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Assessment of landfill leachate effects on heavy metal contamination in water and sediments of Garan River, Marivan County, Iran

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ABSTRACT

This study assessed heavy metal contamination in the Garan River's water and sediments to evaluate the impact of landfill leachate. Pollution levels and ecological risks were quantified using established indices, including the contamination factor (CF), pollution load index (PLI), geoaccumulation index (Igeo), enrichment factor (EF), and potential ecological risk index (PERI). Spatially, peak dissolved metal concentrations in water occurred at points S1 and S4, while maximum sediment concentrations were found at S3, directly downstream of the leachate outfall. Water quality analysis indicated that arsenic posed both acute and chronic human health risks at all stations, and chromium posed an acute risk at all stations. Sediment assessment revealed that barium, nickel, and mercury consistently exceeded the threshold effect concentration (TEC). Furthermore, nickel and mercury surpassed the probable effect concentration (PEC) at 100% and 85.71% of stations, respectively, signaling a significant potential for adverse effects on aquatic life. Index analysis showed higher CF values in sediments than in water. The PLI trend increased in sediments but decreased in water, whereas EF was higher in water and spatially stable. The average Igeo for all metals remained below zero, classifying the area as unpolluted. The PERI indicated medium ecological risk in sediments only at stations S3, S6, and S8. While the collective indices suggest that severe, widespread pollution is not present, the identified contamination hotspots and specific risks from metals like arsenic, nickel, and mercury warrant vigilant attention. Consequently, periodic monitoring of leachate-derived metals and proactive preventive measures are critically recommended to preserve aquatic ecosystem quality.

Highlights

* Leachate contamination elevates Ba, Ni, and Hg in river sediments above safety thresholds.
* Arsenic in water poses acute and chronic human health risks at all sampling stations.
* Pollution indices confirm higher heavy metal accumulation in sediments than in the water column.
* Geoaccumulation index classifies the area as unpolluted despite localized contamination hotspots.
* Medium ecological risk from metals is identified in sediments at three downstream locations.

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1. Introduction

The natural and evolutionary balance of ecosystems, including rivers, has always been altered by human activities, and their natural environment is changing and deteriorating (Salati & Moore, 2010). Over the past half-century, pollutants from waste have been a growing trend in the formation of the structure of surface water pollution (Wieczorek et al., 2021).

Surface water is considered the primary source of drinking water in most countries worldwide. Therefore, preventing the pollution of freshwater resources such as rivers and streams, while providing water of optimal quality, in addition to reducing treatment costs and providing human benefits, also brings environmental and biodiversity conservation (Edokpayi et al., 2016). Of the amount of waste produced annually, only 2.3% is disposed of hygienically and without harming the environment (Yang & Xu, 2020). As a result of this mismanagement, highly contaminated leachate is produced from the landfill, which, by migrating and discharging directly or as runoff into water resources, leads to soil, surface, and groundwater contamination. Concentrations of leachate pollutants higher than appropriate standards endanger the natural balance of the environment, living organisms, and general human health, which indicates the need to assess potential risks and reduce adverse effects from landfill leachate in order to achieve sustainable development and management of the environment (Atta et al., 2015; Benfridja et al., 2021; Enitan et al., 2018; Hredoy et al., 2022; Maiti et al., 2015; Naveen et al., 2018; Nta et al., 2020).

NH_3--N, NO_3^-, BOD, COD, SS, and heavy metals are the main pollutants in river basins that can be exacerbated by landfill leachate. The exacerbated effects of these pollutants on surface and groundwater resources have been investigated and evaluated in various studies (Atta et al., 2015; Benfridja et al., 2021; Iresha et al., 2023; Noerfitriyani et al., 2018; Olivares-Rieumont et al., 2005; Shaylinda et al., 2020). Heavy metals enter surface waters and rivers either naturally (geogenic), including weathering of rocks, soil erosion, atmospheric precipitation, wind, and dissolution of soluble salts in water, or through human activities, including various industries, untreated wastewater, landfills, and agricultural activities (Ali et al., 2022; Olivares-Rieumont et al., 2005; Salati & Moore, 2010; Wardhani et al., 2024; Zaranyika et al., 1993). The entry of heavy metals from landfill leachate into groundwater and surface waters has faced developing countries with the most severe environmental challenges (Boateng et al., 2019). Non-biodegradability, bioaccumulation in microorganisms, aquatic plants, and animals, and subsequent circulation in the food chain, environmental stability, persistence, and biotoxicity properties have made heavy metal pollution in rivers a significant issue for both aquatic ecosystems and human health (Dwanga et al. 2024; Varol, 2011). Heavy metals in leachate are affected by this environment through entering the river's aquatic environment and causing pollution through mechanisms of dilution by rainwater during flushing events, adsorption on solid sediment/soil compounds, hydrolysis and also precipitation of newly formed phases (which incorporate these pollutants into their crystal structure), as a result of which most of the metals settle in bottom sediments (Ettler et al., 2006; Islam et al., 2015; Varol, 2011). The mobility, solubility, and bioavailability of heavy metals are changed in combination with dissolved organic matter in leachate and speciation in aquatic environments, causing toxic effects on the environment and human health (Alsaffar et al., 2016; Naveen et al., 2018; Parvin & Tareq, 2021). Heavy metals in carbonate, sulfide, and organic compounds are more toxic and pose greater risks due to their greater bioavailability (Karbassi et al., 2008). Excessive intake of essential heavy metals and excessive intake of toxic heavy metals through direct consumption of drinking water, consumption of agricultural products irrigated with contaminated water, and consumption of aquatic animals that have accumulated these metals in their bodies expose humans to various diseases (Alsaffar et al., 2016). For example, continuous consumption of arsenic-contaminated water can lead to chronic effects such as neurological problems, cancer, cardiovascular disease, and high blood pressure (Boateng et al., 2019). In addition, accumulation of heavy metals from continuous consumption of contaminated fish has caused Itai-Itai disease (caused by chronic cadmium poisoning) and Minamata disease (caused by chronic mercury poisoning) in humans (James, 1977).

Sediments are important ecological components in the river environment, transferring between 30 and 98% of the total metal load of a river in a sediment-dependent manner (Varol & Şen, 2012). Continuous discharge of leachate into the river flow and deposition of persistent pollutants, including heavy metals, in riverbed sediments by increasing their concentration in sediments, changes this part of the river that, under variable physical and chemical conditions, cause sediments to act as both a source of nutrition, a storage reservoir, and a secondary source of pollutants, and also cause disturbances in aquatic life (Karbassi et al., 2008; Sayadi et al., 2015a). For example, iron can cause problems for aquatic life by covering the sediments of the riverbed. In addition, some of these deposited metals are used by aquatic plants and animals, and after decay and decomposition, they re-enter the water. The composition of bed sediment, the composition of suspended sediment, and water chemistry determine the behavior of metals in the river and lead to different amounts of metals and concentration differences between the two environments of river water and sediment (James, 1977).

The Garan River in Marivan, west of Iran, is affected by the leachate discharged from the Garan landfill and changes its physicochemical and chemical properties, including the concentration of heavy metals in the river water and the accumulation of these metals in the riverbed sediments. Therefore, considering the agricultural and horticultural uses of the river water downstream and aquaculture activities in the riverbed receiving reservoir, the ecological importance of this water body, and the possibility of direct human uses, it is vital to assess the concentration of heavy metals in the river after the point of leachate entry into the river to ensure its ability to withstand the load imposed on the river. Considering the above, the present study was conducted to determine the spatial distribution of heavy metals in the two environments of water and sediments of the Garan Riverbed, to confirm or reject the human origin of the pollution, and to determine the intensity of heavy metal pollution in water and sediments using relevant pollution indices.

2. Materials and Methods

2.1 Study area

The Garan River, located in Marivan County, Kurdistan Province, Iran, has geographical coordinates of 46°14'16" to 46°28'54" East and 35°29'45" to 35°47'2" West, and is approximately 25 km long in this county. The rainfall in this area is 800 mm. Agricultural and horticultural activities, fishing and aquaculture, and recreational activities are among

the various uses of this river by humans. As a result of these activities, this river faces the problem of the entry of domestic wastewater flows, pollution with solid waste from residents of the river basin and waste from commercial and industrial activities, and the entry of landfill leachate. The study area is characterized by a Mediterranean climate, featuring hot, dry summers and wet, rainy winters. The average temperature in this area is 14 °C.

2.2 Data collection and sample analysis

In order to investigate the effect of Garan landfill leachate on the heavy metal content of Garan River water, 8 points were selected for sampling, as well as the heavy metal content of 7 points in river bottom sediments. These points included one point upstream of the stream (S1) to determine the background metal concentration as a control point, one point at the leachate entry point into the river, and 6 points downstream of the river (S3, S4, S5, S6, S7, S8). Samples taken downstream of the river were selected based on access to the river for sampling at intervals of approximately 1 km and to investigate the quality conditions of the river water after the incoming leachate mixed with the river water. At the second point, a leachate sample was taken from the drainage area for analysis, so there was no sediment to sample. The geographical coordinates of the sampling points were recorded using a GPS device and are presented in Table 1.

Table 1 Geographic coordinates of sampling points

Sample Site	Zone	Elevation	Longitude	latitude
S1 (Upstream)	38 S	1343	617804	3932998
S2 (Conjunction)	38 S	1311	617894	3930353
S3 (Downstream)	38 S	1312	617300	3930441
S4 (Downstream)	38 S	1212	616930	3930829
S5 (Downstream)	38 S	1194	616567	3931304
S6 (Downstream)	38 S	1227	616684	3931730
S7 (Downstream)	38 S	1194	616730	3932147
S8 (Downstream)	38 S	1221	617697	3932817

Polypropylene containers were used for water sampling to prevent heavy metal contamination of the sample. The water and leachate sampling containers were first washed with nitric acid and then rinsed with distilled water. All containers were rinsed three times with river water before sampling and collecting the samples. At each station, 500 ml of sample was collected with three sampling times at 4-hour intervals, and real samples were extracted from them. 1 ml of pure nitric acid was added to each sample to reduce the pH to approximately 1 (to minimize precipitation and adsorption of metals on the container wall) (Bazrafshan et al., 2016; Islam et al., 2015). After recording the sample characteristics on the sampling containers, these containers were transported to the laboratory at 4°C for analysis. (Association, 1926). The standard protocol of the US Environmental Protection Agency was used to collect composite samples of riverbed sediment (US EPA, 2001). A portable Ekman sampler was used to sample riverbed sediments at a depth of 0 to 5 cm. Three composite cores (200 g) of sediment were collected from each sampling site. The top 2 cm of each core was removed from the center of the receiver with an acid-washed plastic spatula to eliminate any contamination from the metal parts of the corer. For the analysis of background metals, the sediment was cored using a percussion hammer corer (50–80 cm in length) (Schottler & Engstrom, 2006). Air-tight polyethylene bags were used to collect composite sediment samples in the field and then transported to the laboratory. The sediment samples were dried in an oven at 45°C for 48 h to a constant weight. The dried samples were then ground with a mortar and pestle and sieved using a 106-µm mesh. The smaller particle size fraction was homogenized by grinding in an agate mortar and stored in labeled glass bottles for chemical analysis.

2.3. Sample digestion and extraction of metals

In this study, all chemicals used were analytical grade reagents, and purified water (Milli-Q) (Elix UV5 and MilliQ models, manufactured by Millipore, USA) was used to prepare the solutions. Laboratory containers (Teflon and polypropylene) were first carefully cleaned, and after soaking in a 5% nitric acid (HNO_3) solution for more than 24 hours, they were washed with Milli-Q water and dried completely. To calibrate the devices, the required standard concentrations of heavy metals were prepared by diluting 1000 mL stock solutions (Merck Certificate AA standard (Merck)). For metal analysis, 20 mL of water sample and 0.5 g of sediment sample were mixed with 5 mL of HNO_3(69% purity) and 2 mL of hydrogen peroxide (30% purity) in a Teflon-capped vessel and digested in a microwave digestion system. After digestion, the resulting solution was filtered using a syringe filter (DISMIC® – 25HP PTFE model, 0.45 µm pore size, Toyo Roshi Kaisha, Japan) and stored in 50 mL polypropylene tubes (Nalgene, New York). Finally, the vessels were washed with Milli-Q water and dried by air flow.

2.4 Instrumental analysis and quality assurance

For the analysis of heavy metals in the samples, an inductively coupled plasma spectrometer (ICP-OES EOP) (SPECTRO ARCOS) was used. The calibration curve was prepared using the multi-element standard solution XSTC-13 (Spex CertiPrep®, USA) and the calibration curves with a coefficient of determination (R^2) greater than 0.999 were accepted for calculating the metal concentrations. Moreover, a multi-element solution (Agilent Technologies, USA) with a concentration of 1.0 µg/L was used as a tuning solution, which covered a wide range of element masses. The quality of the results was assessed using an internal quality control method and the experimental batches were confirmed if they met the internal quality control criteria (IQCs). In each experiment, blanks, certified reference materials (CRMs), and samples were analyzed twice to eliminate batch-specific errors. Relative standard deviation (RSD) of less than 5% was checked using the calibration solution, and then the analysis process was started. For quality control of the laboratory methods, reference materials NMIJ CRM 7303 and river sediment were used. Comparison of the measured mean values and standard deviations for elemental values in the reference materials with the verified values showed that the sample preparation conditions and instrument settings had the desired accuracy and precision.

2.5 Heavy metal evaluation indices in sediments

In order to analyze chemical and geochemical data, it is of great importance to determine the background values to estimate heavy metal pollution indices and perform

calculations. For this purpose, the concentrations measured at the first sampling station, which is located before the leachate enters the river, were considered as background values. Then, to determine the level of heavy metal pollution caused by the leachate entering the river, the contamination factor (CF), pollution load index (PLI), geoaccumulation index (I_{geo}), enrichment factor (EF), and potential ecological risk index (PERI) were calculated and evaluated.

2.5.1 Contamination Factor

The *CF* is used to show the sediment contamination with heavy metals (Eq. 1)

$$CF = C_m/C_b \quad (1)$$

Where C_m is the concentration of the metal of interest in the sediment at the sampling site, and C_b is the background concentration of the metal of interest in the sediment before the leachate enters the river. The *CF* is categorized into four levels: low contamination level ($CF<1$), moderate contamination level ($1<CF\leq3$), significant contamination level ($3<CF\leq6$), and very high contamination level ($CF>6$) to describe the contamination of each metal individually in the sediment (Schottler & Engstrom, 2006).

2.5.2 Pollution Load Index (PLI)

In order to determine the combined effect of each specific metal on the overall water quality and sediments in the water body, the pollution load index was calculated. Eq. 2 was used to calculate PLI.

$$PLI = \sqrt[n]{CF_1 \times CF_2 \times \ldots \times CF_n} \quad (2)$$

Where *n* represents the number of metals and CF represents the contamination factor associated with each metal. To describe this index, four pollution levels were defined: non-polluted level (PLI < 1), moderate pollution level (1 < PLI < 2), high pollution level (2 < PLI < 3), and very high pollution level (3 < PLI) (Zarei et al., 2014).

2.5.3 Geoaccumulation index

The geoaccumulation index (Igeo) was used to assess the level of heavy metal contamination from leachate entering the river at various concentrations of these metals in the river sediments. This index has been used by various researchers to investigate the contamination of soils and sediments. This method determines the level of contamination by comparing the current amount of heavy metals in samples with their background (natural) levels. To take into account natural changes in materials in the environment, as well as minor impacts of human activities, the background amount of each heavy metal is multiplied by a constant factor of 1.5. The I_{geo} value was calculated using Eq. 3.

$$I_{geo} = log_2(C_n/1.5B_n) \quad (3)$$

where C_n is the measured heavy metal concentration at the sampling sites in river sediments (mg/kg), B_n is the measured background heavy metal concentration in river sediments before leachate contamination (mg/kg), and 1.5 is a constant used to calculate the effect of lithological changes on heavy metal background values. The pollution level calculated with the I_{geo} index was divided into seven classes (0-6) as uncontaminated ($I_{geo} \leq 0$), uncontaminated to almost contaminated ($0 < I_{geo} < 1$), almost contaminated ($1 < I_{geo} < 2$), contaminated ($2 < I_{geo} < 3$), highly contaminated ($3 < I_{geo} < 4$), highly contaminated to very highly contaminated ($4 < I_{geo} < 5$), very highly contaminated ($I_{geo} \geq 5$) based on the Mueller classification (Sayadi et al., 2015b; VESALI et al., 2012).

2.5.4 Enrichment Factor

Enrichment factor (*EF*) is considered one of the useful tools for assessing the level of heavy metal pollution in rivers in environmental studies. The *EF* is calculated based on the comparison of the tested elements with a reference element. The reference element is selected in such a way that there are few changes in its presence in nature. Among conservative elements such as aluminum (Al), iron (Fe), or silicon (Si), Al and Fe are used as geochemical normalizing elements of metal data due to the insignificant contribution of human resources in their production compared with natural resources. Having the least variability in their presence and existence in very high concentrations plays a role in the selection of the reference element. These conditions neutralize the effect of minor changes in heavy metal concentration or interactions with other elements on the results. In the present study, iron was used to normalize metal pollutants to separate natural components from components caused by human activities due to its strong association with fine solid surfaces and geochemical behavior similar to many rare metals. To assess human impacts on the studied heavy metals, the *EF* value for each metal was calculated using Eq. 4.

$$\mathrm{EF} = \frac{(C_n/C_{Fe})_S}{(C_n/C_{Fe})_B} \quad (4)$$

Where $(C_n/C_{Fe})_S$ is the ratio of the concentration of the n^{th} heavy metal to the concentration of iron in the sample, and $(C_n/C_{Fe})_B$ is the ratio of the concentration of the nth metal to the iron in the background sample.

Heavy metal contamination was classified using the *EF* into seven categories: 1. no enrichment (< 1), 2. low enrichment (1-2.9), 3. moderate enrichment (3-4.9), 4. relatively severe enrichment (5-9.9), 5. severe enrichment (10-24.9), 6. very severe enrichment (25-49.9), and 7. very, very severe enrichment (> 50) (Babu et al., 2021; Zarei et al., 2014).

2.5.5 Potential Ecological Risk Index

The Potential Ecological Risk Index (PERI) is a qualitative assessment tool for assessing the ecological risks posed by pollutants in various environments, including sediments. This index was introduced by Hakanson in 1980 (Hakanson, 1980) and is used to assess the ecological risks of heavy metals and other pollutants in sediments. This index is based on the presence of toxic metals in the environment and the ecological responses to the toxicity of these metals in assessing the level of heavy metal contamination in sediments. This index was calculated according to Eq. 5.

$$PERI = \sum_{n=1}^{n}(T_n * CF_n) \quad (5)$$

where T_n is the toxic response index for the n^{th} heavy metal ((Zn = 1), (Cu = 5), (Cr = 2), (Pb = 5), (Mo = 15), (Ni = 2), (Cd = 30), (Hg = 40), (As = 10)) (Sheykhi & Moore, 2013) and CF_n is the calculated *CF* for the n^{th} heavy metal. Evaluation of sediment quality with the *PERI* of metals at four levels including: 1. low ecological risk level (PERI < 150) 2. medium

ecological risk level (150 < PERI < 300) 3. high ecological risk level (300 < PERI < 600), and 4. very high ecological risk level (PERI > 600) was investigated (Sayadi et al., 2015b; Wardhani et al., 2024).

3.1 Results and Discussion

3.1 Heavy metal concentration in water and sediment

The concentrations of heavy metals in water and sediments of the Garan River at the eight studied locations are presented in Tables 2 and 3, respectively. Heavy metal concentrations range for water (µg/l) and sediment (mg/kg) was 15.97-91.74, and 9400-13200 for Al, 1-1.06 and 5.56-14.99 for As, 10.25-12.39 and 45.76-173.91 for Ba, <0.042 and 0.52-0.72 for Be, <0.049 and 0.09-0.4 for Cd, <0.043 and 12.34-16.21 for Co, 1.88-3.13 and 21.72-35.91 for Cr, 18.95-21.36 and 18.22-28.27 for Cu, 13.71-135.21 and 7000-7100 for Fe, <0.351 and <20 for Hg, 5.73-6.31 and 11.73-22.46 for Li, 0.34-3.28 and 0.49-1.52 for Mn, <1.67 and <40 for Mo, 0.29-0.74 and 42.97-156.58 for Ni, <2.17 and 23.31-30.83 for Pb, <0.46 and <10 for Sb, 0.16-1.73 and <20 for Sn, 47.06-50.52 and 2.67-12.12 for Sr, <0.064 and 13.09-21.69 for V and 3.47-28.42 and <100 for Zn. According to Table 2, Be, Cd, Co, Hg, Mo, Pb, Sb, and V had the lowest concentrations in water samples at all sampling points, such that their values were lower than the detection limit of the device. The concentration of As was also lower than the detection limit of the device in all samples except for sample S5 (1.06 µg/L). The highest concentration of Al, Ba, Fe, Mn, and Zn was obtained in sample S5, and their lowest concentrations were obtained in samples S5, S8, S8, S8 and S7, respectively. The highest concentrations of Cr, Li, Ni, Sn, and St were also recorded in sample S4, and among these metals, the lowest values of Cr, Li, Sn, and St were recorded in samples S8, S7, S7 and S2, respectively, and the minimum concentration of Ni was estimated to be below the detection limit in 5 samples (samples S3, S5, S6, S7 and S8). The maximum and minimum concentrations of Cu were also recorded in samples S2 and S5, respectively. The results of heavy metal concentrations in Table 3 show that Hg, Mo, Sb, and Sn were below the detection limit in all samples. The minimum amounts of Al, As, Cd, Co, Cu, Li, Pb, and V were found in sample S1, and Be was also found in samples S1 and S4. Among these metals, the maximum amount of Be, although it did not change much along the sampling points, was found at point S8, and the maximum amount of Li was also recorded in sample S6; the rest of the heavy metals had their maximum amounts in sample S2. The lowest concentrations of Ba and Mn were found in sample S7, and Cr, Ni, and St were found in samples S4, S5, and S6, respectively. The concentration of Zn in all samples and Fe in all samples except sample S6 remained constant. The concentration of heavy metals was compared with the criteria for heavy metals in EPA river water and the California State Sediment Quality Assessment Guidelines (Table 4) (Florida Department of Environmental Protection, January 2003; U.S. Environmental Protection Agency; U.S. Environmental Protection Agency). As can be seen in Table 5, the As concentration exceeded the threshold effect concentration in 28.57% of sediment samples, posing a risk to sediment-dwelling organisms at these points and causing acute and chronic effects on human health through river water at all points. Ba, Hg, and Ni exceeded the threshold effect concentration in sediment at all points studied and exceeded the probable effect concentration at 28.57, 100, and 85.71% of points, respectively. Cr also had acute effects on human health only in water samples and at all points, and did not pose a risk to aquatic life. Adi Septiono and Roosmini investigated the distribution of heavy metals in the Citarum River. In this study, they found that the concentration of Cr in water at two points was higher than the relevant standard and was probably due to the discharge of wastewater from various industries. The higher-than-standard concentration of Cu was also attributed to the synergistic effect of agricultural and industrial activities. In sediments, the concentration of Cu exceeded the standard values compared, indicating moderate pollution levels at the first three points and severe pollution levels at the other points. This increase in Cu concentration was attributed to the deposition of suspended matter that absorbed Cu (Septiono & Roosmini, 2015). The order of heavy metal concentrations in the Cisadane River sediments in the study conducted by Sulistyowati et al. was Pb > Cr > Cd. The results of this study showed a positive and significant Pearson correlation between the metals, indicating a common origin of the metals (natural or anthropogenic). The concentration of metals increased as the river moved downstream, which was a result of point and non-point sources (soil leaching and urban and industrial wastewater discharge) (Sulistyowati et al., 2023). Those elements, having values of zero or non-calculable, are not shown in Table 5.

The speed and type of flow are hydrodynamic aspects of the river that contribute to the deposition of heavy metals in laminar flows and cause resuspension of these materials in turbulent flows. Additionally, the interaction between water and sediment, as well as the deposition and dissolution of heavy metals and their compounds, is influenced by factors such as the topography and geography of the region, as well as soil type, discharge, slope, and vegetation cover (Iresha et al., 2023).

3.2 Indices Analysis

3.2.1 CF

The results of calculating *CF* in river water and sediments are presented in Table 6, and Fig. 1 shows the ratio of sediment pollution to water. In the studied water samples, the *CF* for all metals in the background sample (S1), as well as for Be, Cd, Co, Hg, Mo, Pb, Sb, V, and As (except S5 with *CF* = 1.057 and medium pollution level), which were below the detection limit, was calculated to be 1. The metals of Al, Ba, Fe, Mn, and Zn had a CF less than 1 in all sampling points and were classified as low pollution level. The *CF* higher than 1 was obtained for three metals Cr (1.113), Li (1.016), and Ni (2.053), in S4, and were placed in the group with medium pollution level. Moreover, the three metals Cu, Sn, and St each had values higher than 1 and medium contamination levels at two points of S2 (1.061) and S8 (1.039), S4 (1.215) and S5 (1.136), and S4 (1.05) and S6 (1.001), respectively.

Table 2 Concentration of heavy metals in water samples

Water Sample	Descriptive	Al	As	Ba	Be	Cd	Co	Cr	Cu	Fe	Hg	Li	Mn	Mo	Ni	Pb	Sb	Sn	Sr	V	Zn
	Parameters	ppb	ppb	ppb	ppb	ppb	ppb	ppb	ppb	ppb	ppb	ppb	ppb	ppb	ppb	ppb	ppb	ppb	ppb	ppb	ppb
S1	Min	91.7	1.00	12.24	0.04	0.05	0.04	2.66	19.95	134.95	0.35	6.13	3.15	1.67	0.24	2.17	0.46	1.18	47.79	0.06	28.15
	Max	92	1.00	12.56	0.04	0.05	0.04	2.94	20.25	135.47	0.35	6.31	3.41	1.67	0.48	2.17	0.46	1.70	48.41	0.06	28.69
	Mean	91.7	1.00	12.40	0.04	0.05	0.04	2.80	20.10	135.21	0.35	6.22	3.28	1.67	0.36	2.17	0.46	1.44	48.10	0.06	28.42
	Stdev	0.48	0.00	0.23	0.00	0.00	0.00	0.20	0.21	0.37	0.00	0.13	0.18	0.00	0.17	0.00	0.00	0.37	0.44	0.00	0.38
S2	Min	38.67	1.00	10.82	0.04	0.05	0.04	2.13	21.29	42.68	0.35	5.61	0.96	1.67	0.27	2.17	0.46	0.76	46.71	0.06	8.74
	Max	39.13	1.00	11.18	0.04	0.05	0.04	2.47	21.51	43.06	0.35	5.99	1.42	1.67	0.41	2.17	0.46	1.04	47.41	0.06	9.10
	Mean	38.90	1.00	11.00	0.04	0.05	0.04	2.30	21.40	42.87	0.35	5.80	1.19	1.67	0.34	2.17	0.46	0.90	47.06	0.06	8.92
	Stdev	0.33	0.00	0.25	0.00	0.00	0.00	0.24	0.16	0.27	0.00	0.27	0.33	0.00	0.10	0.00	0.00	0.20	0.49	0.00	0.25
S3	Min	22.06	1.00	11.42	0.04	0.05	0.04	2.58	19.72	26.62	0.35	5.75	1.22	1.67	0.29	2.17	0.46	0.97	47.67	0.06	5.10
	Max	22.34	1.00	11.58	0.04	0.05	0.04	3.02	20.08	27.10	0.35	6.09	1.60	1.67	0.29	2.17	0.46	1.31	48.15	0.06	5.30
	Mean	22.20	1.00	11.50	0.04	0.05	0.04	2.80	19.90	26.86	0.35	5.92	1.41	1.67	0.29	2.17	0.46	1.14	47.91	0.06	5.20
	Stdev	0.20	0.00	0.11	0.00	0.00	0.00	0.31	0.25	0.34	0.00	0.24	0.27	0.00	0.00	0.00	0.00	0.24	0.34	0.00	0.14
S4	Min	23.71	1.00	12.16	0.04	0.05	0.04	2.92	18.99	23.88	0.35	6.18	0.26	1.67	0.60	2.17	0.46	1.47	50.33	0.06	4.99
	Max	24.09	1.00	12.44	0.04	0.05	0.04	3.28	19.41	24.22	0.35	6.44	0.46	1.67	0.88	2.17	0.46	2.01	50.71	0.06	5.31
	Mean	23.90	1.00	12.30	0.04	0.05	0.04	3.10	19.20	24.05	0.35	6.31	0.36	1.67	0.74	2.17	0.46	1.74	50.52	0.06	5.15
	Stdev	0.27	0.00	0.20	0.00	0.00	0.00	0.25	0.30	0.24	0.00	0.18	0.14	0.00	0.20	0.00	0.00	0.38	0.27	0.00	0.23
S5	Min	15.91	0.98	11.04	0.04	0.05	0.04	2.58	18.67	17.34	0.35	5.67	0.42	1.67	0.29	2.17	0.46	1.50	47.51	0.06	4.21
	Max	16.09	1.22	11.36	0.04	0.05	0.04	2.82	19.13	17.72	0.35	5.97	0.66	1.67	0.29	2.17	0.46	1.76	48.17	0.06	4.59
	Mean	16.00	1.10	11.20	0.04	0.05	0.04	2.70	18.90	17.53	0.35	5.82	0.54	1.67	0.29	2.17	0.46	1.63	47.84	0.06	4.40
	Stdev	0.13	0.17	0.23	0.00	0.00	0.00	0.17	0.33	0.27	0.00	0.21	0.17	0.00	0.00	0.00	0.00	0.18	0.47	0.00	0.27
S6	Min	22.43	1.00	11.09	0.04	0.05	0.04	2.62	19.96	16.49	0.35	5.64	0.30	1.67	0.29	2.17	0.46	0.20	47.98	0.06	3.54
	Max	22.77	1.00	11.51	0.04	0.05	0.04	2.78	20.24	16.91	0.35	6.00	0.46	1.67	0.29	2.17	0.46	0.36	48.26	0.06	3.82
	Mean	22.60	1.00	11.30	0.04	0.05	0.04	2.70	20.10	16.70	0.35	5.82	0.38	1.67	0.29	2.17	0.46	0.28	48.12	0.06	3.68
	Stdev	0.24	0.00	0.30	0.00	0.00	0.00	0.11	0.20	0.30	0.00	0.25	0.11	0.00	0.00	0.00	0.00	0.11	0.20	0.00	0.20
S7	Min	19.69	1.00	10.27	0.04	0.05	0.04	2.46	19.56	15.49	0.35	5.59	0.76	1.67	0.29	2.17	0.46	0.16	47.79	0.06	3.35
	Max	19.91	1.00	10.53	0.04	0.05	0.04	2.74	20.04	15.95	0.35	5.87	1.02	1.67	0.29	2.17	0.46	0.16	48.15	0.06	3.59
	Mean	19.80	1.00	10.40	0.04	0.05	0.04	2.60	19.80	15.72	0.35	5.73	0.89	1.67	0.29	2.17	0.46	0.16	47.97	0.06	3.47
	Stdev	0.16	0.00	0.18	0.00	0.00	0.00	0.20	0.34	0.33	0.00	0.20	0.18	0.00	0.00	0.00	0.00	0.00	0.25	0.00	0.17
S8	Min	19.07	1.00	10.10	0.04	0.05	0.04	1.74	20.77	13.53	0.35	5.67	0.25	1.67	0.29	2.17	0.46	0.21	47.47	0.06	3.67
	Max	19.33	1.00	10.50	0.04	0.05	0.04	2.06	21.03	13.89	0.35	5.87	0.43	1.67	0.29	2.17	0.46	0.43	47.79	0.06	3.97

	Mean	19.20	1.00	10.30	0.04	0.05	0.04	1.90	20.90	13.71	0.35	5.77	0.34	1.67	0.29	2.17	0.46	0.32	47.63	0.06	3.82
	Stdev	0.18	0.00	0.28	0.00	0.00	0.00	0.23	0.18	0.25	0.00	0.14	0.13	0.00	0.00	0.00	0.00	0.16	0.23	0.00	0.21

