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ORIGINAL ARTICLE

Pollution and Health Risk Assessment of Heavy Metals in the Soil Around an Open Landfill Site in a Developing Country (Kazerun, Iran)

Abdulmannan Rouhani[1] · Shabnam Shadloo[2] · Abbas Naqibzadeh[3] · Michal Hejcman[1] · Morad derakhsh[4]

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Abstract

Solid waste has always been an integral part of human life, and the generation of these substances is increasing with the growth of the human population. One of the most critical environmental problems of the present age is the release of pollutants from landfills into the soil, surface water, and groundwater of the surrounding environment. Thus, the main objective of this study is to evaluate the extent of soil pollution and potential ecological and health risks related to the disposal of municipal solid waste (MSW) near a landfill site in Kazerun, Iran. Soil samples were collected from inside the landfill (ILA), agricultural land (AGL), and nursery land (NUL) and analyzed for six heavy metals (HMs), including arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), nickel (Ni), and zinc (Zn). Results revealed that the HMs in ILA soils had moderate to high pollution risk. As and Cd had the highest concentration in the study area, and all HMs were higher than the background value. Cluster analysis (CA) showed that studied metals might be characterized as two groups: group 1 (Ni, Cu, As, and Cd) related to anthropogenic activities in the study area. In contrast, group 2 (Cr and Zn), was associated with parent materials. The health risk assessment results showed that oral ingestion was the primary exposure path for elements, and children were more vulnerable to harmful health effects. Metals, namely Cu for adults and As, Cr, Cu, and Ni for children, pose maximum cancer risks. Hazard Quotient (HQ) and Hazard Index (HI) values were lower than the threshold limit, indicating no non-carcinogenic health risk to humans. Moreover, possible harmful impacts of HMs accumulation in nature and soil near human settlements call for appropriate planning for discharging toxic waste in these areas.

Keywords Heavy metals · Soil pollution · Ecological risk · Kazerun · Landfill

✉ Abdulmannan Rouhani
a.rohani70@gmail.com

Shabnam Shadloo
sh.shadloo93@gmail.com

Abbas Naqibzadeh
a.naqibzadeh@gmail.com

Michal Hejcman
Michal.Hejcman@ujep.cz

Morad derakhsh
moradderakhsh28@gmail.com

[1] Department of Environment, Faculty of Environment, University of Jan Evangelista in Ústí nad Labem (UJEP), Ústí nad Labem, Czech Republic

[2] Department of Environmental Sciences, Faculty of Natural Resources and Environment Sciences, University of Guilan, Sowmehsara, Guilan, Iran

[3] Department of Environmental Science, School of Natural Resources and Desert Studies, Yazd University, Yazd, Iran

[4] Maneh and Samalghan County, North Khorasan, Iran

1 Introduction

The global industrial revolution has released the exceptional value of toxic elements (contaminants) in the environment. The majority of pollutant sources are characterized by rapid urban sprawl and increased growth in population [1, 2]. This exponential growth of the world population has trained a phenomenal rise in municipal solid wastes worldwide [3]. In developing countries like Iran, non-engineered landfills are the only waste management technique. They are considered an economical and convenient option to dispose of municipal solid waste (MSW) [4, 5]. However, this convenient option has some disadvantages, including solid waste mismanagement and landfill leachate, which causes soil pollution, and surface and groundwater contamination, especially from the harmful and toxic HMs [6–9]

Heavy metal is a significant parameter of concern in landfill leachate due to its cumulative, persistent, and toxicity. Pollution resulting from HMs has been a worrying

concern even in the ocean environment for a substantial period [10, 11]. Unlike organic matter in leachate, which may decrease the methane production from solid waste [12], HMs can remain within the landfills for about 150 years if they are leached at a rate of 400 (mm/year) [13]. Soils can accumulate multiple contaminants, including Pb, Zn, Cr, Ni, Se, Cd, As, Hg, and Cu. These HMs have been classified by the United States Environmental Protection Agency (USEPA) as priority control pollutants because of their toxicity, bioaccumulation, and low degradability [14, 15]. Also, As, Cd, Cr, and Ni are categorized as group 1 carcinogens based on the International Agency for Research on Cancer [16], which are also toxic in the environment.

Limited attention has been paid to the effect of the landfill and its leachate on the geochemical properties of the soils in Iran. However, several kinds of research [9, 17–20] have been conducted to evaluate the pollution of HMs in the soil around a municipal waste dumpsite all over the world. In Iran, Azizpour et al. [21] evaluated the impact of HMs concentration on soils nearby a landfill in the Tonekabon region, northwest of Iran. The results indicated that the HM pollution ranged from uncontaminated to highly contaminated, which showed significant HM pollution in the study area associated with MSW disposal in Tonekabon. Jawed Pazhmaan et al. [22] examined the HM contamination and the spatial distribution of soil pollution nearby a landfill site in the southwest of Gorgan, Iran. Their study showed that soil in the landfill area had a moderate ecological risk index and the central parts of the study area had elevated concentrations of Cd and Pb than the southern parts because of the slope and runoff of the waste leachate. Additionally, the concentration of Cd and Pb in the suspicious sites was higher than in the control site.

