850.80
Sc	45	63.30	82	70.38 ± 5.46	67.80
As	45	13.51	41.57	27.22 ± 7.53	26.36
Ba	39	119.52	124.25	121.90 ± 2.53	121.92
Be	45	1.47	4.94	2.37 ± 0.84	2.25
Bi	45	6.55	14.28	10.32 ± 2.25	10.14
B	42	65.42	67.16	66.24 ± 0.91	66.16
Cd	45	2.38	6.44	3.92 ± 1.27	3.94
Co	45	7.93	33.57	16.40 ± 7.87	12.06
Cu	45	4.14	79.68	29.14 ± 18.77	23.57
Ga	45	68.16	125.26	87.84 ± 17.35	80.69
In	45	12.66	49.41	22.20 ± 7.88	20.23
Pb	45	22.19	60.52	32.43 ± 10.54	30.41
Li	45	15.20	34.21	22.07 ± 5.34	20.43
Ni	45	18.67	60.01	33.54 ± 12.41	28.13
Se	45	18.80	55.85	31.89 ± 10.28	28.95
Ag	45	8.29	11.06	9.86 ± 0.60	9.86
Sr	42	132.87	140.06	136.40 ± 3.80	136.32
Tl	45	14.46	38.23	22.71 ± 6.79	20.33
V	42	103.31	107.80	105.58 ± 2.41	105.64
Zn	45	21.78	112.55	47.59 ± 24.71	40.42
Au	45	14.83	106.02	29.10 ± 22.16	23.11
Ge	45	10.05	44.84	31.61 ± 8.09	32.19
Mo	45	16.73	34.27	22.53 ± 5.14	22.22
Nb	45	42.85	128.57	67.16 ± 22.82	60.52
Pd	45	52.66	70.38	61.68 ± 3.74	61.21
Pt	45	61.66	94.40	72.74 ± 9.49	69.84
Sb	45	43.30	130.07	74.68 ± 25.44	66.58
Sn	45	21.87	73.47	39.78 ± 12.75	37.44
Ti	45	425.80	3,789.38	1,700.27 ± 1,036.32	1,311.83
Zr	45	15.19	102.58	46.73 ± 25.38	35.91

3.1.1 Correlation analysis

This indicates a relationship between the two parameters. The correlations between the measured elements are shown in Table 2, and a scatter plot matrix of Mn, As, Cd, Co, Cu, Pb, Ni, Zn, Sn, and Ti is shown in Fig. 3a. The highest positive relationships were found to be between Co-Sn (0.9117), As-Cd (0.8524), Pb–Zn (0.8462), Co-Ti (0,8351), and Sn-Ti (0.8405).

3.1.2 Factor analyses (FA)

The Kaiser–Meyer–Olkin (KMO) measure of sample adequacy is approximately 0.70, higher than 0.6. The consequence of Bartlett's Test of Sphericity is 0.00, less than 0.05. Eight metals were classified into three categories, with the three main components accounting for 85.573% of the variance. The first and second components were 53.543% and 32.031%, respectively. Accordingly, the rotation composition matrix (Table 3) shows that As, Cd, Co, Sn, and Ti have higher values in the first factor, while Mn, Cu, and Zn have higher values in the second factor.

Mn and Cu had higher concentrations in the iron factory (8), while As and Zn had higher concentrations in the cement factory (3). In the recycling factory (4), Co, Ti, and Sn also had higher concentrations. The main sources of As, Cd, Co, Sn, and Ti were industry and transportation, while the primary sources of Mn, Cd, Cu, and Zn were industry, traffic, tires, and soil. The 2D scatterplot shows the factors for eight PTEs in various locations, as depicted in Fig. 3b.

3.1.3 Principal Components Analysis (PCA) and Hierarchical Cluster Analyses (HCA)

HCA was performed to explore further the correlation between the metals studied in the soil shown in Fig. 3c. PCA and HCA results showed the possibility that Mn, Cu, and Zn came from one source, while As, Cd, Co, Sn, and Ti came from another source in Fig. 3d.

