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ORIGINAL ARTICLE

Heavy Metal Contamination, its Transfer from Dumpsite Soils to Vegetables and its Risk to Human Health in Benin City, Nigeria

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ABSTRACT

The indiscriminate location of dumpsites within residential areas and the consumption of vegetables cultivated around such placement poses significant environmental and public health concerns. This study investigated heavy metal contamination in soil and vegetables (Amaranthus hybridus) cultivated around three selected dumpsites from December 2021 to February 2022 to determine the level of metal contamination and assess potential human health risks associated with vegetable consumption. Soil and vegetable samples were collected and analysed for heavy metals using Atomic Absorption Spectrophotometer (PG Instruments AAS 500F). Heavy metal concentrations differed significantly among the sampling station ($p < 0.05$) with Pb and Cu showing elevated levels at some locations. The Transfer Factor (TF) values indicated exceptionally high Fe accumulation (8.15) at station 1, while Mn exhibited a high TF value at station 3 (1.28). Hazard Quotient (HQ) values revealed variations in metal-specific contributions to non-carcinogenic risk between children and adults, while Hazard Index (HI) values exceeded the acceptable limit of 1 across all sampling locations, indicating potential non-carcinogenic health risks from prolonged exposure. The Cancer Risk Factor (CRF) and Cumulative Cancer Risk Index (CRI) indicated that Cd was the major contributor to carcinogenic risk. These findings demonstrate that consumption of vegetables cultivated around dumpsites may pose significant non-carcinogenic and carcinogenic health risks, particularly to children, underscoring the need for stricter waste management and routine monitoring of food crops.

Keywords: Dumpsites; Heavy metals; Health risk assessment; Transfer factor; Vegetables, Benin City.

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1.0 INTRODUCTION

Waste is an inevitable byproduct of human activity and development, encompassing materials discarded or intended for disposal, regardless of their potential for reuse or recycling (Viljoen et al., 2021). Global waste generation has increased substantially in response to population growth and rapid urbanisation. According to the World Bank (2022), the average per capita waste production rate is 0.74 kg/day, albeit this varies depending on socioeconomic and geographic factors. Domestic, industrial, agricultural, commercial, and institutional activities all contribute to waste production. Many developing countries, including Nigeria, continue to struggle with waste management because of weak regulatory frameworks, insufficient funding, inadequate infrastructure, and low levels of public awareness, all of which contribute to ineffective and unsustainable waste management practices (Omokaro et al., 2025). Open dumpsites remain the predominant method of waste disposal in many parts of Nigeria, despite growing concerns regarding their environmental and public health implications. Municipal wastes commonly contain hazardous materials such as electronic waste, batteries, metal scraps, plastics, paints, household chemicals, and industrial residues, which serve as important sources of heavy metals (Owonikoko & Alimba, 2024). The term heavy metals is used to describe both essential micronutrients (Fe, Zn, Cu and Mn) and potentially toxic elements (Pb and Cd), which may accumulate in environmental media and pose ecological or human health risks when present at elevated concentrations. During waste decomposition, rainfall infiltration and microbial degradation generate leachates capable of mobilising heavy metals from waste materials into surrounding soils, groundwater, and surface waters. These contaminants may subsequently disperse through runoff, atmospheric deposition, and subsurface migration, thereby increasing their persistence and potential uptake by plants (Teng et al., 2021; Angon et al., 2024). Consequently, soils surrounding dumpsites often function as sinks for heavy metals, where contaminants may persist for extended periods due to their non-biodegradable nature. The accumulation of heavy metals in plants around dumpsites has become an environmental health hazard due to the ability of these toxic metals to move into food chains (Balkhair & Ashraf, 2016). Heavy metals, originating from the dumpsites seep into surrounding soils and water bodies, thus causing environmental pollution and posing risks to plants, animals, and humans (Angon et al., 2024). In polluted soils, heavy metals may be absorbed by plant roots through passive diffusion, ion exchange, and active transport systems that normally facilitate nutrient uptake. Once absorbed, these metals are translocated through the xylem and phloem of the plant to various parts such as leaves, stems, fruits and other edible plant parts (Rai et al., 2019). The extent of the metal accumulated in the plants will depend on many factors including the pH of the soil, amount of organic matter in the soil, speciation of the metals, type of plant, nature of roots of the plants, and environmental conditions. The plants may accumulate the metals in their edible parts after growing in contaminated soils for a long time, hence causing a greater likelihood of human exposure to metals when consuming these vegetables. Health implications associated with heavy metal exposure through contaminated food crops include renal failure, bone depletion, neurological impairment, and cardiovascular illnesses (Ruzaidy & Amid, 2020; Munir et al., 2022). Therefore, the contamination of the plants grown in the vicinity of waste dumping areas poses a serious environmental and public health concern, particularly in urban and peri-urban communities where agriculture frequently occurs in close proximity to dumpsites. Studies in the Nigerian literature support the evidence of increased risk exposure of humans to heavy metals through the intake of

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vegetables grown in areas near dumpsites (Ogunkunle et al., 2017; Akintan et al., 2019; Otugboyega et al., 2024). Collectively, these studies suggest that dumpsites are important sources of heavy metal pollution in Nigerian agricultural environments, although not all geographical regions have been covered equally. Although previous studies have investigated heavy metal contamination around selected dumpsites in Benin City, these investigations have largely focused on individual environmental media or single locations. To the best of our knowledge, no previous study has undertaken a comparative assessment of multiple dumpsites by simultaneously evaluating heavy metal concentrations in soils and edible vegetables, their transfer from soil to plants, and the associated non-carcinogenic and carcinogenic human health risks. Consequently, information on heavy metal transfer from contaminated soils into commonly consumed vegetables and the resulting dietary exposure risks to residents of Benin City remains limited.