Table 3 Concentration of heavy metals (mg/kg) in sediment samples

Sediment Sample	Descriptive parameters	Al	As	Ba	Be	Cd	Co	Cr	Cu	Fe	Hg	Li	Mn	Mo	Ni	Pb	Sb	Sn	Sr	V	Zn
S1	Min	9396.0	5.5	55.4	0.4	0.1	12.2	27.2	18.1	7100.0	20.0	11.6	0.7	40.0	85.2	23.2	10.0	20.0	12.0	13.0	100.0
	Max	9404.0	5.7	55.8	0.6	0.2	12.5	27.5	18.3	7100.0	20.0	11.8	0.9	40.0	85.6	23.5	10.0	20.0	12.3	13.2	100.0
	Mean	9400.0	5.6	55.6	0.5	0.1	12.3	27.4	18.2	7100.0	20.0	11.7	0.8	40.0	85.4	23.3	10.0	20.0	12.1	13.1	100.0
	Stdev	5.7	0.2	0.3	0.1	0.0	0.2	0.2	0.2	0.0	0.0	0.2	0.1	0.0	0.3	0.2	0.0	0.0	0.2	0.2	0.0
S3	Min	13198.0	14.9	173.8	0.5	0.3	16.0	35.4	28.1	7100.0	20.0	19.8	1.4	40.0	62.4	30.7	10.0	20.0	3.4	21.5	100.0
	Max	13202.0	15.1	174.0	0.7	0.4	16.4	35.8	28.4	7100.0	20.0	20.0	1.7	40.0	62.7	30.9	10.0	20.0	3.8	21.8	100.0
	Mean	13200.0	15.0	173.9	0.6	0.4	16.2	35.6	28.3	7100.0	20.0	19.9	1.5	40.0	62.5	30.8	10.0	20.0	3.6	21.7	100.0
	Stdev	2.8	0.1	0.2	0.2	0.1	0.2	0.3	0.2	0.0	0.0	0.2	0.2	0.0	0.2	0.2	0.0	0.0	0.3	0.2	0.0
S4	Min	9594.0	8.2	51.0	0.5	0.1	12.6	21.5	20.7	7100.0	20.0	17.3	0.6	40.0	83.9	24.5	10.0	20.0	4.4	14.5	100.0
	Max	9606.0	8.4	51.2	0.6	0.2	12.9	21.9	20.9	7100.0	20.0	17.6	0.8	40.0	84.5	24.9	10.0	20.0	4.8	14.8	100.0
	Mean	9600.0	8.3	51.1	0.5	0.1	12.8	21.7	20.8	7100.0	20.0	17.5	0.7	40.0	84.2	24.7	10.0	20.0	4.6	14.7	100.0
	Stdev	8.5	0.2	0.2	0.1	0.1	0.2	0.3	0.2	0.0	0.0	0.2	0.1	0.0	0.4	0.3	0.0	0.0	0.3	0.2	0.0
S5	Min	11497.0	9.3	70.1	0.4	0.1	14.1	32.3	22.0	7100.0	20.0	15.8	0.7	40.0	42.8	27.0	10.0	20.0	7.0	16.9	100.0
	Max	11503.0	9.6	70.5	0.7	0.2	14.4	32.6	22.6	7100.0	20.0	16.1	0.9	40.0	43.1	27.5	10.0	20.0	7.2	17.1	100.0
	Mean	11500.0	9.4	70.3	0.6	0.1	14.2	32.4	22.3	7100.0	20.0	15.9	0.8	40.0	43.0	27.3	10.0	20.0	7.1	17.0	100.0
	Stdev	4.2	0.2	0.3	0.2	0.1	0.2	0.2	0.4	0.0	0.0	0.2	0.1	0.0	0.2	0.3	0.0	0.0	0.1	0.1	0.0
S6	Min	12799.0	12.3	46.5	0.5	0.3	15.3	31.9	22.6	6997.0	20.0	22.4	0.5	40.0	156.2	28.7	10.0	20.0	2.3	14.5	100.0
	Max	12801.0	12.7	47.0	0.7	0.4	15.7	32.2	22.9	7003.0	20.0	22.6	0.8	40.0	156.9	28.9	10.0	20.0	3.0	14.8	100.0
	Mean	12800.0	12.5	46.8	0.6	0.4	15.5	32.1	22.8	7000.0	20.0	22.5	0.6	40.0	156.6	28.8	10.0	20.0	2.7	14.6	100.0
	Stdev	1.4	0.3	0.4	0.2	0.1	0.3	0.2	0.2	4.2	0.0	0.1	0.2	0.0	0.5	0.2	0.0	0.0	0.5	0.2	0.0
S7	Min	10797.0	5.5	45.6	0.4	0.0	12.8	25.6	19.3	7100.0	20.0	18.3	0.4	40.0	132.1	24.8	10.0	20.0	6.0	14.8	100.0
	Max	10803.0	5.7	45.9	0.7	0.2	13.1	25.9	19.6	7100.0	20.0	18.7	0.6	40.0	132.5	25.1	10.0	20.0	6.3	15.2	100.0
	Mean	10800.0	5.6	45.8	0.6	0.1	13.0	25.7	19.5	7100.0	20.0	18.5	0.5	40.0	132.3	24.9	10.0	20.0	6.2	15.0	100.0
	Stdev	4.2	0.1	0.2	0.2	0.1	0.3	0.2	0.3	0.0	0.0	0.3	0.1	0.0	0.3	0.2	0.0	0.0	0.2	0.3	0.0
S8	Min	13098.0	9.1	57.6	0.6	0.3	15.1	35.8	21.9	7100.0	20.0	19.2	0.6	40.0	107.6	30.7	10.0	20.0	4.6	17.8	100.0
	Max	13102.0	9.4	57.8	0.9	0.5	15.4	36.0	22.4	7100.0	20.0	19.4	0.8	40.0	108.0	31.0	10.0	20.0	5.0	18.0	100.0
	Mean	13100.0	9.2	57.7	0.7	0.4	15.2	35.9	22.2	7100.0	20.0	19.3	0.7	40.0	107.8	30.8	10.0	20.0	4.8	17.9	100.0
	Stdev	2.8	0.2	0.1	0.2	0.1	0.2	0.2	0.3	0.0	0.0	0.2	0.2	0.0	0.3	0.2	0.0	0.0	0.3	0.1	0.0

Table 4 EPA River water heavy metal (µg/L) criteria for aquatic life and human health, and California State sediment quality assessment guidelines

Heavy metal	SQAG		WQC_ALC		WQC_HHC	
	TEC	PEC	acute	chronic	acute	chronic
Al	*	*	*	*	*	*
As	9.8	33	340	150	0.018	0.14
Ba	20	60	*	*	1000	*
Be	*	*	*	*	*	*
Cd	1	5	1.8	*	*	*
Co	50	*	*	*	*	*
Cr	43	110	16	11	total	*
Cu	32	150	*	*	1300	*
Fe	*	*	*	1000	*	*
Hg	0.18	1.1	1.4	0.77	*	*
Li	*	*	*	*	*	*
Mn	*	*	*	*	50	100
Mo	*	*	*	*	*	*
Ni	23	49	470	52	610	4600
Pb	36	130	65	2.5	*	*
Sb	*	*	*	*	5.6	640
Sn	*	*	*	*	*	*
Sr	*	*	*	*	*	*
V	*	*	*	*	*	*
Zn	120	460	120	120	7400	26000

Table 5 Percentage of samples exceeding EPA river water heavy metal criteria (µg/L) for aquatic life and human health, and California State Sediment Quality Assessment Guidelines

Heavy metal	SQAG		WQC_ALC		WQC_HHC	
	TEC	PEC	acute	chronic	acute	chronic
As	28.57	0	0	0	100	100
Ba	100	28.6	_	_	0	_
Cr	0	0	0	0	100	_
Hg	100	100	0	0	_	_
Ni	100	85.7	0	0	0	0

The *CF* of Hg, Mo, Sb, and Sn due to concentrations below the detection limit, Zn in all samples, and Fe in all samples (except S6, which faced a slight decrease in concentration and its *CF* decreased (0.99) were equal to 1 due to constant concentrations at all sampling points and therefore had a low contamination level in river sediments. The metals Al, Be, Co, Cu, Li, Pb, V, As, and Cd (except at S7, where their *CF* for As and Cd was ≤1) had *CF* values in the range of medium contamination levels. Cr in S4 and S7, Ba in S4, S6, and S7, and Ni in S3, S4, and S5 showed *CF* less than 1, indicating low contamination levels; in other locations, they had a medium contamination level. Mn also had *CF* > 1only in two points, S3 and S5, showing a medium pollution level; in the rest points, the contamination level was low. The maximum *CF* of Al, As, Ba, Be, Co, Cr, Cu, Mn, Pb, and V was recorded at S3 (the first point after leachate discharge), St at S5, Li, and Ni at S6, and Cd at S8. The other six heavy metals also had constant *CF* due to constant concentrations or below the detection limit. Mir Mohammad Ali et al. found in their study that the CF of Pb, As, and Cr was low in most of the sampling points in winter (<1), and only Cd had medium contamination level. In winter, Cr and Pb also showed medium contamination level in some points. The overall average value of *CF* in the two seasons was also Cd > Pb > Cr > As, respectively, which did not match the present study, which had the highest average for As (Ali et al., 2022). Varol also concluded in his study on the Tigris River that the highest *CF* values of the 10 metals studied were at the first point, which was the place of copper mine wastewater discharge. Different metals at this point showed different levels of contamination. Moderate contamination levels were also obtained for other metals at other points except for Pb. High *CF* values were also obtained at S3 and S7 due to the discharge of wastewater from different cities, which was consistent with the present study (Varol, 2011). Al, Cr, Fe, Co, Ni, and U had *CF* in the low contamination level range in the seven rivers studied in the study by Ustaoğlu and Saiful Islam, Pb and Cd had CF with significant contamination levels in three rivers, and Cu, Zn, As, Mn, Pb and Cd metals had *CF* values at moderate contamination levels in some rivers (Ustaoğlu & Islam, 2020).

3.2.2 PLI

PLI in river water varied between 0.59 (S8) and 0.78 (S2), which confirms the unpolluted level at all points along the river. This index in sediments also varied from 0.98 to 1.28, and at two points, S7 and S4, with PLI values of 0.98 and 0.99, respectively, there was an unpolluted level, and at other points with values higher than 1, it indicated an average level of pollution. The highest PLI value in sediments was also obtained at S3, the first point after the leachate discharge site. The average PLI value in sediments (1.1) indicates the average level of pollution caused by heavy metals in the river course. A comparison of the PLI values in the two water and sediment sections is shown in Fig. 2. According to Fig. 2, it can be seen that PLI in river water has a decreasing trend along the course, but in river sediments, the PLI value has gradually increased; this trend of changes could be due to the transfer of metals leached from upstream to downstream and their accumulation in the sedimentary environment of the river. In a study conducted by Amirul Islam on the Halda River, the value of this index was greater than 1 in all the studied points in sediments and had a moderate pollution level (1 < PLI < 2), which was consistent with other points except for S4 and S7 in this study (Islam et al., 2020). Saiful Islam et al. also found that the PLI was higher than 1 in all points and in both seasons of the study for heavy metals Cr, Ni, Cu, As, Cd, and Pb in the Buriganga River, and very high values of this index in points B4 to B8 were attributed to the possible effects of various industries and urban activities (Islam et al., 2018).

Table 6 Contamination factor for heavy metals in different river environments

Type Of Environment	Sampling Site	Al	As	Ba	Be	Cd	Co	Cr	Cu	Fe	Hg	Li	Mn	Mo	Ni	Pb	Sb	Sn	Sr	V	Zn	Ave
Water	S1	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
	S2	0.42	1.00	0.89	1.00	1.00	1.00	0.82	1.06	0.32	1.00	0.93	0.36	1.00	0.95	1.00	1.00	0.63	0.98	1.00	0.31	0.83
	S3	0.24	1.00	0.93	1.00	1.00	1.00	0.99	0.99	0.20	1.00	0.95	0.43	1.00	0.80	1.00	1.00	0.80	1.00	1.00	0.18	0.83
	S4	0.26	1.00	0.99	1.00	1.00	1.00	1.11	0.95	0.18	1.00	1.02	0.11	1.00	2.05	1.00	1.00	1.21	1.05	1.00	0.18	0.91
	S5	0.17	1.06	0.91	1.00	1.00	1.00	0.96	0.94	0.13	1.00	0.94	0.17	1.00	0.80	1.00	1.00	1.14	0.99	1.00	0.15	0.82
	S6	0.25	1.00	0.91	1.00	1.00	1.00	0.96	1.00	0.12	1.00	0.94	0.11	1.00	0.80	1.00	1.00	0.20	1.00	1.00	0.13	0.77
	S7	0.22	1.00	0.84	1.00	1.00	1.00	0.93	0.98	0.12	1.00	0.92	0.27	1.00	0.80	1.00	1.00	0.11	1.00	1.00	0.12	0.77
	S8	0.21	1.00	0.83	1.00	1.00	1.00	0.67	1.04	0.10	1.00	0.93	0.10	1.00	0.80	1.00	1.00	0.22	0.99	1.00	0.13	0.75
	Ave	0.35	1.01	0.91	1.00	1.00	1.00	0.93	1.00	0.27	1.00	0.95	0.32	1.00	1.00	1.00	1.00	0.66	1.00	1.00	0.28	0.83
Sediment	S1	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
	S3	1.40	2.69	3.13	1.13	2.56	1.31	1.30	1.55	1.00	1.00	1.70	1.92	1.00	0.73	1.32	1.00	1.00	0.30	1.66	1.00	1.43
	S4	1.02	1.49	0.92	1.05	1.01	1.04	0.79	1.14	1.00	1.00	1.49	0.90	1.00	0.99	1.06	1.00	1.00	0.38	1.12	1.00	1.02
	S5	1.22	1.69	1.26	1.12	1.03	1.15	1.19	1.22	1.00	1.00	1.36	1.04	1.00	0.50	1.17	1.00	1.00	0.58	1.30	1.00	1.09
	S6	1.36	2.24	0.84	1.13	2.54	1.25	1.17	1.25	0.99	1.00	1.91	0.79	1.00	1.83	1.23	1.00	1.00	0.22	1.12	1.00	1.24
	S7	1.15	1.00	0.82	1.09	0.67	1.05	0.94	1.07	1.00	1.00	1.58	0.62	1.00	1.55	1.07	1.00	1.00	0.51	1.14	1.00	1.01
	S8	1.39	1.65	1.04	1.37	2.80	1.24	1.31	1.22	1.00	1.00	1.64	0.87	1.00	1.26	1.32	1.00	1.00	0.40	1.37	1.00	1.24
	Ave	1.22	1.68	1.29	1.13	1.66	1.15	1.10	1.21	1.00	1.00	1.53	1.02	1.00	1.12	1.17	1.00	1.00	0.48	1.24	1.00	1.15

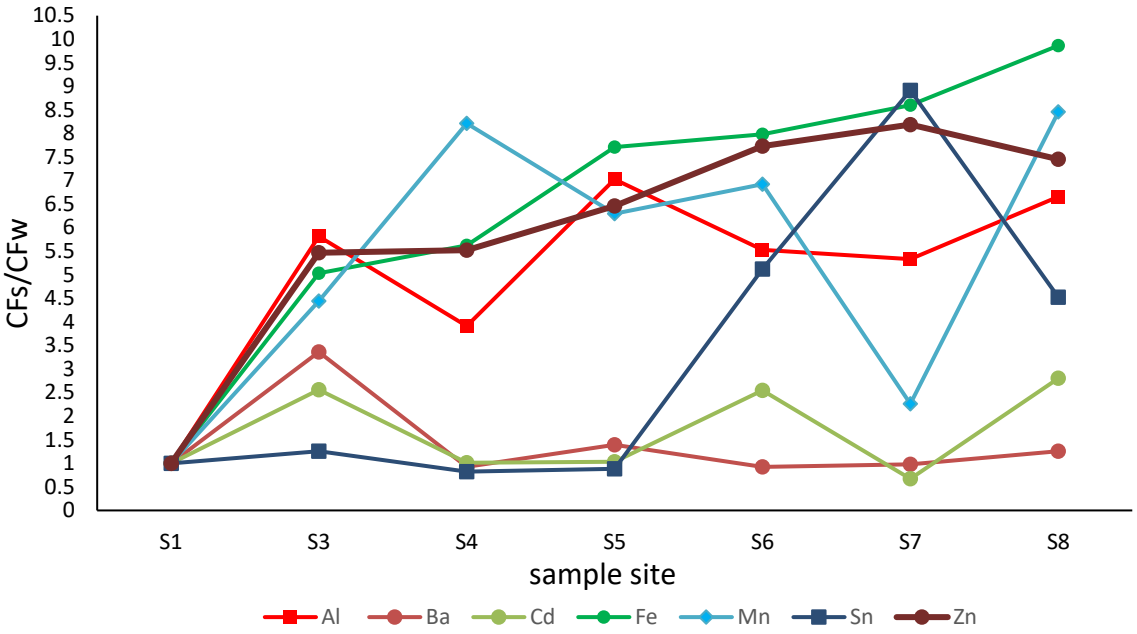


Fig. 1 Changes in the ratio of sediment to water contamination factor in the river for heavy metals

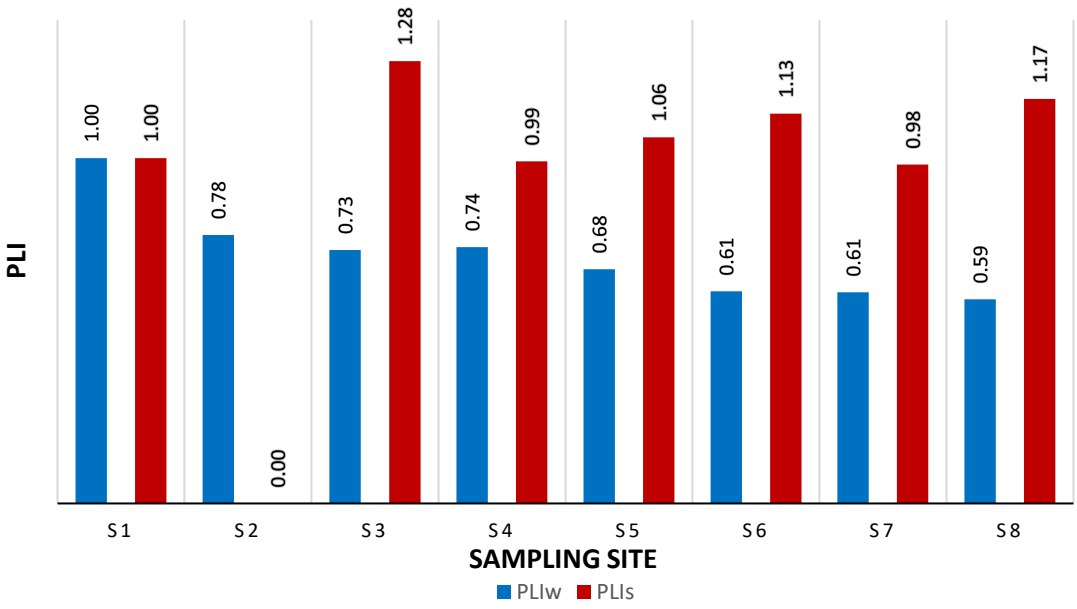


Fig. 2 Comparison of PLI index between water and sediment

3.2.3 EF

The *EF* value in S1 (background concentration) and also for Fe (reference metal) was calculated to be equal to 1. The lowest *EF* value in water samples was obtained for Al and Zn at S3, for Mn at S4, and for Sn at S7, and for other metals at the leachate entry point, i.e., S2. The maximum value of this factor was also recorded for Cr and Mn at S7, Sn at S5, and Ni at S4, and for other metals at S8. The trend of *EF* changes in the water at the studied sites is shown in Fig. 3. The *EF* value for all metals along the river course and at all points generally increased except for four metals: Cr, Mn, Ni, and Sn; among which the EF value of Cr decreased after S7 and Mn also showed a trend of consecutive increase and decrease and after reaching its maximum value at S7, it finally decreased at S8. The EF value of Ni also continued its gradual increase after a sudden increase and reaching a maximum value at S4, and then a sudden decrease at S5. The trend of changes in the *EF* of Sn was also reversed and was a gradual increase to a maximum value at S5, and then a sudden decrease and reaching a minimum value at S7, and an increase again at S8. The overall average *EF* of all metals at each point along the sampling route was S8 (7.41) > S7 (6.58) > S5 (6.31) > S6 (6.24) > S4 (5.09) > S3 (4.15) > S2 (2.63) > S1 (1.00). Accordingly, S1 (background) was considered without enrichment, S2 had low enrichment, S3 had moderate enrichment, and S4 to S8 had relatively severe enrichment.

The trend of changes in *EF* of the studied heavy metals in sediments is shown in Fig. 4. In sediments, the *EF* of Hg, Mo,

Sb, and Sn due to concentrations below the detection limit and Zn due to constant concentrations at all sampling points and also the stability of the amount of Fe present in the sediments as a reference metal (except at S6, which was accompanied by a slight decrease and caused the recording of the maximum *EF* for these metals) was equal to 1. The maximum EF in sediments was obtained for St in S5 and for Be, Cd, Cr, and Pb at S8, Ni, and Li at S6, and for other metals at S3. The minimum *EF* value was obtained for Li Ni at S5 and for St at S6.

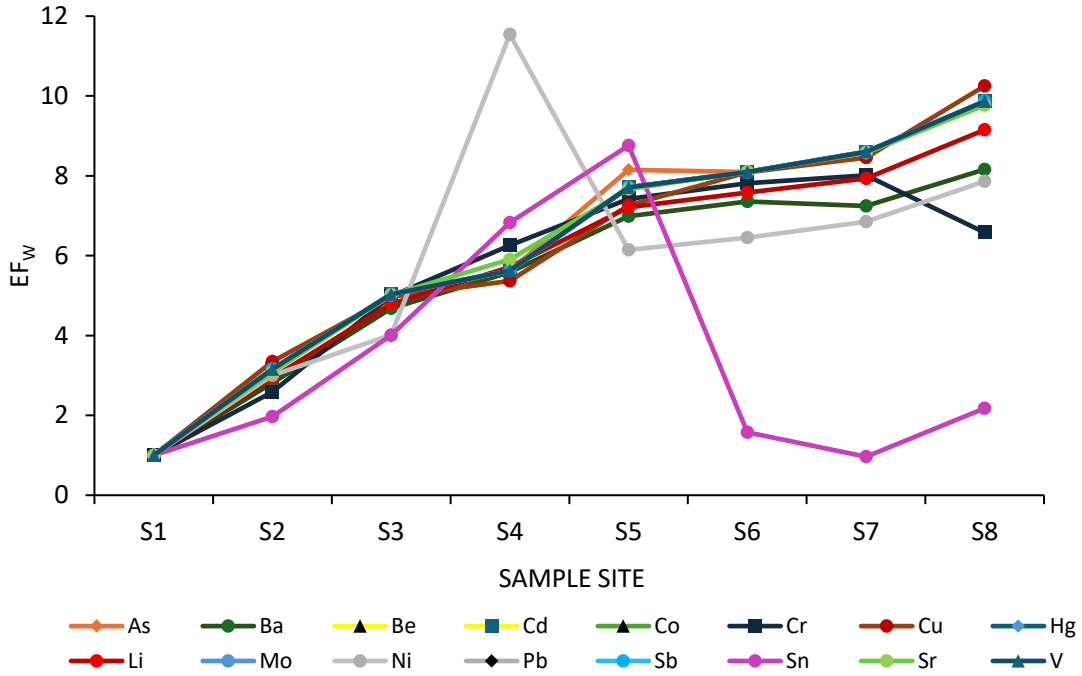


Fig. 3 EF Change trends for heavy metals in river water

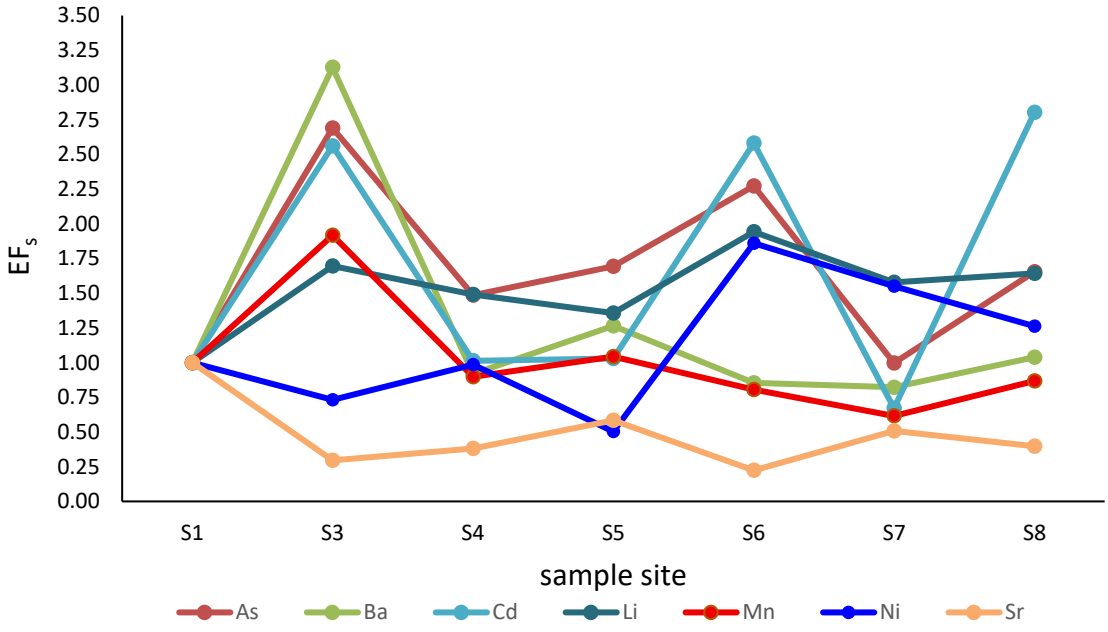


Fig. 4 EF change trends for heavy metals in river sediment

The *EF* value for As, Ba, Cd, Cu, and Mn was also recorded at the minimum value at S7, and the other metals had the minimum *EF* value at S4. The *EF* values for different metals in river sediments followed different patterns. Some metals, such as arsenic and barium, initially had the maximum *EF* value (at the point of leachate entry) and then followed a decreasing trend along the river course. Some other metals, such as Cr and Pb, had the opposite trend, and their *EF* values changed from low to high. Others, such as Ni and Li, first increased and then decreased. The average *EF* of total metals at different sampling points for sediments was S3 (1.43) > S6 (1.26) > S8 (1.24) > S5 (1.09) > S4 (1.02) > S7 (1.01) > S1

(1.00), respectively. Therefore, S1 was without enrichment, and all other points had low metal enrichment. EF values greater than 1.5 indicate metal contamination as a result of human activity, and values around 1 indicate the natural concentration of metals and the effect of weathering processes and entry due to crustal compounds (Islam et al., 2015). *EF* study of both aquatic and sedimentary environments shows that most heavy metals in water have human sources, while the amount of this factor in sediments for most metals is less than 1.5 in most places, indicating the natural origin of these metals. The contradictory trend of this factor for some metals and its increase and decrease in sampling points can indicate the presence of polluting sources downstream of the leachate entry point. Field studies showed that in the study route, there are rural areas, cultivated lands, asphalt plants, and sand and gravel plants. Therefore, EF changes can be caused by the entry of rural wastewater into the river or the entry of agricultural wastewater containing agricultural toxins and pesticides. It is also possible that the asphalt plant and sand and gravel plants play a major role in causing *EF* changes because sand and gravel extraction activities can cause the release and washing of heavy metals, or the wastewater from their processing can enter water sources. Asphalt production through asphalt binder, asphalt additives, and recycled concrete aggregate can also introduce various metals, including V, Ni, Zn, Mo, Cu, Fe, Ag, and As into the river through the resulting effluent (Park, 2024; Rentier & Cammeraat, 2022; Spreadbury et al., 2021; Von Gunten et al., 2020). In their study on Karun River sediments, Keshavarzi et al. examined EF of nine heavy metals and concluded that Cd, Cr, As, and Cu were in the pollution-free range at most sampling points. Moderate enrichment (2<EF<5) of Ni in all samples, moderate enrichment of Pb (caused by municipal wastewater and oil refinery), and Zn in most samples, as well as severe to very severe enrichment due to Hg in most samples were other results of this study that were attributed to the discharge of municipal and industrial wastewater containing these metals. Although these results were in conflict with the present study, they can confirm the effect of human activities on the increase of heavy metals in river sediments (Keshavarzi et al., 2015). Varol and Şen found in their study in the upper reaches of the Tigris River that the discharge of metal-containing wastewater from point sources (copper mining) into the river can increase the concentration and, consequently, the enrichment factor of metals such as Cd, Co, Cu, Ni, Pb and Zn in sediments at downstream points of the receiving river (Varol & Şen, 2012).

3.2.4 I_{geo}

I_{geo} of Hg, Mo, Sb, and Sn was recorded as constant and negative due to concentrations below the detection limit in all sample points and Zn in all points and Fe in all points except S6 (although the value of this index was negative at this point too) due to constant concentrations in the sediments, which indicates the lack of contamination of the sediments by these metals. Al, Be, Cr, Co, Pb, and St were negative at all points and therefore were free of contamination in the river sediments. I_{geo} values of Ba (1.06), Cu (0.05), Mn (0.35), and V (0.14) were positive, and the highest value at S3, and Ba was in the almost contaminated group, and the other three metals were in the non-contaminated to almost contaminated group. Cd at S3 and S8 and Ni at S6 and S7 had values between 0 and 1 and were in the non-contaminated to almost contaminated group.

Table 7 Geoaccumulation index of heavy metals in river sediment

Sampling site	1	3	4	5	6	7	8
Al	-0.58	-0.10	-0.55	-0.29	-0.14	-0.38	-0.11
As	-0.58	0.84	-0.01	0.17	0.58	-0.59	0.14
Ba	-0.58	1.06	-0.71	-0.25	-0.83	-0.87	-0.53
Be	-0.58	-0.40	-0.52	-0.42	-0.41	-0.46	-0.13
Cd	-0.58	0.77	-0.57	-0.54	0.76	-1.16	0.90
Co	-0.58	-0.19	-0.53	-0.38	-0.26	-0.51	-0.28
Cr	-0.58	-0.20	-0.92	-0.34	-0.36	-0.67	-0.19
Cu	-0.58	0.05	-0.39	-0.29	-0.26	-0.49	-0.30
Fe	-0.58	-0.58	-0.58	-0.58	-0.61	-0.58	-0.58
Hg	-0.58	-0.58	-0.58	-0.58	-0.58	-0.58	-0.58
Li	-0.58	0.18	-0.01	-0.15	0.35	0.07	0.13
Mn	-0.58	0.35	-0.74	-0.52	-0.92	-1.28	-0.79
Mo	-0.58	-0.58	-0.58	-0.58	-0.58	-0.58	-0.58
Ni	-0.58	-1.03	-0.60	-1.58	0.29	0.05	-0.25
Pb	-0.58	-0.18	-0.50	-0.36	-0.28	-0.49	-0.18
Sb	-0.58	-0.58	-0.58	-0.58	-0.58	-0.58	-0.58
Sn	-0.58	-0.58	-0.58	-0.58	-0.58	-0.58	-0.58
Sr	-0.58	-2.34	-1.98	-1.36	-2.77	-1.56	-1.92
V	-0.58	0.14	-0.42	-0.21	-0.43	-0.39	-0.13
Zn	-0.58	-0.58	-0.58	-0.58	-0.58	-0.58	-0.58

As was found to be uncontaminated to almost contaminated at S3, S5, S6, and S8, and at uncontaminated levels at other sites. Li was also found to be negative at S4 and S5 and at uncontaminated to almost contaminated levels at other sites. The order of I_{geo} of total metals at the sampling sites was S3 (-4.57) > S8 (-7.14) > S6 (-8.2) > S5 (-10.02) > S1 (-11.7) > S4 (-11.98) > S7 (-12.25), and thus the I_{geo} of total metals at all sites was found to be uncontaminated. The values of this index for the studied elements are presented in Table 7.

In a study conducted by Karbasi et al., it was concluded that the values of the I_{geo} calculated for the average concentration of Pb (3.78), Cd (0.57), Zn (4.36), and Cu (10.01) in the sediments of the Shoor River were in the range of highly polluted, unpolluted to almost polluted, highly polluted to very very polluted, and very very polluted, respectively, which was in contrast to the results of the present study (Karbassi et al., 2008). Salati and Mori also found in a study conducted on heavy metals (Cr, Mn, Ni, Cu, Zn, Pb, and Cd) in the dry River that except for manganese, which had negative values at all sampling points and was at the unpolluted level, and Cd, which had values less than zero only at S1, the other metals showed different levels of contamination with this index at other sampling points (Salati & Moore, 2010). Nur Fadlillah et al., in their study, assessed the ecological risk of heavy metals Pb, Cu, Cd, Cr, Fe, and Al in the Winongo River and concluded that the land accumulation index for aluminum and iron had different levels of contamination, and the remaining metals were at a level without contamination (Fadlillah et al., 2023).

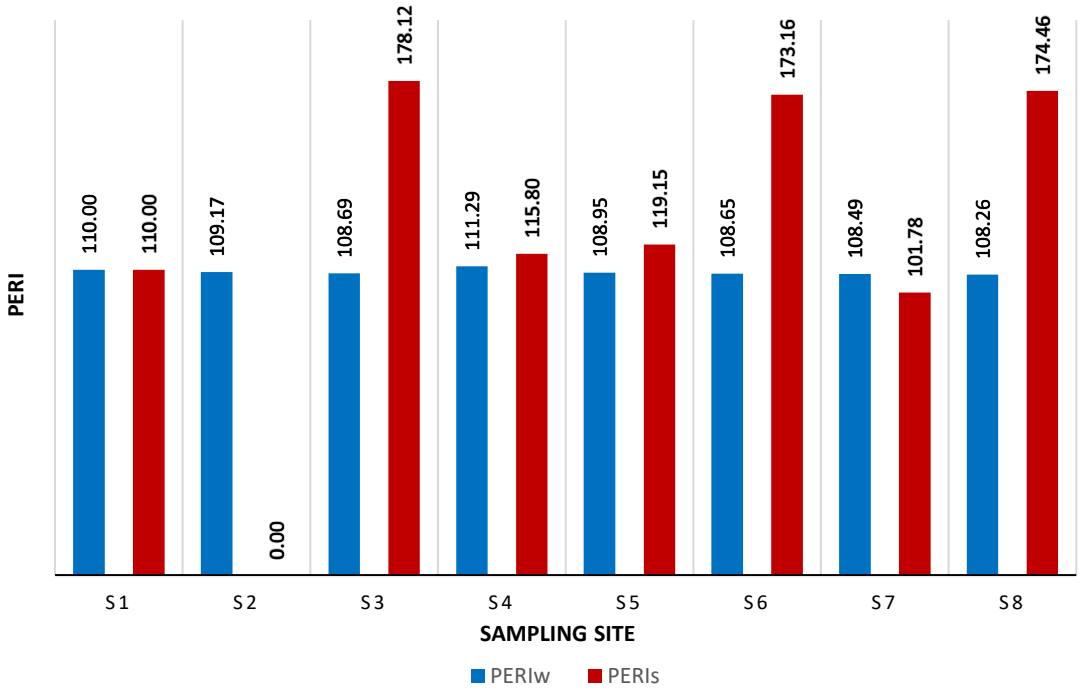


Fig. 5 Comparison of PERI index between the water and sediment of the river

3.2.5 PERI

The comparison of the results of PERI calculations in the two environments of water and river sediments is presented in Fig. 5. In performing this parameter, values lower than the detection limit of metals were considered equal to the detection limit. According to this figure, the PERI at all points in the river water was recorded in an almost constant range (108.26 - 111.29) and was placed in the group with a low ecological risk level. However, the level of PERI in river sediments at S3 (the highest value), S6, and S8 increased and were placed in the range with a medium ecological risk level. Sheykhi and Moore, in a study they conducted on the Kor River, found that river sediments at points 4, 5, 8, and 9 had a very high ecological risk degree and at points 2 and 6 had a high ecological risk, which did not correspond to the findings of the present study (Sheykhi & Moore, 2013). Banu et al. also concluded in their study on the Turag River that the PERI of the five metals studied was low in the first four points and moderate in only point 5 (Banu et al., 2013). In the four rivers affected by mining activities studied by Jimenez-Oyola et al., the PERI from heavy metals was high in the Fermı´n River and very high in the other rivers, with the highest contribution of As and then Cd (Jiménez-Oyola et al., 2021).

