

Human Health Risk Assessment of Heavy Metals in Municipal Dumpsite Soils Using Monte Carlo Simulation: A Case Study From Kuje, Nigeria

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**HUMAN HEALTH RISK ASSESSMENT OF HEAVY METALS IN MUNICIPAL DUMPSITE
SOILS USING MONTE CARLO SIMULATION: A CASE STUDY FROM KUJE, NIGERIA**

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Abstract

This study assessed human health risks associated with heavy metal (HM) contamination in soils from the Agwan-Tiv municipal dumpsite, Kuje Area Council, Abuja, Nigeria. A total of ten soil samples (five from the dumpsite and five from a nearby uncontaminated control site) were collected using a stratified composite sampling design and analyzed for physicochemical properties and concentrations of arsenic (As), lead (Pb), cadmium (Cd), iron (Fe), and zinc (Zn) using atomic absorption spectrometry (AAS). Human health risk assessment (HHRA) was conducted for adults and children via ingestion and inhalation exposure pathways following the United States Environmental Protection Agency (USEPA) framework. Probabilistic modeling using Monte Carlo simulation (MCS; 10,000 iterations) was applied to quantify variability and uncertainty in carcinogenic risk estimates. Heavy metal concentrations at the dumpsite exceeded those at the control site for all metals, with Cd surpassing FAO/WHO permissible limits. Average daily intake (ADI) values via ingestion were substantially higher than via inhalation for both age groups, confirming ingestion as the primary exposure pathway. Hazard index (HI) values exceeded unity for As, Pb, Cd, and Fe in both adults (range: 1.76 – 62.1) and children (range: 0.012 – 40.40), indicating potential non-carcinogenic health risks. Incremental lifetime cancer risk (ILCR) values for As, Pb, and Cd ranged from Level V (medium-high) to Level VII (extremely high) for adults and Level III (low-medium) to Level VII (extremely high) for children. MCS results corroborated these findings, with children at the 95th percentile showing considerably elevated exposure levels. These outcomes reveal the need for soil remediation, improved waste management practices, further site-specific research and targeted public health interventions, particularly for children residing near the dumpsite.

Keywords: Health Risk Assessment; Monte Carlo Simulation; Heavy Metal Contamination; Hazard Index; Probabilistic Risk Assessment; Kuje; Nigeria

1. Introduction

Rapid urbanization and industrial growth have introduced large quantities of potentially toxic elements (PTEs) into soil environments worldwide. As concentrations of these contaminants rise, human exposure through incidental ingestion and inhalation can result in considerable public health concerns [1]. Traditionally, health risk assessments (HRAs) have relied on total PTE concentrations, however reliance on only total concentrations may lead to overestimation of risks [1, 3].

In Nigeria, approximately 32 million tonnes of solid waste are generated annually, yet only 20–30% is properly collected and disposed of [2]. The majority originates from industrial processes, wastewater discharge, and vehicle emissions, releasing persistent, non-biodegradable, and bioaccumulative pollutants that threaten both the environment and public health [3–5]. Improper waste handling in many developing nations frequently involves open dumping rather than engineered sanitary landfills. According to the Yale Center for Environmental Law and Policy, Nigeria ranks among the 30 worst-performing countries globally in waste management, with waste generation projected to increase at 2.4% annually [6]. Typical Nigerian municipal solid waste comprises approximately 57% food residues, 29% inorganic materials, 5% glass and metals, and 4% miscellaneous fractions [2].

Heavy metals (HMs) are elements with densities exceeding 5 g cm⁻³ in elemental form [7–10] and are ubiquitous in industrial applications and consumer products. Common toxic HMs including Pb, Cd, and As—originate primarily from industrial operations, mining, and agricultural inputs [11]. Excessive accumulation triggers severe adverse physiological effects: Pb is linked to impaired neurodevelopment in children and cardiovascular and renal disorders in adults [14, 15]; Cd exposure, mainly via diet and inhalation, causes renal dysfunction, skeletal damage, and increased cancer risk [16,17]; As is a well-established human carcinogen [3]. Beyond human health, HM contamination disrupts ecosystems by accumulating in biota and altering ecological balance [18–20].

Previous investigations in Kuje and adjacent Abuja districts have documented elevated HM levels in dumpsite soils [32–35], confirming that these sites pose persistent contamination risks. However, comprehensive probabilistic health risk evaluations for the Kuje area remain limited; available studies have been predominantly deterministic and have not explicitly quantified exposure uncertainty or variability. This study therefore addresses the following specific research gap: no probabilistic Monte Carlo-based HHRA has been conducted for the Agwan-Tiv dumpsite in Kuje, despite its importance as a populated peri-urban waste site.

