



# Assessing heavy metal contamination in soils using improved weighted index (IWI) and their associated human health risks in urban, wetland, and agricultural soils

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## Abstract

Contamination of nine heavy metals (HMs) Zn, Pb, Cu, Cd, As, Co, Cr, Mo, and Ni in agricultural, urban, and wetland soils from Western and Rift Valley parts of Kenya was assessed using improved weighted index (IWI) and pollution loading index (PLI). Non-carcinogenic risks posed by the HMs were assessed using hazard quotients (HQ) and hazard index (HI), while carcinogenic risks were assessed using cancer risks (CR) and total cancer risks (TCR). The average concentration of Zn, Cr, Ni, Pb, Co, Cu, As, Mo, and Cd was 94.7 mg/kg, 43.6 mg/kg, 22.3 mg/kg, 21.0 mg/kg, 19.8 mg/kg, 18.0 mg/kg, 16.3 mg/kg, 1.83 mg/kg, and 1.16 mg/kg, respectively. IWI ranged from 0.57 to 6.04 and categorized 6.82% of the study sites as not polluted, 27.3% as slightly polluted, 43.2% as moderately polluted, and 22.7% as seriously polluted. PLI ranged from 0.38 to 3.95 and classified 15.9% of the sites as not polluted, 61.4% as slightly polluted, 20.5% as moderately polluted, and only 2.3% as seriously polluted. Wetlands retained more HMs from both urban and agricultural runoff and were therefore the most polluted. The heavy metals did not pose any risks via inhalation and dermal contact, but HQ<sub>ingestion</sub> for As for children was >1 in 2.3% of the sites studied. CR via ingestion and TCR for As were above the allowable limits for children and adults indicating high risks of cancer. Intensive agriculture and urbanization should be closely monitored to prevent further HM pollution.

**Keywords** Improved weighted index · Pollution loading index · Heavy metals · Agricultural soils · Health risks · Urban soils

## Introduction

Accumulation of heavy metals (HMs) in soils is a major global concern due to their persistence and non-biodegradable characteristics, as well as the health risks they pose to human populations (He et al. 2015; Hu et al. 2017; Chonokhuu et al. 2019; Kumar et al. 2019a; Kapwata et al. 2020). HMs are ubiquitous with both natural and human sources (Chen et al.

2015; Kamunda et al. 2016; Mungai et al. 2016; Kapwata et al. 2020) and are deposited in soils making soils major sinks for HMs (Kowalska et al. 2018). Although they may occur naturally in soils at low concentration which is essential for biological systems (Chen et al. 2015; Kamunda et al. 2016; Setia et al. 2021), anthropogenic activities have caused significant deposition of HMs in soils therefore accelerating polluting of soils (Mungai et al. 2016; Hu et al. 2017; Chonokhuu et al. 2019). Further, HMs deposited in soils may be transported over long distances through runoff therefore polluting water resources (Liang et al. 2017; Alidadi et al. 2019; Emenike et al. 2020). On the other hand, those bound to dust particles are transported via atmospheric transportation and deposition thus enhancing nonpoint source pollution (Nabulo et al. 2006; Kowalska et al. 2018; Githaiga et al. 2020).

Human exposure to HMs in soils occurs via three main pathways: direct ingestion, dermal contact, and inhalation (Hu et al. 2017; Chonokhuu et al. 2019). However, it is worth noting that HMs are also capable of bioaccumulating and biomagnifying in food chains, which affects food quality and increases potential threat to human health (Nabulo et al.

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2006; Mungai et al. 2016; Liang et al. 2017; Zwolak et al. 2019). Agricultural activities are some of the main human activities polluting soils (Kumar et al. 2019b), and significant amounts of heavy metals have been found accumulating in crops and vegetables cultivated on polluted soils (Emurotu and Onianwa 2017; Tomno et al. 2020). As a result, HM pollution especially in agricultural soils is a global concern due to the pivotal role of food production (Kowalska et al. 2018; Mehmood et al. 2019; Setia et al. 2021). The health risks associated with HMs are well documented (Chen et al. 2015; Kamunda et al. 2016; Hu et al. 2017; Kinuthia et al. 2020). Cadmium (Cd), lead (Pb), and mercury (Hg), for instance, affect the central nervous system during prenatal development leading to lower intelligent quotient, whereas Pb is largely attributed to kidney and liver dysfunctions (Zwolak et al. 2019; Setia et al. 2021). Arsenic (As), on the other hand, has been known to cause liver cancer (Wang et al. 2014). Therefore, monitoring the levels and spatial distribution of HMs in agricultural, urban, and wetland soils is important to prevent bioaccumulation and biomagnification, to find sustainable management approaches, and to safeguard human health (Keshavarzi et al. 2021).

Anthropogenic activities in Kenya, such as intensive agricultural practices, urbanization, solid waste, vehicular emissions, and industrialization, have led to significant increase of HMs in the environment (Ochieng et al. 2007; Mungai et al. 2016; Maina et al. 2019; Ndungu et al. 2019; Githaiga et al. 2020; Tomno et al. 2020). This has accelerated soil and water pollution due to urban and agricultural runoff, and majority of the Kenyan population is currently at a high risk of health effects brought by exposure to HMs. In order to prevent and control HM pollution in soils, understanding the contamination characteristics is important (Chen et al. 2015). As the success to effective assessment of soil pollution depends on the use of pollution indices (Kowalska et al. 2018), it is therefore necessary to adopt assessment tools that accurately and precisely evaluate HM contamination (Abraham and Parker 2008). This also helps to improve the formulation of prevention and remediation policies, as well as recuperation of contaminated soils (Kumar et al. 2019c). So far, many methods of evaluating HM contamination in soils have been proposed, including geoaccumulation indices, enrichment factors, and pollution loading indices, among others (Blaser et al. 2000; Hu et al. 2013; Kowalska et al. 2018; Heidari et al. 2019; Ndungu et al. 2019; Githaiga et al. 2020). However, majority of these pollution indices have flaws and are limited in their use (Kowalska et al. 2018; Yang et al. 2021). For instance, the use of enrichment factor requires application of conservative elements, e.g., aluminum (Al), zirconium (Zr), iron (Fe), scandium (Sc), titanium (Ti), and manganese (Mn), but application of different conservative element may give totally different results (Loska et al. 2005; Poh and Tahir 2017; Githaiga et al. 2021). The geoaccumulation index is also limited in its

use as it does take into account natural geochemical variation, whereas Nemerow index does not include the weighing factor (Kowalska et al. 2018). Further, majority of the current models used to assess HM contamination do not consider the characteristics of those HMs at specific sites (Yang et al. 2021).