4. Conclusion

The present study investigated the effects of landfill leachate on heavy metal pollution in the water and sediments of the receiving water body, the Garan River, in Marivan County, Iran. Using heavy metal pollution indices (CF, PLI, I*geo*, EF, and PERI), it was found that although the overall level of heavy metal pollution in water and sediments of the river is not very high, some metals such as arsenic and chromium in water have acute and chronic effects on human health. In addition, arsenic in sediments exceeded the threshold concentration, and barium, nickel, and mercury exceeded the threshold concentrations, indicating possible effects and potential ecological risks for aquatic organisms. The trend of changes in pollution indices such as CF, PLI and EF indicates the gradual

accumulation of metals in sediments and their transfer to the downstream of the river. PLI and PERI indices showed that sediments in some places are exposed to moderate ecological risk. Given the ecological importance of the Garan River and its human uses, periodic monitoring and preventive measures are recommended to reduce the entry of pollutants and maintain the quality of the river. Given the increasing trend of PLI in sediments and EF in river water in the downstream reaches of the river, further metal surveys are recommended at further distances, including at the Garan Dam.

Statements and Declarations

Ethical considerations

Ethical issues (Including plagiarism, informed consent, misconduct, data fabrication and/or falsification, double publication and/or submission, redundancy, etc.) have been completely observed by the author.

Data availability

Data will be made available on request.

Conflicts of interest

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

Author contribution

B. Ahmadi: Writing the main Draft; H. Mohammadi, H. Salehzadeh, B. Ghaderi, and P. Yonessi: Conceptualization; Writing – Review & Editing; and B. Shahmoradi: Conceptualization, Methodology, Review, and Final Proof.

AI Use Declaration

During the preparation of this manuscript, the authors used AI tools for designing the Graphical Abstract. All content has been carefully reviewed and revised by the authors, who take full responsibility for the final version of the manuscript.

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Assessment of environmental health indicators in hospitals in southern Kerman, Iran: A cross-sectional study (2021)

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ABSTRACT

Hospitals are fundamental components of healthcare systems and play an important role in providing medical care, preventing disease, and protecting population health. Monitoring environmental health indicators in hospitals is essential for identifying deficiencies and improving healthcare environments. This study aimed to assess environmental health indicators in six teaching hospitals affiliated with Jiroft University of Medical Sciences in southern Kerman Province, Iran. This descriptive-analytical, applied, cross-sectional study was conducted in 2021. The study population consisted of six teaching hospitals affiliated with Jiroft University of Medical Sciences, and all hospitals were included using a census sampling method. Data were collected using the checklist of the National Evaluation Program of Public Hospitals (Health and Hygiene Department), which consisted of 99 questions with a maximum score of 1,180 points. The overall environmental health status of the six hospitals varied across the evaluated domains. Water and wastewater health and occupational health were among the least favorable domains, whereas patient health and infection control showed comparatively better performance. Hospital C had the highest overall score (1,144 points), while Hospital F had the lowest overall score (974 points). The environmental health status of the evaluated teaching hospitals was generally at a moderate level. Particular attention should be given to water and wastewater management and occupational health. Regular assessment and targeted interventions may help address identified deficiencies and improve environmental health conditions in hospitals.

Highlights

- A comprehensive environmental health assessment covered all six teaching hospitals.
- Participating hospitals showed an overall moderate status in environmental health.
- Water, wastewater, and occupational health areas required the greatest improvement.
- Patient safety and infection prevention indicators showed the strongest performance.
- Findings guide priorities for stronger environmental health management in hospitals.

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1. Introduction

The primary responsibilities of governments toward their populations include ensuring security, providing education, and protecting public health. Health promotion and maintenance are fundamental components of public health and

play an important role in advancing health equity and social justice. The Ottawa Charter for Health Promotion emphasizes that health is influenced by a broad range of social, economic, and environmental conditions and identifies equity as a fundamental principle of health promotion (World Health Organization, 1986; Edwards, 2009). Accordingly, achieving and maintaining optimal health should be regarded as an important societal and public health priority. Health is a complex and multidimensional concept influenced by numerous determinants, including socioeconomic conditions, environmental factors, health-related behaviors, and the organization and performance of health systems (Edwards, 2009; Terris, 1994).

An effective and efficient health system is therefore essential for improving population health and reducing health inequalities. According to the World Health Organization (WHO), a health system comprises all organizations, people, and actions whose primary purpose is to promote, restore, or maintain health (Michel et al., 2021; Rifkin, 2014). Health systems extend beyond the provision of clinical care and include activities aimed at addressing the broader determinants of health and creating supportive environments for disease prevention and health promotion (Michel et al., 2021; Rifkin, 2014). Within this context, hospitals represent a critical component of healthcare systems and are central to the delivery of specialized and acute healthcare services. However, healthcare delivery in hospital settings is also associated with risks, particularly healthcare-associated infections (HAIs), which can adversely affect patient outcomes, quality of care, and healthcare costs (Muller et al., 2016; Storr et al., 2017). Environmental contamination within healthcare facilities can contribute to the transmission of pathogenic microorganisms and the occurrence of HAIs, highlighting the importance of environmental hygiene and effective infection prevention and control (IPC) practices (Muller et al., 2016; Tartari et al., 2022). Consequently, adherence to established hygiene standards and implementation of evidence-based IPC measures are essential for protecting patients and healthcare workers, improving the safety and quality of healthcare, and reducing the burden of preventable healthcare-associated infections (Storr et al., 2017; Tartari et al., 2022).

The World Health Organization (WHO) describes the health-promoting hospital as a healthcare setting that extends beyond the provision of medical treatment to actively promote the health of patients, healthcare workers, and the surrounding community. Hospitals play a critical role in the delivery of healthcare services and serve as important settings for health education, clinical training, and medical research. Their mission therefore extends beyond the diagnosis and treatment of disease to include the restoration and promotion of patients' health, the education and training of healthcare professionals, and the advancement of knowledge aimed at improving population health (Afshari et al., 2016; Whitehead, 2005). Accordingly, health promotion within hospital settings is a complex and multidimensional process that depends on the interaction of organizational, environmental, and individual factors (Johnson & Baum, 2001; Whitehead, 2005).

Failure to adequately address these factors may compromise the hospital's ability to fulfill its healthcare mission and provide a safe, high-quality care environment (Johnson & Baum, 2001).

Adherence to appropriate hygiene and infection prevention and control (IPC) standards is a fundamental component of safe and effective hospital care (World Health Organization, 2016). Inadequate implementation of such measures can contribute to the occurrence and transmission of healthcare-associated infections (HAIs), which remain a major challenge for healthcare systems worldwide (Allegranzi et al., 2011; World Health Organization, 2016). HAIs are associated with increased morbidity and mortality, prolonged hospital stays, impaired quality of care, and substantial additional healthcare costs (World Health Organization, 2016; Zimlichman et al., 2013). Evidence from different healthcare settings indicates that HAIs affect a considerable proportion of hospitalized patients, with the burden generally being higher in low- and middle-income countries than in high-income countries (Allegranzi et al., 2011; Magill et al., 2014). The WHO estimates that approximately 7% of patients in acute-care hospitals in high-income countries and up to 15% of patients in low- and middle-income countries acquire at least one HAI during their hospital stay (Storr et al., 2017; World Health Organization, 2016). These infections therefore represent a major patient-safety and public-health concern.

The development and transmission of HAIs are influenced by complex interactions among susceptible hosts, infectious agents, and the healthcare environment (Allegranzi et al., 2011; Otter et al., 2011). Hospitalized patients are often particularly vulnerable because of underlying illnesses, impaired immune responses, invasive medical procedures, and prolonged exposure to healthcare environments (Allegranzi et al., 2011; Magill et al., 2014). At the same time, hospital environments may harbor a wide range of pathogenic microorganisms, including antimicrobial-resistant organisms, which can persist on surfaces, medical equipment, and other environmental reservoirs (Otter et al., 2011). Contaminated environmental surfaces have been recognized as potential reservoirs for healthcare-associated pathogens and may contribute to their transmission, particularly when appropriate environmental cleaning and disinfection practices are not adequately implemented (Leas et al., 2018; Otter et al., 2011). Consequently, the healthcare environment can play an important role in the transmission of infectious agents, particularly when environmental cleaning and disinfection, ventilation, water and sanitation, waste management, and other infection prevention measures are inadequate (Johnson & Baum, 2001; Leas et al., 2018; Otter et al., 2011).

Environmental health in hospitals encompasses a broad range of measures aimed at maintaining a safe and hygienic healthcare environment and reducing opportunities for pathogen transmission. These measures include appropriate environmental cleaning and disinfection, access to safe water and sanitation, effective healthcare-waste management, adequate ventilation, proper management and

decontamination of healthcare equipment and environmental surfaces, and compliance with established IPC practices (Johnson & Baum, 2001; Leas et al., 2018; Storr et al., 2017). Environmental cleaning is particularly important because contaminated surfaces can serve as reservoirs for healthcare-associated pathogens, while appropriate cleaning and disinfection practices can reduce environmental contamination and contribute to infection prevention (Leas et al., 2018; Otter et al., 2011). Therefore, systematic assessment and continuous improvement of environmental health conditions within hospitals should be considered integral components of comprehensive infection prevention strategies and may contribute substantially to reducing the occurrence and transmission of HAIs (Johnson & Baum, 2001; Leas et al., 2018; Storr et al., 2017).

Achieving and maintaining an appropriate level of environmental health in hospitals requires compliance with established standards, regulations, and technical requirements. Hospital environmental health standards provide a framework for maintaining safe and hygienic conditions and establishing minimum acceptable requirements for healthcare facilities. Compliance with these standards is particularly important because environmental hygiene encompasses multiple domains, including cleaning and disinfection, water and sanitation, healthcare-waste management, ventilation, and the appropriate management of environmental surfaces and equipment (Carling, 2016; Storr et al., 2017).

Despite the recognized importance of environmental hygiene in preventing healthcare-associated infections (HAIs), evidence indicates that deficiencies in environmental health practices remain a concern in healthcare facilities (Carling, 2016; Peters et al., 2022). Environmental surfaces can serve as reservoirs for pathogenic microorganisms, and inadequate cleaning and disinfection may increase opportunities for pathogen transmission within healthcare settings (Carling, 2016; Otter et al., 2011). A systematic review of environmental hygiene interventions found that most included studies reported reductions in healthcare-associated infections or patient colonization following environmental hygiene interventions, further emphasizing the importance of maintaining appropriate environmental conditions in healthcare facilities (Peters et al., 2022).

Evidence from Iran has similarly identified deficiencies in several components of hospital environmental health. Studies conducted in Iranian hospitals have reported shortcomings in healthcare-waste segregation, collection, storage, treatment, and disposal, as well as inadequate implementation of relevant management requirements (Farzadkia et al., 2009; Malekahmadi et al., 2014). For example, a study conducted in teaching hospitals affiliated with Iran University of Medical Sciences identified deficiencies in waste segregation, collection, transportation, treatment, and disposal and emphasized the need for appropriate standards, regulations, and personnel training to improve existing conditions (Farzadkia et al., 2009). More recent evidence from Kermanshah has also highlighted persistent challenges in hospital waste management and the need for more sustainable and systematic approaches (Alighardashi et al., 2024). Similarly, recent evidence from Isfahan has highlighted excessive infectious waste generation, gaps in waste treatment, and the need for more sustainable approaches to hospital waste management (Mohammadi, 2025).

These findings suggest that hospital environmental health should not be considered a static condition but rather as an ongoing quality and safety priority requiring regular monitoring, periodic assessment, and continuous improvement (Peters et al., 2022; Storr et al., 2017). Systematic evaluation of compliance with environmental health standards can help identify deficiencies, guide corrective interventions, and support the maintenance of a safe healthcare environment for patients, healthcare workers, and visitors (Carling, 2016; Storr et al., 2017).

Given the potential role of hospital environmental health monitoring in reducing healthcare-associated infections and associated healthcare costs, maintaining and regularly assessing environmental health conditions is an important component of hospital quality and patient safety. However, despite the importance of this issue, there is limited evidence regarding the status of environmental health indicators in teaching hospitals in southern Kerman Province. Therefore, considering the importance of maintaining appropriate environmental health conditions and the lack of comprehensive evidence in this setting, the present study aimed to assess the status of environmental health indicators in teaching hospitals affiliated with Jiroft University of Medical Sciences in 2021.

2. Materials and Methods

This study is a descriptive-analytical and applied approach that uses cross-sectional data from 1400. The study population is comprised of teaching hospitals affiliated with Jiroft University of Medical Sciences, including six teaching hospitals. In accordance with the number of hospitals, the census method was used in this study, and all teaching and medical hospitals in the south of Kerman province were investigated. These data were collected using the checklist of the National Evaluation Program of Public Hospitals (Health and Hygiene Department). The Ministry of Health and Medical Education has prepared this checklist, whose content validity has been approved.

This study tool includes 99 questions and 1180 points that aim to assess the health status of the construction and the environment in terms of health Index (30 questions), Health mission of manpower (9 questions), sanitation (27 questions), waste management (9 questions), water quality and sewage (2 questions), infection control (10 questions), and occupational health (12 questions).

According to the standard checklist, each of the questions had three answers (yes, somewhat, and no). According to the type of questions, the option "yes" received 20, 15, and 10 points, and the option "no" earned zero points, and "somewhat" was given a score of 1 to 19, 1 to 14, and 1 to 10. In order to control measurement errors, checklists were completed, in all six hospitals, by a trained environmental health expert through observation and interview. Inter-rater reliability was not

separately assessed in the present study because the checklist was completed by a single trained environmental health evaluator. To minimize measurement error, the evaluator followed the standardized scoring criteria and procedures specified in the national checklist.

The maximum standard limit for each of the six indices of the health status of hospitals was equal to the sum of the highest scores for each question. The minimum standard limit was equal to 20% of the maximum standard limit.

Finally, scores between 80-60, and over 80%, or the difference between the highest and lowest standards in each dimension, were divided into three groups: weak, medium, and good. Due to the principle of trustworthiness and confidentiality of information, hospitals were not named.

3.1 Results and Discussion

In this study, health indices of six hospitals in the south of Kerman province were analyzed. In order to determine the health status of these hospitals separately, the following data analysis has been conducted. The total number of hospital scores (A) obtained was 1087, which, according to Table 2, represents an average level of health. The water and wastewater hygiene index has the greatest percentage difference compared to another Index. As for environmental health and cleanliness in the hospital, a difference in points is also found with the highest standard level being 37.6%. In order for a sanitation index to be calculated, there must be the lowest percentage difference between the highest score and the highest standard. This hospital has favorable construction conditions and is in a healthy state of sanitation.

In the present study, water and wastewater health and occupational health were among the less favorable domains. These findings highlight that hospital environmental health extends beyond healthcare waste management and requires simultaneous attention to wastewater treatment, water quality, occupational safety, infection prevention, and environmental hygiene. The observed variation among hospitals also suggests that hospital-level organizational and managerial factors may influence compliance with environmental health standards. Therefore, interventions should be tailored to the specific deficiencies identified in each hospital rather than applying a uniform corrective strategy.

Table 1 The values of the lowest and highest scores in different health indices

Study index	Number of questions	The minimum standard	Maximum standard	Difference between Max and Min standard	Unfavorable	Moderate	Favorable
The health mission	9	100	125	25	less than 115	115.1-120	more than 120
Sanitation condition	27	244	305	61	less than 288	288.1-293	more than 293
Hospital waste	9	72	90	18	less than 83	83.1-86	more than 86
Water and wastewater	2	16	20	4	less than 18	18.1-19	more than 19
Occupational Health	12	124	155	31	less than 143	143.1-149	more than 149
Patient health and infection control	10	96	120	24	less than 97	97.1-101	more than 101
Environmental health status	30	292	365	73	less than 340	340.1-346	more than 346
Total	99	944	1180	236	less than 1084	1084.6 - 1114	more than 1114

The hospital obtained a total of 1109 points (B), which is the average score of the hospital's indices (Table 2). Consequently, this hospital has the highest standard of sanitation and control of nosocomial infections, and its water and sewage health index differs the most from the desired standard. There is a difference of 28% in environmental cleanliness and 9% in occupational health from the standard limit. It is evident from the results of this study that hospital (C) possesses a comprehensive desirableness; its acquired score was 1144, and it is considered to be one of the most reputable hospitals in terms of health quality. A total of 1014 points were obtained by Hospital (D), which is one of the weaker hospitals according to Table (2) based on a total of 7 indexes. There was an unfavorable level of cleanliness and hygiene in this hospital compared to the highest standards. In terms of occupational health, water and sewage health, and sanitation status, this hospital differs the most from the standard. A total of 1085 points have been earned by Hospital (E). This score indicates the overall health of the hospital based on indexes such as cleanliness and environmental health.

When comparing water and wastewater hygiene, the hospital had the highest percentage difference from the highest

standard. In terms of sanitation, it is only 8% different from what is desirable. Hospital (F) has the lowest score among the six hospitals studied. A score of 974 indicates that this hospital is weak in terms of health indexes. Hospital (F) in terms of water and wastewater health index was 125% lower than the highest standard. There was a large difference between the two indexes of environmental health and the health mission within this hospital, at 86.4% and 72%, respectively. Inefficient management is responsible for the poor performance of this hospital.

Table 2 Amount of acquired points and percentage of points acquired with the highest limit by different Indices

Hospital	Score and percentage	The health mission	Sanitation condition	Hospital waste	Water and wastewater	Occupational Health	Patient health and infection control	Environmental health status	Total points
A	Score	118	300	85	16	148	102	318	1087
	Differential percentage	28	8	28.7	100	22.6	14.3	37.6	90
B	Score	114	300	90	18	152	105	330	1109
	Differential percentage	44	8	0	50	9.7	0	28	15.5
C	Score	120	302	90	18	152	105	357	1144
	Differential percentage	13.6	4.8	0	50	9.7	0	6.4	0
D	Score	115	270	87	18	140	98	286	1014
	Differential percentage	39.6	56	16.7	50	50.2	33.3	63.2	125
E	Score	116	300	85	16	138	102	336	1085
	Differential percentage	35.2	8	27.8	100	55.6	14.3	23.2	96
F	Score	107	281	82	15	137	92	260	974
	Differential percentage	72	38.4	44.4	125	58.1	60	86.4	150

Table 3 The average score of different Index of the studied hospitals

Index of the study	Average score	Weak	Average	Good
The health mission	2.5±115	2 (33.3)	4 (6.66)	0 (0)
Sanitation condition	13.4±292.16	2 (33.3)	0 (0)	4 (66.6)
Hospital waste	2.5±87.5	1 (16.16)	2 (33.3)	3 (50)
Water and wastewater	17.6 ±1.8	3 (50)	3 (50)	0 (0)
Occupational Health	144±0.6	3(50)	1 (16.16)	2 (33.3)
Patient health and infection control	100.6±1.7	1 (16.16)	1 (16.16)	4 (66.6)
Environmental health status	314.5±14.8	5 (83.8)	0 (0)	1 (16.16)
Total	1068.8 ±2.4	3 (50)	2 (33.3)	1 (16.16)

Although the data were collected in 2021, the findings provide a baseline assessment of environmental health conditions in teaching hospitals in southern Kerman Province. The study was based on a standardized national hospital health checklist and included all six teaching hospitals affiliated with Jiroft University of Medical Sciences, thereby documenting the

environmental health status of these hospitals at the time of assessment. The findings can serve as a reference point for subsequent evaluations and for identifying areas that require continued monitoring and improvement. However, because environmental health practices and hospital management systems may change over time, the findings should not be interpreted as representing the current environmental health status of these hospitals. The time elapsed since data collection is therefore acknowledged as a limitation of the present study.

3.1 Limitations

This study has several limitations. First, the data were collected in 2021; therefore, changes in hospital infrastructure, environmental health practices, and management systems after the study period may limit the generalizability of the findings to the current situation. Second, the checklist was completed by a single trained environmental health evaluator. Although standardized scoring procedures were followed, the absence of multiple independent evaluators prevented assessment of inter-rater reliability and may have introduced observer bias. Future studies should consider repeated assessments and the use of multiple independent evaluators to improve the reliability and temporal relevance of the findings.

4. Conclusion

As the most important health centers, hospitals should be excellent in terms of health indicators. However, this study has revealed that the level of environmental health and indicators evaluated by 6 hospitals under the auspices of Jiroft University of Medical Sciences is reasonable. This is consistent with previous studies conducted in Isfahan, Qom, Fars, Kermanshah, and Karaj hospitals. It appears that the country's hospitals, despite their health mission, have received little attention and that basic measures must be taken to improve performance. According to the various indexes, indicators related to environmental health status, water quality and sewage health, and occupational health are poorly evaluated. In contrast, the indicators related to patient health and infection control are in reasonable shape. The reason for this is the numerous measures taken in recent years to prevent nosocomial infections in the country. It is commendable that the medical staff has taken such measures. A weakness of some hospitals is their failure to explain the health mission of human resources. In the analysis of the data, we found that a hospital that has been weak in this area does not fare well in other areas. The serious mission hospitals have in improving society has been ignored due to reasons such as clients not adhering to health standards and inadequate supervision. The most unfavorable indicator is related to the environmental health status of hospitals, and a major cause of this is the lack of regular cleaning. The evidence shows that the principles of hospital waste collection are appropriate, and that only temporary landfills are unsanitary. As it relates to wastewater, the most critical issue is the management of treatment plants, which must also be overseen by environmental health. There are 33% of hospitals that are considered to be weak due to the age of the construction and defects in the floors, laundry and kitchen, which are costly to maintain. According to the points earned, Jiroft University of Medical Sciences hospitals are in a moderate state, and comprehensive management is required in order to achieve the desired conditions. Therefore, Jiroft University of Medical Sciences is expected to take appropriate measures in order to turn the current situation into a favorable one.

Statements and Declarations

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Ethical considerations
Ethical issues (Including plagiarism, informed consent, misconduct, data fabrication and/or falsification, double publication and/or submission, redundancy, etc.) have been completely observed by the author.

Data availability
Data will be made available on request.

Conflicts of interest
The author declares that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Author contribution
MM. Soori: Writing Main Draft; S. Akhoundi Taraadeh, F. Eslami, and R. Noroozpoor: Conceptualization; Writing – Review & Editing; and A. Hosein Panahi: Conceptualization, Methodology, Review, and Final Proof.

AI Use Declaration
During the preparation of this manuscript, the authors used AI tools for designing the Graphical Abstract. All content has been carefully reviewed and revised by the authors, who take full responsibility for the final version of the manuscript.

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Assessment of Environmental Health Status and Compliance with Health Standards in Dental Clinics in Ardabil, Iran (2026)

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ABSTRACT

Dental clinics, as public centers, must comply with environmental hygiene principles to reduce the risks of contamination and transmission of important diseases such as hepatitis and AIDS. The present study was conducted with the aim of determining the environmental hygiene status of dental clinics in Ardabil in 2026. In this descriptive cross-sectional study, 13 active dental clinics in Ardabil were surveyed using a census method. The required information was collected from dental offices based on a health inspection checklist with form code 920318/111. The data were analyzed using Excel software. The status of the health indicator studied in the section of specific criteria of dental clinics was in accordance with the regulations, with a frequency of 90.38%, and the section of hygiene of tools and equipment and waste management violated the regulations, with frequencies of 73.37, and 72.12%, respectively. Overall, the level of compliance with the regulations in the present study in 1404 was assessed with a frequency of 80.48%. To improve the health indicators of the dental clinics studied, it is recommended to hold training courses on the principles of disinfection and sterilization of equipment, continuous monitoring by environmental health experts, implementation of a self-control and self-declaration process, raising awareness about the use of standard personal protective equipment, and providing the necessary platforms to utilize the capacity of private companies active in the field of decontamination of infectious waste produced in the dental clinics studied.

Highlights

* Environmental health compliance in surveyed Ardabil dental clinics reached 80.48%.
* Clinics achieved 90.38% compliance with the assessed environmental health criteria.
* Instrument and equipment sterilization showed 73.37% non-compliance across clinics.
* Infectious waste management was a major risk, with 72.12% of clinics non-compliant.

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1. Introduction

The management, maintenance, expansion, and promotion of health in human societies is one of the basic policies for achieving social justice in countries, and this goal has always been a national priority as an important human need and an individual and social value (Vishteh et al., 2010; World Health Organization [WHO], 2000). The principles of the Iranian Constitution also emphasize the provision of health and treatment as one of the main needs of the people (Hatami et al., 2015). Health systems include all organizations, institutions, and resources that provide health services, and their main goal is to promote the health of the community (WHO, 2000). According to the definition of the Environmental Health Regulations approved by the Council of Ministers in 2013, dental offices and clinics provide health and treatment services

to patients as public places (Alihoseini et al., 2016; Environmental and Occupational Health Centre, 2013). In providing dental services, there is a possibility of dentists and patients coming into contact with contaminated blood, mucus, or saliva, which can expose them to severe contamination.Failure to control risk factors associated with these services can lead to the spread of dangerous diseases such as hepatitis and AIDS (Rezaeipandari et al., 2015). Compliance with environmental hygiene principles, including the provision of safe drinking water, sanitary disposal of wastewater, waste management, and proper disinfection of instruments, is of great importance. Also, employees' attention to hygiene standards such as proper hand washing and the use of disposable items is effective in promoting the health of clients (Mokhtari & Babaei, 2006). According to Article 7 of the Environmental Health Regulations, the Ministry of Health is required to control public places such as dental clinics in terms of environmental hygiene principles (Zakerjafari & Mohammadi Salami, 2008). In recent years, numerous studies have been conducted to examine the environmental hygiene indicators of medical and dental offices, each of which has examined a part of these environmental hygiene standards and principles.

Among other things, in the section on waste management and proper waste disposal, it has been stated that very large amounts of infectious and hazardous waste are produced daily in medical clinics around the world. In the 1980s and 1990s, exposure to the AIDS and hepatitis B viruses led to an increase in the potential risk of exposure to medical waste, and caused the waste produced in these centers to be a threatening factor for the health of doctors, patients, staff, and even buildings adjacent to these centers (Bokhoree et al., 2014). Numerous other studies have also examined microbial and infectious contamination resulting from failure to comply with hygiene standards in offices and clinics, especially dental and medical offices, which is related to the environmental health index of ensuring the health and hygiene of clients and relevant staff (Al-Omari & Al-Dwairi, 2005; Alipour et al., 2009; Askarian et al., 2007; Sarir & Zandinezhad, 2006).

In a study by Shahbazi et al. in Nahavand city in 2005, it was evident that medical offices did not meet the expected environmental health indicators. In this study, there were problems in the correct principles of disinfection of tools and equipment, the construction of sanitary facilities, unsanitary disposal of sewage, and improper management of production waste (Shahbazi et al. 2009). In another study, Ali Hosseini et al. in Bushehr city in 2012 studied the indicators of building sanitation, the condition of sanitary facilities, waste management, ensuring the health of clients, safety, sewage disposal, and the condition of sanitary facilities, and in general, the centers providing health and medical services had favorable conditions in terms of hygiene in most of the indicators studied (Alihoseini et al. 2016). Now, considering the importance of the subject and the lack of coherent research in evaluating the environmental health indicators of dental clinics in Ardabil city, the results of this study can play an effective role in providing basic information to help managers and planners of health and treatment centers in Ardabil city, to establish preventive measures in adverse health cases, and to inform citizens. Therefore, efforts to improve and promote the level of environmental health of dental clinics, through making appropriate decisions, providing practical solutions, and allocating sufficient resources, especially in the environmental health sector, are expected to achieve the desired situation. This study was conducted with the aim of determining the environmental health status of dental clinics in Ardabil city-2026.

2. Materials and Methods

In this descriptive-cross-sectional study, 13 dental clinics in Ardabil city were surveyed in 1404 using a census method. The required information was collected by reviewing the documents available in the comprehensive urban health service centers and based on the health inspection checklist for dental offices with form code 920318 / 111. This checklist was developed based on the guidelines of the Health and Workplace Center and includes 62 questions in the fields of personal health of employees (13 questions), hygiene of tools and equipment (13 questions), health and safety of buildings (20 questions), waste management (8 questions), and specific regulations of dental clinics (8 questions). The collected data were analyzed using Excel software. The following relationship was used to obtain the health status of each part of the checklist and the percentage of compliance and non-compliance of the evaluated parameters (Alihoseini et al., 2016; Nourmoradi et al., 2017).

$$Percentage\ of\ compliance\ with\ health\ regulations = \frac{Regulation\ compliance\ frequency\ data}{Total\ frequency\ data} * 100 \qquad (1)$$

3. Results and Discussion

Tables 1 to 5 show the health status of the studied indicators, broken down into the personal health status of employees, hygiene of tools and equipment, health and safety of the building, waste management, and specific regulations of dental clinics in Ardabil city. In examining the personal health indicators of the studied dental clinics, the existence of health deficiencies in the areas of not using healthy, clean, and brightly colored gowns and not using personal protective equipment in 38.46 Percent and not properly disinfecting the suction unit and its components in 46.15 percent of the studied dental clinics was evident (Table 1).
The present study was conducted with the aim of determining the environmental health indicators of dental clinics in Ardabil in 1404. In examining personal health indicators, the results showed that the failure to use a clean, bright-colored gown and

the failure to use personal protective equipment were in 38.46% of cases that were contrary to the regulations. In a study conducted by Bastami et al. on the subject of investigating the status of infection control in dental offices and clinics in Qazvin province in 1393, the results indicated that the use of a clean gown and personal protective equipment during work by dentists was in 55.80 and 33.70% of cases that were contrary to the regulations, respectively (Bastami et al. 2014). In another study on the subject of knowledge, understanding, and attitude of dentists about COVID-19 and infection control among Jordanian dentists in 2019, the results showed that the use of gowns and personal protective equipment (dental glasses, masks, and gloves) was observed by dentists in 82.6% and 92.9% of cases, respectively (Khader et al., 2020). Also, in this section, the lack of proper disinfection of the suction unit and its components was evident in 46.15% of the dental clinics studied. In the excellent study by Hosseini et al. on the subject of environmental hygiene in private offices, laboratories and radiology in Bushehr city in 2014, the results showed that disinfection of the dental unit and its other accessories was carried out correctly in 97.80% of the units studied (Alihoseini et al. 2016). The Infection Control Guide for Dental Clinics, developed by the Centers for Disease Control (CDC), emphasizes the use of gloves, masks, and changing them for each patient, the wearing of gowns, eye protection by dentists, and measures such as vaccination, sterilization, and proper disinfection of dental instruments and surfaces to reduce the possibility of infection transmission (Rezaeipandari et al., 2015).

Table 1 The status of personal health indicators in dental clinics in Ardabil city in 2026

Parameters	Health Status (%)	
	Compliance with regulations	Non-Compliance with regulations
*Proper hand washing	100	0
Hand washing guide	92.31	7.69
Prohibition of work for employees with infectious diseases	100	0
Prohibition of sale and use of tobacco products	100	0
Installation of a label prohibiting the use of tobacco products	84.62	15.38
Use of clean, healthy, and brightly colored gowns	61.54	38.46
A designated place for storing equipment	92.31	7.69
Vaccination of medical staff against diseases	76.92	23.08
Infection control training certificate specific to medical staff	76.92	23.08
Use of personal protective equipment	61.54	38.46
Use of sterile disposable equipment	100	0
Decontamination of the suction unit and its components	53.85	46.15
A sterile examination set for each patient	100	0

* *Critical cases*

In the section on hygiene of instruments and equipment, the most important critical non-conformities include failure to observe the correct principles of disinfection and sterilization of instruments and equipment used in 46.15 and 53.85 percent of the units studied, respectively. Also, according to the surveys conducted in none of the dental clinics studied in Ardabil city, the self-control and self-declaration process of hygiene was carried out (Table. 2).

Table 2 Status of health indicators of tools and equipment in dental clinics in Ardabil city in 2026

Parameters	Health Status (%)	
	Compliance with regulations	Non-Compliance with regulations
*Use of disposable razors and needles	100	0
*Adherence to the correct principles of disinfection of instruments and equipment	53.85	46.15
*Sterilization of the instruments used	46.15	53.85
Use of a standard periapical radiography device	100	0
Arrangements for washing and disinfecting instruments and equipment	69.23	30.77
Standard disinfectant and disinfection solutions	76.92	23.08
Separate sink for hand washing for clients	100	0
Existence of a separate sink for washing instruments	76.92	23.08
Standard materials and medicines	100	0
Condition of shelves, drawers, etc.	76.92	23.08
Condition of chairs, furniture, etc.	76.92	23.08
Installation of a first aid kit with necessary items	76.92	23.08
Self-control and self-declaration process of hygiene by the owner	·	100

* *Critical cases*

In the section on hygiene of instruments and equipment, and among the most important critical non-conformities in this section, we can mention the failure to observe the correct principles of disinfection of instruments and equipment in 46.15% and the failure to sterilize the instruments used in 53.85% of the units studied. In a study conducted in 2008 to compare environmental health indicators of private clinics in Chaharmahal and Bakhtiari province, Fadaei et al. stated that disinfection of tools and equipment was not performed correctly in 66.40% of the units studied (20). It seems that expanding the necessary training by environmental health inspectors, focusing on the correct principles of hand disinfection and disinfection and sterilization of dental instruments and tools can be effective in improving the knowledge and performance of dentists. Also, in the present study, there was apparently no requirement from the

comprehensive health service centers of Ardabil city to implement the self-control and self-declaration process. Using the capacity of self-control and self-declaration, health companies can be useful in improving the performance of centers providing health and medical services such as dental clinics. In a study in Dareh Shahr in 1402, Mirzaei et al. stated that implementing the self-control and self-declaration process of hygiene is necessary and essential and can only be achieved with the follow-up of health authorities (Mirzaei et al. 2023).

In the building health and safety section, the water supply system was in compliance with regulations in 100 Percent of cases. Also, among the health deficiencies in this section, we can mention the discrepancy in the condition of the toilets and the condition of proper ventilation with a frequency of 46.15 Percent, as well as the discrepancy in the condition of the fire extinguisher with a valid charge with a frequency of 53.85 percent (Table. 3).