Soil analysis in regard to toxic metals concentration is useful for environmental pollution evaluation and provides essential information on the value of anthropogenic activities and possible threats to inhabitants living in close proximity tdumpsites [23–25]. People living nearby to these landfills are thought to be in serious danger of being exposed to harmful metals. Therefore, in this study, we investigated soil chemical characteristics at specific MSW landfill sites in Kazerun. The aims of the survey can be declared as follows: (i) to evaluate the concentration of As, Cd, Co, Cu, Cr, Mo, Ni, and Zn in the soils polluted with MSW, (ii) to evaluate the pollution level and ecological risks based on several pollution indices and factors, (iii) to assess the human health risk assessment models to analyze whether the exposure to HMs of any dose could result in an adverse effect to human health.

2 Materials and Methods

2.1 Study Area

This study was conducted around a landfill site in Kazerun, Fars province, Iran, located at latitude 29° and 37 min 4 s north and longitude 51° and 39 min and 12 s East (Fig. 1). Based on the census of 2016, the population of this city was 142,057 people, which in this regard is the 53rd city in the country in terms of population [26]. The landfill of Kazerun is located in the geographical position of 29° 43′ 30″ north latitude and 51° 29′ 20″ east longitude and at an altitude of 1329 m above sea level. The area is generally hilly-mountainous, and disposal is done in a flat area between the hills.

Disposal is mainly in the form of depots at the site. Trenches have been constructed in which its capacity is filled, and now no trench has been dug. The waste is stored on the surface of the filled trench and deposited on the site's surface. Then it is spread daily by a loader and covered with soil. The cover soil is from the same soil of the region (fine-grained marl) available in sufficient quantity. The created cover is almost enough and has prevented the spread of waste in the area and reduced the effects of odors and insects. Also, waste incineration was precluded in the landfill. Hazardous hospital waste is incinerated at the hospital site, but clinic wastes are mixed with municipal waste and disposed of at this landfill site. Due to the depots' relatively large volume and height, there is a problem of leachate and gas.

Consequently, leachate traces are also seen on the site due to low soil permeability and the prevention of waste incineration. A large number of scavenger people are illegally collecting recyclable materials at the site. After filling the trench and creating the final cover layer in some parts of the site, tree planting has been done.

2.2 Sample Collection and Analysis Methods

In this survey, to study the soil contamination with HMs, 29 topsoil samples were collected from inside landfill site (ILA), surrounding nursery (NUL), and agricultural (AGL) lands from a depth of 0 to 20 cm by a plastic shovel and transferred to the laboratory. Soil samples were dried at 25 °C and grounded under 100 mesh. Six trace elements (As, Zn, Cr, Ni, Cu, and Cd) were considered for analyzing the soil pollution risk. Then 1 g of individual soil sample was weighed and placed in a polyethylene beaker, and by adding HCL and HF (7 cc), the samples were heated in a water bath at 100 °C until near drying. After cooling the samples, 7 cc of HNO_3 and HCL were added to each in the