Benin City, the capital of Edo State, represents one of Nigeria's fast growing urban centres that generates substantial quantities of municipal solid waste due to increasing population density, urbanisation, commercial activities, and industrial development (Emeribe et al., 2025). There are several active and inactive dumpsites located near residential and farming areas in the city. Urban agriculture is practiced in these surroundings, where vegetable crops are grown for domestic use and sale in the market. Although the dependence on these crops is increasing, limited information is available about heavy metal accumulation in soils and plant samples around these dumpsites in the Benin City metropolis. This study was undertaken to determine whether soils and edible plants collected around dumpsites in Benin City contain elevated concentrations of heavy metals that may pose potential human health risks. In light of these concerns, this study aims to assess the concentrations of heavy metals in soils and edible plants around selected dumpsites within the Benin City metropolis, with the goal of evaluating potential environmental and public health risks associated with the consumption of vegetables cultivated around dumpsites.

2.0 MATERIALS AND METHODS

2.1 Study area

The study was carried out in Benin City, Edo State in Southern Nigeria. Benin City lies in the equatorial region between latitudes 6° 11' and 6° 29' N, and longitudes 5° 33' and 5° 47' E and is made up of five Local Government Areas (LGA) which are Ovia North-East, Oredo, Ikpoba-Okha, Egor and Uhunmwonde LGAs. The town has a total land area of about 112.552 km^2, with an average elevation of 77.8 m above sea level in the humid tropical rainforest belt of Nigeria. In Benin City communities, the predominant language is Bini and overall annual rainfall between 2,000 and 3,000 mm has been recorded. The rainy season lasts from April to October, with a brief dry spell around August. The dry season lasts from November to April, with a sultry and dusty harmattan phase between December and January. In addition, the region is frequently subjected to high relative humidity levels ranging from 75 to 85 percent

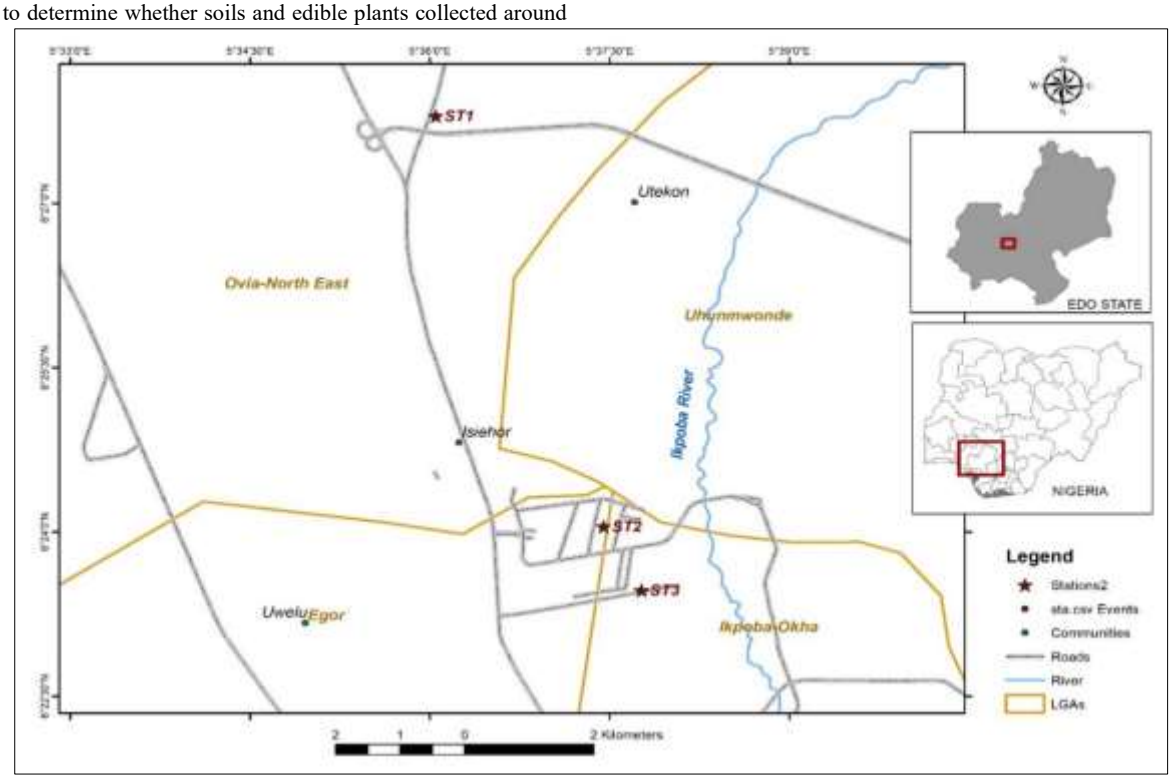


Figure **1.** Map of the study area.

2.2 Sampling and sampling locations

This study employed a cross-sectional observational field design involving multi-station spatial comparison of heavy metal concentrations in soils and edible plants collected from selected dumpsite associated locations within Benin City metropolis**.** Samples collected during each monthly visit were analysed separately. For each station, heavy metal concentrations obtained from the three-monthly sampling events were averaged to obtain a single mean concentration value. Consequently, the values reported in this study represent the mean ± standard error of measurements obtained over the three-month sampling period. Sampling was conducted from December 2021 to February 2022, corresponding to the dry season in Southern Nigeria. The study was designed to evaluate spatial variation in heavy metal concentrations among dumpsite-associated locations rather than seasonal variability. Consequently, the findings represent dry-season conditions only. Three sampling locations were established: Station 1 Farmland located adjacent to a dumpsite where vegetables are cultivated commercially for human consumption (6°24'03" N and longitude 5°37'27" E); Station 2: Otofure a municipal dumpsite managed by the Ministry of Environment anchored by Edo State Waste Management Board (6°27'48" N and longitude 5°36'03" E); Station 3: Hospital waste disposal dumpsite receiving biomedical waste (Station 3: 6°23'28" N and longitude 5°37'46" E), Figure 1. These dumpsites were located in close proximity to residential areas and received waste from multiple sources, including household, commercial, and industrial activities. Plastic bottles, polythene bags, food scraps, leaves, tree branches, papers, bricks, and biomedical waste were among the waste products observed around the dumpsites. Evidence of occasional incineration activities at the dumpsites was also observed. In addition, various edible fruits and vegetables, including pawpaw (*Carica papaya*), plantain (*Musa musaceae*) waterleaf (*Talinum fruticosum*) and green leaf (*Amaranthus hybridus*) were observed growing within and around the dumpsite.