Table 3 Status of health and safety indicators of dental clinic buildings in Ardabil city in 2026

Parameters	Health Status (%)	
	Compliance with regulations	Non-Compliance with regulations
* Water supply system	100	0
Sewage collection and disposal system	84.62	15.38
Necessary preparations for controlling insects and rodents	84.62	15.38
Standard heating and cooling system	100	0
Standardization of the dental unit space	84.62	15.38
Rest room condition	84.62	15.38
Toilet condition	53.85	46.15
Healthy flush tank in the toilet	76.92	23.08
Wall covering of the washing area	69.23	30.77
Floor covering of the building	92.31	7.69
Wall covering of the building	84.62	15.38
Roof covering of the building	84.62	15.38
Doors and windows	92.31	7.69
Temperature and humidity	84.62	15.38
Optimal lighting	92.31	7.69
Optimal ventilation	53.85	46.15
Fire extinguisher with valid charge	46.15	53.85
Sanitary condition of the water supply building	92.31	7.69
Compliance with hygiene principles in the water supply	84.62	15.38
Safety compliance status	92.31	7.69

* *Critical cases*

General principles and criteria of environmental hygiene, such as the provision and supply of safe drinking water, proper disposal of sewage and waste, provision of humidity, proper light and ventilation, standard physical space, toilets and sanitary facilities, and compliance with hygiene standards by medical staff are effective and important in ensuring public health and controlling infections (Pasquarella et al., 2010). In the present study and in the building health and safety section, discrepancies in the condition of sanitary facilities and proper ventilation were evident in 46.15% and the condition of fire extinguishers with valid charges in 53.85%. In a study by Shahbazi et al. in Nahavand in 2005, the poor condition of sanitary facilities was reported to be in violation of regulations in 18.85% of the units studied (Shahbazi et al. 2009). Also, in a comprehensive study by Hosseini et al. in Bushehr in 2012, the condition of sanitary facilities and toilets violated regulations in 18% of the units studied, the condition of air conditioning in 20.09, and the condition of fire extinguishers in 40.48% of the units studied (Alihoseini et al. 2016). Failure to address the condition of sanitary facilities and toilets, failure of flush tanks, improper cleaning of toilets, unpleasant odors due to failure of hoods, and failure to properly place and charge fire extinguishers were among the reasons for the low level of compliance with regulations in the health and safety section of the buildings in the units studied. Continuous monitoring by environmental health experts, decisive action, and identification of offending centers with health deficiencies are recommended to address these non-conformities. In the section on waste management indicators, in the context of the critical situation related to the proper disposal of sharps and cutting waste in 30.77% of dental clinics, there was a clear violation of the regulations. The collection of surgical blades and used needles in a safe box in accordance with health regulations should be considered (Hokmabadi et al., 2013). In the excellent study by Hosseini et al. in Bushehr in 2012, the failure to use a safe box and the unprincipled disposal of sharps and cutting waste in 11.90% and the unprincipled disposal of infectious waste in 7.60% of the units studied were in violation of the regulations (Alihoseini et al. 2016). In the present study, records of waste decontamination were evident in only 46.15% of the units studied.It seems that using the capacity and services of companies trusted by the Ministry of Health in the collection and disposal of infectious waste generated by medical centers such as the dental clinics studied can be effective in improving the waste management process. Poor management of medical waste exposes medical and healthcare staff, waste collection staff, and communities around these centers to infections, effects, and injuries from the toxicity of medical waste, which has led to serious health problems in most developing countries (Alihoseini et al. 2016). It is essential to observe the principles of temporary waste storage, separate hazardous infectious waste from regular waste, and transport waste according to a continuous schedule in health care centers. Other inconsistencies in the waste management sector include the poor condition of temporary storage facilities and the frequency of daily waste collection in 38.46% of the units studied. In a study by Pak Fetrat et al. on the subject of investigating the status of waste management in Iranian dental centers in 2018, the results showed that by separating the recyclable part of dental waste from other parts, the volume

of waste and consequently the costs related to waste collection and disposal are reduced (Pakfetrat et al. (2018).

In the section related to waste management indicators, the critical parameter of proper disposal of sharps and cutting waste was found to violate the regulations in 30.77 percent of dental clinics. Also, among the non-conformities in this section, we can mention the inadequate condition of the temporary storage location and the frequency of daily waste collection in 38.46 percent and the lack of contract records with waste decontamination companies in 53.85 percent of the units studied (Table 4).

Table 4 Status of waste management indicators in dental clinics in Ardabil city in 2026

Parameters	Health Status (%)	
	Compliance with regulations	Non-Compliance with regulations
* Proper disposal of sharp and cutting waste	69.23	30.77
Use of colored bags to separate waste	76.92	23.08
Contract records with waste disposal companies	46.15	53.85
Lid, pedal-operated and washable trash cans	84.62	15.38
Temporary waste storage area	61.54	38.46
Daily waste collection frequency	61.54	38.46
Separation of infectious and non-infectious waste	76.92	23.08
Proper and principled disposal of amalgam and other consumable materials	100	0

* Critical cases

In the specific regulations section of dental clinics, the status of users' training records on working with radiation and the lack of use of personal protective equipment against radiation were evident in 38.46 and 23.08 percent of cases, respectively, in violation of the regulations (Table. 5).

Waste produced in dental clinics in Ardabil city was stored in temporary waste storage facilities in an unprincipled manner, sometimes for 2 to 3 days, which shows the disregard for the importance of principled waste management by the units studied. In the excellent study by Hosseini et al. in Bushehr in 2012, the frequency of waste collection was appropriate in only 89% of the units studied (Alihoseini et al. 2016). Also, in the study by Shahbazi et al. in Nahavand in 2005, 12.50% of the units studied lacked a suitable place for temporary storage of waste (Shahbazi et al. 2009). Expanding educational programs and distributing brochures on waste management, continuous monitoring of the process of waste separation and disposal by environmental health experts working in health centers, and also communicating the comprehensive waste management policy by the provincial health center to all clinics and dental offices operating in Ardabil city are recommended to improve the performance of the units studied.

Table 5 Status of indicators related to specific criteria of dental clinics in Ardabil city in 2026

Parameters	Health Status (%)	
	Compliance with regulations	Non-Compliance with regulations
Status of construction of a dental radiography center on the lower floors	100	0
Installation of a lead screen or lead chamber in the dental radiography center	100	0
Radiography device with quality control approval	100	0
Records of training course on working with radiation for users	61.54	38.46
Use of personal protective equipment against radiation	76.92	23.08
Establishment of the center in a non-residential building	100	0
Sketch and plan of the unit building	92.31	7.69
Installation of a compressor in a space away from rooms and waiting rooms	92.31	7.69

* *Critical cases*

In Fig. 1, the overall and separate averages of the total indicators of personal hygiene, hygiene of tools and equipment, building hygiene and safety, waste management, and specific criteria of dental clinics in Ardabil city in 2026 are expressed as percentages.

As can be seen in Fig. 1, the status of the health indicator studied in the specific criteria section of dental clinics was in accordance with the regulations with a frequency of 90.38 Percent, and the hygiene of tools and equipment and waste management sections conflicted with the regulations with a frequency of 73.37 and 72.12 Percent, respectively. Overall, the level of compliance with the regulations in the study of environmental health indicators of dental clinics in Ardabil in 2026 was assessed with a frequency of 80.48 Percent (Fig. 1).

In the section on specific criteria for dental clinics, the lack of records of users' presence in the radiation work period was evident in 38.46% of the units studied. Radiation training courses for users who work with ionizing radiation are held in the form of introductory and advanced courses under the supervision of the Radiation Protection Office of the Atomic Energy Organization of Iran; the validity of these courses is one year, and users must participate in refresher courses. Also, in this section, the use of personal protective equipment by dentists while working with radiation was contrary to regulations in 23.08% of the units studied. Hokamabadi et al. in North Khorasan Province in 2012 stated that there were no gonadal and thyroid shields in 2 hospitals and one hospital lacked radiation warning signs (Hokmabadi et al. 2013). In a study by Hosseini et al. in Bushehr in 2012, the results showed that the use of personal protective equipment (such as lead aprons, gonadal and thyroid protectors) by 76.6% of dentists and 100% of personnel working in the radiology department

of the studied units was in accordance with the regulations (Alihoseini et al. 2016). Given the importance of observing safety and security issues in this sector, environmental health experts should, in accordance with the provisions of the regulations for inspecting dental offices, provide necessary and continuous training on the principles of protection and safety of working with radiation for employees working in dental clinics in Ardabil. According to the results of the present study, the status of the health index under study in the specific regulations of dental clinics was in accordance with the regulations with a frequency of 90.38%, and the health of tools and equipment and waste management violated the regulations with a frequency of 73.37% and 72.12%, respectively. In the excellent study by Hosseini et al. on the subject of environmental health of private offices, laboratories, and radiology in Bushehr city in 2012, the status of building health indicators, health services, waste management, ensuring the health of clients, safety, sewage, and health facilities was in a desirable state with an average of 85.3, 82, 87.2, 75.8, 61.1, 95.3, and 99.1%, respectively (Alihoseini et al. 2016).

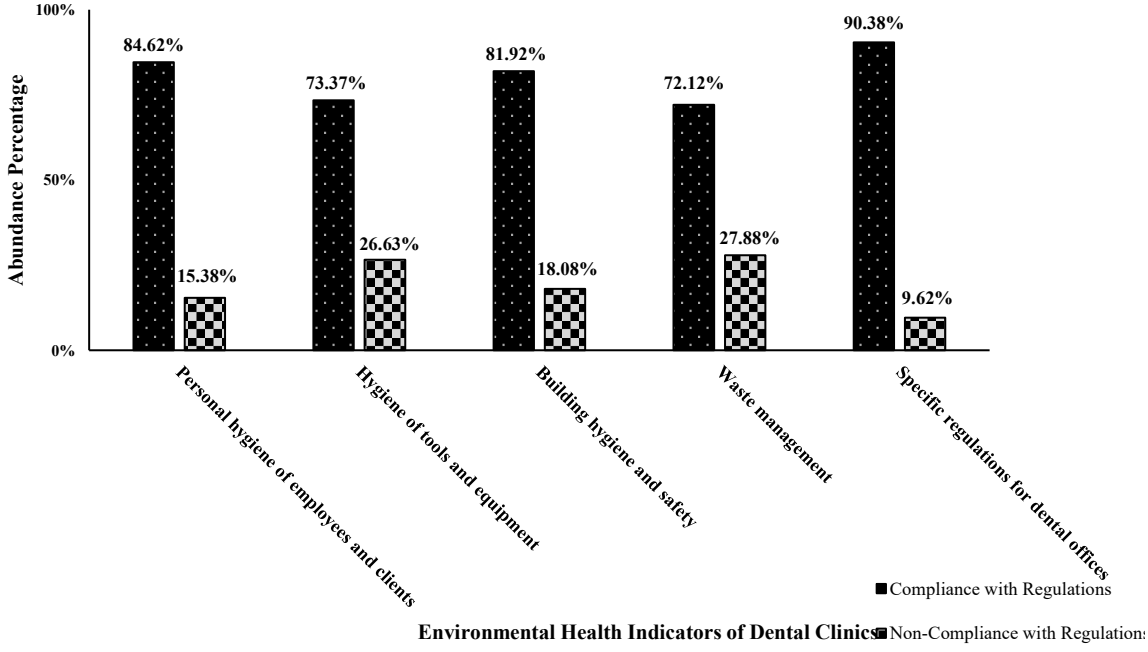


Fig. 1 General status of environmental health indicators of dental clinics in Ardabil city in 2026

4. Conclusion

In the present study, and in general, the level of compliance with the regulations in the environmental health indicators of dental clinics in Ardabil city in 1404 was assessed at 80.48%. The results indicated that dental centers in Ardabil city are involved in environmental health challenges such as the lack of use of personal protective equipment by medical staff, lack of attention to disinfection and sterilization of used tools and equipment, poor condition of sanitary facilities, ventilation and fire extinguishing equipment, improper management of waste produced in the centers, and lack of training records on working with radiation for users. Therefore, to improve the environmental health indicators of the dental clinics under study and eliminate health non-conformities, especially in the areas of instrument and equipment hygiene and waste management, it is recommended to hold training courses on familiarizing oneself with the correct principles of disinfection and sterilization of equipment, continuous monitoring by environmental health experts, implementation of a self-control and self-declaration process, raising awareness about the use of standard personal protective equipment, and providing the necessary platforms to utilize the capacity of private companies active in the field of management and decontamination of infectious waste produced in the dental clinics under study

Statements and Declarations

Ethical considerations

Ethical issues (Including plagiarism, informed consent, misconduct, data fabrication and/or falsification, double publication and/or submission, redundancy, etc.) have been completely observed by the author.

Data availability

Data will be made available on request.

Conflicts of 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.

Author contribution

MH. Abbasi: Writing the main Draft; F. Nami, M. Nami, and A. Abbasi: Conceptualization; Writing – Review & Editing, Conceptualization, Methodology, Review, and Final Proof.

AI Use Declaration

During the preparation of this manuscript, the authors used AI tools for designing the Graphical Abstract. All content has been carefully reviewed and revised by the authors, who take full responsibility for the final version of the manuscript.

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Stabilization and solidification of electroplating sludge in cement-based mortars: effects of rice husk ash on compressive strength and heavy-metal leachability

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ABSTRACT

Electroplating generates large volumes of heavy-metal-bearing sludge whose secondary environmental burden can exceed that of the wastewater itself, yet a unified, reliably safe disposal route is still lacking. This study examined the stabilization/solidification (S/S) of electroplating sludge (ES) in cement-based mortars and assessed rice husk ash (RHA) as a complementary matrix for immobilizing the contained metals. Twelve mortar mixes were prepared: a plain Portland cement mortar (mix F) and eleven mixes organized into three groups, comprising ES introduced into the binder phase at 0-75 % of the cement mass together with RHA at 5 % (Group A) or 10 % (Group B) of the cement mass, and ES used as a partial fine-aggregate replacement at 15-35 % by volume (Group C). Fresh-state behavior (slump, setting time, specific gravity), unconfined compressive strength at 7, 28 and 90 days, and the leaching of Cd, Pb, Ni, Cr, As and Cu were measured. Replacing fine aggregate with ES raised the slump from 75 to 150 mm, whereas increasing the ES content of the binder progressively retarded setting, an effect amplified by a higher RHA content. Compressive strength fell sharply once ES entered the binder, from 16.3-29.0 MPa for the sludge-free mixes to 0.30-6.1 MPa at ES/CEM ratios of 25-75 %, and the 28- and 90-day values were almost identical, so that no late-age recovery should be expected. Used as a fine aggregate, ES was far less damaging, with the mortars retaining 22-53 % of the control strength. Leaching tests conducted under an identical protocol on the raw sludge and on the mortars showed that Cd, Pb and As were immobilized below their US EPA regulatory levels, whereas Cr remained above the 5 mg/L limit in every sludge-bearing mix. The results delimit the loadings over which ES can be incorporated into cementitious materials and identify chromium leaching as the principal constraint on safe reuse.

Highlights

- Electroplating sludge was stabilized in cement mortars with rice husk ash.
- Raw sludge exceeded regulatory limits for every heavy metal measured.
- Compressive strength fell sharply as sludge replaced cement in the binder.
- Using sludge as fine aggregate preserved higher strength and workability.
- Cd, Pb, and As were immobilized, but Cr leaching exceeded the limit.

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1. Introduction

The rapid expansion of industrial activity over recent decades has markedly increased human exposure to heavy metals, and even low environmental concentrations are now recognized as a serious threat to public health (Balali-Mood et al. 2021). Among the principal anthropogenic sources, the electroplating industry is especially problematic because hazardous metal

ions such as copper (Cu), nickel (Ni), chromium (Cr) and lead (Pb) are routinely discharged in its wastewater and are readily taken up by living organisms (Rajoria et al. 2021). Electroplating and surface-treatment operations generate large volumes of chemically complex wastewater (Wang H et al. 2024), and when this is discharged without adequate treatment it can give rise to toxic metal–organic complexes that pose long-term ecological risks. A range of treatment technologies has accordingly been developed and refined, including chemical precipitation, adsorption, coagulation, flocculation, ion exchange, electrocoagulation, membrane filtration and biorecovery (Kamar et al. 2024).

Despite this diversity, conventional treatment routes are frequently constrained by high capital and operating costs and by the difficulty of handling and disposing of the resulting sludge (Rajoria et al. 2021), so that the complete removal of toxic metals from industrial effluents remains a persistent challenge (Adenuga et al. 2019). Advanced oxidation processes (AOPs), originally developed in the 1980s for drinking-water applications, have more recently attracted attention for the destruction of persistent organic pollutants in wastewater (Deng and Zhao 2015; Momotko et al. 2025). Even so, the variable composition and high contaminant load of electroplating wastewater often lead to incomplete metal removal during treatment (Rajoria et al. 2025), and an estimated 2–20 % of the chemicals and metals used in the process are leached into the wastewater stream (Rajoria et al. 2021). Reducing the environmental footprint of this sector therefore requires complementary strategies that address not only the liquid effluent but also the solid residues generated during its treatment.

A particularly important issue in this respect is the large quantity of metal-bearing sludge produced during the treatment of electroplating wastewater, which is classified as highly toxic and hazardous (Yu et al. 2022). The secondary environmental burden associated with this sludge is, in many respects, greater than that of the wastewater itself, and at present no unified and reliably safe disposal route exists (Wang H et al. 2024). Current management practices for such hazardous residues include thermal treatment, bioremediation, electrochemical treatment, chemical precipitation and stabilization/solidification (S/S) (Wang J et al. 2025). Among these, S/S has become a well-established and versatile technology for rendering contaminated wastes harmless and enabling their reuse (Chen et al. 2020; Petrillo et al. 2025). During the S/S process, cement hydration generates a crystalline matrix that physically entraps the contaminants within the sludge (Chen et al. 2009), and the incorporation of industrial byproducts as secondary raw materials in such systems is consistent with the principles of the circular economy (Monosi et al. 2016).

Portland cement (PC) remains the most widely used binder for S/S applications (Ouhadi et al. 2021), yet its manufacture is energy-intensive and releases large amounts of carbon dioxide (CO_2), dust, sulfides, and nitrogen compounds, contributing to several pressing global environmental problems (Miao et al. 2025). The partial replacement of PC by industrial or agricultural byproducts has therefore gained increasing attention as an environmentally safer, mechanically stable, durable and economically attractive alternative (Chiu et al. 2019). Rice husk ash (RHA) is a particularly relevant candidate in Iran, where husk accounts for approximately 23 % of the mass of the paddy grain and more than 0.5 million tonnes are generated each year, principally in the northern rice-growing provinces; at the ash yields typical of combustion, of the order of 18–20 % by mass, this corresponds to an annual RHA potential on the order of 10^5 tonnes (Yaghoubi et al. 2019). Most of this residue is presently directed to low-value routes such as combustion for rice drying and domestic heating, use as animal feed, mulching material and soil amendment, or is disposed of by open burning, while the ash that remains has historically had no application and has commonly been discharged to water streams (Yaghoubi et al. 2019). A multicriteria assessment carried out for Guilan Province nevertheless ranked industrial valorization as the most appropriate use of the husk and disposal by burning as the least appropriate (Yaghoubi et al. 2019), so that conversion of the ash into a construction material is consistent with national waste-management priorities.

This role is underpinned by the chemistry of the material, which consists predominantly of silica (Della et al. 2002). Where combustion is controlled, that silica is retained in a largely amorphous, porous and high-surface-area form which reacts with the calcium hydroxide released during cement hydration to precipitate additional calcium silicate hydrate (C-S-H) and to refine the pore structure of the hardened matrix. Its reactivity is not an intrinsic property, however, but varies with the final combustion temperature and its duration, the heating and cooling regimes, the type of kiln and the fineness of grinding, as well as with the rice variety, the moisture content of the husk and the geographical origin of the material (Chandrasekhar et al. 2003). RHA has accordingly been examined in numerous studies as a mineral additive capable of enhancing the performance of cement-based systems (Asavapisit and Macphee 2007), and recent Iranian work confirms its pozzolanic contribution to the strength and durability of high-performance and self-compacting concretes produced with locally available materials (Ziyari and Mostakhdemin Hosseini 2025; Fathollahi et al. 2025; Momeni and Ghorbani Kamsari 2025). Its usefulness extends to S/S matrices specifically, having been assessed for the solidification/stabilization of heavy-metal-contaminated slurry (Chiu et al. 2019), while Chen et al. (2022) showed that modified RHA can act as an eco-friendly binder for the effective stabilization of waste clay in road-construction applications, further supporting its potential as a low-carbon alternative to PC. Despite this evidence, and despite domestic interest extending back several decades (Jafarpour et al. 1997), the industrial valorization of rice husk ash in Iran has so far remained limited.

Building on these developments, a growing body of research has examined the stabilization and beneficial reuse of metal- and sewage-derived sludges in construction materials, using a wide range of binder systems and producing outputs as varied as cement clinkers, structural and decorative mortars, fired clay bricks and ceramics, concrete paving and paver blocks, subgrade backfill and dedicated matrices for the encapsulation of heavy metals.

Given the heterogeneous and chemically reactive nature of plating sludge, important questions remain regarding the technical specifications of the resulting cementitious product, in particular its strength development and long-term leaching

behavior, and a systematic evaluation of these aspects represents a novel contribution of this study. Within this framework, the present study investigates the feasibility of incorporating electroplating sludge into cementitious mortars and evaluates rice husk ash as a complementary matrix for immobilizing the heavy metals it contains. Specifically, the work assesses the influence of the sludge, introduced either into the binder phase alongside RHA or as a partial replacement for the fine aggregate, on the fresh-state behavior, the compressive strength, and the leaching behavior of the resulting stabilized products, in line with the broader move towards sustainable construction practices based on the valorization of industrial and agricultural byproducts.

2. Materials and Methods

2.1 Raw materials collection and preparation

Ordinary Portland cement (OPC) was used as the principal binder, rice husk ash (RHA) as a supplementary cementitious material, and electroplating sludge (ES) either as a component of the binder phase or as a partial fine-aggregate replacement. The origin of each constituent and the treatment applied to it before mixing are described below.

2.1.1 Electroplating sludge

The electroplating sludge was collected from the sludge-dewatering unit of the Pirouz Shomal industrial wastewater treatment plant, which treats effluents generated by electroplating operations. The sludge is the metal-bearing precipitate separated during the treatment of chromium and nickel plating baths and of the associated rinse waters. Detailed information on the configuration of the treatment train that produced the sludge was not released by the plant operator and is therefore not reported here. The material was collected as a wet filter cake, homogenized and kept in sealed containers so that its moisture content remained unchanged until characterization.

The as-received cake contained approximately 49 % of free water (Section 2.2.2) and was therefore oven-dried at 64 °C to constant mass before mix proportioning; the reason for selecting this temperature is given in Section 2.2.2. After drying, the cake was disaggregated by hand in a laboratory mortar with a pestle until the agglomerates were completely broken down. No mechanical milling, air classification or sieving was applied, so the sludge was introduced into the mortars with its natural grain-size distribution, which is reported in Section 2.2.3.

2.1.2 Rice husk ash

The rice husk was obtained from a local supplier in the city of Ilam and was calcined in an electric laboratory furnace at 550 to 600 °C for 6 h. This temperature window was selected because it is high enough to burn off the organic fraction of the husk but remains below the range in which the silica crystallizes into cristobalite and tridymite, so that the ash retains the amorphous silica responsible for its pozzolanic activity. After calcination, the ash was cooled to ambient temperature, hand-ground in a mortar to break down the residual husk skeleton, and stored in sealed containers until use. The mineralogy and the fineness of the ash were not determined; this limitation is discussed in Section 3.6.

2.1.3 Cement and fine aggregate

The binder was an ordinary Portland cement produced by the Hegmatan Cement Company (Hamedan, Iran), and the fine aggregate was standard sand conforming to DIN EN 196-1 (2016). Both materials were used as received, without any further treatment.

2.1.4 Superplasticizer

A commercial high-range water-reducing admixture was used to restore a comparable consistency in the sludge-bearing mixes of Groups A and B, in which the water/binder ratio falls sharply as the sludge dosage increases (Section 2.3). It was added at 0.20-0.65 % of the cement mass; the individual dosages are given with the mix proportions in Section 2.3. No admixture was used in mix F or in the mixes of Group C.

2.2 Materials characterization

2.2.1 Chemical composition

The chemical compositions of the three constituents were determined by X-ray fluorescence (XRF) spectrometry; the corresponding oxide contents are reported in Table 2 together with the relevant limits of ASTM International C618-05 (2005).

Table 1 Chemical composition (wt %) of OPC, RHA and ES, with the relevant limits of ASTM International C618-05 (2005) for natural pozzolans

Chemical compound	OPC	RHA	ES	ASTM C618 limit
SiO_2	21.90	92.90	9.35	-
Al_2O_3	6.27	3.25	16.12	-
Fe_2O_3	3.48	0.77	1.63	-
$SiO_2 + Al_2O_3 + Fe_2O_3$	31.65	96.92	27.10	min. 70
CaO	59.26	0.94	9.92	-
MgO	1.03	1.01	1.14	-
Na_2O	0.02	0.29	0.86	-
K_2O	0.80	2.54	0.15	-
SO_3	2.35	2.50	2.35	max. 4
LOI	4.14	3.60	33.94	max. 10

As shown in Table 1, RHA satisfies the ASTM International C618 requirements for natural pozzolans, with a combined $SiO_2 + Al_2O_3 + Fe_2O_3$ content of 96.92 % (≥ 70 %), an SO_3 content of 2.50 % (≤ 4 %), and a loss on ignition of 3.60 % (≤ 10 %). In contrast, ES does not meet the pozzolanic threshold, with a summed oxide content of only 27.10 % and an unusually high loss on ignition of 33.94 %. ES therefore cannot be regarded as a conventional pozzolan and was investigated primarily as a candidate matrix for the stabilization/solidification of its heavy-metal content rather than as an active mineral additive. It should be noted that the oxides listed in Table 1 account for the great majority of the OPC but for only about three-quarters of the ES; the balance is made up of the heavy-metal oxides, principally those of chromium, nickel, copper and zinc, which are the reason for classifying the material as hazardous and whose concentrations are reported in Section 3.1 rather than as oxide equivalents.

2.2.2 Moisture content of the sludge

Because the as-received electroplating sludge contained a substantial fraction of free water, its moisture content was determined gravimetrically prior to mix proportioning. Three replicate subsamples of the homogenized sludge were placed in tared crucibles, dried to constant mass at 64 °C and reweighed. A drying temperature of 64 °C was chosen in preference to the more common 105 °C so that free water would be fully removed while minimizing the loss of volatile and thermally labile metal species, in particular Hg compounds and the oxyhydroxide-bound water associated with the precipitated heavy-metal phases, thereby preserving the composition reported in Table 1. The average data are summarized in Table 2.

Table 2 Gravimetric determination of the moisture content of the as-received electroplating sludge (mean of three replicate determinations)

Parameter	Value
Container mass (g)	12.897
Container + wet sample, before drying (g)	17.118
Container + dry sample, after drying (g)	15.048
Wet sample mass (g)	4.221
Dry sample mass (g)	2.151
Mass of water lost (g)	2.070
Moisture content, wet basis (%)	≈ 49

The moisture content, calculated on a wet-mass basis (2.070 / 4.221 × 100), was approximately 49 %, confirming that the as-received material was a wet paste rather than a granular solid. To remove any associated uncertainty in the mixing-water content and in cement hydration, the sludge was fully oven-dried at 64 °C before incorporation in the mixes.

2.2.3 Particle-size distribution of the sludge

Because the dried sludge is cohesive and tends to form agglomerates, its grain-size distribution was determined by wet sieving, the procedure normally adopted for fine-grained soils, rather than by dry sieving. A representative subsample was first dried at 64 °C for 24 h and broken down with a rubber hammer to the smallest achievable size, so that the distribution reported here corresponds to the disaggregated material rather than to the as-received cake. The material was then placed on a sieve stack comprising the No. 4, 8, 16, 30, 40, 50, 100 and 200 sieves (4.75, 2.36, 1.18, 0.60, 0.425, 0.30, 0.15 and 0.075 mm) and washed with a jet of water, the washing being continued until the water leaving the sieves ran completely clear. The fractions retained on each sieve were subsequently oven dried at 110 °C for 24 h and weighed; the higher temperature is acceptable at this stage because only the mass of each fraction is required, whereas the 64 °C used elsewhere was chosen to preserve the chemical composition. The cumulative passing percentages were calculated and plotted as the grain-size distribution curve shown in Fig. 1, together with the nominal grading of the CEN standard sand specified in DIN EN 196-1 (2016). Wet sieving was preferred because the washing separates particles that would otherwise remain bound to each other, so that every particle passes the sieve at its true size and the resulting curve represents the actual fineness of the material more reliably than one obtained by dry sieving. The analysis resolves the distribution down to the No. 200 sieve (0.075 mm) only; the fraction finer than that size was not subdivided, and the curve of Fig. 1 is therefore truncated at 0.075 mm.

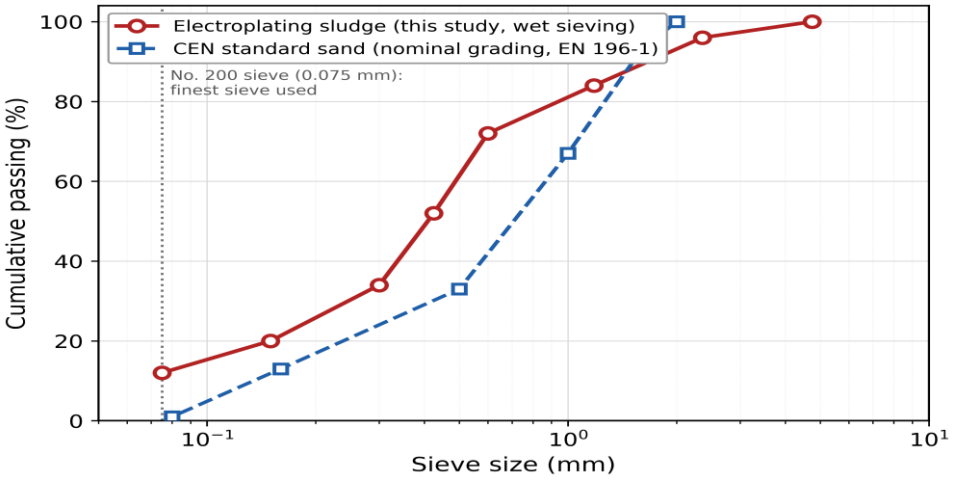


Fig. 1 Grain-size distribution of the electroplating sludge determined by wet sieving, compared with the nominal grading of the CEN standard sand specified in DIN EN 196-1 (2016). The sludge curve is truncated at the No. 200 sieve (0.075 mm), the finest sieve used

The curve confirms that the sludge is a predominantly fine-grained material. The whole of the sample passes the 4.75 mm sieve (No. 4), 96 % passes the 2.36 mm sieve (No. 8), 52 % passes the 0.425 mm sieve (No. 40), 34 % passes the 0.30 mm

sieve (No. 50) and 12 % passes the 0.075 mm sieve (No. 200), so that more than half of the mass is finer than about 0.4 mm. Comparison with the standard sand in Fig. 1 shows that the sludge is the finer of the two materials over the whole range covered: at 0.425 mm the sand passes about 28 % against 52 % for the sludge, and at the finest sieve the sand passes about 1 % against 12 %. The sludge is therefore fine enough to act as a filler in the mixes of Groups A and B and, in Group C, replaces the standard sand with a distinctly finer material.

2.3 Mix proportions and justification

Twelve mortar mixes were prepared: a plain Portland cement mortar, designated F, and eleven mixes organized into three groups, A, B and C. Mix F contains neither ES nor RHA. Because its binder content is identical to that of Group C but differs from that of Groups A and B, mix F serves as the direct reference for Group C and as a general benchmark elsewhere; within Groups A and B, the internal references are mixes A1 and B1, which contain 5 % and 10 % of RHA, respectively, but no sludge, and therefore isolate the contribution of the pozzolanic addition from that of the sludge.

In Groups A and B, the sludge was introduced into the binder phase together with RHA. The RHA content was fixed at 5 wt % of the cement in Group A and 10 wt % in Group B, and the sludge was dosed at 0, 25, 50, and 75 % of the cement mass (ES/CEM). It should be emphasized that the cement content was held constant within each group, at 490 g in Group A and 465 g in Group B, so that the sludge was an addition to the binder phase and not a substitution for the cement; the term "replacement" is therefore avoided for these mixes throughout the paper. One consequence of this design is that, although the water/cement ratio was held at a nominal 0.485, the water/binder ratio computed on the total powder content, cement plus RHA plus sludge, fell from 0.463 to 0.270 in Group A and from 0.440 to 0.261 in Group B as the sludge dosage increased. Both ratios are reported in Table 4, and it is this fall in the water/binder ratio, as much as the water demand of the sludge itself, that made a superplasticizer necessary at ES/CEM ≥ 25 %.

Table 4 Mix proportions of the twelve mortar designs prepared in this study

Group	Mix	ES level (%)[a]	RHA/CEM (%)	Cement (g)	RHA (g)	ES (g)	Sand (g)[c]	Water (g)	w/c[b]	w/b[b]	Super-plasticizer (g)
Control	F	0	0	413	0	0	1350	200	0.484	0.484	0
A	A1	0	5	490	24.5	0	1350	238	0.486	0.463	0
A	A2	25	5	490	24.5	123	1350	238	0.486	0.373	1
A	A3	50	5	490	24.5	245	1350	238	0.486	0.313	3
A	A4	75	5	490	24.5	368	1350	238	0.486	0.270	3
B	B1	0	10	465	46.5	0	1350	225	0.484	0.440	0
B	B2	25	10	465	46.5	116	1350	225	0.484	0.359	1
B	B3	50	10	465	46.5	233	1350	225	0.484	0.302	3
B	B4	75	10	465	46.5	349	1350	225	0.484	0.261	3
C	C1	15	0	413	0	143	1207	200	0.484	0.484	0
C	C2	25	0	413	0	238	1112	200	0.484	0.484	0
C	C3	35	0	413	0	334	1016	200	0.484	0.484	0

[a] For Groups A and B the sludge level is the mass of ES expressed as a percentage of the cement mass (ES/CEM); the cement content is constant within each group, so the sludge is an addition to the binder phase and not a substitution for the cement. For Group C it is the nominal replacement of the fine aggregate on a volume basis, equivalent to 10.6, 17.6 and 24.7 % of the sand by mass; the sand mass was reduced by the mass of sludge introduced. [b] w/c is the ratio of the mixing water to the cement. w/b is the ratio of the mixing water to the total powder content: cement + RHA + ES for Groups A and B, where the sludge is part of the binder phase, and cement + RHA for mix F and Group C, where the sludge acts as an aggregate; [c] The standard sand was batched to ± 5 g.