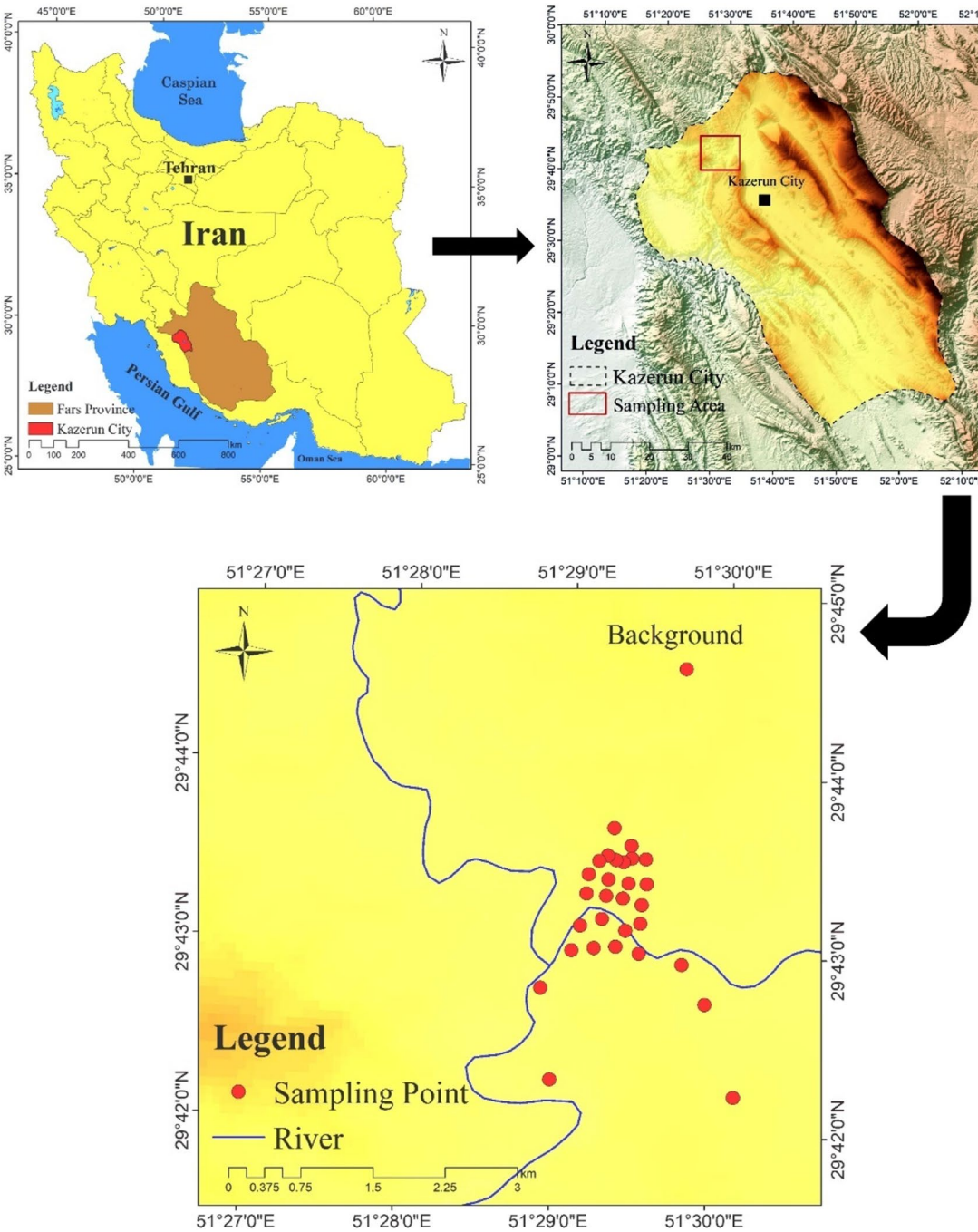

Fig. 1 Location of the study area and sampling sites (coordinate system in this map is WGS 84/UTM zone 39N)

form of Aqua regia and heated in a water bath until near drying. After chemical digestion, all soil samples were analyzed by ICP-OES (Spectro Arcos, Germany), which was credited following ISO 17025 standards by the Iranian National Standards Organization (ISIRI). Then the pH was measured in 1:2.5 soil to distilled water w/w ratio with a pH meter (Cybershot PCD 6500 pH meter, Eutech) [27].

2.3 Pollution and Ecological Risk Assessment

The environmental impact of HM and the level of soil pollution were measured using the geo-accumulation index (I_{geo}). Soil pollution index was obtained by applying HM composition from lithogenic contents of HMs in soil, whose values were not influenced by pedogenic processes [28].

2.3.1 Geo-accumulation Index (I_{geo})

We used I_{geo} to assess the pollution of individual heavy metals. It can evaluate soil pollution according to the ratio between the current metal contents in soil with the reference geochemical background for metals [9, 29–31]. Initially, this index was introduced by Muller [32]. It was calculated using the following Eq. 1

$$\text{Igeo} = \log_2[\frac{C_n}{1.5B_n}] \quad (1)$$

where C_n represents the mean concentration of heavy metal in the topsoil, B_n is the average (crustal) concentration of element n in the upper continental crust (background). The value of 1.5 was used to background matrix correction because of lithological effects. Different classes of I_{geo} are as followed: (I-geo < 0) unpolluted; (0 ≤ I-geo < 1) unpolluted to moderately polluted; (1 ≤ I-geo < 2) moderately polluted; (2 ≤ I-geo < 3) moderately to strongly polluted; (3 ≤ I-geo < 4) strongly polluted; (4 ≤ I-geo < 5) strongly to very strongly polluted; (I-geo ≥ 5) very strongly polluted [33].

2.4 Health Risk Assessment

In this study, human health risk, including carcinogenic risk (CR) and non-carcinogenic risk (HQ) exposure to heavy metals, was evaluated based on the USEPA [34] guidelines. We carried out risk assessment measuring associated with soil by computing the Average Daily Dose (ADD, mg element kg^{-1} body weight day^{-1}) for adults and children (5–16 years). The dose–response was separately measured by ingestion, inhalation, and dermal absorption as follows Eq. (2–4) [35]:

$$\text{ADD}_{\text{ing}} = C \times \frac{\text{IngR} \times \text{EF} \times \text{ED}}{\text{BW} \times \text{AT}} \times 10^{-6} \quad (2)$$

$$\text{ADD}_{\text{inh}} = C \times \frac{\text{InhR} \times \text{EF} \times \text{ED}}{\text{PEF} \times \text{BW} \times \text{AT}} \times 10^{-6} \quad (3)$$

$$\text{ADD}_{\text{derm}} = C \times \frac{\text{SA} \times \text{AF} \times \text{ABS} \times \text{EF} \times \text{ED}}{\text{BW} \times \text{AT}} \times 10^{-6} \quad (4)$$

where C (mg/kg) stands for the soil heavy metal concentration; IngR(mg/day) is the ingestion rate; InhR (m3/day) is the inhalation rates; EF(day/$year^{-1}$) is exposure frequency; ED (year) is exposure duration; BW (Kg) and AT (day) are bodyweight and averaging time, respectively; 10^{-6} represents the unit conversion; SA(cm2) and AF(mg/cm^2) represent the exposed skin surface area and skin adherence factor, respectively; ABS and PEF are the dermal absorption factor (unitless) and the particle emission factor (m^3 kg^{-1}), respectively.