2.3 Soil and plant samples

At each sampling station, five sampling points were established using a systematic random sampling approach. Soil samples were collected from

each point at a depth of 0-15 cm using a soil auger. During each monthly sampling event, five soil subsamples collected from each station were homogenised to produce one representative composite sample for heavy metal analysis. Sampling points were established systematically within a 50 m radius from the centre of each dumpsite with approximately equal spacing to ensure representative spatial coverage. The composite samples were placed in pre-cleaned polyethylene bags, appropriately labelled, and transported to the laboratory for heavy metal analysis. The sampling stations and samples were allocated codes (ST1, ST2, ST3) and station coordinates were acquired with the use of a Garmin GPSMAP76S GPS. A total of 45 soil samples (5 sampling points × 3 stations × 3 monthly sampling events) were collected during the study. For each monthly sampling event, the five soil subsamples collected within each station were homogenized to produce one composite sample for heavy metal analysis, resulting in nine composite soil samples (3 stations × 3 months).

Samples were collected from different locations within the sampling area to account for variability in metal uptake. Plant species' leaves and stems (*Amaranthus hybridus*) were collected at all sampling points and transported to the laboratory for identification and analysis. The plant was identified by a taxonomist in the Department of Botany; University of Benin and a voucher specimen was deposited in the University Herbarium under Voucher No. UBH=A188. *Amaranthus hybridus* was selected because it is a commonly consumed leafy vegetable in the study area, consistently available across all sampling stations, grows naturally around dumpsites, and has been reported to accumulate heavy metals from contaminated soils.

The soil samples were initially air-dried for 3-5 days, and then oven-dried at 80°C for 48 hours to eliminate residual moisture (Van Reeuwijk, 2006). To ensure sample homogeneity, the dried soil samples were ground to a fine texture with a porcelain mortar and pestle, after which they were pulverised and sieved through a 2 mm mesh sieve. To avoid cross-contamination, strict measures were taken during the drying, grinding, sieving, and storage procedures. Before being analysed, the plant samples were cleaned with distilled water, dried, and ground with a mortar and pestle. They were then stored in sterile containers prior to analysis.

2.4 Heavy metal analysis of soil and plant samples

2.4.1 Samples preparation and digestion

One gram (1g) of the prepared soil sample was weighed using a top loading analytical balance and placed into a 250 mL beaker which had previously been washed with distilled water. 5 mL nitric acid (HNO_3) was added, and the beaker was covered with a watch glass for 15-30 mins. The mixture was heated gently on a hot plate at 80 -120ºC. A 15 mL concentrated sulfuric acid (H_2SO_4), and 0.3 mL perchloric acid ($HClO_4$) was added inside a chemical fume cupboard. The heating continued on a hot plate for approximately 1 to 2 h until a clear, pale-yellow digest was obtained, indicating complete digestion of the sample. The mixture was then set aside to cool and diluted with distilled water, filtered into a 50 mL volumetric flask and distilled water was added to make up to the 50 mL mark (Sobukola et al., 2010). The metal extraction procedure was based on the USEPA 3050B acid digestion method (USEPA, 1997). Plant samples were oven-dried at 65°C for 24 h and ground to a fine powder. 1g of the prepared plant sample was weighed into a conical flask and 15 mL of 20% HNO_3 was added. The mixture was allowed to predigest for 30 min at room temperature, after which 10 mL of 30% H_2O_2 and 10 mL HCl of 37% were added gradually. The samples were then heated on a hot plate at 95 ± 5°C for 2 hrs. Digestion was considered complete when a clear, light-coloured solution was obtained and no further evolution of brown fumes was observed during heating. The samples were filtered and made to volume (100 mL) using distilled water after cooling to room temperature (24°C). The plant digestion procedure was adapted from USEPA method 3050B (USEPA, 1997) for plant tissue digestion with minor modifications.

2.4.2 Atomic absorption spectrometry (AAS)

Heavy metal concentrations in soil and plant digests were determined using an Atomic Absorption Spectrophotometer (PG Instruments AAS 500F). Each sample digest was analysed in triplicate, and the mean of the three instrumental readings was used as the final concentration value for statistical analysis. Metal concentrations were expressed in mg kg^{-1}. Determination of the metals was carried out using single-beam hollow cathode lamps specific to each element: Lead (Pb), Cadmium (Cd), Copper (Cu), Zinc (Zn), Iron (Fe), and Manganese (Mn), operated at wavelengths of 217.0, 228.8, 324.8, 213.9, 248.3, and 279.5 nm, with minimum detection limits of 0.05, 0.001, 0.004, 0.001,0.002 and 0.002 mg/L, respectively. Flame atomic absorption spectrophotometry was employed for the analysis of all metals.

2.4.3 Quality assurance procedure

Glassware was washed and rinsed properly with distilled water. Reagents of pure analytical grade were used. Reagent blank determinations (deionized water and acids) were used to correct the instrument readings. The most sensitive wavelength for each element was selected for analysis, and calibrated. Accuracy of the instrument was checked through the determination of Limit of Detection (LOD) and Limit of Quantification (LOQ) using digested reagent blanks.