In Group C, the sludge was used exclusively as a partial replacement for the fine aggregate, at nominal levels of 15, 25 and 35 % defined on a volume basis. Because the sludge is appreciably less dense than the standard sand, these levels correspond to sludge masses of 143, 238 and 334 g, that is 10.6, 17.6 and 24.7 % of the sand by mass. The mass of standard sand was reduced by the mass of sludge introduced, so that the total mass of the granular skeleton was held at 1350 g in every mix, while the cement content, the mixing water and therefore the paste volume were left unchanged and no superplasticizer was added. A nominal water/cement ratio of 0.485 was adopted throughout, and the mortars were prepared in accordance with ASTM International C305-11 (2011). The complete mix proportions are given in Table 4.

The dosage levels were selected to cover as wide an interval as possible rather than to reproduce a single expected optimum. For the sludge introduced into the binder, the ES/CEM ratio was increased in equal steps of 25 % up to 75 %, so that the series spans the whole range from a conservative dosage to a deliberately excessive one. This is consistent with the objective of a stabilization/solidification study, in which the practical question is the maximum waste loading that the binder can accept while still meeting the mechanical and leaching requirements; a narrower range would not have allowed the threshold beyond which performance becomes unacceptable to be identified. For the sludge used as a fine aggregate, the levels of 15, 25 and 35 % were adopted because the material is finer than the standard sand and larger substitutions would appreciably modify the grading of the granular skeleton; the intermediate level of 25 % coincides with the ES/CEM ratio of mixes A2 and B2 and thus permits a direct comparison of the two incorporation strategies at nominally the same sludge content.

The RHA contents of 5 and 10 % lie within the interval usually reported as effective for rice husk ash in cementitious systems and were deliberately kept moderate. Higher contents markedly increase the water demand of the mixture, which at the fixed water/cement ratio of 0.485 would have required superplasticizer dosages large enough to mask the effect of the sludge. Two levels differing by a factor of two are sufficient to show whether the pozzolanic contribution of the RHA compensates for the dilution and the retardation caused by the sludge.

2.4 Testing program

2.4.1 Fresh-state properties

The workability of the fresh mortar was evaluated by the slump test in accordance with ASTM International C143-03 (2003), and the fresh density in accordance with ASTM International C138-09 (2009), the latter being reported below as a specific gravity in g cm^{-3}. The slump cone was preferred to the mortar flow table because the mixes of Group C were highly fluid and because the resulting values are directly comparable with the concrete-based stabilization/solidification literature. The initial and final setting times of the binder phase were measured with the Vicat apparatus in accordance with ASTM International C191-08 (2008), for both the increasing ES and the increasing RHA series, and were compared with the corresponding sludge-free mix of the same group.

2.4.2 Hardened-state properties

The unconfined compressive strength of the mortars was determined on 50 mm cubic specimens in accordance with ASTM International C109/C109M-16a (2016). Tests were carried out at curing ages of 7, 28, and 90 days, with three to four specimens per mix and curing age; the arithmetic mean of the replicates is reported.

2.4.3 Leaching characterization (TCLP)

The leaching behavior was evaluated using the Toxicity Characteristic Leaching Procedure (TCLP) described in US EPA Method 1311 (US Environmental Protection Agency 1992), which simulates the leaching expected if a solidified waste were disposed of in a sanitary landfill and is widely used to determine whether a material can be classified as hazardous. Specimens cured for 28 days were crushed, screened, and contacted with the prescribed acetic acid extraction fluid under continuous end-over-end agitation for the standard duration. The leachates were filtered through a 0.45 µm membrane and split into two aliquots: one for pH determination and one, after acidification, for the analysis of Cu, Ni, Cr, Pb, Cd and As by inductively coupled plasma mass spectrometry (ICP-MS). The same procedure was applied to the raw, oven-dried sludge, so that the leachability of the untreated waste (Section 3.1) and that of the stabilized mortars (Section 3.5) are directly comparable and are expressed on the same basis.

3. Results and Discussion

3.1 Heavy-metal leachability of the raw electroplating sludge

To place the stabilization/solidification program in context, the raw electroplating sludge was itself subjected to the TCLP, and the resulting leachate concentrations were compared with the regulatory levels of the US EPA, the guideline for on-site reuse, the criterion for on-site disposal and the European Union limit values, where available (Table 5). Because the same extraction protocol was applied to the untreated waste and to the stabilized mortars, Table 5 constitutes the baseline against which the leaching results of Section 3.5 are to be read.

Table 5 TCLP leachate concentrations (mg/L) of the raw electroplating sludge, compared with the applicable regulatory thresholds

Heavy metal	TCLP leachate of the raw sludge	US EPA regulatory level[a]	On-site reuse	On-site disposal	EU limit
Pb	45	5	0.75	50	-
Zn	280	[b]	4.3	250	-
Cd	2.8	1	0.11	10	-
Cr	> 1 wt %[c]	5	0.6	50	-
As	11	5	5	50	-
Ni	> 3 wt %[c]	[b]	11	250	0.12
Cu	4125	[b]	-	250	0.6

[a] *US EPA regulatory levels for the toxicity characteristic from US Environmental Protection Agency (1992).*

[b] *Zinc, nickel and copper are not among the eight metals for which a toxicity-characteristic regulatory level is defined (As, Ba, Cd, Cr, Pb, Hg, Se and Ag).* [c] *The leachate concentrations of chromium and nickel were not quantified; the values given are the total contents of the dried sludge (> 10 000 and > 30 000 mg/kg respectively) and are not directly comparable with the leachate-based thresholds in the remaining columns.*

3.2 Fresh-state behavior

3.2.1 Slump

The slump test was carried out on the Group C mixes, in which ES partially replaces the fine aggregate, together with the plain reference mortar F, and the results are plotted in Fig. 2. The slump increased monotonically with the sludge content, from 75 mm for the control to 100, 135 and 150 mm at 15, 25 and 35 % replacement, and the four points are well described by the linear relationship $y = 2.24x + 72.94$ ($R^2 = 0.98$), where y is the slump in mmm and x the volumetric replacement level in percent. Each percentage point of replacement thus raised the slump by about 2.2 mm, even though the water/cement ratio was held at 0.485 and no superplasticizer was used. All the sludge-bearing mixes fall within the range conventionally

regarded as high-workability mortar, indicating that, over this composition range, the partial substitution of fine aggregate by ES does not reduce the workability of the fresh mortar but rather improves it.

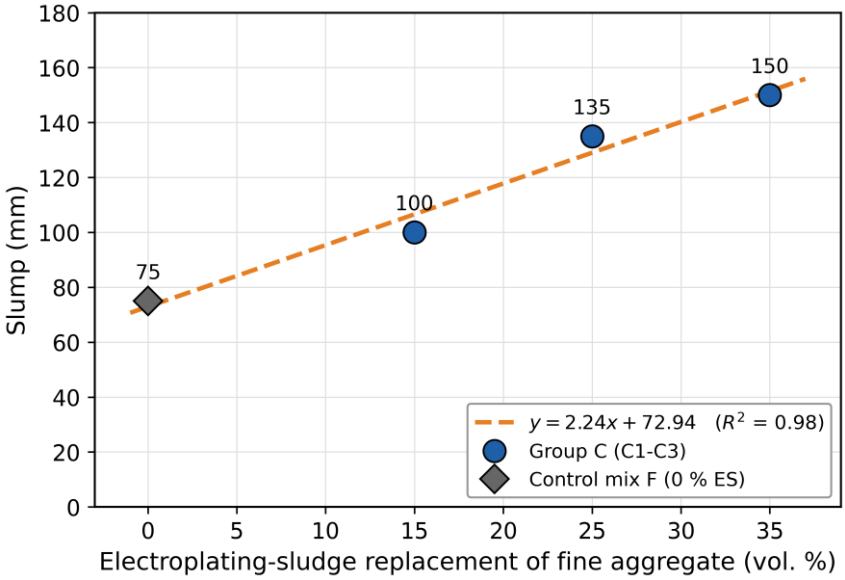


Fig. 2 Slump of the Group C mortars, in which electroplating sludge partially replaces the fine aggregate, together with the plain reference mortar F at 0 % replacement. The fitted line is the least-squares regression through all four points.

The two roles of the sludge act through different mechanisms. When ES is introduced into the binder (Groups A and B), it dilutes the clinker without contributing reaction products of its own, since its combined $SiO_2 + Al_2O_3 + Fe_2O_3$ content is only 27.10 % (Table 2) and it therefore does not meet the compositional requirement for a pozzolan. At the same time its loss on ignition of 33.94 % reflects a large fraction of metal hydroxides, bound water and carbonates, so the particles are internally porous and of high specific surface and retain part of the mixing water instead of releasing it to lubricate the mixture; the metals that pass into the pore solution also precipitate on the cement grain surfaces, a process documented both for cement-solidified electroplating sludge and for cement-treated lead-contaminated clay (Chen et al. 2009; Ouhadi et al. 2021), and modify the early cohesion of the paste. Together with the fall in the water/binder ratio from 0.463 to 0.270 in Group A and from 0.440 to 0.261 in Group B (Section 2.3, Table 4), this is why 1 to 3 g of superplasticizer were required at ES/CEM ≥ 25 % to maintain a comparable consistency.

When ES replaces part of the fine aggregate (Group C), the cement content, the mixing water and therefore the paste volume are unchanged, and only the grading of the granular skeleton is modified. The particle-size distribution measured by wet sieving (Fig. 1) shows that the sludge is markedly finer than the standard sand it replaces, with more than half of its mass finer than about 0.4 mm and 12 % finer than 0.075 mm against about 1 % for the sand. These fines fill the voids between the sand grains and release the water that would otherwise be trapped there, while reducing interparticle friction; the lower specific gravity of the Group C mixes (Table 6) further indicates that, for an equal replaced mass, the porous sludge occupies a larger volume than the sand and increases the volume of fine, paste-like material in the mortar. Both effects raise the flowability and account for the trend in Fig. 2. The two contributions are nevertheless in competition, because the high specific surface of the fines also adsorbs water, so the linear relationship should not be extrapolated beyond 35 % replacement.

Table 6 Specific gravity of the Group C mixes and of the control mortar

Mix code	ES replacement of fine aggregate (vol. %)	Specific gravity (g/cm³)	Change vs control (g/cm³)	Change vs control (%)
F (control)	0	2.145	reference	reference
C1	15	2.085	−0.060	−2.8
C2	25	2.056	−0.089	−4.2
C3	35	2.026	−0.119	−5.6

3.2.2 Setting time

The influence of the ES and RHA contents on the setting behavior of the binder is shown in Fig. 3. Two consistent trends emerge. First, at any fixed RHA content, increasing the proportion of sludge in the binder markedly lengthened both the initial and the final setting times. In Group A the initial set rose from 140 min at 0 % ES to 230, 340 and 400 min at ES/CEM ratios of 25, 50 and 75 %, and the final set from 295 to 390, 470 and 495 min; in Group B the corresponding values were 180, 250, 350 and 420 min for the initial set and 305, 400, 480 and 510 min for the final set. This is consistent with the inhibition of cement hydration commonly reported in the presence of heavy metals, in particular Cu, Cd and Pb, all abundant in the sludge characterized in Table 5, which retard the early hydration of C_3S and C_3A by forming metal hydroxide or metal silicate layers on the cement grain surfaces that delay precipitation of the C-S-H gel (Chen et al. 2009; Chen et al. 2020; Ouhadi et al. 2021). Second, at any given

sludge dosage the mortars containing 10 % RHA set later than their 5 % RHA counterparts, with the initial setting time delayed by 10-40 min and the final setting time by 10-15 min. This RHA-induced retardation is consistent with the slow pozzolanic kinetics of RHA-blended binders: partial substitution of OPC by RHA dilutes the rapidly reacting clinker phases, while the pozzolanic reaction of the amorphous silica with the calcium hydroxide released during hydration is intrinsically slower. The effect is moderate in absolute terms (≤ 40 min on the initial set, ≤ 15 min on the final set), so that practical workability windows are retained at both 5 % and 10 % RHA, even at the highest sludge dosage.

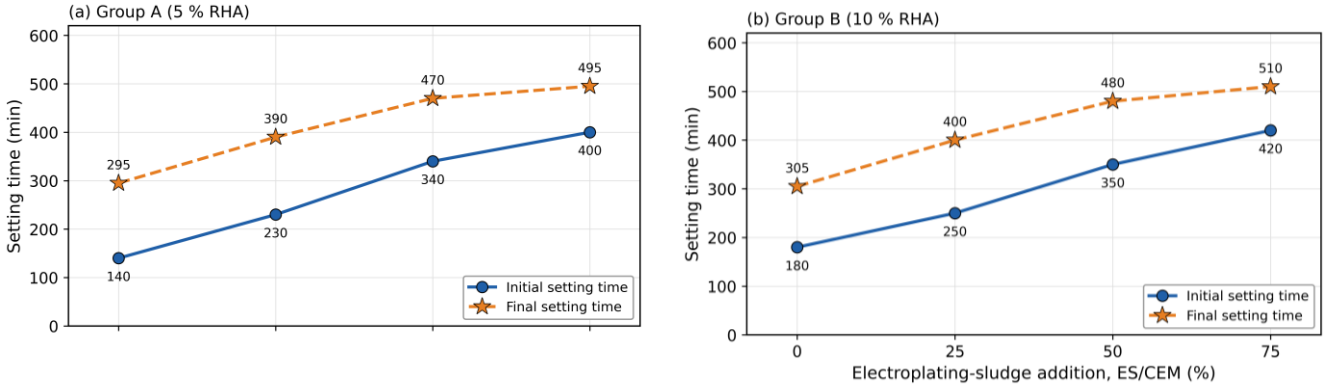


Fig. 3 Initial and final setting times of the binder as a function of the electroplating-sludge dosage: (a) Group A (5 % RHA) and (b) Group B (10 % RHA). The 0 % points correspond to the sludge-free reference mixes A1 and B1.

3.3 Specific gravity

The specific gravity of the Group C mixes decreased steadily as the sludge content increased, from 2.085 g cm^{-3} at 15 % replacement to 2.056 and 2.026 g cm^{-3} at 25 and 35 % respectively (Table 6). Relative to the control mix F (2.145 g/cm³), these are reductions of 0.060, 0.089 and 0.119 g/cm³, about 2.8, 4.2 and 5.6 %. Because Group C and mix F share the same cement content, the same mixing water and the same total mass of granular skeleton, the comparison isolates the effect of the sludge itself. The decrease is consistent with the replacement of dense natural sand by lighter, more porous sludge particles, whose high loss on ignition (33.94 %, Table 2) reflects the hydroxide-rich, low-density character typical of the precipitates produced by electroplating wastewater treatment (Chen et al. 2009; Ouhadi et al. 2021). The change is modest, at most about 6 %, so the partial substitution of fine aggregate by ES lowers the specific gravity of the mixture only slightly.

3.4 Compressive strength

The unconfined compressive strength of the mixes in which sludge was introduced into the binder is shown in Fig. 4a for Group A (5 % RHA) and in Fig. 4b for Group B (10 % RHA), while Fig. 4c compares the fine-aggregate replacement mixes of Group C with the plain reference mortar F at 28 days.

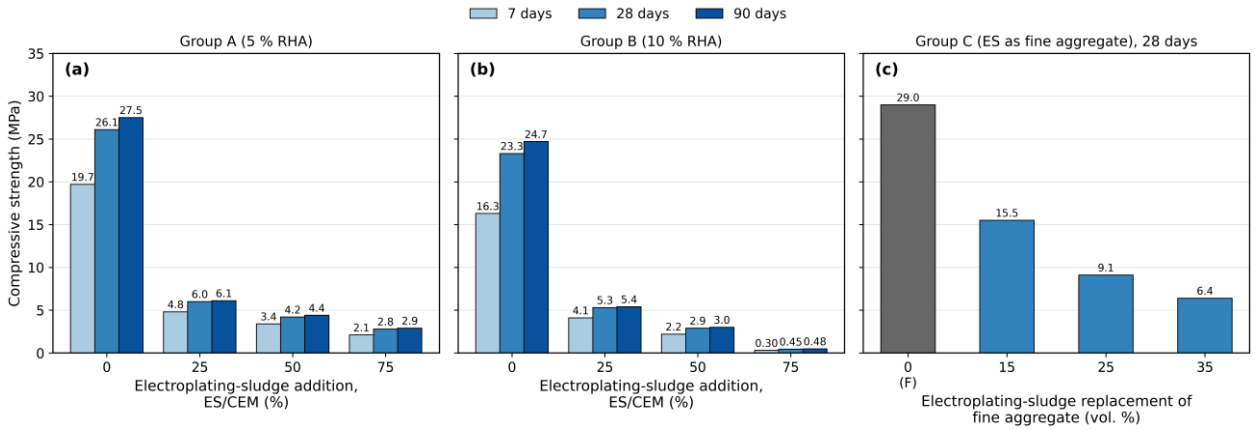


Fig. 4 Unconfined compressive strength of the mortars: (a) Group A (5 % RHA) and (b) Group B (10 % RHA), in which the sludge is added to the binder, at 7, 28 and 90 days; and (c) Group C, in which the sludge replaces part of the fine aggregate, together with the plain reference mortar F, at 28 days.

Before the effect of the sludge is considered, the two sludge-free blended mixes may be compared with the plain control. At 28 days, mix F reached 29.0 MPa against 26.1 MPa for A1 (5 % RHA) and 23.3 MPa for B1 (10 % RHA), so that the pozzolanic addition cost about 10 and 20 % of the 28-day strength even though the cement contents of A1 and B1 exceed that of F. Two factors contribute. The higher total powder content of the blended mixes raises the water demand at a fixed water/cement ratio, and the pozzolanic reaction of the RHA is slow, so that by 28 days it has not yet compensated for the dilution of the clinker; the trend with age supports this interpretation, the 90-day strengths of A1 and B1 recovering to 27.5 and 24.7 MPa. Mix F is therefore used below as the direct reference for Group C, whose binder is identical to it,

while A1 and B1 serve as the internal references within Groups A and B.

For both RHA levels the strength fell steeply once sludge entered the binder. The sludge-free mixes reached 19.7, 26.1 and 27.5 MPa at 7, 28 and 90 days for 5 % RHA (Group A) and 16.3, 23.3 and 24.7 MPa for 10 % RHA (Group B), whereas at ES/CEM = 25 % the 90-day strength dropped to 6.1 MPa (Group A) and 5.4 MPa (Group B), and at 75 % it fell to 2.9 MPa in Group A and 0.48 MPa in Group B. Strength increased with curing age at every dosage, reflecting the slow but continuing development of the binding phases, but the gains were insufficient to offset the dominant effect of the sludge. The steeper loss in Group B indicates that the higher RHA content, while beneficial for pozzolanic refinement in the longer term, further diluted the early strength-giving clinker phases and, combined with the hydration-retarding metals in the sludge, left very little load-bearing capacity at the highest dosage.

The marked strength reduction is attributable to the same mechanisms that retarded setting: the heavy metals interfere with cement hydration (Chen et al. 2009; Chen et al. 2020; Ouhadi et al. 2021), and the high loss on ignition and porous nature of the sludge increase porosity and weaken the matrix; the fall in the water/binder ratio documented in Table 4 also leaves an increasing fraction of the powder unwetted and therefore inert. Reported S/S practice indicates that cement- and lime-based binders dominate the field, with Portland cement generally outperforming lime in compressive and tensile strength, and that an optimal cement dosage of about 6 % has been proposed on economic and engineering grounds (Wang DX et al. 2012); the waste-to-cement ratio in the present mixes was varied between 25 and 75 % to span the range of practical interest.

A further feature of Fig. 4a, b is that the strength gain between 28 and 90 days, which is clear in the sludge-free mixes, almost disappears once sludge is present: at every dosage the two ages give practically the same value, 6.0 against 6.1 MPa at 25 % ES, 4.2 against 4.4 MPa at 50 % and 2.8 against 2.9 MPa at 75 % in Group A, with the same pattern in Group B, so that the 28-day strength is effectively the ultimate strength of these mixes. Three factors combine to produce this plateau. At high ES/CEM ratios, the clinker fraction of the total powder is small and is largely consumed within the first 28 days, leaving little unreacted cement to form further C-S-H. The metal hydroxide and metal silicate layers described in Section 3.2.2 are stable in the pore solution and progressively isolate the residual anhydrous cores, so that hydration is arrested rather than merely delayed and the strength lost to retardation is never recovered at later ages (Chen et al. 2009; Chen et al. 2020). Finally, the precipitation of the metals consumes calcium hydroxide and lowers the alkalinity of the pore solution, which suppresses the pozzolanic reaction of the RHA precisely in the mixes that would benefit most from it, in agreement with the plateau being more pronounced in Group B. Strength in these mixes is therefore governed by a coarse pore structure that later hydration can no longer refine, and no late-age recovery should be expected from sludge-bearing binders.

When ES instead replaced the fine aggregate (Group C, 28-day curing, Fig. 4c), the compressive strengths were 15.5, 9.1 and 6.4 MPa at 15, 25 and 35 % replacement, against 29.0 MPa for the plain reference mortar F, that is retentions of about 53, 31 and 22 % of the control strength. These values are well above those of the mixes in which the sludge entered the binder at a comparable loading; at 25 %, for example, Group C retains 9.1 MPa whereas A2 and B2 retain 6.0 and 5.3 MPa. The binder phase remains intact when ES is used as an aggregate, so hydration is not directly inhibited and the loss of strength arises mainly from replacing a dense, strong sand by weaker and more porous particles, in line with the reduction in specific gravity reported in Table 6. The loss is not proportional over the whole range: the first 15 % of replacement costs about 0.90 MPa per percentage point of sludge, against roughly 0.46 MPa per percentage point between 15 and 35 %, which suggests that once the granular skeleton is dominated by the sludge fines, further replacement has a smaller marginal effect.

3.5 Leaching behavior (TCLP)

The leaching of the stabilized mortars, measured by TCLP on 28-day specimens, is reported in Table 7 and plotted in Fig. 5, which shows the leachate concentrations of Cr, Ni, Cu and Pb against the ES/CEM ratio for (a) Group A (5 % RHA) and (b) Group B (10 % RHA) on a logarithmic scale, together with the US EPA regulatory level of 5 mg/L applicable to chromium and lead. Cadmium and arsenic are not plotted because they lay below the detection limit in all but one determination: arsenic remained below 0.005 mg/L in every mix and cadmium in five of the six, the single exception being mix B4 at 0.1 mg/L, while lead did not exceed 3.0 mg/L in any mix.

Table 7 TCLP leachate concentrations of the stabilized mortars (28-day specimens; mg/L). Group A contains 5 % RHA and Group B 10 % RHA

Mix	ES/CEM (%)	Cd	Pb	Ni	Cr	As	Cu	pH
A2	25	< 0.005	0.1	26.0	9.0	< 0.005	1.8	6.1
A3	50	< 0.005	0.3	44.6	9.7	< 0.005	2.0	5.9
A4	75	< 0.005	1.0	86.6	13.1	< 0.005	2.8	5.7
B2	25	< 0.005	0.4	19.4	5.6	< 0.005	1.0	6.2
B3	50	< 0.005	0.3	41.7	8.5	< 0.005	1.6	5.9
B4	75	0.1	3.0	61.0	11.0	< 0.005	2.0	5.8
US EPA regulatory level	-	1	5	-	5	5	-	-

All three elements are therefore below their respective US EPA regulatory levels (Cd 1, Pb 5, As 5 mg/L; Table 5), indicating effective immobilization by the cementitious matrix. Chromium, by contrast, exceeded the 5 mg/L limit in every sludge-bearing mix, ranging from 5.6 to 13.1 mg/L and increasing with the sludge dosage, so that chromium leaching remains the principal constraint on safe reuse of the products despite the very large reduction relative to the raw sludge (Table 5). Nickel and copper, for which the US EPA does not specify a regulatory level, were released at 19.4-86.6 and 1.0-2.8 mg/L respectively, again rising with the sludge dosage. The eluate pH remained mildly acidic (5.7-6.2), reflecting the buffering imposed by the acetic acid extraction fluid. As Fig. 5 shows, chromium, nickel and copper increase approximately log-linearly with the sludge dosage in both groups; lead follows the same trend in Group A but is non-monotonic in Group B, where the 25 and 50 % mixes are indistinguishable (0.4 and 0.3 mg/L) before a sharp rise to 3.0 mg/L at 75 %.

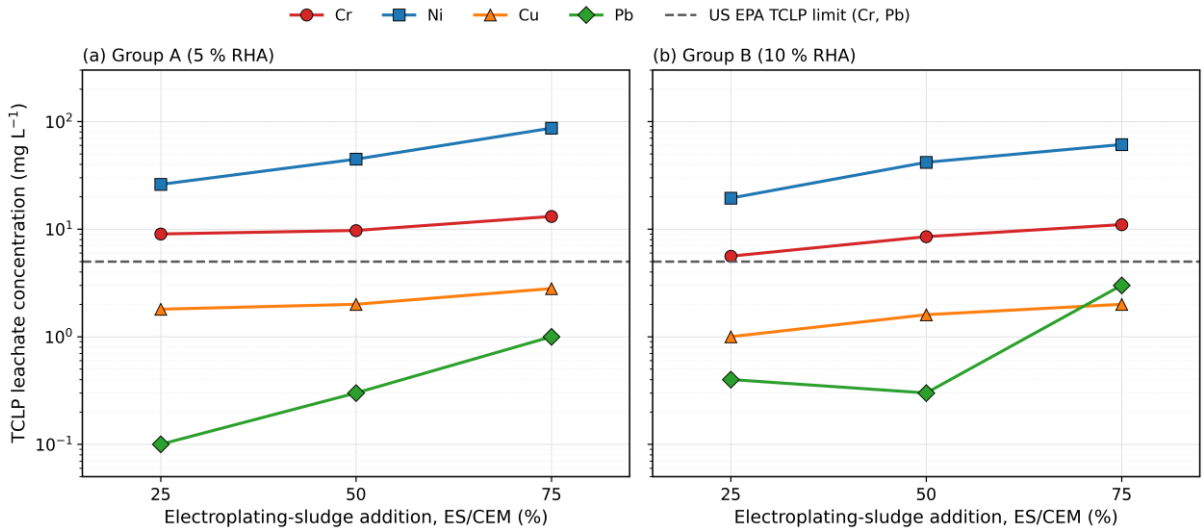


Fig. 5 TCLP leachate concentrations of Cr, Ni, Cu and Pb for the 28-day stabilized mortars as a function of the electroplating-sludge dosage: (a) Group A (5 % RHA) and (b) Group B (10 % RHA). The dashed line is the US EPA regulatory level of 5 mg/L, which applies to chromium and lead but not to nickel or copper. Cadmium and arsenic are omitted because they lay below the detection limit in all but one determination.

The contrast between the elements follows from the way a cement matrix retains them. Cadmium and lead are divalent cations and are held by three mechanisms that all benefit from the alkalinity of the binder: precipitation as sparingly soluble hydroxides and, in the case of lead, as hydroxy carbonate phases; sorption on the C-S-H gel with partial substitution for calcium in its structure and interlayer; and physical encapsulation within the dense hydrate matrix, which limits the access of the extraction fluid to the metal-bearing phases (Chen et al. 2009; Ouhadi et al. 2021). Arsenic is fixed by a different route because it occurs as an oxyanion: it precipitates as calcium arsenate compounds, which the large calcium hydroxide reserve of the matrix makes possible, and is taken up by anion exchange in ettringite and the AFm phases, where arsenate substitutes for sulfate. Both routes are efficient at the concentrations involved here, and the small rise in the lead released at the highest sludge dosages (Fig. 5) simply reflects the greater metal load and the more porous matrix obtained at those levels.

Chromium fails for a specific chemical reason. In electroplating sludge, it derives largely from chromic acid and chromate plating baths and is therefore present to a significant extent as Cr(VI), which exists as the chromate and hydrogen chromate oxyanions (Chen et al. 2020). These species do not precipitate as hydroxides, so the alkalinity that immobilizes Cd, Pb, Ni and Cu offers them no advantage, and chromate remains soluble in alkaline media. The only sinks available in an OPC/RHA system are substitution for sulfate in ettringite and the AFm phases and a limited, reversible uptake by the C-S-H gel, and the sulfate already supplied by the cement and by the sludge (SO_3 = 2.35 % in both, Table 2) competes for the same sites. Under the TCLP protocol the acetic acid extraction fluid drives the eluate to pH 5.7-6.2 (Table 7), a range in which ettringite and the AFm phases are no longer stable and the C-S-H is progressively decalcified, so that chromate previously bound is released; the sensitivity of cement-solidified electroplating sludge to acid attack, and the associated loss of acid neutralization capacity, have been documented directly by Chen et al. (2009). No reducing agent is present to convert Cr(VI) into Cr(III), which would precipitate as the highly insoluble $Cr(OH)_3$. These factors together explain why chromium exceeds the limit in every sludge-bearing mix and why the exceedance widens as the sludge dosage increases in both panels of Fig. 5.

Nickel and copper are cationic and are retained by the same mechanisms as cadmium and lead, yet they are released at much higher concentrations. The difference is one of loading rather than of mechanism: the raw sludge contains more than 3 wt % nickel and leached 4125 mg/L of copper (Table 5), so the retention capacity of the matrix is saturated, and nickel hydroxide is appreciably more soluble than lead hydroxide in the mildly acidic range imposed by the extraction fluid. Fig. 5 also shows that, at a given sludge dosage, the mixes containing 10 % RHA release less chromium and less nickel than their 5 % RHA counterparts, for example 5.6 against 9.0 mg/L of chromium at ES/CEM = 25 % and 11.0 against 13.1 mg/L at 75 %. This is consistent with the pozzolanic formation of additional C-S-H of lower calcium-to-silicon ratio and with a refined pore structure, both of which increase the number of sorption sites and reduce the accessibility of the matrix (Asavapisit and Macphee 2007; Chiu et al. 2019); the effect is

systematic but far too small to bring chromium below the regulatory threshold. The benefit does not extend to every element, however: at the highest dosage, mix B4 released more lead and more cadmium than A4 (3.0 against 1.0 and 0.1 against < 0.005 mg/L), which is consistent with the much weaker and more porous matrix obtained in Group B at that loading, where the 28-day strength was only 0.45 MPa.

Overall, the matrix immobilized Cd, Pb, and As effectively but did not bring Cr within regulatory limits. Because the cause is the oxidation state of the chromium rather than the quality of the binder, the most direct route to compliance is the reduction of Cr(VI) to Cr(III) before or during solidification, for example by adding ferrous sulfate, a sulfide-bearing ground granulated blast furnace slag or zero-valent iron, after which chromium can be retained as an insoluble trivalent hydroxide; alkali-activated slag binders have been shown to perform well for chromium-bearing electroplating sludge for related reasons (Chen et al. 2020). This is an outcome that should be addressed before field application.

3.6 Limitations

First, the mechanistic interpretation offered in Sections 3.2 to 3.5 rests on the particle-size distribution of Fig. 1, on the oxide and loss-on-ignition data of Table 2, and on the strength and leaching behavior itself. Scanning electron microscopy and X-ray diffraction were outside the scope of the present work, so the microstructural changes and the hydrate assemblages invoked here — the metal hydroxide and metal silicate layers on the clinker grains, the substitution of chromate for sulfate in ettringite and the AFm phases, and the refinement of the pore structure by the pozzolanic reaction — were not observed directly. Confirming them would require a combined SEM/EDS and XRD program, ideally supplemented by thermogravimetric analysis of the calcium hydroxide reserve.

Second, the rice husk ash was characterized chemically but neither mineralogically nor physically. The combustion window of 550–600 °C was selected to preserve amorphous silica, and the behavior inferred from the strength and leaching results is consistent with that intention, but no X-ray diffraction pattern was recorded to confirm the absence of cristobalite and tridymite, and neither the Blaine fineness nor the specific surface area of the ash was measured. The pozzolanic contribution of the RHA is therefore established here only indirectly, through its effect on the mortars, and comparison with ashes produced elsewhere should be made with that in mind.

Third, the leaching assessment rests on a single batch extraction at one curing age. The TCLP is a short-term, worst-case screening test conducted under acetic acid conditions on crushed material, and it does not describe the release that would occur from a monolithic product over the service life of a structure. Dynamic or semi-dynamic tank leaching, sequential extraction to identify the binding fractions, and testing at later curing ages and after carbonation would be needed before a durable, field-scale conclusion on long-term containment could be drawn. Extending the program in these directions, and coupling it with the Cr(VI) reduction strategies identified in Section 3.5, is the natural continuation of this work.

4. Conclusion

This study examined the stabilization/solidification of electroplating sludge in cement-based mortars and the role of rice husk ash, evaluating fresh-state behavior, compressive strength, and heavy-metal leaching across twelve mixes. The raw electroplating sludge is unambiguously hazardous. Under the same TCLP protocol later applied to the mortars, its leachate exceeded at least one regulatory threshold for every metal determined, which justifies a stabilization/solidification approach. The effect of the sludge on the fresh mortar depended on its role. Used as a fine-aggregate replacement, it raised the slump from 75 mm for the control mortar to 150 mm at 35 % replacement, improving workability, whereas when added to the binder, it retarded both the initial and the final setting times, an effect amplified by a higher RHA content (10 % against 5 %); the delays nonetheless remained within practical limits, at most 420 min for the initial and 510 min for the final set. Compressive strength fell sharply once sludge entered the binder, from 16.3-29.0 MPa for the sludge-free mixes to 0.30-6.1 MPa at ES/CEM ratios of 25-75 %, and the 28- and 90-day values were almost identical in the sludge-bearing mixes, so that no late-age recovery should be expected. Used as a fine-aggregate replacement, ES preserved markedly higher strength, from 15.5 MPa at 15 % to 6.4 MPa at 35 %, against 29.0 MPa for the plain reference mortar. TCLP testing showed that cadmium, lead and arsenic were immobilized below their US EPA regulatory levels, whereas chromium exceeded the 5 mg/L limit in all sludge-bearing mixes because it is present as the chromate oxyanion, which the alkalinity of a cement matrix cannot precipitate; reduction of Cr(VI) to Cr(III) is therefore the priority for further work.

Statements and Declarations

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Ethical considerations
Ethical issues (Including plagiarism, informed consent, misconduct, data fabrication and/or falsification, double publication and/or submission, redundancy, etc.) have been completely observed by the author.

Data availability
Data will be made available on request.