3.1.4 HRA

The risk assessment of human health for children and adults from ingestion, inhalation, and dermal contact with PTEs present in RD across various functional areas is shown in Table 4. The HQs and HI values for both children and adults exposed to PTEs in RD were considered. The functional areas were ranked as rigid pavement, Cd, Pb, and Ni, with values significantly higher than those of Zn and Cu. The HQs and HI were <1, indicating that no non-carcinogenic significant risk was found in the study area. The results indicated that no substantial non-carcinogenic risk was identified in the study regions for any of the assessed PTEs, as both HQs and HI values were less than 1. The cancer risks (CR) of Cd, Ni, and Pb were estimated for children and adults. The Pb values were lower than 1×10^{-6}, whereas the Cd and Ni values were found between 10^{-6} and 10^{-4}. This conclusion is based on the reports of Bourliva et al. (2017) and Taiwo et al. (2020). The lower HI value (<1) for all metals in children and adults shows no non-carcinogenic risks for other PTEs. The HI values for adults and children in the study area decreased in the following order: Cu > Ni > Cd > Zn > Pb. The LADD for adults ranged from 7.64E-04 to 5.68E-03, whereas for children, it ranged from 2.05E-05

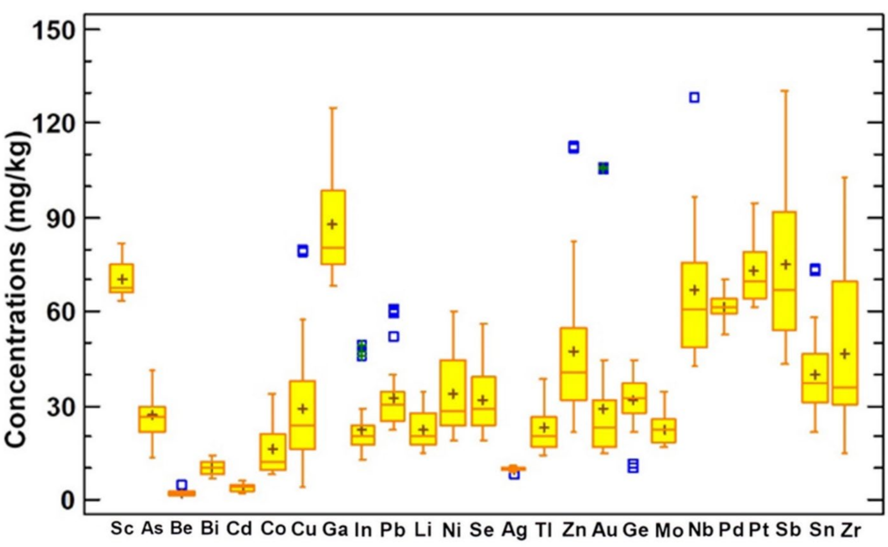


Fig. 2 Box & Whisker plot of metals

Table 2 Correlation chart between measured metals

	As	Cd	Co	Cu	Pb	Ni	Zn	Sn	Ti
Mn	0.1518	0.5466**	−0.0383	0.7841**	0.1376	−0.1796	0.4861**	−0.0217	0.0011
As		**0.8524****	0.7799**	0.5237**	0.3379*	0.4055**	0.3639**	0.7958**	0.5239**
Cd			0.6797**	0.754**	0.2527	0.1871	0.4973**	0.646	0.4975**
Co				0.1166	−0.0748	0.1045	−0.1166	**0.9117****	**0.8351****
Cu					0.4663**	0.1791	0.7484**	0.2279	0.0443
Pb						0.0993	**0.8462****	0.0247	−0.179
Ni							0.1659	0.0838	−0.263
Zn								−0.0881	−0.2604
Sn									**0.8405****