2.4.4 Transfer factor (TF)

The transfer factor (TF) is a measure of heavy metal movement from the soil to the vegetables. The following formula was used to determine the TF values for the chosen heavy metals. (Cui et al., 2004).

$$TF = \frac{\text{metal concentration in vegetable}}{Metal\ concentration\ in\ the\ soil} \quad (1)$$

The transfer factor in this study was calculated using the metal concentration of the whole edible parts of the plant.

2.5 Human health risk of heavy metals in plant samples

2.5.1 Non-carcinogenic risk

Non-carcinogenic risk of heavy metals was calculated using the following formulas as described by Ekere et al. (2020).

2.5.2 Estimated daily intake (EDI)

The Estimated Daily Intake (EDI) of heavy metals through the consumption of vegetable was calculated using the following equation:

$$EDI = \frac{(C \times D)}{BW} \quad (2)$$

Where, EDI = estimated daily intake (mg/kg/day), C = concentration of the metal in sample (mg/kg), D = daily intake of vegetables (kg/day dry weight - 0.32kg) (USEPA, 2010) and BW = bodyweight (18 kg for children and 70 kg for adults).

2.5.3 Hazard quotient (HQ)

The hazard quotient, of each individual metal is calculated as the ratio of the estimated daily intake (EDI) to oral reference dose (RfD) for each metal (USEPA, 2010).

$$HQ = \frac{EDI}{RfD} \quad (3)$$

Where, HQ = hazard quotient, EDI = estimated daily intake, and RfD is the reference dose in mg/kg/day (Pb = 0.004; Zn = 0.3; Cu = 0.04; Fe = 0.7; Cd = 0.001; Mn = 0.14). The Hazard Quotient (HQ) estimates the potential non-carcinogenic health risk posed by exposure to an individual heavy metal. An HQ value less than 1 indicates that adverse health effects are unlikely, whereas an HQ greater than 1 suggests a potential health concern.

2.5.4 Hazard index (HI)

$$HI = \sum_{i=1}^{n} HQ \quad (4)$$

HI = Hazard Index; HQ= Hazard Quotient of the metal; n = Number of heavy metals considered

2.6 Carcinogenic risk

Carcinogenic risk of heavy metals in the consumption of vegetable was calculated using the following formulas;

2.6.1 Carcinogenic risk factor (CRF)

The carcinogenic risk associated with exposure to heavy metals through vegetable consumption was estimated using the Cancer Risk Factor (CRF) according to Alsafran et al. (2021).

$$CRF = EDI \times Cancer\ Slope\ factor \quad (5)$$

Where, CRF = cancer risk factor (Pb = 0.0085, Cd = 6.3); Estimate Daily Intake (EDI); Cancer Slope Factor (CSF)

2.6.2 Cancer risk index (CRI)

The cumulative carcinogenic risk from exposure to multiple heavy metals was calculated as:

$$CRI = \sum_{i=1}^{n} CRF \quad (6)$$

CRI = Cancer Risk Index; CRF = Cancer Risk Factor for the heavy metal; n = Number of carcinogenic heavy metals evaluated.

The CRI represents the total lifetime cancer risk arising from simultaneous exposure to multiple carcinogenic heavy metals. Generally, a CRI between 10^{-6} and 10^{-4} is considered acceptable, whereas values greater than 10^{-4} indicate an unacceptable carcinogenic risk.

2.7 Statistical analysis

Data obtained during the three-monthly sampling were averaged for each station and presented as mean ± standard error. Significant differences ($p < 0.05$) among stations were determined using one-way analysis of variance (ANOVA), followed by Duncan's Multiple Range Test (DMRT). Analysis of the obtained data was carried out using Microsoft Excel 2016 and IBM SPSS Statistics Version 23.0 (IBM Corp., Armonk, NY, USA).

3.0 RESULTS AND DISCUSSION

3.1 Physicochemical properties

The physicochemical properties of the soil samples (Table 1) varied across the different sampling stations; however, the observed differences were not statistically significant ($F(2,5) = 2.12$, $p = 0.35$, $\eta^2 = 0.46$), although the moderate effect size suggests that sampling location contributed to some variation in soil pH. The soil pH values fell within the range recommended for agricultural soils (6.5-6.8) by WHO (1989) and were similar to those reported by Obasi et al. (2012). pH values were also higher than those reported in a previous study by Ekere et al. (2020), who reported a pH range of 6.1 to 6.9. The near-neutral to alkaline nature of the soils may influence metal mobility and bioavailability, as increased pH generally reduces the solubility of some heavy metals through adsorption onto soil surfaces and precipitation reactions (Uchimiya et al., 2020). The EC of the soil recorded in the current study varied between 1178.0 ± 796.6 µS/cm at station 1 and 5696.67 ± 1177.03 µS/cm at station 2. Although EC varied among the sampling stations, there was no statistical significance ($F(2, 5) = 2.08$, $p = 0.21$, $\eta^2 = 0.45$), indicating that sampling location had no significant effect on soil EC. Such variations, however, may be due to differences in salt deposition, decomposition of organic materials, and waste material deposits (Nta et al., 2020). The total organic carbon (TOC) values ranged from 1.58 ± 0.23% to 2.02 ± 0.68%, with the highest TOC value observed at station 1. The observed differences were not statistically significant (F (2, 5) = 0.67, $p = 0.55$, $\eta^2 = 0.21$), indicating relatively similar TOC levels across the sampling stations. The lowest TOC value was detected at station 3. Likewise, soil organic matter (SOM) was within the range of 2.68 ± 0.40% - 3.02 ± 0.18%, with station 2 recording the highest SOM content. The differences among stations were not statistically significant ($F(2, 5) = 0.44$, $p = 0.66$, $\eta^2 = 0.15$), suggesting that SOM content was relatively consistent across the study area. Based on the similar TOC and SOM values recorded at the different stations, it may be inferred that the organic matter properties in the study area were fairly constant. Soil organic matter can affect the transport and mobility of heavy metals through metal adsorption and complex formation (Violante et al., 2010)

3.2 Metal concentrations in soil samples

The concentrations of heavy metals in soil (Table 2) and corresponding plant samples (Table 3) exhibited significant spatial variations among sampling stations, reflecting differences in local geochemical characteristics and metal bioavailability. Lead (Pb) concentrations in soil ranged from 23.06 mg/kg at station 3 to 84.7 mg/kg at station 1.