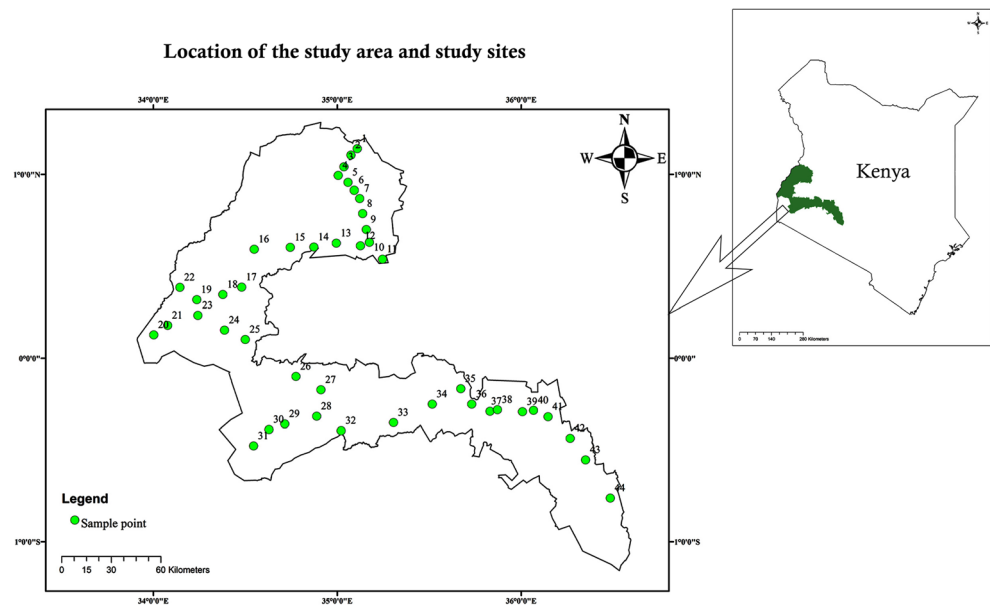
Here, we used the improved weighted index (IWI), a new method proposed by Yang et al. (2021), to investigate HM pollution in soils from Western and Rift Valleys parts of Kenya. IWI is able to eliminate various pitfalls of the traditional indices as it uses multivariate statistical analysis to assign weights to each HM at each study site and therefore is able to incorporate the characteristic of each HM at each of the polluted sites. For comparison purposes, pollution loading index (PLI) was also applied to determine the degree of HM contamination. In addition, non-carcinogenic health risks posed by the HMs in the soils were evaluated using hazard quotients (HQ) and hazard index (HI), whereas cancer risks (CR) and total cancer risks (TCR) were used to calculate carcinogenic risks for adults and children population. This study is important as it examines the efficacy of the new improved weighted index method to assess the level of HM pollution in soils and evaluates if the human population is at any risks of health risks due to HMs in the soil.

## Materials and methods

### Study area

The study area and sites are shown in Figure 1. The Western and Rift Valley parts of Kenya constitute areas with high agricultural potential. Regions like Eldoret, Nakuru, and Kitale are the country's top producer of maize which is the staple food in Kenya. In Busia and Mumias, sugarcane production is the major cash crop, and the area hosts several sugar processing companies such as Nzoia and Mumias. Parts of Ahero and Kendu Bay are known for rice production, whereas Kericho and Londiani are tea-growing regions. As a result of the intensive agriculture in these areas, different urban centers have been developed, including Kitale, Eldoret, Nakuru, and Busia towns, causing increased urbanization. A recent study by Kiprotich et al. (2021) indicated that population dynamics in Kenya were changing, and a large population had shifted from rural areas to urban areas causing population outburst in urban areas. Increased urbanization is usually accompanied by increased solid waste, wastewater release into rivers, and vehicular and industrial emissions that may add significant amounts of HMs into the environment (Thorpe and Harrison 2008; Liu et al. 2018; Kumar et al. 2019a). In addition, due to land-use change, wetlands that act as buffer zones between land and aquatic ecosystems have been lost leading to increased inflow of HMs into water resources. Lake Victoria,

**Fig. 1** Location of the study area and sites in the Western and Rift Valley parts of Kenya



for instance, has flooded in the recent past as a result of increased urban and agricultural runoff that would have otherwise be controlled if wetlands were still intact (Mafaranga 2020). Majority of the sites were close to Lake Victoria and some close to Lake Nakuru; therefore, any contamination could easily find itself in the lakes thus affecting water quality. To avoid further soil and water contamination, it was therefore important to assess the current pollution levels in these areas so as to formulate control policies and regulations.

### Soil sampling and analysis

Triplicate sampling was conducted using a stainless steel trowel. Subsurface soil samples (10–15cm in depth) were collected from a total of 44 sites, 29 sites being agricultural lands, seven from urban centers, and eight from wetlands. The soils were packed into polythene zip bags, where they were preserved in an ice box at  $-4^{\circ}\text{C}$  and transported to the Crop Nutrition Laboratory Services Limited for elemental analysis.

Soils were digested using 4 ml  $\text{HNO}_3$  and 2 ml HF in a microwave (Milestone ETHOS ONE) (Ndungu et al. 2019). HMs zinc (Zn), Pb, Cd, copper (Cu), As, cobalt (Co), chromium (Cr), molybdenum (Mo), and nickel (Ni) were then analyzed using inductively coupled plasma optical emission spectroscopy (ICP-OES Perkin Elmer Inc.). An elaborate QC procedure incorporating both internal and external quality control was employed. Blanks, 1 control sample, and 2 reference samples at the beginning and after every 10 samples were run. For the calibration standards, the solutions were NIST traceable and were sourced from Inorganic Ventures, VA, USA. Water for dilution was deionized and during analysis had a conductivity of 2 microS/cm. All acids were acquired from Sigma Aldrich. The limit of detection was

$\text{As}=\text{Cd}=\text{Cu}=\text{Pb}=\text{Zn}=0.2$  mg/kg and  $\text{Co}=\text{Cr}=\text{Mo}=\text{Ni}=0.1$  mg/kg, and the rate of recovery for the HMs ranged from 98 to 103%.

### Pollution assessment

#### IWI method

The IWI was developed using principal component analysis (PCA) and hierarchical cluster analysis (HCA) to examine the pollution traits of the HMs and the relationships among them at individual sites. IWI was calculated in three main steps: (1) We performed HCA involving the HMs in all the 44 sites, and this was to group the sites into different clusters depending on their pollution status. Ward's linkage method and Euclidean distance approach was used, and data standardization was done using Z-scores. (2) PCA was applied to each of the HCA clusters in order to determine the relative contributions of each HMs to soil contamination at each site. Here, principal components (PC) were extracted using varimax rotation method following Kaiser normalization method, and eigenvalues  $>1$  were considered for this study. (3) The assigned weight for each HM in each site was determined as a product of the corresponding relative eigenvalue and the relative loading value. The weights assigned to the HMs in each HCA cluster and PC considered the contributions of the HMs and the site attributes to the overall pollution level. Relative eigenvalues for each factor represented the relative contributions of the factors to soil pollution and were determined as the proportions of the corresponding eigenvalues, the sum of all relative eigenvalues being one (Yang et al. 2021). The relative loading values of HMs in each factor which represent the relative contributions of metals to the corresponding factor were calculated in the

same way as the relative eigenvalues. (4) The IWI is an integrated index and was therefore determined based on single factor pollution index ( $P_i$ ) using Equation 1.

$$IWI = \sum_{i=0}^n (W_i \times P_i) \tag{1}$$

where IWI is integrated weight index,  $W_i$  weight, and  $P_i$  single factor pollution index.

$P_i$  was calculated using Equation 2.

$$P_i = \frac{C_i}{B_i} \tag{2}$$

where  $C_i$  is the concentration of the heavy metal and  $B_i$  heavy metal background values for soils. Since Kenya does not have known background values (Githaiga et al. 2020), the concentrations of the upper continental crust by Wedepohl (1995) were used as the background values for this study. These values were Zn=52 mg/kg, Cr=35 mg/kg, Ni=18.6 mg/kg, Cu=14.3 mg/kg, Co=11.6 mg/kg, Pb=17 mg/kg, As=2.0 mg/kg, Mo=1.4 mg/kg, and Cd=0.102 mg/kg. The pollution levels based on the IWI were classified as no pollution ( $IWI \leq 1.0$ ), slightly polluted ( $1.0 < IWI \leq 2.0$ ), moderately polluted ( $2.0 < IWI \leq 3.0$ ), and seriously polluted ( $IWI > 3.0$ ) (Yang et al. 2021).