It is acknowledged that bioavailability-based assessments would provide more refined risk estimates. However, as bioaccessibility data were not available for this site, total metal concentrations were used as a conservative worst-case estimate, consistent with standard USEPA guidance. This limitation is acknowledged in the discussion.

The objectives of this study are therefore to: (i) quantify HM concentrations in dumpsite and control soils at Agwan-Tiv, Kuje; (ii) apply a USEPA-based deterministic HHRA for adults and children via ingestion and inhalation; and (iii) employ Monte Carlo simulation to probabilistically characterise uncertainty in carcinogenic risk estimates, thereby providing a more robust basis for risk management decisions.

2. Materials and Methods

2.1 Study Area

The study was conducted at the Agwan-Tiv municipal waste dumpsite located in Kuje Area Council ($8^{\circ}52'46.27''\text{N}$, $7^{\circ}03'39.22''\text{E}$), Abuja, Federal Capital Territory, Nigeria (Figure 1). Kuje serves as the food production and distribution district of the FCT. Mean annual rainfall ranges from 900–2,200 mm, and mean daily temperature from 33.3°C to 36.0°C . The major rainy season extends from late March to September, with a secondary season in October–November and a short dry spell between July and August. Site topography is undulating, ranging from 80 m to 250 m above sea level.

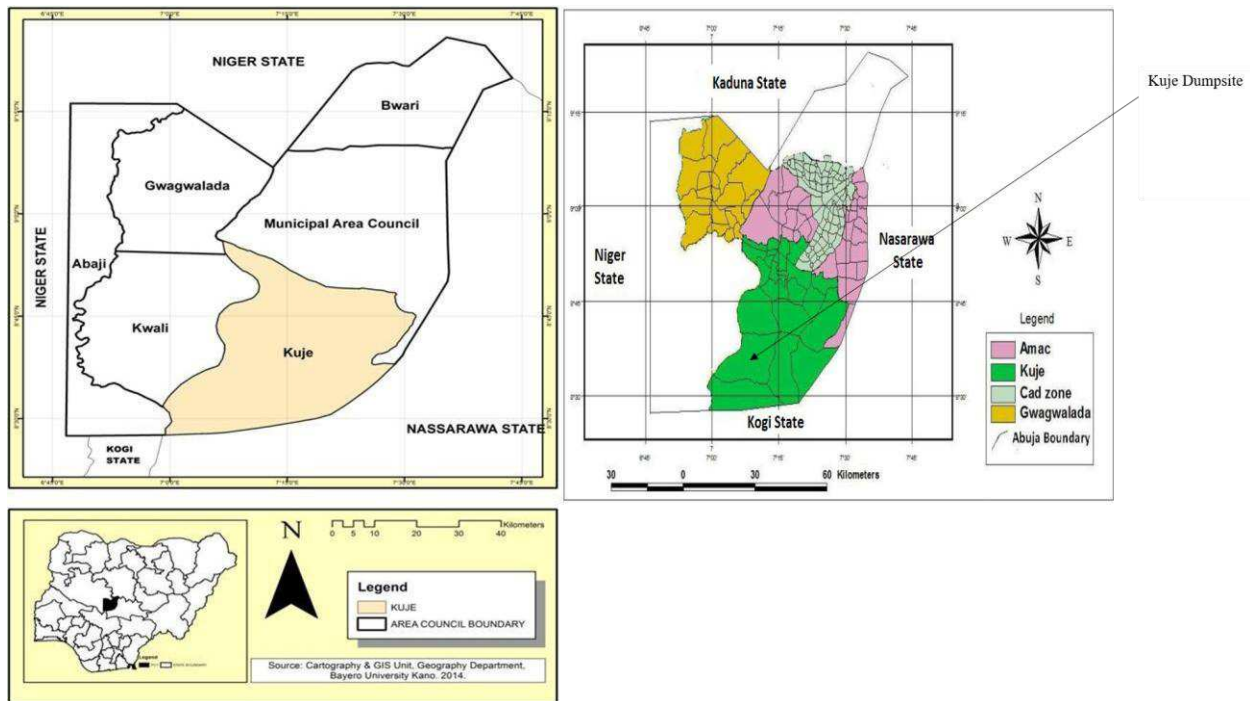


Figure. 1 Map of Kuje showing the approximate area of the dumping site (google map) [49].