After the average daily dose (ADDi) for the three exposure pathways were measured, Hazard Quotient (HQ), Hazard Index (HI), and carcinogenic risk (CR) methods were used to assess the human health risk of heavy metal exposure from polluted landfill soil. The non-carcinogenic health risks of soil heavy metals were characterized as following Eqs. (5)–(6):

$$\text{HQ} = \frac{\text{ADD}_i}{\text{RfD}_i} \quad (5)$$

$$\text{HI} = \sum \text{HQ}_i \quad (6)$$

where HQ_i and HI are the non-carcinogenic hazard quotient and the hazard index, respectively, RfDi (mg/kg-d) represents the reference dose for heavy metals. According to USEPA [34, 36], if values of HQ and HI of > 1, there is a high possibility to occur adverse health risks (non-carcinogenic), while if values of HQ and HI of < 1, there is no apparent health risk to humans [37].

Carcinogenic risk (CR) as a probability of carcinogenic risk was assessed according to Eq. (7). Also, the total carcinogenic risk (TCR) as the sum of Carcinogenic risk of studied heavy metals was as Eq. (8):

$$\text{CR} = \text{ADD}_i \times \text{SF}_i \quad (7)$$

$$\text{TCR} = \sum \text{CR}_i \quad (8)$$

where CR is the total cancer risks through three exposure pathways, and SF is the carcinogenicity slope factor (per mg/kg-d). TCR is the sum of the carcinogenic risk of three exposure pathways. Based on USEPA [36, 38], if CR and TCR < 1×10^{-4}, the carcinogenic risk can be negligible or no effect on human health, if 1×10^{-4} < CR and TCR < 1×10^{-6}, acceptable risk, and if CR and TCR > 1×10^{-6}, the cancer risk is an intolerable range of the human body [37]. The values applied in the measurement of human health risk assessment were adopted from Table S1–2.

2.5 Statistical Analysis

Analyses of the data were conducted applying SPSS version 21. HMs concentration and geochemical fractions among HMs were evaluated using one-way variance analysis (ANOVA). Pearson correlation test was also used to assess the correlation of HMs in soil. Cluster analysis (CA) was employed to characterize the different groups of HMs.

3 Results and Discussion

3.1 Heavy Metals Concentration

The landfill in Kazerun is non-sanitary, in which wastes are directly dumped on the ground surface. Consequently, the soil is probably highly affected by the wastes and leachate produced. Table 1 indicates the HM contents in the waste impacted soils and natural soil inside and around the landfill. The study area's soil was primarily alkaline, with a pH between 7.52 and 8.17. The results show that the levels of HMs in the affected soil showed variation in metal contents among sampling points. Analyzed HMs inside the landfills exceed the background values. A similar conclusion can be made compared with the findings of Du and Li [39] and Bernardo et al. [40]. The mean concentrations of HMs in all areas were below the BA values. The pollution levels were generally higher in soils from the ILA dumpsite locations than in other locations around the landfill site. The concentration ranges of HMs in topsoil samples were As, 20.80–49.92 ppm; Cd, 0.59–24.30 ppm; Cu, 40.5–70.50 ppm; Cr, 90.97–137.50 ppm; Ni, 48.2–74.9 ppm and Zn, 90.10–136.84 ppm. The maximum levels of all HMs were found in ILA, located inside the landfill. In this part of the landfill, higher amounts of waste are buried, and MSW was the primary pollution source of the HMs discharged to the ILA soils, which is consistent with the results of Wang et al. [41] and Wu et al. [42].

Moreover, the high contents of As, Cd, Cu, and Cr in the soil might be the cause of anthropogenic activities around the landfills, including arsenical pesticides, use of fertilizers, used batteries, leaded gasoline, petroleum, tire, and cement factory, chemicals, and electronics manufacturing, woods and steelworks along with municipal runoffs and atmospheric deposition [43, 44]. Impacted soil by high organic landfill leachate plumes may also discharge the HMs, particularly As, into the surrounding environment [45, 46]. As, Cr, Cu, and other elements in soil were significantly correlated with iron and manganese oxides [47]. In our study area, high levels of HMs in soils inside and around the landfill site could be attributed to the earlier mismanagement of landfill operations, including illegal dumping of mixed wastes, unsuitable separation of general and dangerous waste, and inadequate leachate collection systems.