### Pollution loading index

Pollution loading index is widely used to assess HMs pollution and was used in this study as well. PLI determines total degree of contamination in soils and provides an easy way to prove the deterioration of the soil conditions as a result of the accumulation of HMs (Kowalska et al. 2018). PLI was calculated using Equation 3.

$$PLI = \sqrt[n]{P_{i1} + P_{i2} + P_{i3} \dots P_{in}} \tag{3}$$

where  $P_i$  is the calculated value for the single pollution index and  $n$  is the number of analyzed HMs.  $P_i$  larger than unity represents soil contamination, while those below unity represent no contamination. For PLI values, they can be categorized as not polluted ( $PLI \leq 1.0$ ), slightly polluted ( $1.0 < PLI \leq 2.0$ ), moderately polluted ( $2.0 < PLI \leq 3.0$ ), and seriously polluted ( $PLI > 3.0$ ) (Kowalska et al. 2018).

### Health risk assessment

Non-carcinogenic health risks posed by HMs were evaluated using hazard quotients (HQ) and hazard index (HI). The non-carcinogenic risk posed to children and adults was evaluated separately. Three pathways via ingestion ( $HQ_{ingestion}$ ), inhalation ( $HQ_{inhalation}$ ), and dermal contact ( $HQ_{dermal}$ ) were evaluated and were determined using Equations 4, 5, and 6 (Hu et al. 2017; Kusun et al. 2018; Chonokhuu et al. 2019).

$$CDI_{ingest} = \frac{C \times IngR \times EF \times ED}{BW \times AT} \times CF \tag{4}$$

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

$$CDI_{dermal} = \frac{C \times SA \times AF_{soil} \times ABS \times EF \times ED}{BW \times AT} \times CF \tag{6}$$

where CDI is the chronic daily intake of the HMs through ingestion, inhalation, and dermal contact exposure pathways; BW is body weight which is 15 kg for children and 70 kg for adults; EF exposure frequency which is 350 days per year for children and adults; ED exposure duration which is 6 years for children and 30 years for adults; IngR ingestion rate which is 200mg/day for children and 100 mg/day for adults; InhR inhalation rate which is 10 m<sup>3</sup>/day for children and 20 m<sup>3</sup>/day for adults; SA skin surface area which is 2100 cm<sup>2</sup> for children and 5800 cm<sup>2</sup> for adults; AF soil adherence factor which is 0.2 mg/cm<sup>2</sup> for children and 0.07 mg/cm<sup>2</sup> for adults; ABS is dermal absorption factor which is 0.1 for children and adults; PEF particulate emission factor which is 1.3×10<sup>9</sup> m<sup>3</sup>/kg for children and adults; CF conversion factor which is 10<sup>-6</sup> kg/mg for adults and children; and AT average time which is 365×ED for children and adults when determining non-carcinogenic risk and 365×70 for children and adults when calculating carcinogenic risks. These values are adopted from Liang et al. (2017) and Kusun et al. (2018).

HQs were calculated using the Equation, 7 while HI were calculated using Equation 8 as described by Chen et al. (2015) and Kusun et al. (2018). The reference doses, RfD (oral reference dose, inhalation reference dose, and dermal reference dose), for each HM are given by Kamunda et al. (2016).

$$HQ = \frac{CDI}{RfD} \tag{7}$$

$$HI = \sum_{i=0}^n HQ = HQ_{ingestion} + HQ_{dermal} + HQ_{inhalation} \tag{8}$$

Carcinogenic risk (CR) is the incremental probability of an individual developing any kind of cancer throughout life as a result of exposure to HMs capable of causing cancer (Pandit et al. 2020). We calculated carcinogenic risk for both children and adults, and this was evaluated using Equation 9.

$$CR = CDI \times SF \tag{9}$$

where CDI is the chronic daily intake obtained from Equations 4, 5, and 6 and SF is the carcinogenicity slope factor given in literature (Kamunda et al. 2016; Kusun et al. 2018). Total cancer risk (TCR), which is the contribution of individual heavy metals considering all exposure pathways, was evaluated using Equation 10. Generally, the US Environmental

Protection Agency (USEPA) considers CR values of between  $1.00E-06$  and  $1.00E-04$  as acceptable and those below  $1.00E-06$  as negligible (Kusin et al. 2018).

$$TCR = CR_{ingestion} + CR_{dermal} + CR_{inhalation} \quad (10)$$

## Results and discussion

### HM concentrations

The average concentration of Zn, Cr, Ni, Pb, Co, Cu, As, Mo, and Cd was 94.7 mg/kg, 43.6 mg/kg, 22.3 mg/kg, 21.0 mg/kg, 19.8 mg/kg, 18.0 mg/kg, 16.3 mg/kg, 1.83 mg/kg, and 1.16 mg/kg, respectively, from the highest concentrated to the lowest concentrated HM (Table 1). The HMs were widely distributed in all the study sites, except As and Mo which were detected in 15.9% and 94.45% of the sites studied, respectively. The concentrations of the HMs on each land use are also given in Table 1. Soils collected from agricultural lands had the highest average concentration of Cr, As, Ni, Cu, Mo, and Cd which was 48.4 mg/kg, 32.1 mg/kg, 24.6 mg/kg, 17.5 mg/kg, 1.82 mg/kg, and 1.19 mg/kg, respectively. This implied that majority of the HMs could have originated from agricultural-based chemicals, including fertilizers, pesticides, and fungicides. Significant sources of Cd, Cu, and Co in agricultural lands have previously been nitrate fertilizers (Setia et al. 2021). Only Zn was highest in urban areas ranging between 74.2 and 272.5 mg/kg with a mean of 134.2 mg/kg, and its increase due to urbanization, particularly traffic-related activities, has been reported before (Yan et al. 2018). The fact that urban lands were characterized with low HM concentrations could be attributed to HMs being washed away as majority of the urban areas are carpeted, a process which reduces water seepage and accelerates surface runoff (Liu et al. 2018). Pb and Co were highest in wetlands with the average concentrations of 21.2 mg/kg and 22.8 mg/kg, respectively, and this could be due to deposition of the HMs originating from agricultural and urban activities in the wetlands through runoff. Wetlands are identified as buffer zones between terrestrial and aquatic ecosystems, and they play an important ecological role in filtering and retaining HMs (Hong et al. 2020). The probable sources of the HMs are discussed in detail in the “Source apportionment of the HMs using multivariate analysis” section. Nevertheless, one-way analysis of variance (ANOVA) results indicated that the HMs did not differ significantly between the three land uses.

We compared our results with other studies and with the recommended international guidelines in soils (Table 2). The

average concentrations of Ni, Cd, and As in our study was higher than those recorded in agricultural soils from Juja and Mai Mahiu in Kenya by Mungai et al. (2016), while Zn, Cr, As, Ni, Cu, and Mo were also higher compared to those from central part of Kenya (Ndungu et al. 2019) to infer that Western and Rift Valley soils were more enriched with HMs. However, soils from West Pokot had higher Ni and Cu levels (Wanjala et al. 2020), while farmlands and urban lands in China (Yuan et al. 2021) had higher levels of Zn, As, and Cd, and this was attributed to more intensive agriculture in West Pokot and urbanization in China. We found Cu concentrations to be similar with those recorded in Iran (Negahban et al. 2021), but the highest levels of Pb were recorded in urban area of Guangdong Province in China (Hu et al. 2013). However, all the HMs did not exceed the WHO permissible levels in agricultural soils, except Cd which exceeded the recommended levels set by Canadian soil guidelines and the Chinese soil guidelines.