Table **1:** Physicochemical properties of the soil samples

	Station 1 x ± SE	Station 2 x ± SE	Station 3 x ± SE	*F* (2, 5)	*P*	η²
pH	7.47 ± 0.35	7.83 ± 0.15	7.47 ± 0.03	2.12	.35	.46
Electrical conductivity (µS/cm)	1178.0 ± 796.6	5696.67 ± 1177.03	4780.33 ± 2577.20	2.08	.21	.45
Total organic carbon (%)	2.02 ± 0.68	1.74 ± 0.07	1.58 ± 0.23	0.67	.55	.21
Soil organic matter (%)	2.68 ± 0.40	3.02 ± 0.18	2.73 ± 0.39	0.44	.66	.15

Values are presented as mean x ± SE (standard error). ANOVA results are reported in APA 7th edition format as F(2, 5), p, and η² (eta squared). Differences were considered statistically significant at p < .05.

Table **2.** Concentrations (mg/kg) of heavy metals in soil samples

	Station 1 x ± SE	Station 2 x ± SE	Station 3 x ± SE	*F*(2, 5)	*P*	η²	CCME (2007) (mg/kg)
Pb	84.70 ± 0.20[a]	44.60 ± 13.00[a]	23.06 ± 6.91[b]	8.80	0.023	0.78	70
Cd	1.25 ± 0.55[a]	3.13 ± 0.45[a]	2.50 ± 0.30[a]	4.57	0.074	0.65	1.4
Cu	35.95 ± 0.05[b]	21.18 ± 1.47[a]	13.85 ± 4.01[a]	13.49	0.010	0.84	63
Zn	49.80 ± 0.30[a]	119.70 ± 7.73[b]	72.75 ± 7.30[a]	24.08	0.003	0.91	250
Fe	41.80 ± 1.10[a]	4722.00 ± 409.55[b]	5741.00 ± 531.79[b]	38.80	0.001	0.94	–
Mn	94.70 ± 5.50[b]	110.71 ± 6.85[b]	58.87 ± 6.04[a]	18.63	0.005	0.88	–

Means within a row with different superscript letters (a, b) differ significantly according to Duncan's multiple range test at p < .05. Statistical differences among sampling stations were evaluated using one-way analysis of variance (ANOVA). ANOVA results are reported in APA 7th edition format as F(2, 4), p, and η² (eta squared)

Table **3.** Concentrations (mg/kg) of heavy metals in plant samples

	Station 1 x ± SE	Station 2 x ± SE	Station 3 x ± SE	*F* (2, 5)	*P*	η²	WHO/FAO* permissible limit (mg/kg)
Pb	1.65 ± 0.15[b]	0.54 ± 0.23[a]	0.27 ± 0.06[a]	15.82	0.01	0.89	0.30
Cd	0.01 ± 0.01[a]	0.06 ± 0.00[b]	0.05 ± 0.02[b]	11.71	0.02	0.85	0.20
Cu	9.10 ± 0.90[b]	11.70 ± 0.96[b]	3.89 ± 1.57[a]	10.71	0.02	0.84	10.00
Zn	0.50 ± 0.01[a]	67.83 ± 1.77[b]	44.15 ± 13.00[b]	13.22	0.01	0.87	50.00
Fe	340.50 ± 15.50[a]	875.43 ± 170.47[b]	258.33 ± 47.02[a]	8.61	0.02	0.81	450.00
Mn	6.13 ± 0.13[a]	32.07 ± 5.61[ab]	75.10 ± 25.15[b]	3.86	0.09	0.66	500.00

*x= average mean generated from values across the stations, ± standard error; Means within a row with different superscript letters (**a, b**) differ significantly according to Duncan's multiple range test at p < .05. Statistical differences among sampling stations were evaluated using one-way analysis of variance (ANOVA). ANOVA results are reported in APA 7th edition format as F(2, 4), p, and η² (eta squared). Differences were considered statistically significant at p < .05. *adapted from Sultana et al. (2022).*

Despite the relatively elevated soil Pb concentration at station 1 (84.7 mg/kg) and the corresponding plant Pb concentration (1.65 mg/kg), the calculated TF value remained low (< 0.02; Table 4). When compared to Canadian Soil Quality Guideline (CCME, 2007) of 70 mg/kg in soil, Pb concentrations at station 1 exceeded the acceptable limits for agricultural soils.

Table **4.** Transfer factor (TF) of heavy metals in the study area

Station	Pb	Cd	Cu	Zn	Fe	Mn
1	0.02	0.01	0.25	0.01	8.15	0.07
2	0.01	0.02	0.55	0.57	0.19	0.29
3	0.01	0.02	0.28	0.61	0.05	1.28

TF greater than 1 (>) indicates that the vegetable is enriched in heavy metals from the soil (bioaccumulation). TF less than 1 (<1) indicates that heavy metals from the vegetable exclude the metals from soil.