**p-value < 0.05, ** p-value < 0.01*

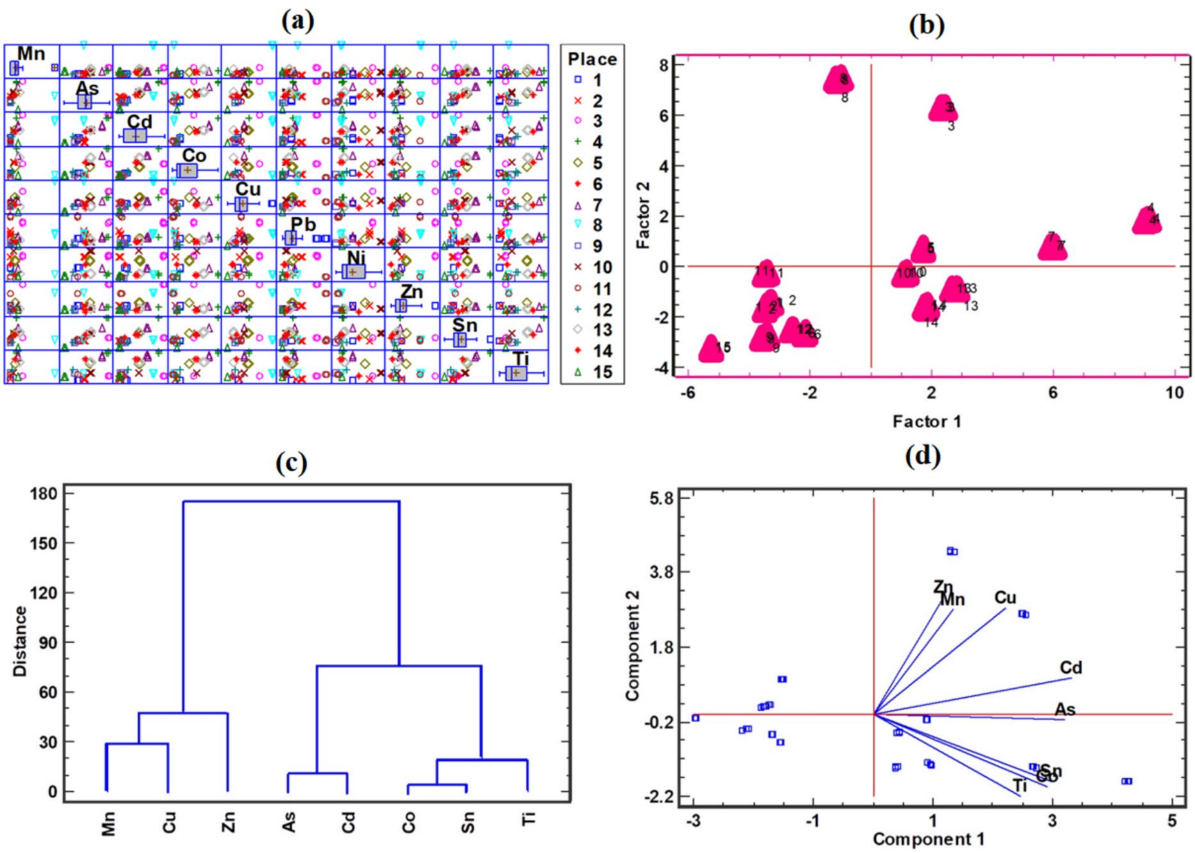


Fig. 3 **a** Scatter plot matrix of PTEs, **b** 2D Scatterplot factors for PTEs in various locations, **c** Dendogram graphic by using Ward's Method, Squared Eulidean, and (**d**) PCA for PTEs (2D Components Plot)

to 1.52E-04. The CR for adults ranged from 1.36E-07 to 6.02E-05, whereas for children, it ranged from 1.17E-08 to 4.08E-05. CR(Ni) = 1.75E-04 exceeds the "acceptable" range (10^{-6} – 10^{-4}). Liu et al. (2023) emphasize that the non-carcinogenic risk of heavy metals, particularly Cr, Cd, and Pb, is significantly higher for children compared to adults, with children facing about ten times the risk in Ma'anshan, China.

Table 3 Factor loading matrix after rotation

Elements	Factor 1	Factor 2
Mn	−0.0246	**0.8167**
As	**0.8132**	0.4210
Cd	**0.6937**	**0.6848**
Co	**0.9734**	−0.0264
Cu	0.1814	**0.9494**
Zn	−0.1074	**0.8659**
Sn	**0.9674**	0.0189
Ti	**0.8881**	−0.1439

Rehman et al. (2023) showed the hazard quotient (HQ) for Zn was greater than 1 in both industrial and heavy transport areas, highlighting its potential health impacts. Health risk research has highlighted a low non-carcinogenic risk (HQ < 1) and probable carcinogenic risk for arsenic (As) through ingestion, with higher risks for children than for adults, as highlighted by Othman and Latif (2020). However, most cancer risk research has focused on Pb and Cd rather than Zn and Cu because inhalation exposure was assessed to be low. The highest risk for non-cancer effects from heavy metal exposure in children and adults was associated with the ingestion of dust particles, particularly for Pb and Cd (Zheng et al., 2010).