To predict the intensity of biological effects of the HMs, we performed a toxicological assessment whereby we compared the recorded HM concentrations with threshold effect concentration (TEC) and the probable effect concentration (PEC) (Macdonald et al. 2000). Respectively, the concentrations of Zn, Cu, As, Cd, Cr, Ni, and Pb in 77.3%, 81.8%, 13.6%, 47.7%, 54.5%, 56.8%, and 93.2% of the study sites were below the TEC. Only As, Cr, and Ni were above the PEC in 2.3%, 4.5%, and 4.5% of the study sites, respectively. Further, we calculated the potential acute toxicity for the heavy metals, which is the ratio of the heavy metal concentration to PEC, using toxic units (TUs) (Mungai et al. 2016). Arranged in the decreasing order, the average TUs for As, Ni, Cr, Cd, Zn, Pb, and Cu was 0.49, 0.46, 0.39, 0.23, 0.20, 0.16, and 0.12, respectively, indicating that As had the highest potential to cause toxicity in the environment and to humans, while Cu had the least potential.

### IWI results

In the calculation of IWI, As was not included because it was only recorded in 16% of the study sites. The HCA was applied to all the sites and resulted to three main clusters. HCA cluster 1, 2 and 3 contained 28, 9, and 7 sites, respectively (Fig 2). Cluster 2 had the highest average concentrations of Cd, Cr, Co, Mo, Ni, and Pb of 1.27 mg/kg, 47.6 mg/kg, 23.7 mg/kg, 2.50 mg/kg, 25.7 mg/kg, and 26.5 mg/kg, respectively. Cluster 3 had the highest average concentrations of Cu (19.7 mg/kg), while cluster 1 had the highest Zn concentration (98.5 mg/kg). When PCA analysis was applied to each cluster, two principal components (PC) resulted from each cluster whose eigenvalue exceeded 1. The Kaiser-Meyer-Olkin measure of sampling adequacy was 0.733 and 0.255 for cluster 1 and 2, while

**Table 1** Descriptive statistics for heavy metals (HMs) in different land uses in Western and Rift Valley parts of Kenya in mg/kg

Heavy metal	Entire study area <i>N</i> = 44		Agriculture land <i>n</i> = 29		Urban lands <i>n</i> = 7		Wetlands <i>n</i> = 8	
	Range	Mean ± SD	Range	Mean ± SD	Range	Mean ± SD	Range	Mean ± SD
Zn	26.6–298.2	94.7 ± 63.6	28.2–234	86.5 ± 52.8	74.2–272.5	134.2 ± 67.9	26.6–298.2	90.2 ± 88.8
Cr	2.69–142.0	43.6 ± 31.3	3.35–142	48.4 ± 34.7	2.69–65.5	27.0 ± 22.6	15.3–71.3	40.7 ± 20.1
As	0.37–91.2	16.3 ± 33.2	0.63–91.2	32.1 ± 51.2	9.24–59.0	26.0 ± 15.6	6.97–47.0	24.1 ± 12.5
Ni	2.62–53.0	22.3 ± 13.8	2.62–53.0	24.6 ± 14.4	2.32–57.1	16.3 ± 18.6	10.9–48.4	23.5 ± 11.6
Co	3.84–64.2	19.8 ± 13.9	3.84–64.2	20.8 ± 15.0	2.67–24.4	11.4 ± 8.16	8.95–82.9	22.8 ± 24.8
Pb	5.80–82.9	21.0 ± 13.9	5.80–39.7	19.3 ± 8.47	4.36–19.5	10.8 ± 5.65	9.53–37.6	21.2 ± 8.45
Cu	2.32–57.1	18.0 ± 13.5	3.23–52.8	17.5 ± 13.5	0.10–4.58	1.95 ± 1.41	7.83–8.63	8.23 ± 0.57
Mo	0.10–7.26	1.83 ± 1.60	0.12–4.95	1.82 ± 1.43	0.60–1.72	1.03 ± 0.41	0.13–7.26	1.73 ± 2.51
Cd	0.25–2.47	1.16 ± 0.50	0.25–2.47	1.19 ± 0.51	0.37–0.97	0.67 ± 0.42	0.52–2.32	1.17 ± 0.56

Bartlett’s test of sphericity was significant for both clusters at  $p < 0.01$ . In cluster 1, PC1 had an eigenvalue of 3.97 and accounted for 49.61% of the variance, while PC2 had an eigenvalue of 2.58 and accounted for 35.63% of the variance. PC1 was highly loaded with Cr, Co, Ni, and Cu, while PC2 was loaded with Cd, Mo, Pb, and Zn. In HCA cluster 2, PC1 eigenvalue was 3.95 and accounted for 45.38% of variance. It was heavily loaded with Cd, Mo, Pb, and Zn and was also similar to PC2 of sites in cluster 1. PC2 eigenvalue was 2.94 and accounted for 36.72% of the variance and was loaded with Cr, Co, Ni, and Cu. In HCA cluster 3, PC1 and PC2 eigenvalues were 4.07 and 3.07 and explained 50.91% and 38.31% of the variance, respectively. PC1 was positively loaded with Cr Co, and Ni and negatively loaded with Cu and Zn, while PC2 of the same cluster was positively loaded with Cd, Mo, and Pb. The loadings of each HM in each site are given in Table 3.

The weights allocated to each HM from each site are also shown in Table 3 and ranged from 0.083 to 0.167. The lowest weight was assigned to Zn in HCA1 PC2, and the highest weight was assigned to Ni in HCA3 PC 1. The IWI was then calculated using Eq.1 and ranged between 0.57 and 6.04 with a mean of 2.53. Based on IWI classification, 6.82% of the sites were categorized as not polluted, 27.3% as slightly polluted, 43.2% as moderately polluted, and 22.7% as seriously polluted. The distribution of IWI across all sites is also shown in Figure 3a. The most polluted site according to IWI was a wetland near Kisumu City, and HMs could have accumulated from urban waste derived from Kisumu City. The rest of the most polluted sites ( $IWI > 4$ ) were in a sugarcane plantation (site 19), a maize plantation (site 34), and in the urban town of Eldoret (site 11), indicating that human activities were the major contributing factors to soil pollution in the studied region.

**Table 2** Comparison of heavy metals in current study with other studies and with soil guidelines by international regulatory bodies

	Zn	Cr	As	Ni	Co	Pb	Cu	Mo	Cd	Reference
Current study	94.7	43.6	16.3	22.3	19.8	21.0	18.0	1.83	1.16	Current study
Agricultural soils from Juja, Kenya	247.4	59.69	8.93	12.56	na	26.87	88.59	na	0.42	Mungai et al. (2016)
Central province, Kenya	12.47	3.054	0.058	4.493	na	na	2.734	na	0.103	Ndungu et al. (2019)
West Pokot, Kenya	73.49	na	na	58.11	na	22.2	46.91	na	na	Wanjala et al. (2020)
Iran	14.11	na	na	12.28	na	93.78	19.39	na	23.78	Negahban et al. (2021)
Guangdong province, China	na	67.2	na	26.0	8.6	51.4	na	na	na	Hu et al. (2013)
Farmland and urban soils in China	85.86	67.37	8.89	27.77	na	30.74	25.81	na	0.19	Yuan et al. (2021)
WHO guideline in agricultural soils	300	100	20	50	50	100	100	na	3	Kamunda et al. (2016)
China guideline in soils	300	300	40	60	na	80	200	na	0.6	He et al. (2015)
Canadian soil guidelines	250	64	12	45	na	70	63	na	1.4	CCME (1999)
Threshold effect concentration (TEC)	121	43.4	9.79	22.7	na	35.8	31.6	na	0.99	Macdonald et al. (2000)
Probable effect concentration (PEC)	459	111	33	48.6	na	128	149	na	4.98	Macdonald et al. (2000)

na not available.