Conflicts of interest
The author declares that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Author contribution
S. Shams: Writing the main Draft; S. Gitipour: Conceptualization; Writing – Review & Editing; and A. M. Baghdadi: Conceptualization, Methodology, Review, and Final Proof.

AI Use Declaration
During the preparation of this manuscript, the authors used AI tools for designing the Graphical Abstract. All content has

been carefully reviewed and revised by the authors, who take full responsibility for the final version of the manuscript.

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Long-term occupational risk assessment and injury severity analysis in municipal solid waste management: A 12-year Fine–Kinney study

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ABSTRACT

This study assessed heavy metal contamination in the Garan River's water and sediments to evaluate the impact of landfill leachate. Pollution levels and ecological risks were quantified using established indices, including the contamination factor (CF), pollution load index (PLI), geoaccumulation index (Igeo), enrichment factor (EF), and potential ecological risk index (PERI). Spatially, peak dissolved metal concentrations in water occurred at points S1 and S4, while maximum sediment concentrations were found at S3, directly downstream of the leachate outfall. Water quality analysis indicated that arsenic posed both acute and chronic human health risks at all stations, and chromium posed an acute risk at all stations. Sediment assessment revealed that barium, nickel, and mercury consistently exceeded the threshold effect concentration (TEC). Furthermore, nickel and mercury surpassed the probable effect concentration (PEC) at 100% and 85.71% of stations, respectively, signaling a significant potential for adverse effects on aquatic life. Index analysis showed higher CF values in sediments than in water. The PLI trend increased in sediments but decreased in water, whereas EF was higher in water and spatially stable. The average Igeo for all metals remained below zero, classifying the area as unpolluted. The PERI indicated medium ecological risk in sediments only at stations S3, S6, and S8. While the collective indices suggest that severe, widespread pollution is not present, the identified contamination hotspots and specific risks from metals like arsenic, nickel, and mercury warrant vigilant attention. Consequently, periodic monitoring of leachate-derived metals and proactive preventive measures are critically recommended to preserve aquatic ecosystem quality.

Highlights

- A decision matrix guided safety actions and resource allocation by injury severity.
- Fine-Kinney analysis assessed 84 accidents at a municipal waste management complex.
- Processing/transfer stations: risk (R = 540), intolerable; >60% of recorded accidents.
- Roads had a high-risk score (R = 450), with intolerable risk due to fatal outcomes.
- Lines, transfer stations, and roads were top priorities for immediate safety action.

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1. Introduction

Municipal solid waste management (MSWM) is a fundamental infrastructure of modern societies and plays a vital role in safeguarding public health and ensuring environmental sustainability. However, the operational nature of MSWM—encompassing waste collection, transportation, processing, recycling, and disposal—exposes workers to a

wide range of occupational hazards, including biological, chemical, and physical agents. If these hazards are not adequately managed, they can lead to irreversible health consequences for workers (Gowda et al., 2023). A systematic review by Thisakya and Dissanayake (2025) identified the MSWM sector as one of the most hazardous occupational settings worldwide, with an occupational accident rate of 35.5 per 100,000 workers (Thisakya & Dissanayake, 2025). However, this rate should be interpreted with caution because it is based on a specific study population and a particular definition of "accident"; therefore, it may not be directly generalizable to other national or regional contexts. Moreover, despite growing awareness of these occupational hazards, research on the root causes of occupational safety and health problems among MSWM workers remains scarce and, in many cases, outdated (Kuijer et al., 2010).

Numerous studies have investigated the occurrence of occupational injuries among waste workers; however, these studies differ substantially in study design, population, outcome definition, recall period, and denominator, limiting direct comparability. A cross-sectional study conducted in Karnataka, India, involving 313 solid waste management workers, reported a prevalence of occupational injuries of 73.2%, with falls accounting for 63.6% (Jayakrishnan et al., 2013). In Addis Ababa, Ethiopia, the 12-month prevalence of occupational injuries was reported as 43.7%, and non-use of personal protective equipment was associated with 2.62-fold increased odds of injury (Odo, 2012). A study by Abd El-Wahab et al. in Alexandria, Egypt, revealed that the prevalence of occupational accidents was 46.5% and that of needlestick injuries 32.7% (Abd El-Wahab et al., 2014). A global systematic review and meta-analysis encompassing 8,138 sanitation workers estimated the pooled prevalence of occupational injuries at 36.49% (Tolera et al., 2024). Furthermore, a study by Martins et al. at the largest landfill site in Latin America found that 68.7% of 999 waste pickers examined had a history of occupational accidents (Martins et al., 2026). It is important to note that these studies report worker-based prevalence estimates (i.e., the proportion of workers who experienced an injury during a defined period), whereas the present study analyzes recorded occupational accidents (i.e., event-based data from an accident registration system). These are conceptually different epidemiological measures and should not be directly compared. Collectively, these studies demonstrate the high occurrence of occupational injuries worldwide and underscore the role of environmental and behavioral factors, but they also highlight the need for event-based analyses that can inform intervention prioritization.

Risk assessment constitutes the core of safety management in occupational settings, and the selection of an appropriate method plays a decisive role in hazard identification and prioritization (Chen et al., 2025). Several semi-quantitative and qualitative methods are available, each with distinct strengths and limitations. The Fine-Kinney method, as one of the semi-quantitative risk assessment techniques, calculates risk as the product of probability (P), exposure, and consequence severity (S), and has been widely applied in occupational health and safety studies (Över Özçelik et al., 2025). However, like other semi-quantitative methods, Fine-Kinney is not inherently superior to alternative approaches such as Job Hazard Analysis (JHA) or Failure Mode and Effects Analysis (FMEA); rather, its appropriateness depends on the research question and available data. JHA is particularly useful for task-specific hazard identification, while FMEA is well-suited for systematic failure analysis of components and processes. Fine-Kinney, by contrast, is advantageous when the objective is to prioritize hazards or zones based on a combination of probability, exposure, and severity, particularly when retrospective accident data are available to inform scoring. Previous studies in the waste management sector have employed JHA (Jalali et al., 2025), FMEA (Babaei Masdaraghi et al., 2018), and FMEA (Keeshavarz Motamed et al., 2022), reflecting the diversity of methodological approaches in this field.

In Iran, concomitant with the ever-increasing urban population and rising waste generation, the development of waste processing and recycling facilities has accelerated considerably in recent years. However, parallel to this expansion, the rate of occupational accidents in this industry has also increased. A review of occupational accident studies in Iran indicates that the waste industry is among those with a high risk of accidents (Mehrifar et al., 2022). Nevertheless, only a limited number of studies have systematically and quantitatively assessed accident risk in waste facilities, and even fewer have covered accident trends over extended time periods. Although the severity of consequences is one of the key components of risk assessment methods such as the Fine-Kinney method, few studies in Iran have analyzed accidents based on injury severity and presented a corresponding prioritization framework. Most studies have confined themselves to hazard identification and risk prioritization, with less attention given to developing an operational framework for optimal resource allocation and safety intervention prioritization grounded in injury severity. To the best of our knowledge, no study in Iran has systematically analyzed occupational accidents in municipal solid waste management facilities based on injury severity over a long-term (12-year) period. This claim is based on our review of the available literature; however, we acknowledge that the absence of evidence is not definitive evidence of absence.

The present study was conducted at a single municipal solid waste management facility in Iran, which includes processing and sorting lines, transfer stations, a maintenance workshop, internal roadways, and administrative sections. This facility was selected because it represents a typical MSWM complex in Iran, with operational characteristics such as mechanical processing, manual sorting, and vehicle traffic that are common to similar facilities in the country. Conventional accident analysis often relies on frequency-based prioritization, which may overlook zones where severe but infrequent events occur. For example, a zone with a high number of minor injuries may receive greater attention than a zone with fewer accidents but more fatalities, even though the latter may pose a greater threat to worker safety. Incorporating injury severity into risk assessment is therefore expected to provide a more informative basis for intervention prioritization, particularly in contexts where resources are limited and must be allocated strategically. This study adopts a reactive approach—analyzing actual occurred accidents rather than relying solely on preventive assessments—to complement existing proactive risk assessment efforts.

However, it is important to clarify that this study does not directly compare reactive and proactive methods, nor does it claim that reactive analysis can identify hazards that proactive assessments cannot detect. Rather, it aims to demonstrate how retrospective accident data, when integrated with severity classification, can inform practical prioritization decisions.

The present study, therefore, seeks to analyze occupational accidents recorded over 12 years at the study facility, classify them by location and injury severity, apply the Fine-Kinney method to calculate risk scores, and develop a prioritization framework to assist managers in allocating limited safety resources. The intended contribution of this study is practical and context-based: it demonstrates how existing tools, when applied systematically to longitudinal accident data, can yield actionable insights for resource allocation in waste management facilities. This study does not claim to introduce a new methodological innovation; rather, it aims to provide a transparent and reproducible framework that may be adapted by similar facilities in other developing countries, with appropriate localization. Whether the proposed framework improves resource allocation compared with existing methods remains an empirical question that requires further validation in future studies.

2. Materials and Methods

2.1 Study Design

The present study is a retrospective, record-based observational study that examined and analyzed occupational accidents recorded at a municipal solid waste management facility in Iran over 12 years (2010–2022). The study was designed with a combined approach encompassing descriptive analysis and semi-quantitative risk assessment.

2.2 Study Population and Sample

A total of 102 accident records were initially identified in the facility's accident registration system for the period 2010–2022. Of these, 18 records were excluded for the following reasons:

- 9 records lacked sufficient detail on the accident description or injury outcome.
- 5 records were duplicates (same accident reported twice)
- 4 records were incomplete (missing date, location, or severity information).

The final sample therefore comprised 84 accidents (82.4% of the initial records). A flow diagram illustrating the record selection process is presented in Figure S1 (Supplementary Material).

2.3 Variable Classification and Operationalization

Two occupational health and safety (OHS) specialists independently coded location and injury severity. Inter-rater agreement was assessed using Cohen's kappa, yielding $\kappa = 0.87$ for location classification and $\kappa = 0.82$ for severity classification, indicating strong agreement. We resolved disagreements through discussion until consensus was reached.

a) Classification of the accident location variable

Based on the operational nature of the facility and the geographical distribution of accidents, locations were disaggregated into five zones (Table 1).

Table 1 Classification of location zones

Zone Code	Zone Name	Description of Included Locations
Z1	Processing and sorting lines	Screen, conveyor belt, sorting lines, granulated fertilizer line, secondary fermentation, sorting room
Z2	Transfer stations and platforms	Khalkhabad, Hesarak, waste reception hall, weighbridge, dog shelter, reject
Z3	Workshop and warehouse	Central workshop, car wash, warehouse, fuel station
Z4	Road, yard, and traffic	Halgheh Darreh road, underpasses, line yard, streets leading to the facility
Z5	Other and administrative sections	Physicians' building, administrative office, administrative staircase, administrative floors, green space

b) Classification of the severity variable

Based on the ultimate consequence of the accident to the injured individual's health, and with reference to standard severity criteria used in risk assessment methods, injury severity was divided into four levels (Table 2). It should be noted that the severity scores assigned in this study (100, 50, 15, and 5) represent a modified version of the standard Fine-Kinney severity scale, adapted to the specific context of occupational accidents in waste management facilities. In the standard Fine-Kinney scale, severity scores range from 1 to 100, with specific anchors for catastrophe, disaster, very serious, serious, important, and noticeable consequences. However, because the standard scale is primarily designed for industrial incidents with property damage components, we adapted the severity anchors to focus exclusively on human health consequences, as follows:

- **S4 (Catastrophic):** Fatality → Score 100 (consistent with standard Fine-Kinney "catastrophe")
- **S3 (Very serious):** Amputation or severe fracture → Score 50 (modified; standard Fine-Kinney assigns 40 for "disaster" and 15 for "very serious")
- **S2 (Serious):** Injury requiring medical treatment and absence from work → Score 15 (consistent with standard Fine-Kinney "very serious")
- **S1 (Minor):** Superficial injury without absence → Score 5 (modified; standard Fine-Kinney assigns 1 for "noticeable")

The rationale for this modification is that the standard Fine-Kinney severity scale includes monetary damage thresholds that are not applicable to occupational accident records in this

context. We therefore retained the human-health anchors and adapted the intermediate scores to reflect the severity gradient observed in our dataset. This modification is explicitly acknowledged as a deviation from the standard method, and its implications for risk scores are discussed in the Limitations section.

2.4 Aggregation rule for zone-level severity

Because each zone contained accidents of varying severity, a single severity score (S) had to be assigned to each zone for the Fine-Kinney calculation. The following aggregation rule was applied consistently across all zones: the severity score corresponding to the most severe accident recorded in that zone was used. This rule was chosen because the objective of the study was to prioritize interventions based on the potential for severe harm, and using the most severe outcome ensures that zones with rare but catastrophic events are not underestimated. Applying this rule:

- Z1 (processing lines): most severe = 1 fatality (S4) → S = 100
- Z2 (transfer stations): most severe = 1 fatality (S4) → S = 100
- Z3 (workshop): most severe = 1 amputation (S3) → S = 50
- Z4 (roadway): most severe = 2 fatalities (S4) → S = 100
- Z5 (administrative): most severe = 1 amputation (S3) → S = 50

Table 2 Classification of Injury Severity

Severity Level	Title	Operational Definition	S Score
S4	Catastrophic	Resulted in death (explicit mention of death or fatality)	100
S3	Very serious (Critical)	Amputation, severe fracture (explicit mention of amputation or fracture)	50
S2	Serious	Injury requiring sutures, dressing, transfer to hospital, and absence from work	15
S1	Minor	Superficial abrasion, contusion, scraping, no need for absence from work	5

2.5 Risk Assessment Method

2.5.1 The Semi-Quantitative Fine-Kinney Technique

To numerically calculate risk and prioritize the different zones, the Fine-Kinney method was employed (Eq. 1). This technique is one of the most comprehensive semi-quantitative risk assessment methods, calculating the risk score (R) by considering three components: probability of occurrence (P), exposure frequency (E), and consequence severity (S) (Gul et al., 2021).

$$R = P \times E \times S \quad (1)$$

The evaluation tables for the concepts used to calculate the risk score in the Fine-Kinney method are presented in Tables 3 to 5 (Gul et al., 2021), and the risk score itself is shown in Table 6 (Moloudi et al., 2021).

Table 3 Probability of Occurrence (P)

Rating/Value	Probability of occurrence (P)
10	Might well be expected
6	Quite possible
3	Unusual but possible
1	Only remotely possible
0.5	Conceivable but very unlikely
0.2	Practically impossible
0.1	Virtually impossible

Table 4 Exposure

Exposure	Rating/Value
Continuous	10
Frequent (daily)	6
Occasional (weekly)	3
Unusual (monthly)	2
Rare (a few per year)	1
Very rare (yearly)	0.5

Table 5 Severity of the Consequence (Standard Fine-Kinney Scale)

Severity of the consequence	Rating/Value
Catastrophe (many fatalities, or >10^7 damage)	100
Disaster (few fatalities, or >10^6 damage)	40
Very serious (fatality, or >10^5 damage)	15
Serious (serious injury, or 10^4 damage)	7
Important (disability, or 10^3 damage)	3
Noticeable (minor first aid accident, or >$100 damage)	1

2.5.2 Determination of Probability (P)

The probability score was determined by the expert panel based on the observed frequency of accidents in each zone over the 12-year study period, combined with the operational nature of the zone. It is important to clarify that accident frequency was not equated with probability; rather, it served as one input among several. The following quantitative criteria (Table 7) were used to translate accident frequency into the categorical P score:

Table 6 Interpretation of Risk Levels in the Fine-Kinney Method

Risk score	Risk level	Suggested action
>400	Very high	Activity should not be performed until corrective action is taken.
200–400	High	Corrective action must be taken immediately.
70–200	Medium	Corrective action is required.
20–70	Low	Attention should be paid as soon as possible.
<20	Acceptable	No corrective action is required.

Table 7 Quantitative criteria for translating accident frequency into the categorical P score

Accident frequency (per zone over 12 years)	Assigned P score	Fine-Kinney description
>20 accidents	6	Quite possible
10–20 accidents	3	Unusual but possible
5–9 accidents	1	Only remotely possible
<5 accidents	0.5	Conceivable but very unlikely

It should be acknowledged that, because the number of exposure opportunities (e.g., number of workers, operating hours) was not available for all zones, the P scores represent expert judgment informed by accident frequency rather than a strictly quantitative probability calculation. This limitation is acknowledged in the Limitations section.

2.5.3 Determination of Exposure

The exposure score was determined based on the number of hours workers were exposed to hazards in each zone, as estimated from shift schedules and operational records. Specifically:

- Continuous exposure (E=10): Workers are present in the zone throughout the entire shift.
- Frequent/daily exposure (E=6): Workers enter the zone multiple times per day.
- Occasional/weekly exposure (E=3): Workers enter the zone a few times per week.
- Unusual/monthly exposure (E=2): Workers enter the zone a few times per month.
- Rare exposure (E=1): Workers enter the zone a few times per year.

We acknowledge that detailed worker-hour data were not available for all zones, and the E scores therefore represent expert estimates based on shift schedules and operational knowledge. This limitation is acknowledged in the Limitations section.

2.5.4 Expert Panel Composition and Procedure

The panel comprised three occupational health and safety (OHS) specialists with the following qualifications:

- **Expert 1:** PhD in Occupational Health, 15 years of experience, including 5 years in waste management facilities.
- **Expert 2:** MSc in Occupational Health, 10 years of experience, including 3 years in waste management.
- **Expert 3:** MSc in Safety Engineering, 8 years of experience, including 4 years in waste management.

Experts were selected based on their demonstrated expertise in occupational risk assessment and their familiarity with waste management operations. Each expert independently reviewed the zone descriptions, accident frequency data, and operational characteristics, and independently assigned P and E scores. Disagreements were resolved through a consensus meeting. The arithmetic mean of the three experts' ratings was used as the final P and E score for each zone, and scores were rounded to the nearest value in the Fine-Kinney scale.

2.5.5 Methodological Contribution and Innovation

While the present study applies the semi-quantitative Fine-Kinney method as its risk assessment framework, the methodological contribution lies in the systematic integration of 12-year longitudinal accident data with expert panel judgment to calibrate the probability (P) and exposure scores, rather than relying solely on expert opinion as in conventional applications of this method. The study does not introduce new weighting factors or a novel algorithmic validation process; rather, it adopts a practical, context-driven approach to risk prioritization. It must be acknowledged that no new weighting method was developed, and no formal inter-rater agreement index (such as ICC or Kendall's W) was calculated. The innovation of this study is therefore practical and context-based, rather than methodological.

2.6 Statistical Analysis Method

The collected data were analyzed after entry into SPSS software version 26 and Microsoft Excel 2019. Given the descriptive analytical nature of the study and the absence of a control group, inferential statistical tests (such as chi-square) were not employed for comparing proportions. Instead, the analysis was conducted based on descriptive indices and the risk score (Moloudi et al., 2022).

3. Results and Discussion

In this section, the results derived from the analysis of 84 occupational accidents recorded over 12 years (2010–2022) at

the municipal solid waste management facility under study are presented. The findings are organized into four sequential steps: descriptive data analysis, examination of the relationship between accident location and injury severity, quantitative risk calculations using the Fine-Kinney method, and finally, derivation of the intervention prioritization matrix.

The analysis of 84 recorded accidents over 12 years (2010–2022) revealed that the study facility faces a high risk of occupational accidents, with 11.9% of all accidents (10 cases) resulting in fatality or amputation. The highest accident frequency was observed in the processing line zone, with 27 cases (32.1%), followed by the transfer station zone with 24 cases (28.6%). According to the semi-quantitative Fine-Kinney method, the processing line and transfer station zones (R = 3600) and the roadway zone (R = 900) were classified as very high risk.

3.1 Descriptive Data Analysis

3.1.1 Frequency distribution of accidents by year of occurrence

The annual distribution of accidents, as shown in Fig. 1, revealed considerable fluctuations over the study period. The year 2019 had the highest number of accidents (21 cases; 25% of all accidents), whereas 2011 had the lowest (one case). In 2018 and 2020, 13 and 12 accidents were recorded, respectively, representing 15.5% and 14.3% of the total.

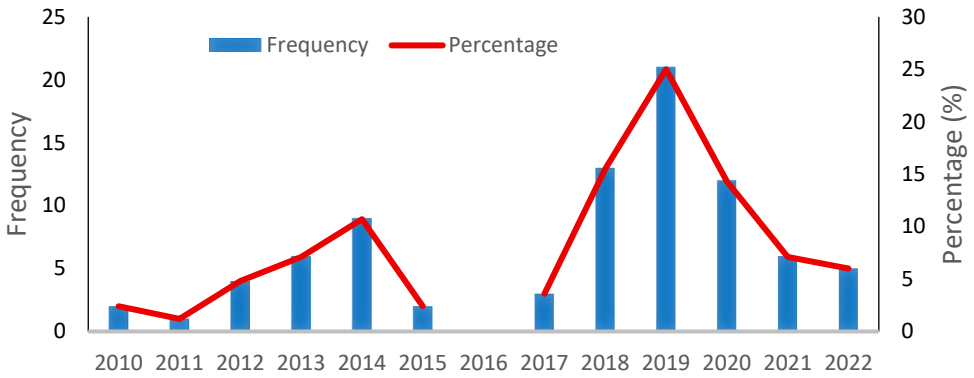


Fig.1 Annual frequency distribution of accidents

A noteworthy observation was the decline in accidents in 2021 and 2022 (6 and 5 cases, respectively) following the peak in 2019. Although this reduction may indicate the positive effect of control measures, it cannot be conclusively attributed to them, as decreased operational activity during the post-pandemic period or changes in the accident recording system may also have contributed to this decline.

3.1.2 Frequency distribution of accidents by accident type and location zone

Injury was the most common type of accident, with 41 cases (48.8%), followed by fire with 11 cases (13.1%) and traffic accidents with 9 cases (10.7%). Amputation, with 6 cases (7.1%), was among the high-severity accidents and occurred specifically in the processing line zone. Dog bites and needlestick injuries, each with 3 cases (3.6%), reflected the specific biological hazards characteristic of this sector. Falls from height, with 5 cases (6.0%), and myocardial infarction, with 2 cases (2.4%), were among the other types of recorded accidents (Fig. 2).

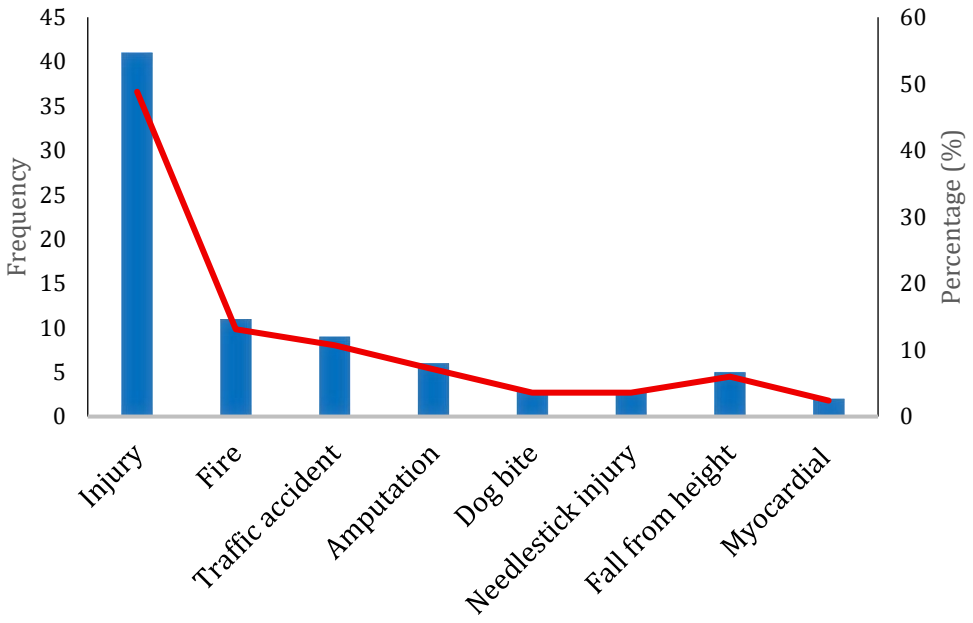
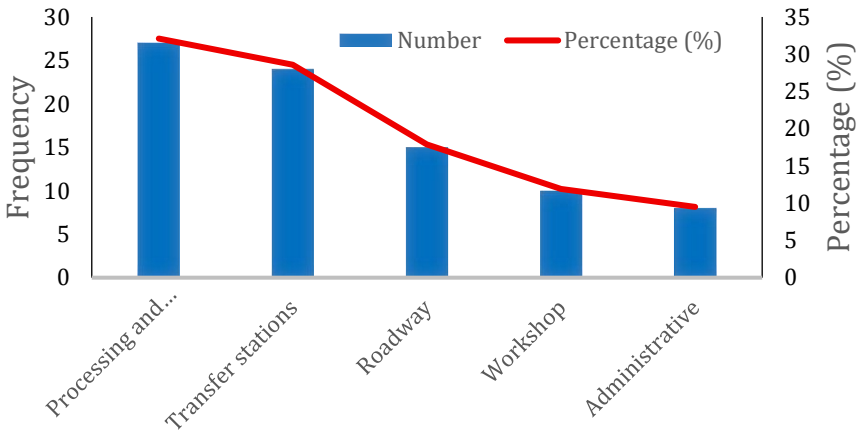


Fig. 2 Frequency distribution of accidents by accident type

A study by Abd El-Wahab et al. in Egypt demonstrated that workers in direct contact with waste were at greater risk of gastrointestinal, respiratory, and musculoskeletal disorders (Abd El-Wahab et al., 2014), This finding is comparable to the high proportion of injuries (48.8%) and the predominance of mechanical hazards observed in the present study. On the other hand, the meta-analysis by Tolera et al. (2024) reported a pooled prevalence of 36.49% for occupational injuries among sanitation workers, which is lower than the 48.8% observed in the present study (Tolera et al., 2024). This difference may be attributable to differences in services as well as in accident recording and reporting systems.

Based on the analysis of location of occurrence (Fig. 3), the processing and sorting line zone, with 27 cases (32.1%), and the transfer station zone, with 24 cases (28.6%), accounted for the highest accident frequencies. The roadway and maintenance workshop zones followed with 15 cases (17.9%) and 10 cases (11.9%), while the administrative sections had the smallest share, with 8 cases (9.5%).

Fig. 3 Frequency distribution of accidents by location zone

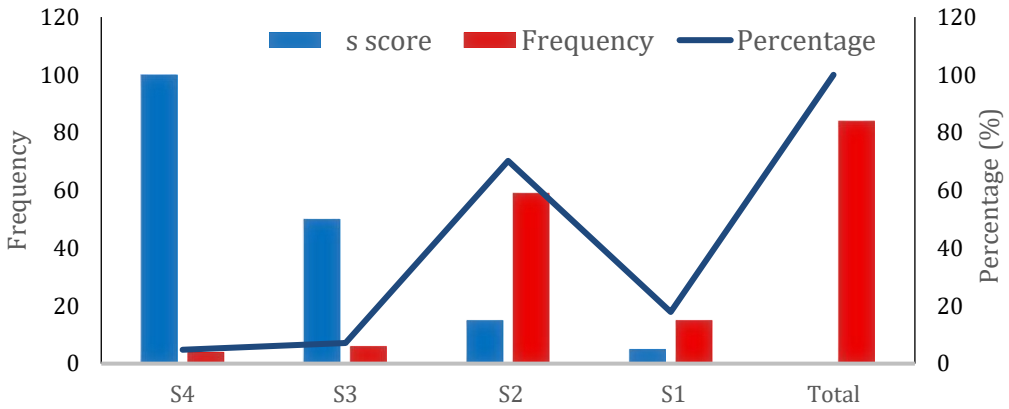


A comparison with previous studies reveals considerable consistency. In the study by Moussiopoulos and Kontogianni (2017), material recovery and composting facilities had the highest hazard levels (Moussiopoulos, 2017). This is consistent with our finding that the processing line zone accounted for 32.1% of all accidents and had an R value of 3600. The study by Moloudi et al. (2022) identified 485 HSE hazards, 38% of which were related to material recovery and composting facilities (Moloudi et al., 2022). Although that study focused on potential hazards, its findings are consistent with the present results.

3.1.3 Frequency distribution of accidents by injury severity

Based on the injury severity classification (Fig. 4), the highest frequency was associated with the serious level (S2), with 59 cases (70.2%). The minor level (S1) accounted for 15 cases (17.9%), the very serious level (S3) for 6 cases (7.1%), and the catastrophic level (S4) for 4 cases (4.8%). Although the S4 level had the lowest number of cases, the occurrence of four fatalities during the study period indicates the presence of severe hazards requiring immediate preventive action.

Fig. 4 Frequency distribution of accidents by injury severity



The proportion of severe accidents (S3 + S4) relative to all accidents was 11.9% (10 out of 84 cases). This high proportion underscores the necessity of devoting special attention to engineering control measures, since many severe accidents primarily arise from deficiencies in equipment design and inherent safety. The accident descriptions indicated that mechanical guards were absent and lockout/tagout procedures were not implemented in the affected zones, suggesting these as contributing factors. This highlights that investment in inherent equipment safety is among the most effective strategies for reducing severe accidents.

Table 8 Intersection Matrix of Location Zone and Injury Severity

Location Zone	S1 (Minor)	S2 (Serious)	S3 (Very serious)	S4 (Catastrophic)	Total
Z1 (Processing lines)	3	21	2	1	27
Z2 (Transfer stations)	5	17	1	1	24
Z3 (Workshop)	2	7	1	0	10
Z4 (Road)	3	9	1	2	15
Z5 (Other/Administrative)	2	5	1	0	8
Total	15	59	6	4	84

3.2 Intersection Analysis of Location Zone and Injury Severity

Based on Table 8, the processing line zone, with 2 cases of amputation (S3) and 1 fatality (S4), had the highest number of severe accidents (3 cases). The transfer station zone, despite its high accident frequency (24 cases), predominantly had S2-level injuries, with 1 amputation (S3) and 1 fatality (S4). The roadway zone, although having a lower frequency (15 cases), was critical because of 2 fatalities (S4). The administrative zone (Z5) recorded 1 amputation (S3) and had the lowest risk score.

3.3 Risk Assessment Results Using the Fine-Kinney Method

Based on the calculations, the processing line zone (Z1) and the transfer station zone (Z2), with a risk score of 3600, were classified at the very high level. This high score is primarily attributable to the combination of high probability, daily exposure, and high severity (fatality). The roadway zone (Z4), with a risk score of 900, was also classified at the very high level. Although the probability and exposure in this zone are lower, the high consequence severity due to 2 fatalities has substantially elevated the risk score. The maintenance workshop zone (Z3), with a risk score of 450, was also classified at the very high level due to the occurrence of one amputation. Finally, the administrative zone (Z5), with a risk score of 100, was identified at the medium level (Table 9).

Table 9 Fine-Kinney Risk Scores and Calculations

Zone	P	Description	E	Description	S	Description	R	Risk Level
Z1	6	Quite possible	6	Daily	100	One fatality	3600	Very high
Z2	6	Quite possible	6	Daily	100	One fatality	3600	Very high
Z3	3	Possible	3	Weekly	50	One amputation	450	Very high
Z4	3	Possible	3	Weekly	100	Two fatalities	900	Very high
Z5	1	Unlikely	2	Monthly	50	One amputation	100	Medium

One of the prominent findings is the high-risk score of the roadway zone (900). This finding is comparable to the study by Demir et al. (2024), which identified 68 risks at a hazardous waste disposal facility in Turkey using the Fine-Kinney method (Demir et al., 2024). In Demir et al.'s study, most risks were classified at the "notable" level, whereas in the present study, four zones were identified at the "very high" level. The study by Jalali et al. (2025), conducted in the same geographical area, utilized the JHA method to identify 8 risk factors for Top Turn machine operators, of which 2 were at the unacceptable risk level (Jalali et al., 2025).

3.3.1 Addressing the Subjectivity of the Fine-Kinney Method

The authors acknowledge that the Fine-Kinney method is inherently susceptible to subjectivity. To mitigate this limitation, several strategies were employed. First, the expert panel comprised three OHS specialists, and the mean of their ratings was used as the final P and E score for each zone, thereby reducing individual bias. Second, the range of expert opinions was examined; however, a formal inter-rater agreement index was not calculated, which is acknowledged as a limitation. Third, the P and E scores were calibrated against 12-year empirical accident data. Fourth, the 12-year accident dataset itself served as a validation mechanism: the risk scores derived from the Fine-Kinney method were cross-checked against the actual distribution of severe accidents across zones. For instance, the processing line zone, which received the highest risk score (3600), also recorded the highest number of severe accidents (3 cases), providing empirical validation. While these measures do not eliminate the inherent subjectivity of the method, they represent a practical and transparent approach to reducing bias.

3.4 Final Decision Matrix for Intervention Prioritization

The decision matrix was constructed using the following explicit rules:

1. Primary criterion: Zones were ranked by their Fine-Kinney risk score (R) in descending order.
2. Tie-breaking criterion: When two zones had the same R value (e.g., Z1 and Z2 both with R=3600), the number of severe accidents (S3+S4) was used as a tie-breaker, with higher numbers receiving higher priority.
3. Action priority categories (Table 10):

- Immediate and emergency: R > 400 (Very high risk)
- High (within 6 months): R = 200–400 (High risk)
- Medium (within 1 year): R = 70–200 (Medium risk)
- Low (periodic review): R < 70 (Low/Acceptable risk)

While severity is already incorporated into R through the S component, the number of severe accidents provides additional information about the cumulative burden of severe events. A sensitivity analysis excluding the number of severe accidents as a criterion showed that the priority ranking remains unchanged.

Table 10 Final Decision Matrix for Intervention Prioritization

Priority	Zone	Risk Score (R)	Risk Level	Number of Severe Accidents	Action Priority
1	Z1 (Processing lines)	3600	Very high	3	Immediate and emergency
2	Z2 (Transfer stations)	3600	Very high	2	Immediate and emergency
3	Z4 (Road)	900	Very high	3	Immediate and emergency
4	Z3 (Workshop)	450	Very high	1	Immediate and emergency
5	Z5 (Other/Administrative)	100	Medium	1	Medium (within 1 year)

Based on the above matrix, four zones processing lines, transfer stations, roadways, and workshop are placed under immediate and urgent priority. In the processing line zone, the most critical action is the installation of standard mechanical guards on rotating machinery and the implementation of a lockout/tagout (LOTO) system. In the transfer station zone, there is a need for fire detection and suppression systems as well as infectious waste management. In the roadway zone, infrastructure improvements and installation of warning signs are prioritized. In the workshop zone, machine guarding and LOTO implementation during maintenance are prioritized.