The Pearson correlation matrix was applied to establish the correlation coefficient of HMs in soils and presented in Table 2. The results revealed that all HMs were positively correlated with each other. Therefore, they possibly had comparable properties and might be originated from the same anthropogenic origin. However, relationships among the HMs were complex and impacted by multiple factors, and therefore the conclusion of Pearson correlation coefficients was not comprehensive. For additional consideration of the relationships between HMs, cluster analysis (CA) was used. The result of CA analysis for the HM contents in soils is indicated in Fig. 2. Two clusters are identified from the dendrograms for the HMs in soils. Cluster I contained Ni, Cu, As, and Cd, while the long distance between Ni and Cu with the other HMs could suggest that this cluster can be addressed separately into two sub-clusters. Cluster II contained Cr and Zn. Therefore, it is recommended by the CA that the studied metals may be categorized into two groups regarding source identification. Group 1 might result from

Table 1 Descriptive statistics of heavy metals in the waste-impacted soils (ppm)

Areas	As	Cd	Cu	Cr	Ni	Zn	pH
ILA							
Max	49.92	2.11	70.50	137.50	74.9	136.84	8.16
Min	24.30	0.75	41.65	99.10	56.88	101.86	7.74
Mean	30.98	1.5	61.69	125.41	69.97	125.99	7.99
AGL							
Max	26.50	0.88	47.91	112.87	67.38	118.72	8.17
Min	22.86	0.59	40.5	97.50	53.94	90.10	7.52
Mean	23.72	0.73	43.64	104.46	59.07	109.29	7.9
NUL							
Max	24.88	0.89	48.50	115.33	67.2	123.02	8.15
Min	20.80	0.66	41.08	90.97	48.2	100.7	7.55
Mean	23.24	0.77	43.72	101.57	56.44	113.22	7.98
BA							
Max	22.90	0.78	37.5	101.5	49.35	102.7	7.38
Min	8.80	0.27	28	74.6	37.23	81	7.35
Mean	15.85	0.53	32.75	88.05	43.29	91.85	7.36

anthropogenic activities in the study area, while group 2 was related to parent materials and, thus, had natural sources.

3.2 Ecological Risk Assessment

The pollution level of HMs was evaluated based on one method: Igeo (Table 3). The Igeo values for each HM are 0.81 to1.18 for As, 0.83 to1.81 for Cd, 0.05 to 0.53 for Cu, 0.09 to 0.15 for Cr, 0.01 to 0.31 for Ni, and − 0.11to 0.04 for Zn. In ILA, the Igeo for As and Cd were in class 1 < I-geo > 2, which means moderately polluted. However, these elements in the AGL and NUL fell in the range of 0–1, suggesting that As and Cd were classified as unpolluted to moderately polluted. The Igeo for trace elements including Cu, Ni, Cr, Ni, and Zn ranged from − 0.08 to 0.53 in the study area and were addressed to be unpolluted to moderately polluted levels, which was consistent with Alam et al. [17] reported that Igeo values of Cu and Zn in sampling sites

Table 2 Pearson correlation coefficient matrix for heavy metals in MSW soil

	As	Cd	Cu	Cr	Ni	Zn
As	1	0.667**	0.627**	0.662**	0.599**	0.535**
Cd		1	0.893**	0.861**	0.749**	0.739**
Cu			1	0.893**	0.817**	0.740**
Cr				1	0.808**	0.646**
Ni					1	0.677**
Zn						1