Elevated Pb concentrations in both soil and plant tissues at station 1, above permissible limits, indicate anthropogenic enrichment, likely associated with waste disposal activities in close proximity to the farmland. However, the very low transfer factor (TF < 0.02) suggests that Pb mobility from soil to plant tissues was limited even with elevated soil concentrations. Despite the presence of Pb contamination in the soil, most of the metal was retained within the soil matrix, and only a small percentage was available for plant uptake. Pb concentrations detected in plant tissues at stations 1 and 2 were above the recommended food safety limit, suggesting a potential health risk to consumers through long-term dietary exposure. The Pb TF values (< 0.02) obtained in this study were lower than those (0.91) reported by Oguh et al. (2019), indicating lower Pb mobility from soil to plant tissues. The soil Cd concentration ranged between 3.13 ± 0.45 mg/kg at station 2 and 1.25 ± 0.55 mg/kg at station 1. The observed differences among stations were not statistically significant ($F(2,5) = 4.57$, $p = 0.07$, $\eta^2 = 0.647$), although the relatively large effect size suggests that sampling location contributed to Cd variability. Cd soil concentrations exceeded the CCME guideline at stations 2 and 3, whereas station 1 remained below the permissible limit. Correspondingly, Cd levels in plant tissues were consistently low (≤ 0.06 mg/kg), although Cd levels in plant tissues were consistently low (≤ 0.06 mg/kg), concentrations differed significantly among stations ($F(2,4) = 11.71$, $p = 0.02$, $\eta^2 = 0.750$), indicating that sampling location significantly influenced Cd accumulation in plant tissues. All recorded plant Cd concentrations were below the WHO/FAO maximum permissible limit for plants. The low TF values (Table 4) for Cd (0.01–0.02) confirm minimal transfer from soil to plant tissues. The low Cd concentrations observed in plant tissues and the low transfer factors observed in the present study are consistent with the findings of Sakiyo et al. (2021), who reported Cd concentrations below detection limits in vegetables cultivated around dumpsites in Adamawa State, Nigeria, suggesting minimal accumulation in vegetables despite the influence of waste disposal activities.

Soil Cu concentration differed significantly among stations, $F(2,5) = 13.49$, $p = 0.01$, $\eta^2 = 0.844$, with the highest mean concentration (35.95 ± 0.05 mg/kg) recorded at station 1. The high effect size ($\eta^2 = 0.844$) indicates that sampling location strongly influenced Cu distribution, possibly due to differences in anthropogenic inputs and metal retention characteristics of the soils. Cu concentration in plant tissues was highest at station 2 (11.7 ± 0.96 mg/kg) with moderate TF values ranging from 0.25 to 0.55. Cu concentration in plant tissues was within the permissible limit at stations 1 and 3, but exceeded the WHO/FAO limit in station 2 (11.7 mg/kg), suggesting enhanced Cu accumulation at the dumpsite location. Elevated accumulation of Cu, Zn and Fe in plants at station 2 is attributed to the continuous anthropogenic inputs of municipal waste and the increased metal availability owing to leachates derived from the waste. Copper is an essential micronutrient, actively taken up by plants, and the observed accumulation reflects both soil availability and physiological demand. However, the moderate values of TF values indicate that Cu uptake by plants was still regulated and did not lead to excessive accumulation even at high concentrations in the soil. This regulated uptake mechanism reduces the risk of Cu-induced phytotoxicity and the associated risks to human health from dietary intake.

Zn contents in the soil at the various stations ranged from 49.8-119.70 mg/kg. Stations differed significantly in soil Zn concentrations, $F(2,5) = 24.08$, $p = 0.003$, $\eta^2 = 0.906$, indicating strong spatial variability in Zn distribution. The large effect size suggests that differences among sampling locations contributed substantially to the observed Zn concentrations, likely due to variation in anthropogenic inputs, waste deposition, and soil characteristics. Zn concentrations in plant tissues ranged from 0.50 to 67.83 mg/kg. Plant Zn concentration at station 2 (67.83 mg/kg) exceeded the WHO/FAO permissible limit of 50 mg/kg, whereas concentrations at stations 1 (0.50 mg/kg) and 3 (44.15 mg/kg) remained below the recommended limit. Although mean soil Zn concentrations across all stations were below the CCME guideline value of 250 mg/kg, the corresponding TF values (0.57–0.61) indicate relatively high Zn bioavailability. Given that Zn plays vital roles in enzymatic functions in plants (Solanki, 2021), its enhanced uptake is not unexpected; however, even though Zn was within the permissible concentration limit in stations 1 and 3, sustained accumulation at the elevated levels observed at station 2 may pose phytotoxic or food chain risks if environmental concentrations increase further. Iron showed a distinct pattern with extremely high soil concentrations at station 2 (4722 ± 409.55 mg/kg) and station 3 (5741 ± 531.79 mg/kg) compared with station 1 (41.80 ± 1.10mg/kg). Fe concentration in plant tissues at station 2 (875.43 ± 170.47 mg/kg) was higher than WHO/FAO permissible limit of 450 mg/kg, indicating significant Fe accumulation. The high Fe burden could be associated with the continuous deposition of metal containing municipal wastes and increased metal affinity due to waste decomposition processes. Despite the high soil Fe concentrations obtained at stations 2 and 3, plant uptake appeared to be influenced not only by total soil Fe but also by metal speciation and bioavailability. The exceedance at station 2 suggests a potential ecological concern and may contribute to increased dietary exposure through consumption of contaminated crops.