In the Addis Ababa study, non-use of personal protective equipment was associated with 2.62-fold increased odds of injury (Odo, 2012), which is consistent with the findings of the present study regarding the role of carelessness and lack of safety equipment in accident occurrence. Furthermore, the study by Abib et al. (2024) in Ethiopia demonstrated that factors such as pre-employment safety training, sleep problems, and temporary workers were significantly associated with occupational injury occurrence (Abib et al., 2024). This finding aligns with the content analysis of accidents in the present study, as in many accident descriptions, drowsiness and carelessness were cited as contributing factors in road accidents.

A deeper interpretation of the findings reveals that the high-risk score of the processing line zone (3600) primarily stems from the combination of three factors: high probability (daily exposure to machinery), continuous exposure (workers are constantly active alongside conveyor belts and screens), and high severity (fatality). Analysis of accident descriptions in this zone indicates that most accidents resulted from workers' hands or clothing becoming caught in machinery and contact with sharp and cutting objects. This pattern points to the absence of standard mechanical guards and the failure to implement lockout/tagout (LOTO) procedures during maintenance and repair. Similar findings have been reported among municipal solid waste workers in India, where occupational health problems are frequently associated with inadequate engineering controls and limited use of personal protective equipment (Chinthapatla & Tiwari, 2024). In other words, despite the existence of international standards for machinery safety, these standards appear not to have been fully implemented in the facility under study. Regarding the transfer station zone, which is classified at the very high level with a risk score of 3600, the high diversity of hazards (fire, dog bites, needlestick injuries, accidents involving machinery, and falls) indicates the multidimensional nature of hazards in this zone. This diversity necessitates a variety of control approaches (technical, behavioral, and administrative) and cannot be addressed through a single control measure alone.

Comparison of the findings of the present study with the research by Imam and Rafizul (2025) at the Rajbandh landfill site in Khulna, Bangladesh, reveals considerable similarities in the vulnerability patterns of waste workers in developing countries (Imam & Rafizul, 2025). In the Khulna study, all informal waste workers surveyed reported that they lacked personal protective equipment. In contrast, in the present study, despite the data being extracted from formal workers at a municipal waste management facility, 15 minor accidents (17.9%) were still recorded, indicating deficiencies in the supply and monitoring system for personal protective equipment use.

The findings of Akyen et al. (2025) demonstrated that informal waste pickers, due to financial constraints, are unable to afford personal protective equipment, thereby endangering their health (Akyen et al., 2025). These findings are considerably consistent with the results of the present study, in which 15 minor accidents (17.9%) were recorded, indicating deficiencies in the supply and monitoring system for personal protective equipment use. Overall, both studies emphasize that safety interventions in the waste industry are not achieved through engineering measures alone, but must also consider the economic, social, and cultural dimensions of workers to bring about sustainable behavioral change.

3.5 Summary of Key Findings

The managerial implications of this study extend beyond the identification of high-risk zones and culminate in the provision of an operational prioritization framework that assists managers in allocating their limited resources in the most optimal manner. Based on this framework, four zones—processing lines, transfer stations, roadways, and workshop—are placed under immediate and urgent priority. For the processing line zone, proposed measures include installation of standard mechanical guards on all screens and conveyor belts, implementation of a lockout/tagout system with practical training for all operators, and installation of emergency stops within easy reach of all workstations. For the transfer station zone, installation of advanced fire detection and suppression systems (such as FM200 gas-based systems), establishment of a management system for infectious and sharps waste, workplace environmental safety measures against stray dogs, and firefighting training are recommended. For the roadway zone, installation of speed bumps and warning signs, improvement of roadway lighting, driver fatigue management programmes, and periodic vehicle technical inspections are prioritised. For the workshop zone, machine guarding and LOTO implementation during maintenance activities are prioritized.

3.6 Limitations

This study has several limitations. Its retrospective design, deficiencies in some recorded accident data, and lack of complete data on demographic characteristics, exposure levels, and organizational factors should be acknowledged. Because detailed denominator data (e.g., number of workers per zone, worker-hours, number of operations, traffic volume)

were not available for all zones, accident frequency could not be converted into incidence rates. Zones with larger workforces or longer operating hours would be expected to have more recorded accidents; therefore, accident counts alone cannot establish that a zone has higher occupational risk. Thus, the risk scores presented in this study should be interpreted as relative prioritization indicators rather than absolute risk measures. Future studies should collect denominator data to enable more precise risk estimation.

The proposed prioritization framework may help managers and safety professionals identify high-risk zones, allocate resources effectively, and plan preventive interventions. If validated in other facilities, it could serve as a model for enhancing occupational risk management systems in the MSWM sector.

4. Conclusion

Certain operational zones of MSWM facilities—particularly processing lines, transfer stations, internal roadways, and maintenance workshops—are exposed to high levels of occupational risk. The fatalities and amputations observed among the reviewed accidents, coupled with the classification of these zones as very high risk by the Fine-Kinney method, underscore the urgent need for immediate, targeted control measures. The prioritization framework developed in this study integrates consequence severity, accident frequency, and risk level to determine intervention priorities and may serve as a practical tool for targeted allocation of limited safety resources. Controlling mechanical hazards and implementing lockout/tagout (LOTO) systems in processing lines, strengthening fire prevention and suppression systems at transfer stations, improving traffic safety on internal roadways, and implementing machine guarding in maintenance workshops are among the highest-priority control measures. Therefore, adopting a risk- and severity-based approach can direct intervention resources toward hazards with potentially more severe consequences for the workforce, rather than focusing solely on accident frequency.

Statements and Declarations

Ethical considerations

Ethical issues (Including plagiarism, informed consent, misconduct, data fabrication and/or falsification, double publication and/or submission, redundancy, etc.) have been completely observed by the author.

Data availability

Data will be made available on request.

Conflicts of interest

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

Author contribution

F. Jalali: Writing the main Draft; N. Ghobadi: Conceptualization; Writing – Review & Editing; and F. Ahmadi Zarajabad: Conceptualization, Methodology, Review, and Final Proof.

AI Use Declaration

During the preparation of this manuscript, the authors used artificial intelligence (AI) tools solely to assist in the design of the graphical abstract. The authors reviewed and approved the final graphical abstract and take full responsibility for the accuracy, integrity, and content of the manuscript.

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Microplastics and human health: A review of sources, exposure pathways, toxicological evidence, and risk assessment challenges

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ABSTRACT

Microplastics (MPs) are plastic particles smaller than 5 mm, detected in aquatic, terrestrial, and atmospheric environments, food, drinking water, and human biological samples, raising concerns about environmental quality and human health. This review examines sources, environmental distribution, human exposure pathways, toxicological evidence, analytical challenges, and health-risk assessment approaches. Ingestion and inhalation are considered the principal exposure pathways in human-health assessments, while evidence for dermal uptake remains less developed. MPs have been reported in human blood, placental tissue, gastrointestinal samples, and other biological matrices, indicating internal exposure; however, detection in human tissues should not be interpreted as evidence that MPs cause a particular disease. Experimental studies report oxidative stress, inflammation, cytotoxicity, immune responses, and reproductive or respiratory effects, but much evidence comes from cellular and animal models. Human epidemiological evidence remains limited and heterogeneous; a 2024 rapid systematic review identified only three human observational studies alongside 28 animal studies addressing reproductive, respiratory, and digestive outcomes. Major uncertainties arise from differences in polymer composition, particle size and shape, surface properties, environmental aging, additives, associated contaminants, sampling procedures, and analytical detection limits, restricting study comparability and complicating quantitative risk assessment. Future research should prioritize harmonized sampling and analytical methods, improved characterization of small MPs and nanoplastics, environmentally relevant exposure scenarios, longitudinal human studies, and integrated exposure-hazard frameworks. Current evidence supports continued environmental monitoring and source-control measures, while conclusions regarding specific human diseases should remain proportionate to available evidence.

Highlights

- Microplastics are widely detected in environmental, food, water, and human matrices.
- Ingestion and inhalation are the main exposure pathways in human-health assessments.
- Experimental studies show oxidative stress, inflammation, cytotoxicity, and immune effects.
- Limited human evidence and methodological heterogeneity hinder quantitative risk assessment.
- Harmonized analytical methods and longitudinal human studies are needed to reduce uncertainty.

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1. Introduction

Plastic pollution has generated a persistent environmental contamination problem because plastic materials can remain in the environment and undergo fragmentation into progressively smaller particles. Microplastics (MPs) are commonly defined as plastic particles smaller than 5 mm. They may be generated directly as small particles or formed

secondarily through the degradation and fragmentation of larger plastic materials. Their physical and chemical characteristics vary according to polymer type, size, shape, surface properties, and environmental history (Gowda et al., 2023; Thisakya & Dissanayake, 2025).

MPs have been reported in water, sediments, soil, atmospheric particles, food, and drinking water. Their occurrence in multiple environmental compartments creates several potential pathways for human exposure. A systematic review of food, drinking water and human samples found substantial variation in the polymer types and shapes reported across studies and showed that the particle-size range investigated differed considerably among environmental and human samples (Kuijer et al., 2010; Jayakrishnan et al., 2013).

The health implications of this exposure remain an active area of investigation. MPs have been detected in human biological samples, including blood and placental tissue, demonstrating that environmental exposure can result in internal contact with plastic particles. At the same time, the detection of MPs in a biological matrix does not establish that they produce a specific adverse health outcome (Akdogan & Guven, 2019; Basumatary et al., 2025).

The distinction between exposure, biological response and disease is particularly important because a substantial part of the current toxicological literature is based on experimental models. In a rapid systematic review, (Carvajal et al., 2026; Chartres et al., 2024) identified three human observational studies and 28 animal studies addressing reproductive, respiratory and digestive outcomes. The authors concluded that microplastic exposure was suspected to adversely affect sperm quality and digestive immunosuppression, while the available human evidence remained limited.

The objective of this review is to evaluate MPs from an environmental-health perspective by linking their sources and environmental occurrence with human exposure, biological effects, analytical limitations, and risk-assessment requirements. Particular attention is given to the distinction between established observations and hypotheses that still require confirmation (Christodoulou et al., 2026; Eberhard et al., 2024).

2. Definition and classification of microplastics

The 5-mm threshold is widely used to define MPs, although terminology and size boundaries vary among scientific studies. The distinction between microplastics and nanoplastics is particularly relevant because analytical methods that can identify larger MPs may not adequately characterize particles at the nanoscale (Akdogan & Guven, 2019; Epeslidou et al., 2026).

MPs are commonly divided according to their origin. Primary MPs are produced or used at microscopic dimensions, whereas secondary MPs result from the fragmentation of larger plastic materials. The environmental characteristics of secondary particles can differ from those of commercially produced particles because weathering changes their surface properties and physical structure. A review of micro- and nanoplastic toxicology noted that many experimental studies have relied on commercially available primary particles rather than particles that have undergone environmental weathering (Fitch et al., 2025; Garcia et al., 2026).

Particle shape is another relevant characteristic. Fragments, fibers, films, foams, and pellets have all been reported in environmental and biological samples. In the systematic review by (Arfin & Nakhate, 2026). fibers were frequently reported in human samples, while fragments and other morphologies were also observed in food and drinking-water studies (Guo et al., 2026). Table 1 summarizes the main characteristics used to describe MPs, including size, polymer type, shape, origin, surface condition, and associated chemicals.

Table 1 Main characteristics used to describe microplastics

Characteristic	Common categories	Relevance
Size	Micro-scale particles; smaller particles approaching the nanoscale	Influences transport, detection, and potential biological interactions
Polymer	PE, PP, PET, PS, PVC, PA and others	Influences density, chemical properties and environmental behavior
Shape	Fragment, fiber, film, foam, pellet	Affects transport, deposition and biological interaction
Origin	Primary or secondary	Reflects production source or environmental fragmentation
Surface condition	Pristine or weathered	Environmental aging can change surface characteristics
Associated chemicals	Additives and environmental contaminants	May modify the chemical exposure associated with particles

Compiled from the reviewed literature (14)

3. Sources and environmental pathways

The release of MPs can occur at different stages of plastic production, use, disposal, and environmental degradation. Larger plastic materials can fragment through mechanical abrasion and weathering, while small particles can also be released directly from products and human activities. Synthetic textiles, plastic materials, urban dust, and industrial activities are among the sources considered in studies of airborne MPs (Hore et al., 2026; Hunt et al., 2024).

Once released, MPs can move among environmental compartments. Water can transport particles from terrestrial sources toward rivers, lakes and marine environments. Atmospheric transport can redistribute particles through deposition, while soils may receive MPs through wastewater-derived materials, agricultural practices, atmospheric deposition, and other pathways. A 2025 review of MP and nanoplastic dynamics emphasized that environmental transport occurs across aquatic and terrestrial systems and that interactions with environmental contaminants can further complicate their behavior (Hore et al., 2026; İçöz et al., 2026). Fig. 1 summarizes the sources, environmental transport routes, and human exposure pathways of MPs.

Airborne MPs have received particular attention because inhalation provides a direct route into the respiratory system

(Jenner et al., 2022). reviewed indoor and outdoor airborne MP and nanoplastic studies and identified major data gaps in exposure measurement and standardization (Johari et al., 2026).

A 2026 review likewise described airborne MPs as particles associated with sources such as textile fibers, urban dust and industrial emissions and emphasized that their atmospheric transport and deposition remain insufficiently characterized (Knol et al., 2010).

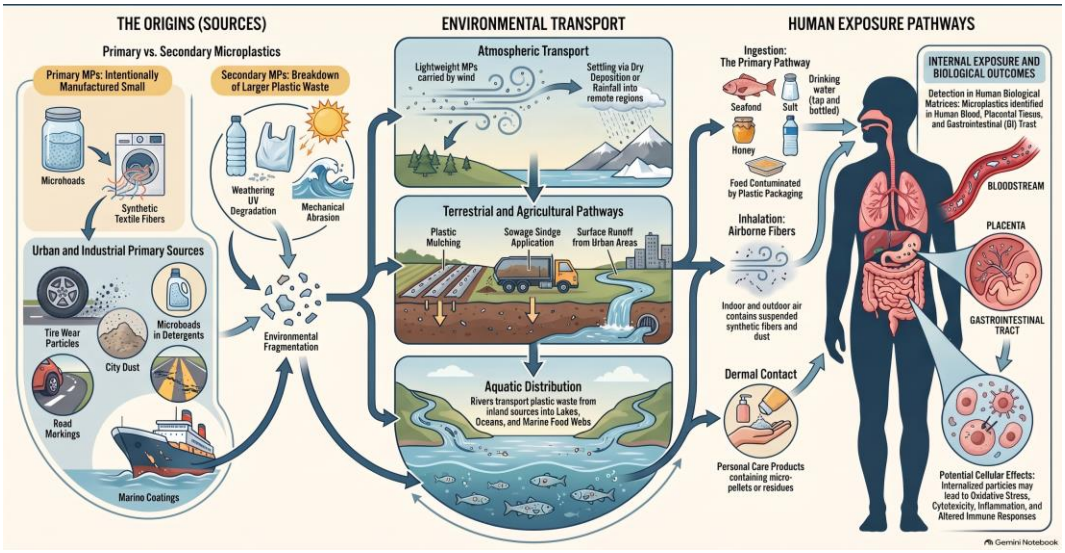


Fig. 1 Sources, environmental transport and human exposure pathways

4. Occurrence in environmental matrices

4.1 Water and drinking water

Drinking water is an important exposure medium because MPs can enter water supplies from environmental sources and potentially persist through parts of the water-treatment and distribution system. A 2024 review of drinking-water studies reported wide variation in MP concentrations among investigations and identified methodological differences as an important reason for the lack of direct comparability (Kumar et al., 2021; Kwon et al., 2020).

The size range considered in monitoring studies is particularly important. Smaller particles can be missed when filtration or identification procedures have relatively high detection limits. The systematic analysis by (Arfin & Nakhate, 2026; Lamas et al., 2026). showed that studies of human samples and drinking water often investigated smaller particles than studies of some food matrices.

4.2 Food

Food can contain MPs originating from environmental contamination, processing, handling, and packaging. Reported particles include several polymer types and morphologies, but quantitative comparisons among food groups remain difficult because analytical procedures differ (Arfin & Nakhate, 2026). Found differences in the types and shapes of MPs reported in food and human samples and emphasized the influence of particle-size detection limits (Li et al., 2024).

The presence of MPs in food is relevant to exposure assessment, but concentration data alone do not provide a direct estimate of absorbed dose. The fate of particles after ingestion depends on particle properties, gastrointestinal conditions, and biological interactions (Basumatary et al., 2025). Reviewed evidence concerning ingestion, distribution, and possible translocation within the human body (McLean et al., 2026).

4.3 Soil and terrestrial systems

Agricultural soils can receive MPs through several pathways, including plastic agricultural materials, wastewater-derived inputs, irrigation and atmospheric deposition. Once present in soil, particles can interact with soil minerals, organic matter, microorganisms, and other environmental constituents. A 2025 review of MP impacts on ecosystems emphasized that MPs can alter soil properties and microbial processes, although the magnitude and direction of these effects depend on particle characteristics and environmental conditions (Hore et al., 2026; Mohasin et al., 2025).

The transfer of MPs from soil to crops and subsequently to humans is an area of continuing investigation. Evidence for environmental occurrence does not automatically establish a quantitatively important contribution to human exposure, because transfer depends on particle size, plant species, soil conditions and other factors (Njema et al., 2026; Novillo-Del Álamo et al., 2026).

4.4 Atmospheric environment

Airborne MPs can originate from synthetic textiles, abrasion, urban dust and industrial sources. They may remain suspended or be deposited on indoor and outdoor surfaces (Jenner et al., 2022). concluded that substantial gaps remain in the quantitative characterization of human inhalation exposure (Parveen et al., 2026). The atmospheric pathway has two implications. First, inhalation can provide direct exposure to humans. Second, atmospheric deposition can redistribute particles into soil and water, connecting the atmospheric compartment with other environmental systems (Knol et al., 2010; Parveen et al., 2026).

5. Human exposure pathways

5.1 Ingestion

Ingestion is one of the principal pathways considered in human MP exposure assessments. Contaminated drinking water, food, and deposited particles can contribute to ingestion. However, exposure estimates vary according to consumption patterns, particle concentrations, particle-size ranges and analytical recovery (Paul et al., 2020; Piroozfar et al., 2026).

Li et al. (2024) reviewed human exposure through ingestion, inhalation and skin contact and discussed the subsequent distribution and possible translocation of MPs (Akdogan & Guven, 2019).

5.2 Inhalation

Airborne MPs can enter the respiratory tract. The available literature includes both indoor and outdoor environments, but estimates of inhaled dose remain uncertain because air-sampling methods, sampling duration, and particle-size detection vary substantially (Jenner et al., 2022). Standardized exposure measurement has been identified as a major research need (Płuciennik et al., 2024).

Evidence that airborne MPs can reach human respiratory tissues has also been reported (Akdogan & Guven, 2019; Rajput et al., 2026). Summarized evidence concerning inhalation and distribution, while earlier human tissue research has reported airborne MPs in lung tissue. These observations support the plausibility of internal respiratory exposure but do not by themselves establish adverse clinical outcomes (Rehman et al., 2026; Saha et al., 2026).

5.3 Dermal exposure

Dermal contact has been considered as a possible exposure route, particularly during contact with contaminated water or products containing plastic particles. However, compared with ingestion and inhalation, the evidence base for systemic uptake through intact human skin is less developed (Akdogan & Guven, 2019; Shao et al., 2026). included skin contact among the exposure pathways considered in their review but also emphasized the uncertainties surrounding MP transport and biological effects (Shiry et al., 2026). Table 2 summarizes the main human exposure pathways considered in this review, including ingestion, inhalation, dermal contact, and internal exposure.

Table 2 Main human exposure pathways (Li et al., 2024; Vdovchenko & Resmini, 2024)

Exposure pathway	Potential sources	Current evidence	Main uncertainty
Ingestion	Food, drinking water, environmental particles	Well documented as an exposure pathway	Absorbed dose and contribution of different sources
Inhalation	Indoor and outdoor airborne particles	Increasing evidence	Long-term inhaled dose and small-particle exposure
Dermal contact	Water and consumer products	Possible exposure route	Systemic uptake through intact skin
Internal exposure	Blood, placenta and other biological matrices	MPs have been detected in human samples	Biological significance of detected particles

6. Microplastics in the human body

The detection of MPs in human biological samples has provided direct evidence that exposure can result in internal contact (Akdogan & Guven, 2019; Vdovchenko & Resmini, 2024). reviewed reports of MPs in different tissues and discussed mechanisms that may permit particle translocation, with particle size identified as one factor that can influence biological transport.

The polymer types and particle morphologies reported in human samples are not uniform (Arfin & Nakhate, 2026). showed that polyester, polypropylene, polyethylene, polyamide and other polymers have been reported across food and human samples, while fibers were frequently observed. The review also emphasized that the detection limit strongly affects which particles are identified (Walker-Franklin et al., 2026).

Pregnancy-related samples have received particular attention. Studies have reported MPs in maternal and fetal-associated biological compartments. Such findings indicate exposure during pregnancy but should not be interpreted as evidence that the detected particles caused adverse pregnancy or developmental outcomes unless such associations are independently demonstrated (Wang et al., 2026; Xia et al., 2026).

The interpretation of human tissue studies requires particular attention to contamination control. Because environmental MPs are ubiquitous and fibers can be introduced during sample collection and laboratory processing, appropriate procedural blanks and contamination-control measures are essential (Xiang et al., 2026).

7. Toxicological mechanisms

7.1 Oxidative stress

Oxidative stress is among the biological responses repeatedly investigated in experimental MP studies. The proposed mechanism involves an imbalance between reactive oxygen species and antioxidant defenses. Experimental evidence supports the possibility that particle characteristics and exposure conditions influence oxidative responses (Fitch et al., 2025). reviewed toxicological evidence and emphasized that particle physicochemical characteristics influence MP bioactivity (Xu et al., 2026).

Oxidative stress findings from cells or animals should not be interpreted as evidence that environmental MP exposure causes oxidative-stress-related disease in humans. Human studies capable of linking measured internal MP exposure with validated oxidative-stress biomarkers remain limited (Yang et al., 2024).

7.2 Inflammation and immune responses

Inflammatory responses have been reported in experimental systems exposed to MPs. These responses are biologically relevant because persistent inflammatory signaling can influence tissue function. However, the magnitude of inflammation depends on particle characteristics, exposure concentration and experimental conditions (Yano et al., 2020).

The 2024 rapid systematic review by Chartres et al. separated evidence from human and animal studies and concluded that human evidence remains limited (Carvajal et al., 2026).

7.3 Cellular injury and genotoxic responses

Experimental studies have reported cytotoxic and genotoxic responses under selected exposure conditions. These effects may result from direct particle-cell interactions, oxidative stress or associated chemical constituents. However, the relevance of experimental doses to environmental human exposure is often uncertain (Zhang et al., 2024).

A major issue is that MPs are not chemically identical across studies. Polymer type, additives, surface oxidation and environmental contaminants can alter biological responses. Therefore, a toxicity result obtained with one type of particle should not automatically be generalized to all MPs (Kumar et al., 2021).

7.4 Reproductive effects

Reproductive toxicity has been investigated extensively in experimental models. The rapid systematic review by Chartres et al. (2024) included 11 animal studies addressing reproductive outcomes and two human observational studies. The review concluded that the overall body of evidence for sperm quality supported suspicion of adverse effects, while recognizing the limitations of the human evidence (Carvajal et al., 2026).

A separate systematic review focused specifically on human reproductive outcomes and examined the available evidence concerning MP exposure and reproductive health concluded that the evidence base was limited and that further research was required to clarify the relationship between exposure and reproductive outcomes (Basumatary et al., 2025).

7.5 Respiratory effects

The respiratory tract is a direct target of inhaled airborne MPs. Experimental studies have reported inflammatory and cellular responses, while human studies have provided evidence of exposure. The main unresolved question is whether environmental inhalation produces doses sufficient to cause clinically meaningful chronic disease (Eberhard et al., 2024).

Eberhard et al. (2024) emphasized that quantitative inhalation exposure data are still insufficient for reliable health-risk characterization (Jenner et al., 2022).

7.6 Gastrointestinal effects

Ingested MPs initially interact with the gastrointestinal tract. Experimental evidence has reported inflammatory and immune responses and changes in intestinal function. In the rapid systematic review by Chartres et al. (2024), the evidence concerning digestive immunosuppression was considered sufficient to suspect adverse effects, although much of the underlying evidence was from animal studies (Carvajal et al., 2026; Chartres et al., 2024). Table 3 summarizes the toxicological endpoints reported in experimental and human studies and the current interpretation of the evidence for each endpoint.

2.5.2 Determination of Probability (P)

The probability score was determined by the expert panel based on the observed frequency of accidents in each zone over the 12-year study period, combined with the operational nature of the zone (Szwed et al., 2006). It is important to clarify that accident frequency was not equated with probability; rather, it served as one input among several. The following quantitative criteria (Table 3) were used to translate accident frequency into the categorical P score:

Table 3 Toxicological effects reported in experimental and human research (Chartres et al., 2024; Yang et al., 2024)

Endpoint	Main evidence base	Current interpretation
Oxidative stress	Mainly cellular and animal studies	Biological mechanism supported experimentally
Inflammation	Cellular and animal studies; limited human biomarker evidence	Plausible mechanism; human causal evidence limited
Cytotoxicity	Mainly experimental studies	Depends on particle and exposure conditions
Genotoxicity	Mainly cellular and animal studies	Requires confirmation under environmentally relevant exposure
Reproductive effects	Animal studies and limited human observational studies	Human evidence remains limited
Respiratory effects	Animal, cellular and limited human studies	Exposure is plausible; chronic health effects uncertain
Gastrointestinal effects	Mainly animal studies, with limited human evidence	Some adverse effects are suspected
Endocrine effects	Experimental and emerging human evidence	Requires further human studies
Chronic disease	Limited and inconsistent human evidence	Causality not established

8. Microplastics as carriers of other contaminants

MP surfaces can interact with chemicals present in environmental matrices. The extent and nature of these interactions depend on polymer type, surface properties, environmental conditions and contaminant chemistry. Weathering can modify particle surfaces and therefore change their interaction with surrounding substances.

This issue is relevant to health assessment because exposure to an environmentally aged MP may involve both particle-related effects and exposure to chemicals associated with the particle. A 2026 review of airborne MPs described their potential to interact with contaminants such as heavy metals and persistent organic pollutants and emphasized their role as mobile environmental particles (Knol et al., 2010).

However, the statement that MPs universally increase the toxicity of co-occurring contaminants would be too broad. The direction and magnitude of combined effects depend on particle properties, contaminant identity, exposure conditions, and biological system. Risk assessment should therefore evaluate these interactions experimentally rather than assuming that adsorption automatically produces greater toxicity (Basumatary et al., 2025).

9. Analytical methods for microplastic identification

Reliable exposure assessment requires analytical methods that can detect and identify MPs with sufficient sensitivity and specificity. Microscopy can provide information on particle number, size, and morphology, while spectroscopic and thermal methods provide chemical information (Vdovchenko & Resmini, 2024; Wang et al., 2026).

Fourier-transform infrared spectroscopy (FTIR) and Raman spectroscopy are widely used for polymer identification. Thermal analytical approaches such as pyrolysis-gas chromatography/mass spectrometry provide chemical information based on polymer decomposition products. These techniques answer different analytical questions and cannot always be considered interchangeable (Vdovchenko & Resmini, 2024; Wang et al., 2026). Fig. 2 presents a general analytical workflow for microplastic determination, from sample preparation to detection and polymer identification.

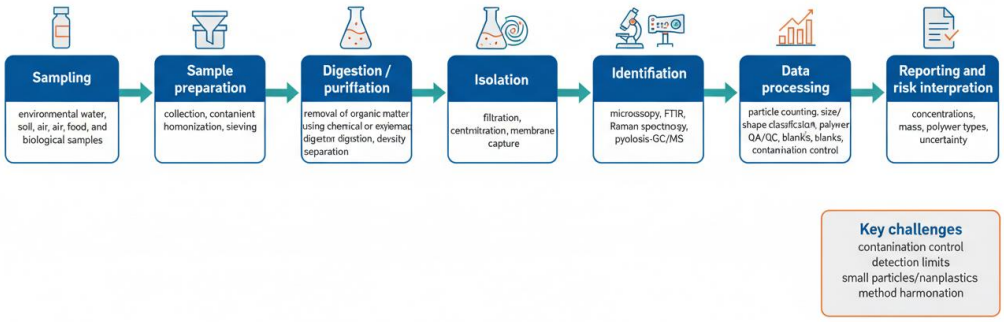


Fig. 2 General analytical workflow for microplastic determination

Particle-size detection is one of the most important limitations (Arfin & Nakhate, 2026). showed that the analytical detection limit strongly influences the size distribution reported in the literature. Studies examining human samples tended to include smaller particles than several studies of food matrices.

The analytical workflow varies with sample type and research objective. No single method currently provides complete characterization of all relevant particle sizes and polymer classes (Vdovchenko & Resmini, 2024; Yang et al., 2024).

10. Quality assurance and contamination control

MP research is particularly vulnerable to laboratory contamination because plastic materials are common in laboratory environments and airborne fibers can enter samples. Contamination can alter particle counts and morphology distributions, especially when the target concentration is low.

Reliable studies therefore require procedural blanks, appropriate field blanks where relevant, controlled sample handling and transparent reporting of contamination-control procedures. These measures are especially important for fiber analysis and for biological samples in which the expected particle concentration may be low (Vdovchenko & Resmini, 2024; Yang et al., 2024).

Differences in quality-control procedures can contribute to variation between studies. For this reason, a numerical concentration should not be interpreted independently of the analytical method, particle-size range, recovery rate and blank correction (Vdovchenko & Resmini, 2024; Yang et al., 2024; Vogel et al., 2024).

11. Human-health risk assessment

Health-risk assessment for MPs presents a different problem from risk assessment for a single chemical substance. MPs are heterogeneous particles that vary in polymer type, size, shape, surface properties, additives, and environmental aging. Consequently, a single concentration value does not fully characterize the exposure (Li et al., 2024; Yang et al., 2023; Vogel et al., 2024).

Vogel et al. (2024) proposed a framework for assessing the risks of micro- and nanoplastic particles to human health. They emphasized the need to consider particle properties, exposure, and hazard together (Vogel et al., 2024).

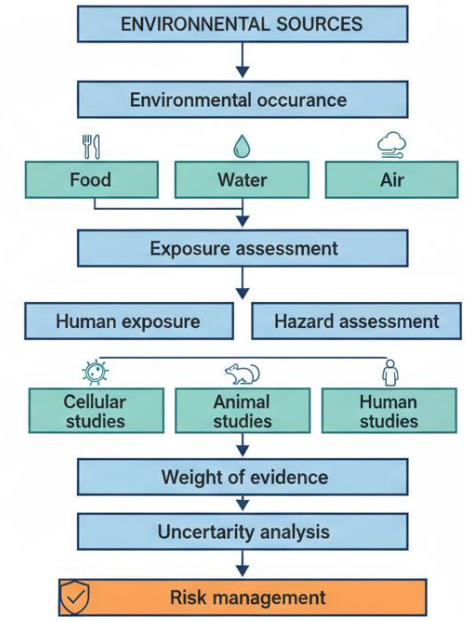


Fig. 3 Conceptual framework for microplastic health-risk assessment

The first component of risk assessment is exposure characterization. This requires information on environmental concentrations, particle size, polymer composition, exposure route, and exposure duration. The second component is hazard characterization, which requires dose-response information and biologically relevant endpoints. Risk characterization then integrates the two while considering uncertainties (Vogel et al., 2024; Yang et al., 2023). Fig. 3 presents a conceptual framework for microplastic health-risk assessment that integrates exposure characterization, hazard characterization,

and risk characterization. In addition, Table 4 summarizes the main components of human-health risk assessment and the principal uncertainties associated with each component.

Table 4 Main components and uncertainties in human-health risk assessment (Vogel et al., 2024)

Component	Required information	Main uncertainty
Exposure assessment	Concentration, size, shape, polymer and exposure route	Inconsistent sampling and reporting
Internal dose	MP concentration in biological matrices	Limited longitudinal measurements
Hazard identification	Biological effects associated with defined particle properties	Heterogeneous experimental conditions
Dose-response relationship	Exposure level and biological response	Limited human dose-response data
Population assessment	Age, exposure patterns and susceptibility	Limited population-specific evidence
Risk characterization	Integration of exposure and hazard	Uncertainty in external-to-internal dose relationships
Uncertainty analysis	Analytical and biological sources of uncertainty	Often incompletely quantified
Risk management	Source reduction and exposure control	Effectiveness differs among sources

12. Human evidence versus experimental evidence

One of the most important issues in MP research is the difference between evidence obtained from humans and evidence obtained from experimental systems (Chartres et al., 2024; Yang et al., 2023).

Animal and cellular studies are useful for identifying possible mechanisms and biological endpoints. They can help determine whether a particle can interact with a biological system and under what experimental conditions a response occurs. They cannot, by themselves, establish that the same effect occurs in environmentally exposed humans (Chartres et al., 2024; Li et al., 2024; Yang et al., 2023).

The rapid systematic review by Chartres et al. (2024) provides a clear example. The review included only three human observational studies compared with 28 animal studies. The evidence was therefore much larger for experimental models than for humans (Eberhard et al., 2024).

More recent human-focused evidence continues to show the same problem. A systematic review of human studies published in 2025 reported associations between MP exposure and inflammatory, oxidative-stress and endocrine-related biomarkers but emphasized that causal relationships with specific diseases remained unestablished (Ririe et al., 2025).

Therefore, statements such as “microplastics cause cancer,” “microplastics cause cardiovascular disease,” or “microplastics cause infertility” would exceed the evidence available for most of these outcomes. A scientifically appropriate formulation is that MPs have been associated with biological changes in some human studies and have produced specific adverse effects in experimental models, while causal relationships with most clinical diseases remain unresolved (Chartres et al., 2024; Ririe et al., 2025; Yang et al., 2023).