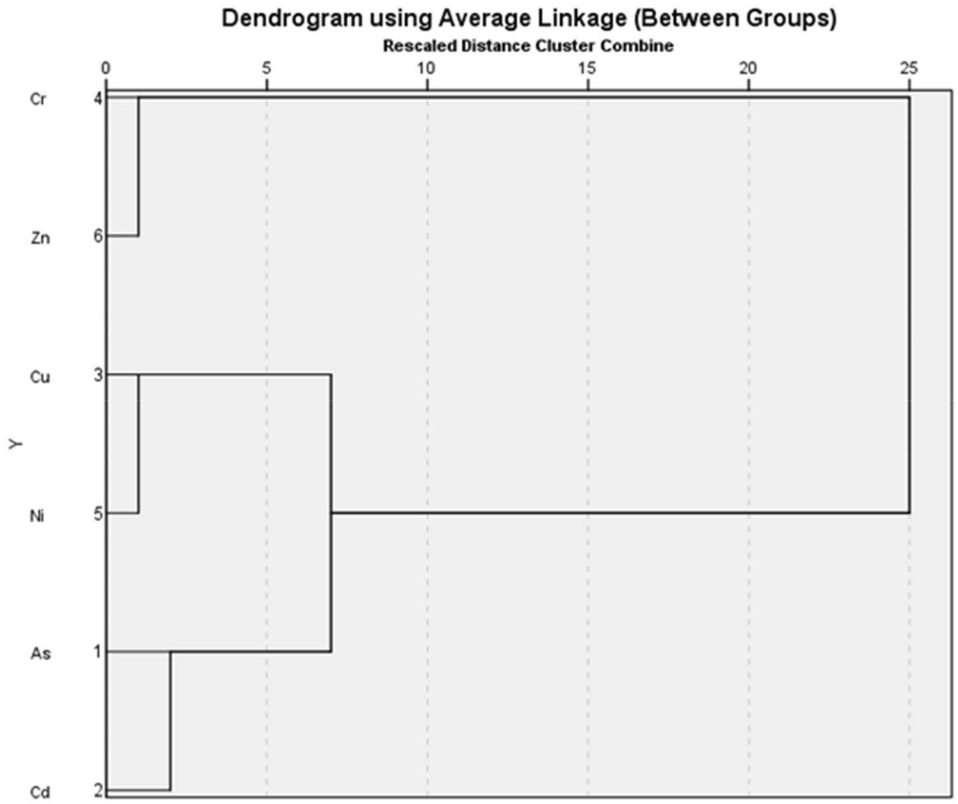

Fig. 2 Dendrogram showing the hierarchical clustering of the heavy metals

Table 3 Geo-accumulation (Igeo) for four HMs in the study area

Land type	Igeo					
	As	Cd	Cu	Cr	Ni	Zn
ILA	1.18	1.81	0.53	0.15	0.31	0.04
AGL	0.84	0.83	0.05	− 0.09	0.08	− 0.15
NUL	0.81	0.93	0.05	− 0.13	0.01	− 0.11

were unpolluted. Furthermore, the computed soil pollution index (Igeo) also implies that HMs pollution in the studied landfills was mainly from anthropogenic sources derived from wastes disposed of instead of lithological sources in soils. Generally, when the content of an HM exceeds a determined threshold value, it would lead to negative results for the environment and human health [48].

3.3 Human Health Risk Assessment

The average daily dose (ADD) of HMs exposed by children and adults through ingestion, inhalation, and dermal contact in soil samples is given in Table S3. The most dosed element through ingestion was Cr in AGL soil and Zn in ILA soil. The ADD values from the three exposure pathways followed the order of ingestion > dermal contact > inhalation.

3.3.1 Non-carcinogenic Risk Assessment

Non-carcinogenic risks due to the HMs in soil polluted with MSW are shown in Table 4 and Fig. 3. As given in Fig. 3a and b, the oral ingestion route contributed 80% of the non-carcinogenic risk for adults, while, for children, the oral ingestion route contributed 85%. Thus, ingestion is defined as one of the primary pathways to the overall health hazard, and other researchers presented similar findings [20, 49, 50]. As and Cr were HMs of significant health concerns through ingestion for children and adults. In this study, non-carcinogenic risks involved in As followed the order as ingestion (6.83E-01) > dermal (1.91E-11) > inhalation (1.41E-01) in children and ingestion (1.42E-01) > dermal (2.08E-11) > inhalation (4.13E-02) in adult at ILA. A similar trend of non-carcinogenic risk was found in Cr with ingestion (5.73E-02) > dermal (8.83E-10) > inhalation (6.86E-02) in adult and ingestion (2.77E-01) > dermal (8.1E-10) > inhalation (3.15E-01) in children at ILA. Children's higher HI and HQ values show that children in polluted areas with HMs were more susceptible to harmful effects.

For all HMs through three routes, the HQs and HI were less than 1.0, indicating no adverse effects, which is consistent with the findings of Nhien and Giao [51]. However, As and Cr were very close to the threshold level, and significant attention should pay to these elements. Earlier, Wang et al. [41] declared a substantial risk to human health due to Cr and As for the same site. The mean HI of an individual HM for adults and children observed the same value arranging trend, As > Cr > Ni > Cu > Cd > Zn (Fig. 3c and d). The HI values of HMs by combined exposure in children and adults indicated the highest level in ILA.

3.3.2 Carcinogenic Risk Assessment

Generally, Zn is classified as non-carcinogenic based on the International Agency for Research on Cancer [16]; thus, carcinogenic risk (CR) for only As, Cd, Cr, and Ni were obtained and shown in Table 5 and Fig. 4. Analogous to none-carcinogenic risk, the oral ingestion CR was significantly higher than the results of dermal and inhalation

Table 4 The results of non-carcinogenic health risk assessment of soil HMs from different sources