The unusually high Fe transfer factor recorded at station 1 (TF = 8.15) indicates that Fe concentration in plant tissues was more than eight times greater than the measured soil concentration. Fe is generally characterized by low mobility in soils and tightly regulated uptake by plants, such a high TF should be interpreted cautiously. One possible explanation is that TF was calculated using total soil Fe concentration rather than the bioavailable Fe fraction, which may not accurately represent the amount of Fe accessible for plant uptake. In addition, localised soil heterogeneity could have resulted in underestimation of soil Fe at the sampling point despite the use of composite samples. Another plausible explanation is atmospheric deposition of Fe-rich dust or particulate matter onto leaf surfaces, particularly in areas affected by waste disposal activities (Liang, 2022), which could increase measured Fe concentrations in plant tissues even after sample preparation. Furthermore, because TF is calculated as the ratio of plant to soil concentrations, unusually low soil Fe values can mathematically produce inflated TF estimates. Therefore, the exceptionally high Fe TF observed at station 1 should be interpreted with caution and may reflect the combined influence of soil heterogeneity, differences between total and bioavailable Fe fractions, and atmospheric deposition rather than exceptionally efficient biological uptake alone. Future studies incorporating bioavailable Fe analyses and foliar deposition assessments would help clarify the dominant pathway responsible for Fe accumulation. The consistently higher concentrations of Cu, Zn and Fe in vegetables collected from station 2 indicate that the managed municipal dumpsite represents the principal source of multi-metal contamination within the study area. Continuous deposition of mixed municipal wastes and the production of metal-rich leachates likely enhanced metal availability in the root zone, resulting in greater uptake by plants than at the other sampling stations. Mn concentrations in the soil differed significantly among stations, $F(2,5) = 18.63$, $p = 0.005$, $\eta^2 = 0.88$, indicating pronounced spatial variability in Mn distribution. The large effect size suggests that sampling location was a major factor influencing Mn levels, possibly due to differences in waste inputs and soil properties. Mn content ranged from 58.87 to 110.71 mg/kg with the highest soil Mn concentration recorded at station 2. In contrast, the Mn concentrations in plants (Table 3) were highest at station 3 (75.10 ± 25.15 mg/kg) with TF value greater than unity (1.28) indicating bioaccumulation. The high Mn TF value at station 3 may be attributed to site-specific soil chemical conditions. Soil pH and other geochemical properties may influence Mn availability but to verify the mechanisms behind this observed accumulation, a further examination of metal speciation, plant physiological characteristics and other soil properties would be necessary.

3.3 Non-carcinogenic risk

3.3.1 Estimated daily intake (EDI)

The Estimated Daily Intake (EDI) values (mg/kg/day) of heavy metals; Pb, Cd, Cu, Zn, Fe and Mn in adult and children through the consumption of

vegetables are presented in Table 5. The EDI values were higher than the corresponding TDI values for some metals particularly Pb, Cu and Fe indicating high exposure potential. However, Cd, Zn and Mn were below the reference doses in several stations. Estimated Daily Intake were higher in children than adults. This may be owing to their lower body weight and higher vegetable consumption per unit body mass leading to higher contaminant intake relative to physiological size. Children have developing metabolic and detoxification mechanism, making them more susceptible to heavy metal contamination (Ferguson et al., 2017). The high level of Fe EDI, particularly at station 2, could be attributed to the abundance of Fe compounds within the municipal solid waste, including metals, building materials, and domestic wastes. However, the health risk of iron contamination is determined by its bioavailable fractions rather than the overall level of contamination. Even though Pb and Cd had lower EDI values compared to Fe, their toxicological importance is greater because they are non-essential metals that can have an adverse impact on humans at low concentrations. Cd has a high tendency to accumulate in body tissues and has been linked with kidney dysfunction and cancer- related illness whereas, Pb can affect cognitive development in children (Jaishankar et al., 2014; WHO, 2021). Table 6 presents the HQ and Hazard Index HI values of heavy metals (Pb, Cd, Cu, Zn, Fe and Mn) for children and adults via consumption of edible vegetables from various dumpsites area. HQ results indicate that children experienced greater non-carcinogenic exposure risk than adults across most metals. Children recorded higher HQ values than adults for all assessed metals, confirming their greater vulnerability to dietary metal exposure. Fe was the major contributor to non-carcinogenic risk at stations 1 and 2 with the highest HQ value at station 2 (children: 22.58; adults: 5.81). At station 3, Mn was the dominant contributor with HQ values of 9.69 in children and 2.49 in adults. Other metals including Pb, Cu and Zn also contributed to the overall risk at certain locations indicating the potential health concerns associated with consumption of vegetables grown near dumpsites. The variation among the stations suggests that differences in waste composition, metal availability, and soil plant transfer processes is influenced by human exposure risk.

HI values obtained in this study (21.34-39.60 in children and 5.50-10.19 in adults) were higher than values reported in some vegetable contamination studies (Orisakwe et al., 2018, Ogunkunle et al., 2022), indicating greater cumulative non-carcinogenic exposure. Among the three stations, station 2 exhibited the highest HI values for both children and adults, indicating the greatest overall non-carcinogenic health risk. The elevated HI values may be associated with cultivation of vegetables near active dumpsites where continuous waste decomposition enhances metal availability. An HI value greater than 1 indicates a potential non-carcinogenic health concern, especially for vulnerable populations such as children. The HI values for both children and adults exceeded 1 across all stations, indicating potential non-carcinogenic health risks from combined metal exposure. Similarly, studies of HI above 1 in children and adults have been reported by Atikpo et al. (2021) and Akinkpelumi et al. (2025).