Differences in PTE pollution across RDs can variably impact human health by age. Our findings indicate that children face a significantly higher HRA than adults, supporting this hypothesis. The health index for children surpassed the safety threshold in all water bodies, unlike adults, suggesting a considerable non-carcinogenic risk from surface dust exposure, which is consistent with prior research (Dey et al., 2024). This increased vulnerability in children, potentially due to weaker immune systems and lower health literacy, aligns with previous studies (Najmeddin et al., 2018; Xu et al., 2025). Therefore, strategies to minimize children's exposure to PTEs in surface water are essential. Our findings were similar to those of Isinkaralar et al. (2024) in Düzce. Previous research has found that five PTEs are the most studied pollutants because of their potential health hazards. Although inhalation is negligible, ingestion poses a higher health risk, particularly in children (Chen et al., 2024). Hou et al. (2019) reviewed research showing that HI values for adults were generally below the safe level of 1.0, except for lead (Pb) in Daye City for children, which had an HI of 1.074, indicating potential health risks. The HI was an effective tool for HRA, and the potential contamination levels were as follows: Pb > Cu > Ni > Cr > Cd. The non-carcinogenic risks of Pb were higher than those of the other metals for both children and adults. The HI values for Cd, Pb, and Ni in children were 1.25–1.61, 2.93–3.74, and 1.00–1.14, respectively, and the HI for Cd was 1.05–2.56. Children exhibited higher HI values than adults, suggesting that industrial activities and traffic density significantly

Table 4 Deterministic Human Health Risk Assessment (HRA) average results for adults and children

Risk	Gender	Metals (mg kg^{-1})				
		Cu	Cd	Ni	Pb	Zn
HQ_{der}	Adult	6.83E-05	1.10E-02	1.87E-05	1.74E-03	2.23E-05
	Children	4.47E-05	7.22E-03	1.14E-03	1.14E-04	1.46E-05
HQ_{ing}	Adult	3.06E-02	1.03E-02	1.86E-02	5.73E-02	2.63E-02
	Children	1.43E-01	4.80E-02	8.66E-02	2.51E-01	2.33E-01
HQ_{inh}	Adult	7.33E-07	9.86E-08	8.43E-07	8.15E-07	1.20E-06
	Children	3.52E-06	4.70E-07	4.05E-06	3.92E-06	5.75E-06
HI	Adult	3.07E-02	2.13E-02	1.86E-02	5.54E-02	2.63E-01
	Children	1.43E-01	5.52E-02	8.67E-02	2.52E-01	2.33E-01
LADD	Adult	5.68E-03	7.64E-04	6.53E-03	6.32E-03	9.27E-03
	Children	1.52E-04	2.05E-05	1.75E-05	1.69E-04	2.48E-04
CR	Adult		6.02E-05	1.75E-04	1.36E-07	
	Children		1.40E-06	4.08E-05	1.17E-08	