**Table 3** Relative eigen and relative loadings used in the determination of weight assigned to individual heavy metals in each study site

HCA cluster	Principal component	Relative eigenvalue	Variable	Loading value	Relative loading	Allocated weight
1	PC1	0.606	Cr	0.936	0.254	0.154
			Co	0.915	0.248	0.150
			Ni	0.961	0.261	0.158
			Cu	0.877	0.238	0.144
	PC2	0.394	Cd	0.814	0.241	0.095
			Mo	0.918	0.272	0.107
			Pb	0.938	0.277	0.109
2	PC1	0.573	Zn	0.711	0.210	0.083
			Cd	0.946	0.251	0.144
			Mo	0.964	0.255	0.146
			Pb	0.978	0.259	0.148
			Zn	0.887	0.235	0.135
	PC2	0.427	Cr	0.827	0.243	0.104
			Co	0.883	0.259	0.111
			Ni	0.827	0.243	0.104
			Cu	0.866	0.254	0.108
3	PC1	0.571	Cr	0.885	0.281	0.160
			Co	0.813	0.258	0.147
			Ni	0.925	0.293	0.167
			Zn	0.530	0.168	0.096
	PC2	0.429	Cd	0.901	0.315	0.135
			Mo	0.967	0.338	0.145
			Pb	0.989	0.346	0.148

### Pollution assessment using pollution index and pollution loading index

All the sites were polluted with Cd as the pollution index ( $P_i$ ) was  $>1$ . Also, 61.4%, 72.7%, 50%, 59.1%, 61.4%, 50%, and 75% of the study sites were found to be polluted with Cr, Co, Mo, Ni, Pb, Cu, and Zn, respectively. PLI ranged from 0.38 to 3.95 with a mean of 1.56. 15.9% of the study sites had PLI values of less than 1 and were indexed as not polluted. 61.4% of the study sites were indexed as slightly polluted, while 20.5% as moderately polluted. Only one site, 34, was indexed as seriously polluted as shown in Figure 3b. This was a maize plantation near Kisumu-Busia highway, and vehicular emission, especially particles from car brakes, could have contributed to the HM pollution in the site (Thorpe and Harrison 2008).

Similar to IWI, the average PLI values were highest in wetlands, followed by agricultural lands and lowest in urban soils. Wetlands may act as sinks of HMs retained from both urban and agricultural runoff (Arias et al. 2005; Zhang et al. 2010), and accumulation of more HMs is expected. In comparison, IWI concluded that more sites were seriously polluted compared to PLI, to indicate that the IWI resulted to precise results about the pollution of an area. With these findings, the

soils in the study area require remediation plans. When we compared PLI and IWI results, there was a very high positive correlation between PLI and IWI of 0.910 at  $p < 0.01$  significance level, therefore ascertaining that the results from both pollution assessment methods were similar.

### Source apportionment of the HMs using multivariate analysis

Pearson's correlation analysis provides important information regarding metal associations. In current study, there was high positive correlation between Co and Cd ( $r=0.626$ ), Co and As ( $r=0.765$ ), Co and Cr ( $r=0.683$ ), Co and Ni ( $r=0.748$ ), and Co and Cu ( $r=0.588$ ), all at  $p < 0.01$  significance level. Zn also had strong positive correlations with Mo ( $r=0.613$ ) and Pb ( $r=0.633$ ) and moderate correlation with Cd ( $r=0.467$ ) as shown in Table 4. However, Zn association with Cr and Ni was negative ( $r=-0.435$  and  $-0.380$  at  $p < 0.05$ , respectively). Other significantly strong correlations existed between Pb and Mo ( $r=0.879$ ), Ni and Cr ( $r=0.883$ ), and Mo and Cd ( $r=0.777$ ). HMs with strong correlations implied that they might have similar sources or their concentrations were influenced by similar activities (Heidari et al. 2019; Kumar et al. 2019a; Githaiga et al. 2020). Principle component analysis