Table **5.** Estimated daily intake (EDI) of heavy metals in children and adult via consumption of vegetables from dumpsites

Metals	TDI ($mgkg^{-1}day^{-1}$)	S1 Children	S1 Adults	S2 Children	S2 Adults	S3 Children	S3 Adults
Pb	0.0036	0.0298	0.00766	0.00975	0.00251	0.00125	0.00488
Cd	0.0010	0.00018	0.00005	0.00108	0.00028	0.00023	0.00090
Cu	0.040	0.1643	0.0423	0.2113	0.0543	0.0181	0.0702
Zn	0.300	0.00903	0.00232	1.2247	0.3149	0.2050	0.7972
Fe	0.700	6.1479	1.5809	15.8064	4.0645	1.1994	4.6643
Mn	0.140	0.1107	0.0285	0.5790	0.1489	0.3487	1.3487

Table **6.** Hazard quotient (HQ) and hazard index (HI) of heavy metals in children and adults via consumption of vegetables from dumpsites

Metal	S1 Children	S1 Adults	S2 Children	S2 Adults	S3 Children	S3 Adults
Pb	7.45	1.92	2.44	0.63	1.22	0.31
Cd	0.18	0.05	1.08	0.28	0.90	0.23
Cu	4.11	1.06	5.28	1.36	1.76	0.45
Zn	0.03	0.01	4.08	1.05	2.66	0.68
Fe	8.78	2.26	22.58	5.81	6.66	1.71
Mn	0.79	0.20	4.14	1.06	9.69	2.49
HI	21.34	5.50	39.60	10.19	22.89	5.87

Table **7.** Cancer risk factor (CRF) and cumulative cancer risk index (CRI) of Pb and Cd in children and adult via consumption of vegetables from dumpsites

	Pb CRF		Cd CRF		CRI (Pb + Cd)	
Station	Adult	Child	Adult	Child	Adult	Child
S1	8.08×10^{-5}	2.17×10^{-4}	3.78×10^{-4}	1.01×10^{-3}	4.59×10^{-4}	1.23×10^{-3}
S2	2.64×10^{-5}	7.14×10^{-5}	2.21×10^{-3}	5.86×10^{-3}	2.24×10^{-3}	5.93×10^{-3}
S3	1.36×10^{-5}	3.57×10^{-5}	1.83×10^{-3}	4.85×10^{-3}	1.84×10^{-3}	4.89×10^{-3}

3.3.2 Carcinogenic risk

Table 7 presents the cancer risk factor (CRF) for Pb and Cd and the cumulative cancer risk index (CRI), which were calculated to assess potential long-term health risks associated with consumption of vegetables from the study areas. CRF for adults who consume these vegetables grown in the study stations ranged between 8.08×10^{-5}, 2.64×10^{-5} and 1.36×10^{-5} for Pb stations 1, 2 and 3, respectively. In contrast, Cd-related CRF values were higher, and ranging from 3.78×10^{-4} at station 1 to 2.21×10^{-3} at station 2. Consequently, the adult CRI ranged from 4.59×10^{-4} at station 1 to 2.24×10^{-3} at station 2, exceeding the acceptable lifetime cancer risk threshold of 10^{-4}.In children, CRF values were consistently higher than those of adults, with Pb CRF ranging from 3.57×10^{-5} to 2.17×10^{-4} and Cd CRF ranging from 1.01×10^{-3} to 5.86×10^{-3}. Although carcinogenic risk values for both adults and children exhibited the same spatial pattern across the three stations, the CRF values for children remained consistently higher than those for adults. This difference is attributable to the higher estimated daily intake per unit body weight in children, as the cancer slope factor is identical for each metal irrespective of age. Consequently, the nearly constant child-to-adult CRF ratio across all stations (approximately 2.7-fold for both Pb and Cd) reflects the age-specific exposure parameters incorporated into the risk assessment model, particularly body weight, rather than differences in contaminant concentrations among the sampling sites. The corresponding child CRI ranged from 1.23×10^{-3} at Station 1 to 5.93×10^{-3} at station 2, indicating the greater vulnerability of children to carcinogenic exposure. Spatially, station 2 exhibited the highest CRI for both adults and children, followed by station 3, with station 1 showing the lowest cumulative risk. This trend aligns with observed soil Cd concentrations and TFs, confirming that dietary intake risk is driven primarily by bioavailable Cd in the soil-plant system. Thus, the higher Cd contribution to CRI reflects both its carcinogenic slope factor and greater persistence in biological systems. Unlike Pb, Cd is efficiently retained in tissues and has a longer biological half-life, leading to cumulative exposure over time.

Conclusion

The results demonstrated that vegetables cultivated around the dumpsites accumulated heavy metals at varying concentrations across sampling locations. Although some metals were within recommended limits, elevated concentrations of Pb, Cu, Zn, and Fe were observed at certain locations, indicating the influence of dumpsite activities on vegetable contamination. The dietary exposure assessment revealed that Fe recorded the highest estimated daily intake (EDI) values among the analysed metals, with the greatest exposure observed at station 2. Although Fe played an important role in non-carcinogenic risk estimation by the observed HQ values, Pb and Cd represented greater toxicological concerns due to their non-essential nature and potential carcinogenic effects. The hazard index values exceeded the acceptable threshold at all sampling stations, indicating potential non-carcinogenic health risks from long-term vegetable consumption. Children were more vulnerable than adults due to their higher exposure relative to body weight. In addition, the carcinogenic risk assessment further indicates that dietary exposure to carcinogenic metals, particularly Cd, contributed to increased potential lifetime cancer risk among consumers. This study provides baseline evidence of dumpsite-associated heavy metal transfer into edible vegetables in Benin City and identifies vegetable consumption as a potential pathway for human exposure. The findings contribute to the limited data available for Benin City and provide scientific evidence to support environmental monitoring, agricultural land-use planning, and public health decision-making. Future research should investigate seasonal variations, expand spatial sampling, evaluate metal speciation to determine metal bioavailability, and incorporate dietary surveys specific to the local population. Such studies would improve understanding of long-term exposure and the associated health risks of consuming vegetables grown in contaminated environments.

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Conflict of Interest

The authors declare that there are no competing interests

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Author Contribution

IIE and OAO conceived the study; IIE conducted data analysis and interpretation, while OAO gathered and organized the data. IIE drafted the initial version of the manuscript, participated in reviewing and commenting on the draft. All authors approved the manuscript for publication after review

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