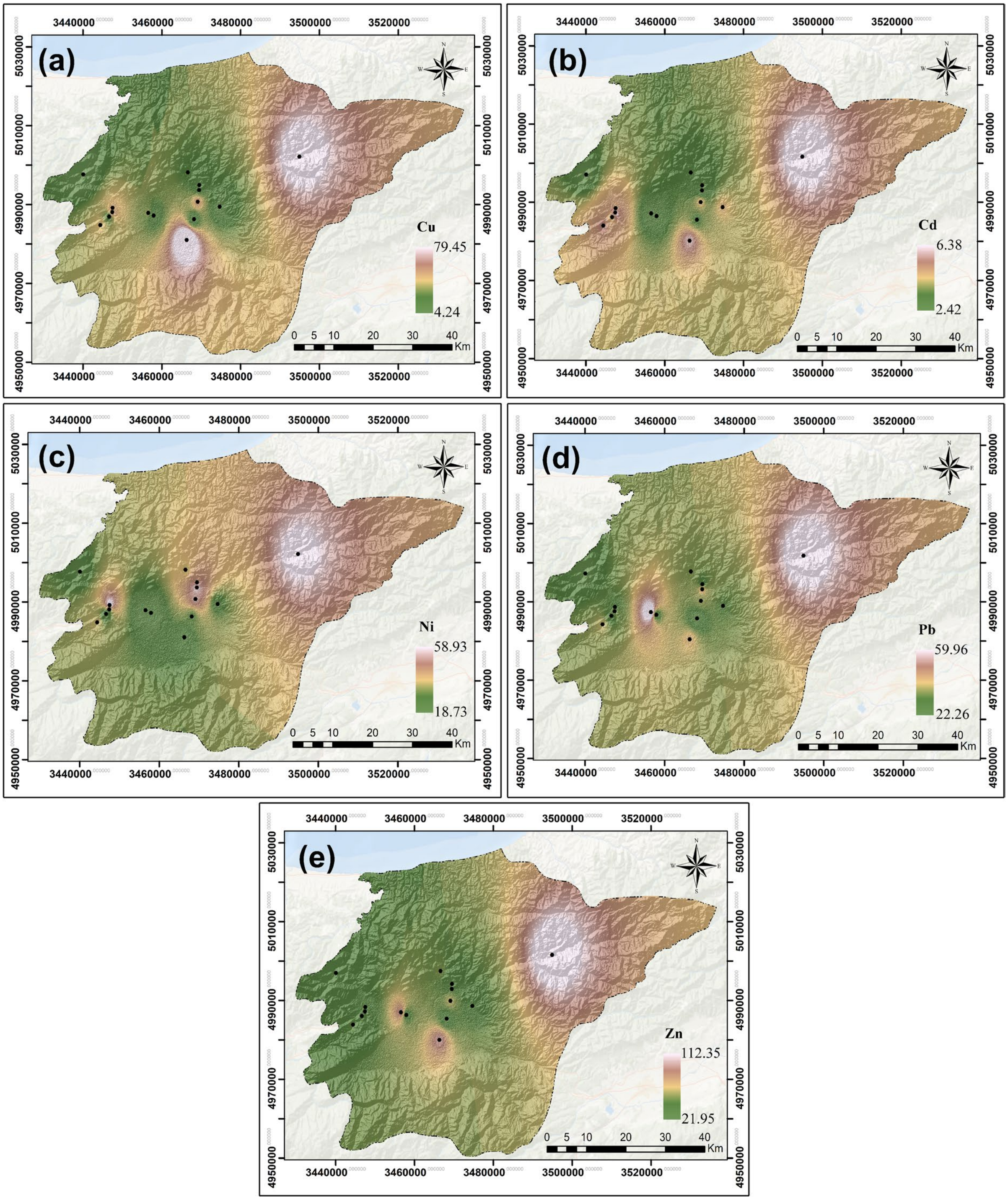


Fig. 4 Spatial distribution patterns of (**a**) Cu, (**b**) Cd, (**c**) Ni, (**d**) Pb, and (**e**) Zn in the RD

contribute to RD pollution. Liang et al. (2025b) found that HI values followed the order: As > Cr > Pb > Mn > Cu > Ni > Zn > Cd. Notably, the HI values for As, Cr, and Pb were approximately 1, indicating potential HRs. Moreover, the total non-carcinogenic risk (HI) for children was ranked As > Pb > Cr > Cd > Zn

> Ni > Cu > Hg, with children experiencing a higher HI than adults, primarily because of hand-mouth intake (Cui et al., 2020). They found that the carcinogenic risk for As, Cd, Cr, and Ni was significantly lower than the threshold of 10^{-6}—10^{-4}, suggesting that respiratory intake does not pose a carcinogenic risk. Due to their high toxicity, even low levels of As, Cr, and Pb require careful regulation to prevent serious health issues. Children are particularly vulnerable to Pb and Cr, with emerging neural health systems and high absorption rates. This is supported by a previous study in Assiut City, Egypt, and Southwest China reported by Soliman et al. (2025) and Huang et al. (2025). These findings do not closely resemble those of a previous study, which indicated the following order for children: As > Pb > Cr > Co > Ni > Cd, and for adults: Pb > As > Cr > Co > Ni > Cd (Okoro et al., 2025). These data support the existing literature, highlighting the ingestion pathway as a primary contributor to adverse health outcomes in humans (Isinkaralar et al., 2026; Jadoon et al., 2025; Sama et al., 2025).