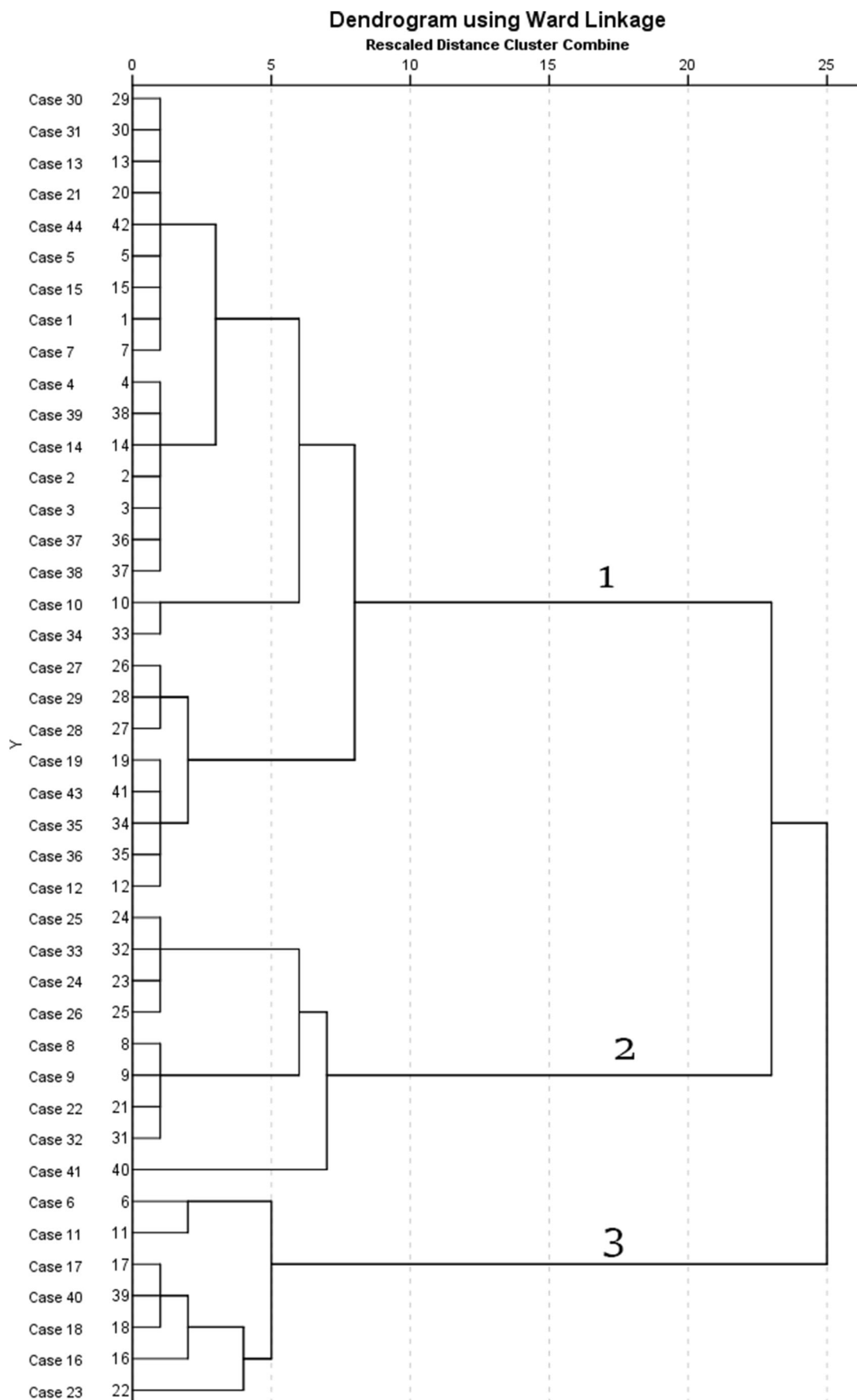
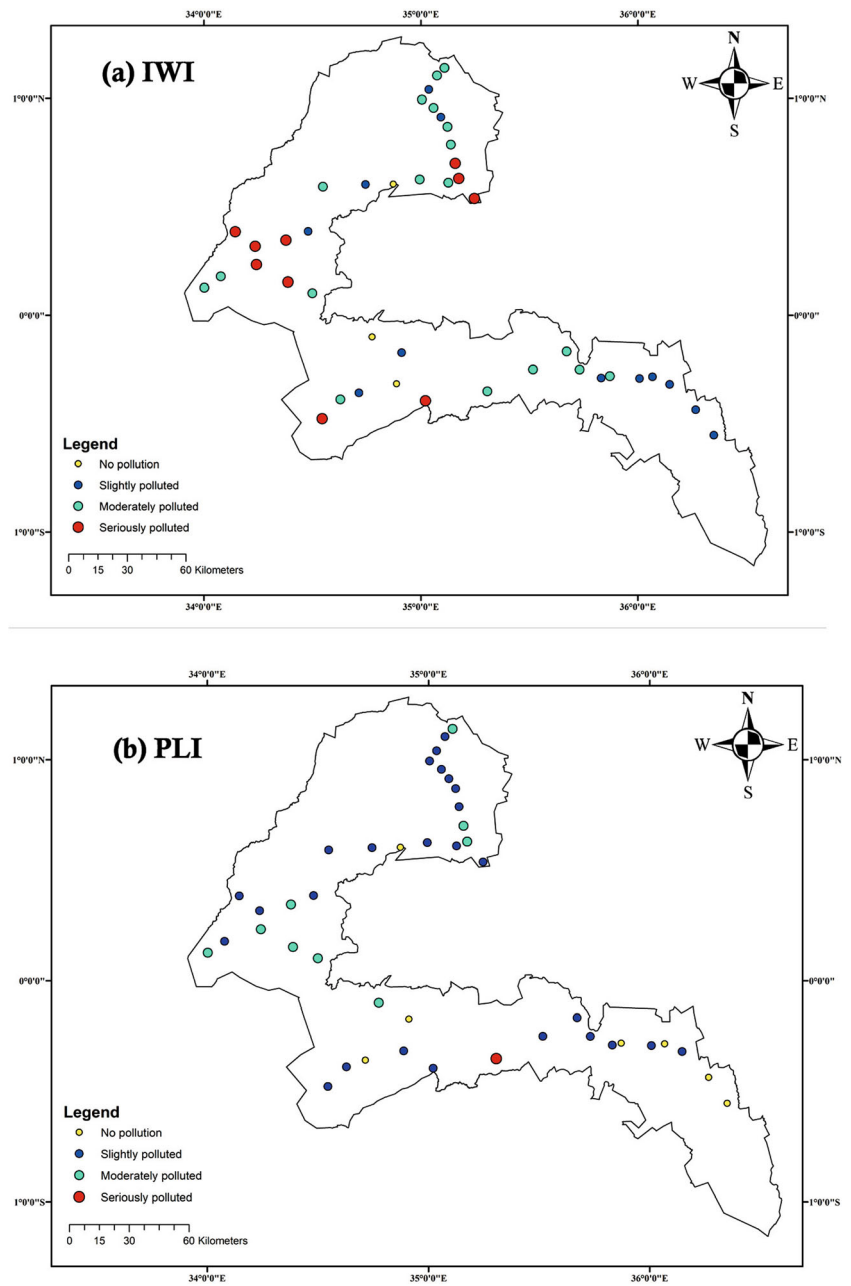


Fig. 2 Study site classification using hierarchical cluster analysis (HCA) with Ward’s linkage and standardized using Z-scores



**Fig. 3** Improved weighted index (IWI) (a) and pollution loading index (PLI) (b) in agricultural, urban, and wetlands soils in Kenya



**Table 4** Pearson’s correlations between and among heavy metals in soils from Western and Rift

	As	Cd	Cr	Co	Mo	Ni	Pb	Cu	Zn
As	1	0.605	0.574	.765*	0.637	0.341	0.264	0.242	-0.254
Cd		1	.316*	.626**	.777**	.411**	.677**	0.227	.467**
Cr			1	.683**	0.006	.883**	-0.066	.527**	-.435**
Co				1	0.180	.748**	0.119	.588**	-0.198
Mo					1	0.086	.879**	-0.100	.613**
Ni						1	-0.037	.482**	-.380*
Pb							1	-0.120	.633**
Cu								1	-0.093
Zn									1

\*Correlation is significant at the 0.05 level (2-tailed).

\*\*Correlation is significant at the 0.01 level (2-tailed).

with Varimax rotation was also performed to deduce the probable sources of all the HMs including arsenic. Eigenvalues >1 was considered for current study. This resulted to two main principal components. Cumulatively, PCA explained 83.3% of the variance. PC1 accounted for 54.1% of the variance and contained with As, Cd, Cr, Co, and Ni with loadings of 0.664, 0.858, 0.966, 0.951, and 0.901, respectively. PC2 accounted for 29.2% of the variance and contained Mo, Pb, Cu, and Zn which had loadings of 0.841, 0.890, 0.842, and 0.668, respectively. We concluded that HMs in PC1 originated human activities. As mentioned in the “HM concentrations” section, higher average concentrations of Cr, As, Ni, and Cd were recorded in agricultural lands, and their possible sources include fertilizers, pesticides, and fungicide application (Keshavarzi and Kumar 2019). Previous studies have shown that As, Cd, Cr, Co, and Ni present in human impacted areas were sourced from anthropogenic activities (Olawoyin et al. 2012; Mungai et al. 2016; Ndungu et al. 2019), implying that the soils in our study area were being polluted by the intensive agriculture and urban activities. Arsenic, nickel, and cadmium may have originated from fertilizer and pesticide application in the agricultural lands, as well as combustion of fossil fuels in the urban areas, more so the vehicular emissions (Shrivastava et al. 2015; Missimer et al. 2018; Kubier et al. 2019). PC2 elements were sourced from natural lithogenic activities. Zinc, for instance, is present in phosphate rocks which are well distributed in Kenya (Yang et al. 2017), whereas Mo and Cu have a tendency of accumulating in highly weathered soils, including sedimentary and sulfide-bearing shales (Heidari et al. 2019; Dinter et al. 2021). On the other hand, lead may accumulate in soils through volcanic emissions which are widespread in the Rift Valley region.