Heavy metal	Land type	Adult				Children			
		HQ_{ng}	HQ_{inh}	HQ_{dermal}	HI	HQ_{ing}	HQ_{inh}	HQ_{dermal}	HI
As	ILA	1.42E-01	2.08E-11	4.13E-02	1.83E-01	6.83E-01	1.91E-11	1.41E-01	8.23E-01
	AGL	1.08E-01	1.59E-11	3.16E-02	1.41E-01	5.23E-01	1.46E-11	1.07E-01	6.30E-01
	NUL	1.06E-01	1.56E-11	3.11E-02	1.37E-01	5.12E-01	1.43E-11	1.05E-01	6.17E-01
Cd	ILA	2.06E-03	5.29E-12	8.19E-04	2.88E-03	9.92E-03	4.85E-12	2.78E-03	1.27E-02
	AGL	1E-03	2.57E-12	3.99E-04	1.41E-03	4.83E-03	2.36E-12	1.35E-03	6.18E-03
	NUL	1.06E-03	2.72E-12	4.21E-04	1.48E-03	5.09E-03	2.49E-12	1.43E-03	6.52E-03
Cu	ILA	2.12E-03	3.09E-13	2.81E-05	2.14E-03	1.02E-02	2.84E-13	9.52E-05	1.03E-02
	AGL	1.5E-03	2.19E-13	1.99E-05	1.51E-03	7.21E-03	2.01E-13	6.73E-05	7.29E-03
	NUL	1.49E-03	2.19E-13	1.99E-05	1.52E-03	7.23E-03	2.01E-13	6.75E-05	7.31E-03
Cr	ILA	5.73E-02	8.83E-10	1.14E-02	6.86E-02	2.77E-01	8.1E-10	3.88E-02	3.15E-01
	AGL	4.77E-02	7.36E-10	9.52E-03	5.72E-02	2.31E-01	6.75E-10	3.22E-02	2.63E-01
	NUL	4.64E-02	7.15E-10	9.25E-03	5.56E-02	2.24E-01	6.56E-10	3.13E-02	2.55E-01
Ni	ILA	4.81E-03	5.42E-13	7.08E-05	4.87E-03	2.31E-02	4.97E-13	2.40E-04	2.34E-02
	AGL	4.05E-03	4.58E-13	5.98E-05	4.11E-03	1.95E-02	4.2E-13	2.03E-04	1.98E-02
	NUL	3.87E-03	4.37E-13	5.71E-05	3.92E-03	1.87E-02	4.01E-13	1.94E-04	1.89E-02
Zn	ILA	5.75E-04	8.46E-14	1.15E-05	5.87E-04	2.78E-03	7.76E-14	3.89E-05	2.81E-03
	AGL	4.99E-04	7.34E-14	9.96E-06	5.09E-04	2.41E-03	6.73E-14	3.37E-05	2.44E-03
	NUL	5.17E-04	7.6E-14	1.03E-05	5.27E-04	2.51E-03	6.97E-14	3.49E-05	2.53E-03

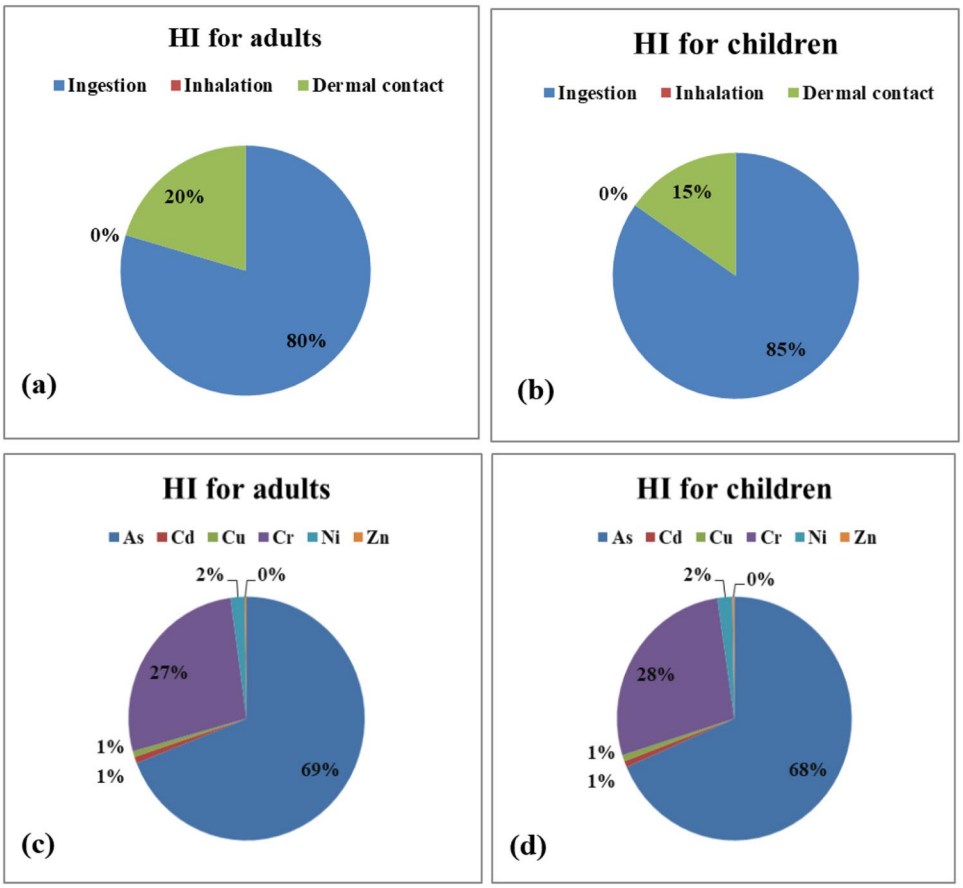

Fig. 3 Contribution of various pathways and different HMs to HI

Table 5 The carcinogenic health risk assessment results of soil HMs from different sources