13. Environmental and public-health management

The management of MP pollution should begin with source reduction. Preventing plastic materials from entering the environment reduces the opportunity for fragmentation and secondary MP formation. Measures addressing plastic waste management, industrial losses, synthetic fibers, and abrasion-derived particles can therefore reduce environmental inputs (Basumatary et al., 2025; Yang et al., 2023).

Water and wastewater treatment can also influence MP concentrations, but removal efficiency varies according to treatment configuration and particle characteristics. Monitoring should therefore consider the entire pathway from source water through treatment and distribution rather than treating the final concentration as independent of upstream conditions (Vdovchenko & Resmini, 2024; Yang et al., 2024).

Airborne exposure requires a different set of controls. Because indoor environments can contain synthetic fibers and other plastic-derived particles, source reduction, ventilation and appropriate filtration may reduce exposure, although the quantitative effectiveness of individual interventions requires further evaluation (Eberhard et al., 2024; Saha et al., 2025).

The environmental-health perspective also requires separation of pollution control from disease-prevention claims. Reducing MP emissions is justified because it reduces persistent plastic contamination. Demonstrating that a specific control measure prevents a specific human disease requires epidemiological evidence that is currently limited (Basumatary et al., 2025; Yang et al., 2023; Chartres et al., 2024; Ririe et al., 2025).

14. Research gaps and priorities

14.1 Standardization of analytical methods

Different studies use different filters, digestion procedures, identification methods, detection limits and reporting units. Standardized methods are needed to make results comparable (Vdovchenko & Resmini, 2024; Yang et al., 2024).

14.2 Detection of smaller particles

Particles at the lower end of the microplastic size range and nanoplastics remain difficult to quantify. Their exclusion from routine monitoring may lead to underestimation of exposure (Chartres et al., 2024). emphasized the effect of detection limits on reported particle distributions.

14.3 Environmentally relevant particles

Many experimental studies use commercially produced particles with relatively uniform characteristics. Environmental particles may be weathered, irregular and chemically different. Toxicological studies using environmentally representative particles would improve the relevance of experimental evidence (Ririe et al., 2025).

14.4 Longitudinal human studies

Cross-sectional studies can identify associations but are limited in their ability to establish temporal relationships. Longitudinal cohorts that measure external exposure, internal

exposure and health outcomes would provide stronger evidence (Chartres et al., 2024; Ririe et al., 2025).

14.5 Vulnerable populations

Pregnancy, infancy, childhood, and occupational exposure require further investigation. Early-life exposure is of particular interest because developing biological systems may respond differently to environmental stressors. Research in this area should combine exposure measurements with appropriate developmental outcomes rather than relying solely on experimental toxicity (Chartres et al., 2024; Eberhard et al., 2024; Hunt et al., 2024; Li et al., 2024; Ririe et al., 2025).

14.6 Integrated risk assessment

Environmental monitoring, human exposure assessment and toxicology are often conducted separately. A more useful framework would connect these components so that measured environmental concentrations can be related to realistic human doses and then evaluated against experimentally and clinically relevant hazard information (Chartres et al., 2024; Li et al., 2024; Vogel et al., 2024; Yang et al., 2023).

Fig. 4 illustrates the proposed research pathway for integrating environmental monitoring, human exposure assessment, and toxicological evidence within an environmental-health assessment framework.

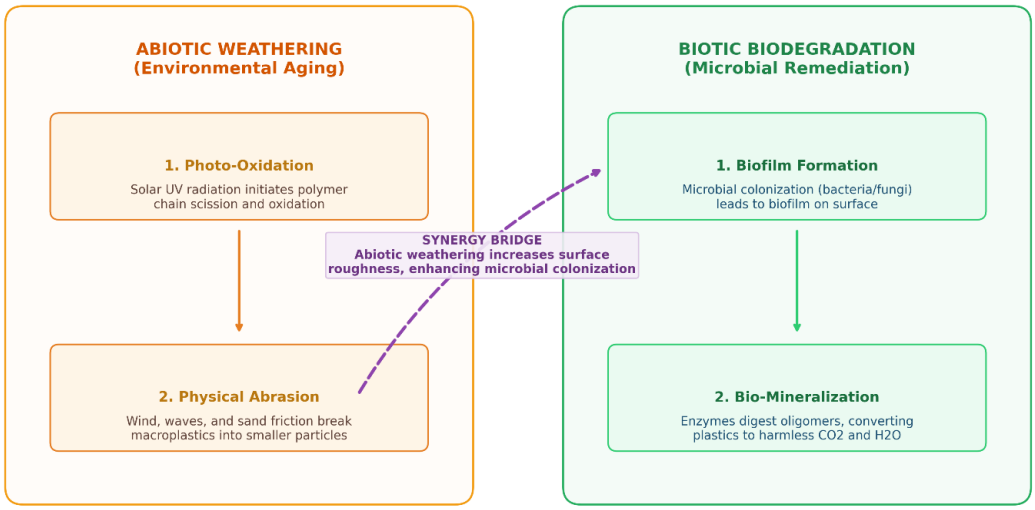


Fig. 4 Proposed research pathway for environmental-health assessment

15. Proposed priorities for future studies

Future MP research would benefit from a shift from isolated concentration measurements toward integrated exposure assessment. Environmental studies should report particle size, shape, and polymer composition together with concentration and analytical recovery. Human studies should report the analytical detection limit and contamination-control procedures in sufficient detail to permit comparison (Li et al., 2024; Vdovchenko & Resmini, 2024; Vogel et al., 2024; Yang et al., 2024).

Toxicological studies should also report particle characteristics in detail. Polymer identity alone is not enough to reproduce an exposure because particles with the same polymer may differ in size, morphology, surface condition and environmental history (Li et al., 2024; Vogel et al., 2024; Yang et al., 2023).

Human epidemiological studies should prioritize measured exposure rather than relying only on assumed environmental concentrations. Where biological samples are used, repeated measurements would help distinguish transient exposure from persistent internal presence (Li et al., 2024; Ririe et al., 2025; Vogel et al., 2024).

Finally, systematic reviews should maintain a clear distinction between human observations, animal evidence and mechanistic findings. This separation makes it possible to identify which conclusions are established and which remain hypotheses (Chartres et al., 2024; Ririe et al., 2025; Vogel et al., 2024; Yang et al., 2023).

16. Conclusions

Microplastics are widely distributed environmental contaminants that can enter human exposure pathways through water, food and air. Their detection in human biological samples demonstrates internal exposure, but detection alone does not establish that MPs cause a particular disease. Experimental evidence indicates that MPs can produce oxidative stress, inflammatory responses, cellular effects and reproductive, respiratory or gastrointestinal alterations under specific conditions. Human evidence is considerably more limited. The 2024 rapid systematic review by Chartres et al. illustrates this difference, with three human observational studies compared with 28 animal studies for the selected health outcomes. The reliability of future risk assessments will depend on better exposure measurements, standardized analytical procedures, and improved characterization of small MPs and nanoplastics. Environmental aging, polymer composition, particle morphology and associated chemicals should be considered when interpreting toxicological results. Current evidence supports continued monitoring and reduction of MP emissions. It also supports further investigation of possible human-health effects. Claims concerning specific chronic diseases should, however, remain limited to what human evidence can currently support.

Statements and Declarations

Ethical considerations

Ethical issues (Including plagiarism, informed consent, misconduct, data fabrication and/or falsification, double publication and/or submission, redundancy, etc.) have been completely observed by the author.

Data availability

Data will be made available on request.

Conflicts of interest

The author declares that he has no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Author contribution

M. Rezaee: Writing the main Draft, Writing – Review & Editing, Conceptualization, Methodology, Review, and Final Proof.

AI Use Declaration

During the preparation of this manuscript, the authors used artificial intelligence (AI) tools solely to assist in the design of the graphical abstract. The authors reviewed and approved the final graphical abstract and take full responsibility for the accuracy, integrity, and content of the manuscript.

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Assessment of medical laboratory waste management in Bam City in 2025

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ABSTRACT

Effective adherence to medical waste management protocols in clinical diagnostic laboratories is crucial for safeguarding public health and preventing the transmission of pathogens and environmental pollution. This descriptive cross-sectional study, conducted in 2025, evaluated waste management practices across all nine active clinical laboratories in Bam City, Iran, using a census method. Data were collected through direct on-site observation, structured interviews with laboratory technical supervisors, and a standardized checklist approved by the Ministry of Health and Medical Education. Daily and monthly waste quantities by category were also obtained from laboratory managers, with analysis performed using Microsoft Excel. Findings revealed an average waste generation of 0.19 kg per patient per day, with the nine laboratories collectively producing 106.48 kg/day and 2,875 kg/month. Compliance with national regulations varied significantly across domains: the highest adherence was recorded in waste collection and segregation (82.54%), whereas the lowest rates were observed in decontamination procedures (66.67%) and training/supervision (59.26%). The overall mean compliance rate across all laboratories was 71.56%, indicating moderate performance with considerable room for improvement. To enhance waste management standards, policymakers should implement continuous professional training programs, provide laboratories with standardized decontamination equipment, establish proper temporary storage infrastructure, and enforce regular supervisory audits by environmental health experts. Adopting a systemic, integrated approach throughout the entire waste management cycle—from generation to final disposal—is critical to ensuring safety and regulatory compliance.

Highlights

- Average medical lab waste generation was 0.19 kg per patient per day in Bam City labs.
- Nine labs produced 106.48 kg/day and 2,875 kg/month of waste in total combined.
- Waste collection and segregation compliance was highest, reaching 82.54% across all labs.
- Decontamination procedures had the lowest compliance rate at just 66.67% of required steps.
- Overall mean compliance was 71.56%, showing moderate performance with room to improve.

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1. Introduction

Amalgam has been the most commonly used dental filling material for over 200 years (Tibau & Grube, 2022). One of its main components, mercury, is of particular concern due to its potential adverse effects on humans and the environment (Joy & Qureshi, 2020). It is estimated that the annual consumption of mercury for dental applications is 3-4% worldwide (approximately 300 metric tons of mercury) (Chin et al., 2000; Khwaja et al., 2014). Although the use of amalgam as a restorative material has recently declined, amalgam restorations and their release into the environment continue to be a widely used dental restorative material. According to recent studies, dental clinics are responsible for a significant amount (10–70%) of daily mercury entering the environment through wastewater treatment plants (Jamil et al., 2016; Molina et al., 2014).

Mercury may bioaccumulate in fish and other organisms and therefore can impose an environmental mercury burden on the entire food chain (Gupta & Yadav, 2024; Qu et al., 2022). Among the groups directly exposed to mercury are dentists and their patients, who have been reported to have significantly increased plasma mercury concentrations compared with controls (Manyani et al., 2021; Warwick et al., 2019). On the other hand, people and the environment are indirectly exposed to this element through mercury emissions from waste incinerators and mercury in wastewater from dental clinics and households (Cheng & Hu, 2012). Several studies have shown that mercury exposure may lead to several health complications, such as disruption of the developing central nervous system, pulmonary and nephritic damage, and impaired osmoregulation functions (Fernandes Azevedo et al., 2012). These complications are usually attributed to the strong affinity of mercury for sulfur and sulfhydryl groups within living organisms (Houston, 2014).

Increased knowledge about the risk of toxic effects from anthropogenic accumulation of mercury in ecosystems has led to increased pressure to reduce the discharge of mercury waste into the environment. As a result, the problem of mercury waste in dental clinics has received increasing attention, and restrictions on the transport and disposal of contaminated waste have been imposed in several countries. Studies show that mercury emissions from dental clinics can be reduced by improved waste disposal system design, the use of high-pressure water cleaning, and frequent replacement of amalgam separators and filters (Eshrati et al., 2024; Musliu et al., 2021). For example, it was reported that the use of mercury separators reduced the mercury content in the wastewater of some dental clinics from 270 mg mercury per dentist per day to only 35 mg mercury per dentist per day. As a result, many countries such as Switzerland, Germany, Sweden, and Denmark have introduced mandatory installation of amalgam separators in dental clinics (Arenholt-Bindslev and Larsen 1996; Arenholt-Bindslev 1998). However, the use of these separators in dental clinics in Iran is still uncommon. Furthermore, despite the fact that a large part of the recycled wastewater is reused for irrigation, it is necessary to investigate the amount of mercury discharged into the wastewater collection system.

Therefore, this study aimed to quantitatively assess the mercury load in wastewater from some dental clinics. The results of this project will contribute to the Ministry of Health's efforts to reduce mercury concentrations in wastewater by providing new measures for the handling and disposal of mercury-containing waste from dental clinics.

2. Materials and Methods

An atomic absorption spectrometer (Spectra AA 220 FS, Varian) equipped with a vapor generation accessory (VGA-77, Varian) and a T-shaped quartz absorption cell was used for mercury determination.

2.1 Reagents and Solutions

All chemicals were of analytical grade. All water used was obtained from a Milli-Q reagent system (EC= 18.2 MΩ cm, Millipore, Bedford, MA, USA). All plastic and glassware were soaked in 4 M nitric acid for at least 12 h, washed with distilled water, and finally rinsed with Milli-Q water before use.

Nitric acid (68.0-70.0%) and sodium chloride (99.5%), sulfuric acid (GPR), hydrochloric acid (37.0%), hydroxylamine hydrochloride (99.0%) and stannous chloride (98.0%), potassium permanganate (99.5%), potassium persulfate (98.0%), mercury (for calibration, solution of 1001 ± 1001 mg/L in 2 mol/L HNO3) were purchased from Merck and Sigma.

2.2 Sample Collection

The samples were collected in acid-washed glass containers from two clinics (each clinic, 15 samples) in Tehran. The samples were collected after a variety of treatments, including amalgam restorations, composite fillings, root canal treatment, cavity preparation plus temporary filling, glass ionomer filling, pulpotomy, scaling plus polishing, fissure sealants, and cementation. Each sample was collected directly from the dental chair exits into collection containers at the end of each treatment.

2.3 Sample Preparation and Testing

The collected samples were stored in HNO_3 (concentrated acid was added to each sample container to obtain a final acid concentration of 1% and stored in a refrigerator at 4°C until analysis). The samples were digested according to USEPA Method 245.1. Briefly, the sample containers were shaken vigorously before taking subsamples for analysis. 50.0 mL of the filtered sample (gravity filtration using Schleicher & Schuell 595 filter papers) was placed in a clean 150 mL plastic bottle, and then 5.0 mL of H_2SO_4, 2.5 mL of HNO_3, 5.0 mL of $K_2S_2O_8$ (5%), and 5 mL of $KMnO_4$ (5%) were added. Then, each sample was kept in an oven at 95°C for 2 hr. After cooling to room temperature, hydroxylamine hydrochloride (12% in 12% NaCl) was added dropwise to each bottle until the $KMNO_4$ color disappeared. If the $KMNO_4$ color reappeared before 15 min after the first addition, more hydroxylamine hydrochloride was added. Approximately 1 mL of hydroxylamine hydrochloride was sufficient to completely reduce the excess $KMNO_4$. Finally, water was added to each bottle to obtain a volume of 100.0 mL. Total mercury was determined using cold vapor atomic absorption spectrometry (CV-AAS).

3. Results and Discussion

The mercury concentration in the wastewater of two dental clinics is given in Table 1. This table also includes the types of treatments provided to each patient. The dental treatments performed are classified into 3 different categories: one is "amalgam restoration only". The second is "amalgam restoration with other types of treatment", and the third is "no amalgam restoration". From Tables 1 and 2, it is clear that the wastewater samples of Clinic 1 have relatively higher mercury than those of Clinic 2. This can be easily explained by the release of mercury from the amalgam itself during the dental treatment or afterwards from amalgam particles deposited inside the drainage pipes. The highest mercury content measured in the effluent was from Clinic 1 (Sample 2), where there were 25 patients, nine of whom had undergone amalgam restorations, and the mercury content was 125.67 µg/L. In addition, in cases treated without amalgam, the mercury content was very low, so that in many cases of treatment without amalgam, the mercury content was below the detectable level. The reason for the higher mercury content in the samples of the second treatment mode compared to the third treatment mode is the use of amalgam, which contains high levels of mercury, while in the third treatment mode, the possible mercury content was very low. Nevertheless, the mercury content in the effluent was higher than the permissible mercury content in the environment according to the EPA recommendation (0.002 mg/l) (Mirlean et al., 2003; Nevado et al., 2003; Vandeven & Mcginnis, 2005). The mean, standard deviation, and range of mercury concentration in the samples versus treatment type are shown in Table 3.

Table 1 Mercury concentration in wastewater samples collected from dental clinics, along with the type of treatment performed

Sample	Type of Treatment	Total No. of patients treated	Treated with amalgam	Hg (µg/L)
1	2	30	7	98.32
2	2	25	9	125.67
3	2	27	6	78.44
4	2	32	5	80.62
5	2	21	6	83.54
6	٣	26	0	3.25
7	2	28	8	114.67
8	2	21	3	31.92
9	2	23	2	19.87
10	2	27	4	43.22
11	2	22	6	75.41
12	2	27	5	71.33
13	3	26	0	1.16
14	3	20	0	ND
15	3	37	0	ND
16	3	39	0	0.84
17	2	35	7	60.42
18	2	32	6	45.63
19	2	40	8	93.74
20	2	42	7	71.23
21	2	48	8	88.55
22	2	46	6	65.92
23	2	39	5	42.25
24	3	40	0	ND
25	3	42	0	0.54
26	3	44	0	n.d.
27	2	43	4	19.95
28	2	37	6	63.64
29	3	43	0	1.29
30	3	41	0	0.21

Table 2 Mean, (standard deviation), maximum, and minimum mercury levels by type of dental treatment

	Type of Treatment	n	Min.	Max.	Mean	SD
Clinic 1	2	11	19.87	125.67	74.82	32.74
	3	4	0	3.25	1.11	1.53
Clinic 2	2	9	19.95	93.74	61.26	23.02
	3	6	0	1.29	0.48	0.51

According to Table 3, the mercury content in treatment mode 2 was higher than in treatment mode 3, and the type of treatment also had a significant relationship with the mercury content. The reason is that most of the mercury in the wastewater of dental clinics comes from amalgam, which is used a lot in treatment mode 2, while amalgam is not used in treatment mode 3 (Mukherjee et al., 2004; Vandeven & Mcginnis, 2005).

Table 3 Relationship between treatment type and mercury content based on *t*-test

Type of Treatment	n	Hg (µg/L)	SD	F	Sig	df
2	20	68.72	28.9	15.64	0.0001	28
3	10	0.73	1.02			19.09

4. Conclusion

This study confirms that wastewater effluent from sampled dental clinics in Tehran contains mercury concentrations substantially exceeding permissible environmental limits, with levels directly correlated to the use of dental amalgam. The findings demonstrate that procedures involving amalgam restorations release significant quantities of mercury into clinic wastewater, whereas treatments that do not utilize amalgam result in negligible, often undetectable, mercury levels. This stark contrast underscores amalgam as the primary source of mercury pollution within this setting. The environmental and public health implications are considerable, given mercury's persistence, capacity for bioaccumulation, and toxicity. The discharge of such contaminated wastewater into municipal systems poses a risk of wider environmental contamination, ultimately affecting ecosystems and human health through indirect exposure pathways. Consequently, these results underscore the urgent need for the widespread adoption and strict enforcement of mitigation measures in Iranian dental practices. Mandatory installation and maintenance of efficient amalgam separators are essential first steps, alongside the implementation of best management practices for amalgam waste. Furthermore, the development and enforcement of specific national regulations for dental effluent are crucial to significantly reduce this preventable source of mercury pollution. Future research should expand the scale of monitoring to quantify the total national mercury load from dental clinics and rigorously evaluate the real-world efficacy of available separator technologies in diverse clinical environments.

Statements and Declarations

Ethical considerations

Ethical issues (Including plagiarism, informed consent, misconduct, data fabrication and/or falsification, double publication and/or submission, redundancy, etc.) have been completely observed by the author.

Data availability

Data will be made available on request.

Conflicts of interest

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

Author contribution

M.H. Abbasi; F. Ahmadizadeh: Investigation, Writing the main Draft; N. Shams-Aldini; N. Fazli: Conceptualization; Writing – Review & Editing.

AI Use Declaration

During the preparation of this manuscript, the authors used ChatGPT for language translation. All content has been carefully reviewed and revised by the authors, who take full responsibility for the final version of the manuscript.

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Composting of Sanandaj Digested sewage sludge with sawdust: changes in heavy metals, PCBs, and selected organic compounds

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ABSTRACT

The co-composting of anaerobically digested sewage sludge from the Sanandaj wastewater treatment plant with wood sawdust (at a 2:3 volumetric ratio) was evaluated over a two-month period using an outdoor windrow composting method. Total heavy metal concentrations (arsenic, cadmium, chromium, copper, lead, mercury, molybdenum, nickel, selenium, and zinc) were measured before and after composting via Atomic Absorption Spectroscopy (AAS). Organic micro-pollutants including PCBs, benzo[a]pyrene, organochlorine pesticides, and semi-volatile compounds were analyzed using GC-ECD, GC-FID, and GC-MS. Post-composting data revealed significant reductions in zinc (83%), copper (81%), lead (79%), and arsenic (62%); conversely, chromium levels increased by 64.1%, while nickel concentrations remained unchanged. These trends are attributed to the loss of organic matter mass combined with physical dilution by the bulking agent. All heavy metal concentrations complied with the guidelines set by the Department of Environment of Iran and the U.S. EPA standards (Part 503), while PCBs, pesticides, and other pollutants remained below detection limits. Due to methodological limitations, especifically the lack of replicate windrows, continuous thermophilic monitoring, and comprehensive Quality Assurance/Quality Control (QA/QC) documentation, these findings should be interpreted as apparent concentration shifts rather than absolute mass removals. Nevertheless, they provide foundational insights for the management of sewage sludge in Sanandaj.

Highlights

- Co-composting with sawdust cut dry-basis Zn, Cu, Pb, and As by 62–83% in 60 days.
- Chromium rose 64.1% and nickel 3.7%, reflecting organic-matter mass loss.
- PCBs, organochlorine pesticides, and benzo[a]pyrene stayed below detection limits.
- Final compost met Iranian DOE and US EPA Part 503 Exceptional Quality limits.
- No dry-mass balance was tracked, so shifts are concentrations, not mass removals.

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1. Introduction

Sewage sludge management is a persistent operational challenge for municipal wastewater treatment plants. While digested sludge is widely applied to agricultural land due to its high nitrogen, phosphorus, and organic matter content (Silva & Santos, 2023; Johnson & Smith, 2023), its agronomic utilization carries substantial environmental and toxicological risks (Zhao et al., 2024). The accumulation of heavy metals, particularly arsenic, cadmium, chromium, copper, lead, mercury, and zinc, presents persistent ecological and soil health hazards (Zhang et al., 2022; Wang et al., 2022; Hušek et al., 2026). Furthermore, municipal sludge serves as a sink for persistent organic pollutants (POPs) and emerging contaminants, including PCBs and organochlorine pesticides, which resist conventional wastewater treatment stages and concentrate in the settled solids (Cui et al., 2024; Wang et al., 2024; Wu et al., 2025). Direct land application of unconditioned sludge thus triggers severe phytotoxicity, long-term soil degradation, and potential groundwater contamination. Co-composting provides an effective, ecologically sound pathway for biological stabilization before

land application. Blending dewatered sludge with carbon-rich lignocellulosic bulking agents, such as wood chips or agricultural residues, improves structural porosity, optimizes the carbon-to-nitrogen (C/N) ratio, and sustains the thermophilic conditions driven by specialized microbial communities (Yang et al., 2023; Sellami & Medhioub, 2022; Ivickytė & Povilaitis, 2026). This intensive bio-oxidative process accelerates the degradation of organic matter and promotes the dissipation of toxic micropollutants (Meena et al., 2023; Qiao et al., 2024). However, conservative inorganic contaminants follow distinctly different dynamics. Heavy metals do not degrade; rather, their total concentrations reflect two competing phenomena: physical dilution from the added bulking matrix versus apparent concentration driven by organic matter mineralization and carbon loss (Liu & Song, 2023; Chen et al., 2023). In addition, thermophilic humification alters metal speciation, facilitating passivation and reducing bioavailable fractions in the mature matrix (Awasthi et al., 2022; Li et al., 2022). Comprehensive monitoring of sludge composition before and after the composting cycle is therefore essential to evaluate both total loads and chemical stability.

The Sanandaj wastewater treatment plant in western Iran operates via a conventional activated sludge process followed by anaerobic digestion. Despite steady sludge production, baseline data regarding trace element redistribution and organic micropollutant persistence during outdoor windrow composting remain scarce. Establishing rigorous regulatory compliance is critical to determining the safety of this material for soil amendment. This study evaluates the concentration profiles of ten heavy metals (As, Cd, Cr, Cu, Pb, Hg, Mo, Ni, Se, Zn) and target organic micropollutants, specifically PCB congeners, organochlorine pesticides, benzo[a]pyrene, and semi-volatile organic compounds—during a 60-day outdoor co-composting trial utilizing wood chips at a 2:3 volumetric ratio. Final concentrations are evaluated against the regulatory standards established by the Iranian Department of Environment (Iranian Department of Environment, 2013, 2015) and the “Exceptional Quality” criteria defined under the U.S. EPA Part 503 biosolids rule (US EPA, 1993, 1994).

2. Materials and Methods

2.1 Composting Process and Operational Setup

The experimental study was conducted at the municipal wastewater treatment plant in Sanandaj, western Iran. Dewatered anaerobically digested sludge was mixed with locally sourced wood sawdust at a volumetric ratio of 2:3 (sludge to sawdust) to adjust the moisture content and achieve an appropriate carbon-to-nitrogen (C/N) ratio. This mixture was arranged into trapezoidal piles on an impermeable concrete base as part of a pilot-scale setup. Aerobic conditions were maintained by manually turning the material two to three times per week over a 60-day operational period. This turning frequency ensured sufficient oxygen supply while limiting excessive heat loss during the thermophilic phase. Representative samples were collected at two distinct stages: the raw dewatered digested sludge prior to the addition of sawdust (Day 0), and the final co-compost following the 60-day maturation period. For the final compost, a composite sample was prepared by combining subsamples taken from various depths (the core and outer layers) throughout the pile, thereby minimizing spatial heterogeneity.

2.2 Analytical Determinations and Quality Limitations

All chemical analyses were conducted in accordance with standard dry-weight protocols (USEPA, 2002). As the certificates issued by the laboratory reported numerical concentrations and upper-bound values without specifying the Method Detection Limit (MDL) or the Limit of Quantitation (LOQ), values marked with a ">" symbol are presented here as laboratory reporting thresholds. Specific analytical conditions (such as acid digestion matrices, calibration verification, standard reference materials, and the relative standard deviations of replicates) were not documented in the test certificates and are therefore considered limitations of this study.

2.2.1 Heavy Metals

Representative composite samples of raw sludge and final compost were analyzed for heavy metal content (As, Cd, Cr, Cu, Pb, Hg, Mo, Ni, Se, and Zn) using Atomic Absorption Spectroscopy (AAS). All analyses were conducted by an accredited laboratory following standard analytical protocols

2.2.2 Organic Micropollutants and Polychlorinated Biphenyls (PCBs)

PCB congeners and six organochlorine pesticides (lindane, heptachlor, chlordane, aldrin, dieldrin, and 4,4′-DDT) were measured by GC-ECD according to standard procedures by an accredited laboratory. Benzo[a]pyrene was determined by GC-FID, and five priority organics (hexachlorobutadiene, hexachlorobenzene, N-nitrosodimethylamine, trichloroethylene, and toxaphene) were analyzed by GC-MS. Testing was performed on two composite samples collected before and after composting

3.1 Results and Discussion

3.1 Trace Metal Concentrations

Table 1 outlines the elemental composition of the digested sludge before mixing and in the cured compost after 60 days. Marked reductions occurred for most metals on a dry-weight basis. Zinc and copper concentrations dropped sharply, decreasing by 83.1% (875 to 148 mg/kg) and 80.8% (211 to 40.5 mg/kg), respectively. Lead and arsenic followed a similar downward path, declining by 79.7% (from 34.4 to 7.0 mg/kg) and 62.2% (from 4.13 to 1.56 mg/kg). Cadmium (initially 1.0 mg/kg) and mercury (0.6 mg/kg) both dropped below their laboratory detection cutoffs (< 0.5 mg/kg for Cd; < 0.1 mg/kg for Hg). Molybdenum similarly fell from 20 mg/kg to under the reporting limit of 20 mg/kg, while selenium concentrations remained constant at 0.5 mg/kg.

In contrast to these reductions, total chromium increased from 56.9 mg/kg in raw sludge to 93.4 mg/kg in the finished compost, representing an apparent 64.1% rise. Nickel levels changed very little, moving from 21.8 to 22.6 mg/kg (+3.7%).

3.2 Concentration Changes and Mass Balance Considerations

Non-volatile heavy metals do not escape through gaseous emission or significant leaching in semi-arid, covered windrow piles. Therefore, the measured shifts reflect the balance between physical dilution by the bulking agent and the

concentrating effect of organic matter loss (Amir et al., 2005; Wang et al., 2016).

The pronounced declines in Zn, Cu, Pb, and As, alongside Cd and Hg falling below detection thresholds, are primarily driven by the addition of wood sawdust at a 2:3 volumetric ratio. Introducing a large volume of low-metal organic material expanded the dry mass of the mixture, effectively diluting baseline sludge concentrations (Alvarenga et al., 2016; Gao et al., 2020).

Conversely, organic matter breakdown concentrates conserved mineral fractions. As microbes consume biodegradable carbon and release CO2 and water vapor, the windrow loses total dry weight (Li et al., 2012), raising the proportion of non-degradable elements per kilogram of dry solid. This mechanism explains the slight rise in Ni and part of the Cr increase. However, the steeper increase in chromium (+64.1%) suggests additional factors, such as trace background metals in the sawdust (particularly if scraps of treated wood were present in the locally acquired batches), hand-turning variability, and localized sampling differences. Because mass loss over the 60-day cycle was not tracked gravimetrically, these values must be treated as dry-basis concentration changes rather than absolute mass changes (Huang et al., 2016).

Table 1 Heavy metal concentration results before and after the composting process (mg/kg)

Element	Before Composting (mg/kg)	After Composting (mg/kg)	Observed Change
Arsenic (As)	4.13	1.56	62% Decrease
Cadmium (Cd)	1	< 0.5	Decrease to < 0.5
Chromium (Cr)	56.9	93.4	Increase 64.1%
Copper (Cu)	211	40.5	81% Decrease
Lead (Pb)	34.4	7	79% Decrease
Mercury (Hg)	0.6	< 0.1	Below Detection Limit
Molybdenum (Mo)	20	< 20	Below Detection Limit
Nickel (Ni)	21.8	22.6	3.7% Slight Increase
Selenium(Se)	< 0.5	< 0.5	No Detectable change
Zinc(Zn)	875	148	Decrease 83%

3.3 Organic Micropollutants and Regulatory Evaluation

Laboratory screening showed low levels of organic micropollutants in the treated material. Total PCBs remained below the 0.2 µg/kg detection threshold in the final compost. Similarly, all six organochlorine pesticides (lindane, heptachlor, chlordane, aldrin, dieldrin, and 4,4′-DDT) were below 0.03 µg/kg, and benzo[a]pyrene was under 0.04 mg/kg. The five monitored semi-volatile compounds (hexachlorobutadiene, hexachlorobenzene, N-nitrosodimethylamine, trichloroethylene, and toxaphene) all registered below 0.5 mg/kg.

These results confirm that industrial chlorinated discharges do not heavily impact the municipal influent at the Sanandaj WWTP. Aerobic thermophilic composting promotes the breakdown of susceptible organics alongside adsorption onto stable humic matter (Cui et al., 2019). Dilution with sawdust further helped maintain these trace contaminants well beneath laboratory limits. Consequently, the final compost complies with Iranian Department of Environment standards (Iranian DOE, 2015), US EPA Part 503 Exceptional Quality criteria (USEPA, 1993), and baseline European sludge thresholds (European Commission, 2001) for heavy metals and organic compounds, demonstrating its chemical suitability for soil improvement and agricultural reuse (Table 2).

Table 2 Comparison of heavy metal concentrations in final compost with environmental and agricultural guidelines (US EPA, 1993; Iranian DOE Compost Standards)

Trace Element	Finished Compost(mg/kg)	Iranian DOE Compost Standards (mg/kg)	US EPA Part 503 (EQ Limits) (mg/kg)	Compliance Status
Arsenic (As)	1.56	< 20	41	Compliant
Cadmium (Cd)	< 0.5	< 3	39	Compliant
Chromium (Cr)	93.4	< 150	1200	Compliant
Copper (Cu)	40.5	< 300	1500	Compliant
Lead (Pb)	7.0	< 150	300	Compliant
Mercury (Hg)	< 0.1	< 1.5	17	Compliant
Nickel (Ni)	22.6	< 50	420	Compliant
Selenium (Se)	< 0.5	< 2	100	Compliant
Zinc (Zn)	148	< 500	2800	Compliant

4. Conclusion

Co-composting dewatered digested sewage sludge with wood sawdust at a 2:3 volumetric ratio produced finished compost compliant with national and international environmental standards for agricultural application. This operational trial demonstrates that the marked decreases in dry-weight concentrations of Zn (83.1%), Cu (80.8%), Pb (79.7%), and As (62.2%) were predominantly driven by physical dilution from the sawdust amendment rather than elemental degradation or loss. Conversely, the apparent increase in Cr (64.1%) and the slight rise in Ni (3.7%) reflect concentration effects caused by organic matter decomposition and windrow dry-mass loss, potentially combined with trace impurities from the bulking agent. Target persistent organic pollutants, including organochlorine pesticides and PCBs, remained consistently below laboratory reporting thresholds. Because routine process parameters and windrow mass balances were

not quantified during this plant-scale monitoring, future studies should establish comprehensive dry-mass balances to strictly decouple physical dilution from biological and chemical transformation mechanisms

Statements and Declarations

Ethical considerations

Ethical issues (Including plagiarism, informed consent, misconduct, data fabrication and/or falsification, double publication and/or submission, redundancy, etc.) have been completely observed by the author.

Data availability

Data will be made available on request.

Conflicts of interest

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

Author contribution

All authors contributed to the study conception, data interpretation, and manuscript drafting and review. All authors read and approved the final manuscript.

AI Use Declaration

During the preparation of this manuscript, the authors used AI tools for designing the Graphical Abstract. All content has been carefully reviewed and revised by the authors, who take full responsibility for the final version of the manuscript.

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