**Assessment of human health risks**

The HMs did not pose any health risks through inhalation and dermal contact as  $HQ_{dermal}$  and  $HQ_{inhalation}$  were <1 for both children and adults (Table 5). Similar case was seen for all the other HMs, except As, via ingestion. The maximum  $HQ_{ingestion}$  for arsenic for children was 2.0 in site 24. This was a maize cultivated agricultural land in Sidindi area, and the potential for arsenic to accumulate in food and cause severe health effects was high. Children in this area were more exposed to arsenic toxicity due to their soil-eating behavior. In a similar study by Kumar et al. (2019b), non-carcinogenic risk via ingestion pathway was higher compared to dermal contact, and children were more susceptible to HM toxicity than adults. Similar results were seen for HI (Table 5), whereby the non-cancer risk for arsenic was higher than the safe level for children. Chronic exposure to As can have severe adverse effects such as dermal lesions, liver cancer, peripheral neuropathy, and peripheral vascular disease (Wang et al. 2014; Chen et al. 2015) and should therefore be closely monitored. In

**Table 5** Hazard quotient (HQ) and hazard index (HI) of heavy metals exposure via ingestion, dermal contact and inhalation in soils from Western and Rift Valley, Kenya

HMs	$HQ_{ingestion}$						$HQ_{dermal}$						$HQ_{inhalation}$						HI					
	Children		Adults		Children		Adults		Children		Adults		Children		Adults		Children		Adults		Children		Adults	
	Max	Average	Max	Average	Max	Average	Max	Average	Max	Average	Max	Average	Max	Average	Max	Average	Max	Average	Max	Average	Max	Average	Max	Average
As	2.00E+0	3.57E-1	2.14E-1	4.24E-2	7.96E-4	1.42E-4	9.92E-4	1.77E-4	5.57E-5	9.95E-6	3.14E-5	5.61E-6	2.00E+0	3.26E-1	2.15E-1	3.38E-2	2.00E+0	3.26E-1	2.15E-1	3.38E-2	2.00E+0	3.26E-1	2.15E-1	3.38E-2
Cd	1.62E-2	7.71E-3	1.74E-3	8.20E-4	8.84E-6	4.17E-6	1.10E-5	5.19E-6	1.89E-4	8.97E-5	1.07E-4	5.06E-5	1.64E-2	7.74E-3	1.86E-3	8.75E-4	1.64E-2	7.74E-3	1.86E-3	8.75E-4	1.64E-2	7.74E-3	1.86E-3	8.75E-4
Cr	3.11E-1	9.82E-2	3.33E-2	1.02E-2	8.47E-5	2.60E-5	1.06E-4	3.24E-5	9.12E-4	2.88E-4	5.14E-4	1.62E-4	3.12E-1	9.58E-2	3.40E-2	1.04E-2	3.12E-1	9.58E-2	3.40E-2	1.04E-2	3.12E-1	9.58E-2	3.40E-2	1.04E-2
Co	2.11E-2	6.71E-3	2.26E-3	6.97E-4	1.44E-7	4.43E-8	1.79E-7	5.52E-8	2.07E-3	6.57E-4	1.17E-3	3.71E-4	2.32E-2	7.14E-3	3.43E-3	1.06E-3	2.32E-2	7.14E-3	3.43E-3	1.06E-3	2.32E-2	7.14E-3	3.43E-3	1.06E-3
Mo	9.55E-3	2.52E-3	1.02E-3	2.58E-4	1.37E-7	3.45E-8	1.70E-7	4.30E-8	2.07E-3	6.57E-4	1.17E-3	3.71E-4	9.55E-3	2.41E-3	1.02E-3	2.58E-4	9.55E-3	2.41E-3	1.02E-3	2.58E-4	9.55E-3	2.41E-3	1.02E-3	2.58E-4
Ni	1.74E-2	7.41E-3	1.87E-3	7.86E-4	3.51E-7	1.48E-7	4.38E-7	1.84E-7	4.73E-7	2.01E-7	2.67E-7	1.13E-7	1.74E-2	7.33E-3	1.87E-3	7.86E-4	1.74E-2	7.33E-3	1.87E-3	7.86E-4	1.74E-2	7.33E-3	1.87E-3	7.86E-4
Pb	1.56E-1	4.14E-2	1.67E-2	4.23E-3	5.65E-6	1.43E-6	7.05E-6	1.79E-6	4.33E-6	1.15E-6	2.44E-6	6.49E-7	1.56E-1	3.95E-2	1.67E-2	4.24E-3	1.56E-1	3.95E-2	1.67E-2	4.24E-3	1.56E-1	3.95E-2	1.67E-2	4.24E-3
Cu	1.70E-7	5.51E-8	1.01E-3	3.17E-4	1.70E-7	5.36E-8	2.12E-7	6.68E-8	2.61E-7	8.45E-8	1.47E-7	4.76E-8	6.02E-7	1.89E-7	1.01E-3	3.17E-4	6.02E-7	1.89E-7	1.01E-3	3.17E-4	6.02E-7	1.89E-7	1.01E-3	3.17E-4
Zn	6.54E-3	2.14E-3	7.00E-4	2.22E-4	1.78E-7	5.65E-8	2.22E-7	7.05E-8	1.83E-7	5.98E-8	1.03E-7	3.37E-8	6.54E-3	2.08E-3	7.01E-4	2.23E-4	6.54E-3	2.08E-3	7.01E-4	2.23E-4	6.54E-3	2.08E-3	7.01E-4	2.23E-4

**Table 6** Cancer risks (CR) and total cancer risks (TCR) of heavy metal exposure via ingestion, dermal contact, and inhalation in soils from Western and Rift Valley, Kenya