Heavy metal	Land type	Adult				Children			
		CR_{ng}	CR_{inh}	CR_{dermal}	TCR	CR_{ing}	CR_{inh}	CR_{dermal}	TCR
As	ILA	6.37E-05	9.36E-14	1.86E-05	8.22E-05	3.07E-04	8.59E-14	6.3E-05	3.70E-04
	AGL	4.87E-05	7.17E-14	1.42E-05	6.3E-05	2.35E-04	6.57E-14	4.82E-05	2.84E-04
	NUL	4.78E-05	7.02E-14	1.39E-05	6.17E-05	2.31E-04	6.44E-14	4.73E-05	2.78E-04
Cd	ILA	1.25E-05	1.9E-15	5E-08	1.26E-05	6.05E-05	1.75E-15	1.69E-07	6.07E-05
	AGL	6.1E-06	9.26E-16	2.43E-08	6.12E-06	2.94E-05	8.5E-16	8.25E-08	2.95E-05
	NUL	6.43E-06	9.77E-16	2.57E-08	6.46E-06	3.11E-05	8.96E-16	8.7E-08	3.11E-05
Cu	ILA	1.44E-04	–	1.43E-05	1.58E-04	6.94E-04	–	4.85E-05	7.42E-04
	AGL	1.02E-04	–	1.01E-05	1.12E-04	4.91E-04	–	3.43E-05	5.25E-04
	NUL	1.02E-04	–	1.02E-05	1.12E-04	4.92E-04	–	3.44E-05	5.26E-04
Cr	ILA	8.59E-05	1.06E-12	1.37E-05	9.96E-05	4.15E-04	9.73E-13	4.64E-05	4.61E-04
	AGL	7.15E-05	8.84E-13	1.14E-05	8.3E-05	3.45E-04	8.11E-13	3.87E-05	3.84E-04
	NUL	6.96E-05	8.59E-13	1.11E-05	8.07E-05	3.36E-04	7.88E-13	3.76E-05	3.73E-04
Ni	ILA	8.72E-05	1.18E-14	1.63E-05	1.03E-04	4.21E-04	1.09E-14	5.51E-05	4.76E-04
	AGL	7.36E-05	1E-14	1.37E-05	8.74E-05	3.55E-04	9.17E-15	4.65E-05	4.02E-04
	NUL	7.04E-05	9.55E-15	1.31E-05	8.35E-05	3.40E-04	8.76E-15	4.44E-05	3.84E-04

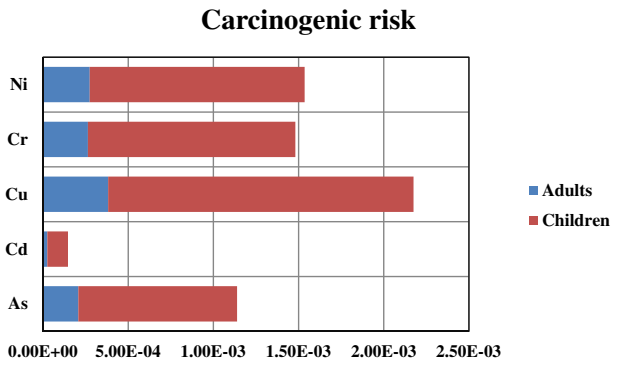

Fig. 4 Carcinogenic risk of HMs for adults and children

(Table 4), signifying that it was the leading portal for HMs exposure. Ingestion of HMs leads to moderate to severe liver abnormalities and gastrointestinal mucosal tissue damage, whereas dermal contact leads to irritant dermatitis and intense allergic responses [52, 53]. Moreover, each element had a lower CR in adults relative to children, comparable with the results of non-carcinogenic risk.

The CR data through the oral ingestion route showed higher values for Cu in adults and As, Cu, Cr, and Ni in children than the allowable limits of 1.0×10^{-4}, implying possible cancer development. However, the CR values for inhalation and dermal contact pathways were less than the acceptable level of 1.0×10^{-4}. In addition, the total TCR values of Cd in children were between 1×10^{-4} and 1×10^{-6}, related to acceptable risk. Through further analysis of Cu, Cr, As and Ni, the TCR value of these metals for children was higher than 1×10^{-4}, which can be considered an adverse cancer risk. However, for adults, TCR values of As and Cr were between 1×10^{-4} and 1×10^{-6}, which means an acceptable risk. The results revealed that the ILA areas had the highest cancer risk due to exposure to HMs.

4 Conclusion

In this study, the concentrations, pollution, and health risk assessment of six HMs (As, Cd, Cr, Cu, Ni, and Zn) in MSW dumpsite-impacted soil samples collected from a landfill site in Kazerun, Iran, were assessed. Results revealed that HMs in ILA (6 soil samples) soils were moderately polluted. Cd (mean concentration: 0.9 ppm) and As (mean concentration: 25.01 ppm) had the highest concentrations in the study area. Moreover, most of the polluted areas by HMs were located in the ILA area. The health risk assessment indicated that oral ingestion was the primary exposure route to metals, and children were more vulnerable to HM toxicity. The high HQ and HI values obtained for As and Cr were close to the permissible limit of 1.0, thus demonstrating possible health effects. Most of the carcinogenic HMs in the study area had CR values higher than the acceptable limit of 1.0×10^{-4} for children, demonstrating the possible development of cancer. As indicated relatively high non-carcinogenic hazards for a single element and Cu showed intolerable carcinogenic risks. In summary, this study provided conclusive evidence for the urgent necessity to establish a safe waste disposal procedure. It recommended selecting a new landfill far from human settlements and agricultural lands.

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Declarations

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Human ethics This article does not contain any studies involving human participants performed by any of the authors.

Consent for publication The participant has consented to the submission of the case report to the journal.

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