3.2 Spatial Distributions of PTEs in RD

Figure 4 shows the spatial distributions of Cu, Cd, Ni, Pb, and Zn in RD throughout the Düzce industrial site. The high concentrations of PTEs were mainly distributed in the steel factory (L13). At the same time, the highest values of Cu, Cd, and Zn are found in the iron steel factory (L8), Cd and Cu are mainly from traffic sources (Amin et al., 2025). In contrast, the lowest Pb concentrations were mainly observed in L1, L2, L5, L6, L8, and L12 from wheel weights and vehicle tires. The concentrations of Cd and Ni varied considerably among L7, L10, L13, and L14, owing to the fact that industrial emissions and urban construction sources were significantly higher in other regions. In addition, Ni may arise from urban construction pollution (Amjadian et al., 2016).

Traffic and industrial emissions, industrial and domestic sewage discharge, and paddy field runoff are potential sources of the PTEs examined (Duan et al., 2020). PCA/FA analysis indicated three primary pollution sources for the eight PTEs: the first source, responsible for Pb, Cu, Zn, and As, likely originated from traffic, vehicle emissions, and industrial activities (Zhu et al., 2021). Local traffic and industrial emissions are known sources of Pb, Cu, and Zn in $PM_{2.5}$-bound heavy metals for air pollution management (Men et al., 2018; Rahat et al., 2025). RDs can contain Pb, Cu, Zn, and As from sources such as decomposed residues, pavements, tires, and emissions (Zhang et al., 2015). Cd may originate from soil erosion or runoff from agricultural land. Cd can originate from mining, emissions, and fertilizers containing Cd, and it can mix with organic matter in the environment, increasing its presence (Okorie et al., 2012). Agricultural soils can contain high levels of Cd due to the use of fertilizers or compost. Pb and Cd had similar distribution patterns, with high pollution in the central, northeastern, and western areas. Ni is mainly found in the central and southern regions, whereas Cu is mostly found in the northeastern and southern parts of the city. Generally, Pb, Cd, Cu, Zn, and Ni concentrations are notably greater in northeastern than in northwest Düzce. The following section describes the factors affecting the spatial distribution of PTEs in RD.

4 Conclusion

Numerous PTEs assessed during the study posed significant risks to HRA and environmental health, even at low levels. Certain PTEs can be toxic, harmful, carcinogenic, and even deadly to humans, even at low concentrations. Inhalation and ingestion by workers in industrial facilities subject to the study can cause significant HRs. Therefore, it is recommended that workers in factories such as recycling, yeast, and chemical factories, where PTEs concentrations were determined to be high within the scope of the study, be careful, warned about HRs, take necessary measures for health, and have regular health checks. The same risk applies to residential areas around the factories. It is necessary to determine which areas are risky in terms of health, conduct analyses around these factories to prevent residence in risky areas if necessary, and warn people living there about HRs. The study was conducted in the region, including industrial establishments, residential areas, and areas where industrial establishments are intertwined with residential areas. Agricultural products grown in these regions can cause serious HRs by absorbing PTEs that are harmful to health. It is recommended that the products grown in this region be controlled, the risky ones be identified and destroyed if necessary, and the cultivation of agricultural products in this region be banned.

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Data Availability The data that support the findings of this study are available from the corresponding author, upon reasonable request.

Declarations

Ethics Approval All authors have read, understood, and have complied as applicable with the statement on "Ethical responsibilities of Authors" as found in the "Instructions for Authors."

Consent to Participate Not applicable.

Consent to Publish Not applicable.

Competing interest The authors declare no competing interests.

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