HMs	Cancer risk via soil ingestion						Cancer risk via dermal contact						Cancer risk via inhalation						Total cancer risks																																																			
	Children			Adults			Children			Adults			Children			Adults			Children			Adults																																																
	Max	Average	Min	Max	Average	Min	Max	Average	Min	Max	Average	Min	Max	Average	Min	Max	Average	Min	Max	Average	Min	Max	Average	Min	Max	Average	Min	Max	Average																																									
As	1.50E-4	2.68E-5	8.03E-5	1.44E-5	5.63E-6	3.26E-5	5.82E-6	1.03E-10	1.24E-8	2.21E-9	1.81E-4	3.24E-5	1.13E-4	2.02E-5	4.06E-5	1.96E-5	2.18E-5	1.05E-5	4.11E-6	8.82E-6	1.56E-10	7.54E-11	3.35E-9	1.62E-9	4.91E-5	2.31E-5	3.06E-5	1.44E-5	7.78E-5	2.51E-5	4.17E-5	1.34E-5	5.26E-6	1.69E-5	2.99E-10	9.64E-11	6.41E-9	2.07E-9	9.41E-5	2.89E-5	5.86E-5	1.80E-5	5.29E-5	2.29E-5	2.83E-5	1.23E-5	4.81E-6	1.15E-5	2.03E-10	8.82E-11	4.36E-9	1.89E-9	6.40E-5	2.69E-5	3.98E-5	1.67E-5	7.72E-7	2.09E-7	4.14E-7	1.12E-7	4.38E-8	1.68E-7	2.97E-12	8.03E-13	6.36E-11	1.72E-11	9.34E-7	2.37E-7	5.81E-7	1.48E-7
Cr	4.06E-5	1.96E-5	2.18E-5	1.05E-5	4.11E-6	8.82E-6	1.56E-10	7.54E-11	3.35E-9	1.62E-9	4.91E-5	2.31E-5	3.06E-5	1.44E-5	7.78E-5	2.51E-5	4.17E-5	1.34E-5	5.26E-6	1.69E-5	2.99E-10	9.64E-11	6.41E-9	2.07E-9	9.41E-5	2.89E-5	5.86E-5	1.80E-5	5.29E-5	2.29E-5	2.83E-5	1.23E-5	4.81E-6	1.15E-5	2.03E-10	8.82E-11	4.36E-9	1.89E-9	6.40E-5	2.69E-5	3.98E-5	1.67E-5	7.72E-7	2.09E-7	4.14E-7	1.12E-7	4.38E-8	1.68E-7	2.97E-12	8.03E-13	6.36E-11	1.72E-11	9.34E-7	2.37E-7	5.81E-7	1.48E-7														
Cd	4.06E-5	1.96E-5	2.18E-5	1.05E-5	4.11E-6	8.82E-6	1.56E-10	7.54E-11	3.35E-9	1.62E-9	4.91E-5	2.31E-5	3.06E-5	1.44E-5	7.78E-5	2.51E-5	4.17E-5	1.34E-5	5.26E-6	1.69E-5	2.99E-10	9.64E-11	6.41E-9	2.07E-9	9.41E-5	2.89E-5	5.86E-5	1.80E-5	5.29E-5	2.29E-5	2.83E-5	1.23E-5	4.81E-6	1.15E-5	2.03E-10	8.82E-11	4.36E-9	1.89E-9	6.40E-5	2.69E-5	3.98E-5	1.67E-5	7.72E-7	2.09E-7	4.14E-7	1.12E-7	4.38E-8	1.68E-7	2.97E-12	8.03E-13	6.36E-11	1.72E-11	9.34E-7	2.37E-7	5.81E-7	1.48E-7														
Co	4.06E-5	1.96E-5	2.18E-5	1.05E-5	4.11E-6	8.82E-6	1.56E-10	7.54E-11	3.35E-9	1.62E-9	4.91E-5	2.31E-5	3.06E-5	1.44E-5	7.78E-5	2.51E-5	4.17E-5	1.34E-5	5.26E-6	1.69E-5	2.99E-10	9.64E-11	6.41E-9	2.07E-9	9.41E-5	2.89E-5	5.86E-5	1.80E-5	5.29E-5	2.29E-5	2.83E-5	1.23E-5	4.81E-6	1.15E-5	2.03E-10	8.82E-11	4.36E-9	1.89E-9	6.40E-5	2.69E-5	3.98E-5	1.67E-5	7.72E-7	2.09E-7	4.14E-7	1.12E-7	4.38E-8	1.68E-7	2.97E-12	8.03E-13	6.36E-11	1.72E-11	9.34E-7	2.37E-7	5.81E-7	1.48E-7														
Pb	4.06E-5	1.96E-5	2.18E-5	1.05E-5	4.11E-6	8.82E-6	1.56E-10	7.54E-11	3.35E-9	1.62E-9	4.91E-5	2.31E-5	3.06E-5	1.44E-5	7.78E-5	2.51E-5	4.17E-5	1.34E-5	5.26E-6	1.69E-5	2.99E-10	9.64E-11	6.41E-9	2.07E-9	9.41E-5	2.89E-5	5.86E-5	1.80E-5	5.29E-5	2.29E-5	2.83E-5	1.23E-5	4.81E-6	1.15E-5	2.03E-10	8.82E-11	4.36E-9	1.89E-9	6.40E-5	2.69E-5	3.98E-5	1.67E-5	7.72E-7	2.09E-7	4.14E-7	1.12E-7	4.38E-8	1.68E-7	2.97E-12	8.03E-13	6.36E-11	1.72E-11	9.34E-7	2.37E-7	5.81E-7	1.48E-7														

general, the HMS can be arranged in a decreasing order: As>Cr>Ni>Pb>Mo>Zn>Cd>Co>Cu for children and As>Cr>Cd>Ni>Pb>Cu>Mo>Zn>Co for adults based on the HI values.

Similar to non-carcinogenic risks, the calculated CR for Cd, Cr, As, and Ni via inhalation and dermal contact were within the allowable range of between 1.00E-06 and 1.00E-04 as shown in Table 6. CR for Pb were below 1.00E-06, and cancer risk due to Pb was therefore negligible. A similar case was observed via ingestion, except that the maximum recorded risk for As via ingestion was 1.50E-4 and 1.44E-5 for children and adults, respectively, which was slightly above the allowable limits. Among the three investigated exposure pathways, exposure via inhalation posed the least risks, while exposure via ingestion of the heavy metals posed the highest risks. This was similar to a study by Setia et al. (2021) which recorded highest CR values for heavy metals via ingestion. Among the heavy metals, Pb had the lowest probability to cause cancer as the CR were below negligible levels of <1.00E-06. It was observed that the maximum TCR for As was 1.81E-4 and 1.13E-4 for children and adults, respectively, and therefore As posed higher cancer risks to children than adults. The high recorded As concentration in site 24 could cause cancer to both adults and children in the region and should therefore be monitored. Based on the average TCR values, As (3.24E-5)>Cr (2.89E-5)>Ni (2.69E-5)>Cd (2.31E-5)>Pb (2.37E-7) for children and As (2.02E-5)>Cr (1.80E-5)>Ni (1.67E-5)>Cd (1.44E-5)>Pb (1.48E-7) for adults.

## Conclusion

The contamination and health risks of HMs in agricultural, urban, and wetland soils from Western and Rift Valley parts in Kenya were successfully assessed. We found that soils collected from agricultural lands had the highest average concentration of Cr, As, Ni, Cu, Mo, and Cd which was 48.4 mg/kg, 32.1 mg/kg, 24.6 mg/kg, 17.5 mg/kg, 1.82 mg/kg, and 1.19 mg/kg, respectively, implying that majority of the HMs could have originated from agricultural-based chemicals, including fertilizers, pesticides, and fungicides. On the other hand, Pb and Co were highest in wetlands with the average concentrations of 21.2 mg/kg and 22.8 mg/kg, respectively, whereas highest average Zn concentrations (134.2 mg/kg) were recorded in urban soils. Using IWI method, 6.82% of the sites were categorized as not polluted, 27.3% as slightly polluted, 43.2% as moderately polluted, and 22.7% as seriously polluted. On the other hand, PLI results indicated that 15.9% of the study sites were not polluted, 61.4% were slightly polluted, 20.5% were moderately polluted, and only 2.3% were seriously polluted, with wetlands being the most polluted among the three land uses. The IWI method found more soils to be seriously

polluted compared to PLI, but the two methods were similar and had a strong relationship between them. The HMs did not pose health risks via dermal contact and inhalation, but arsenic posed non-carcinogenic risks to children as HQ<sub>ingestion</sub> was >1 in one study site. Also, As posed carcinogenic risk to children and adults as Cr and TCR were above the allowable limits. Therefore, As increase in the soils due to urbanization and agricultural activities should be monitored regularly in order to safeguard human health. The information presented in this study is useful to policy makers for making sound plans for reducing soil pollution and recuperation of the polluted soils.

**Availability of data and materials** All data generated or analyzed during this study are included in this published article (and its supplementary information files).

**Author contribution** Kelvin Babu: conceptualization, original draft preparation, investigation, writing, and editing. Samwel Njuguna: validation, investigation, and reviewing. Robert Gituru: supervision, reviewing, and editing. Xue Yan: conceptualization, methodology, supervision, funding acquisition, visualization, reviewing, and editing.

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## Declarations

**Ethics approval and consent to participate** Not applicable.

**Consent for publication** Not applicable.

**Competing interests** The authors declare no competing interests.

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