2.2 Sample Collection

The study area was stratified into two zones: the Dump-site (the active waste disposal area) and the Control site (an uncontaminated control area located approximately 100 m upslope from the dumpsite perimeter, selected to minimize potential leachate influence). These terms are used consistently throughout the manuscript.

Soil samples were collected between 0 and 15 cm depth using a stainless-steel hand trowel, consistent with standard surface soil sampling protocols for contaminated site assessments [44–48]. At the Dump-site, five sampling points were established at the base perimeter (north, south, east, west, and a central arbitrary point) to capture spatial variability across the waste footprint. At the Control site, five sampling points were established at 100 m intervals along a transect line of the dumpsite. At each sampling point, five subsamples were collected within a 1 m^2 quadrat and bulked to form a single composite sample, yielding five composite samples per zone ($n = 10$ total). The limited sample size ($n = 10$) is acknowledged as a constraint on spatial representativeness and statistical power; findings will be interpreted as site-specific indicative estimates rather than spatially exhaustive characterizations. This is one major reason why the Monte Carlo probabilistic methodology is implemented as it gives random sampling concentration data.

All collected samples were air-dried at ambient temperature, crushed, and sieved through a 2 mm stainless-steel mesh prior to chemical analysis. Analyses were conducted at the Chemistry Laboratory, Margaret Lawrence University.

2.3 Analytical Methods

Soil physicochemical properties (pH, total nitrogen, available phosphorus, exchangeable potassium, magnesium, calcium, organic carbon, and electrical conductivity) were determined using standard soil analytical procedures. Bulk density, total porosity, and texture were measured following established protocols.

Heavy metal concentrations (As, Pb, Cd, Fe, Zn) were determined by flame atomic absorption spectrometry (AAS; Model AA-7000, Shimadzu Corporation, Japan) following acid digestion. Approximately 0.5 g of dried, sieved soil was digested in a mixture of concentrated HNO₃ and HClO₄ (4:1 v/v) at 120°C until a clear digest was obtained, then diluted to 50 mL with deionised water prior to AAS analysis.

Quality assurance and quality control (QA/QC) procedures included: (i) analysis of certified reference material (National Institute of Standards and Technology, Standard Reference Material - NIST SRM 2711a Montana II Soil) with recoveries within ±15% of certified values; (ii) procedural blanks processed in every batch to monitor contamination; (iii) duplicate analyses of 20% of samples (relative standard deviation < 10%); and (iv) multi-point calibration using certified standard solutions (Merck, Germany). Instrument detection limits (IDL) were: As 0.01 mg kg⁻¹, Pb 0.05 mg kg⁻¹, Cd 0.005 mg kg⁻¹, Fe 0.05 mg kg⁻¹, and Zn 0.01 mg kg⁻¹. All measured values exceeded the respective detection limits.

2.4 Human Health Risk Assessment Model

Human health risk assessment (HHRA) was conducted following the USEPA framework [38,39], categorizing risks as carcinogenic (CR) and non-carcinogenic (NCR). The population was stratified into adults and children based on age-specific exposure parameters. Two exposure pathways were assessed: ingestion and inhalation.

In this model dermal contact as a pathway was excluded despite being a standard pathway in soil HHRA. This is because it is considered minor to simplify the model and reduce uncertainty, particularly when ingestion already exceeds risk thresholds; this exclusion is acknowledged as a potential source of underestimation of total risk, particularly for children [50].

In this study, exposure assessment was carried out by measuring the average daily intake (ADI) of heavy metals which considered only two pathways: ingestion and inhalation (irrespective of the fact that soil HMs has little risk on inhalation compared to atmosphere). ADI is a key parameter in health risk analysis, it refers to the estimated amount of a substance (such as a pollutant or contaminant) ingested daily by an individual, typically expressed relative to body weight (e.g., mg/kg/day).

ADI (mg kg⁻¹ day⁻¹) for ingestion and inhalation pathways was calculated as [37, 8]:

$$ADI_{Ing} = \frac{C \times I_{ng} R \times EF \times ED}{BW \times AT} \times 10^{-6} \quad (1)$$

$$ADI_{Inh} = \frac{C \times I_{nh} R \times EF \times ED}{PEF \times BW \times AT} \times 10^{-6} \quad (2)$$

where C is the measured heavy metal concentration (mg kg⁻¹), IngR is the soil ingestion rate (mg day⁻¹), InhR is the inhalation rate (m³ day⁻¹), EF is exposure frequency (days year⁻¹), ED is exposure duration (years), BW is body weight (kg), AT is averaging time (days), and PEF is the particle emission factor (m³ kg⁻¹). All exposure parameters and toxicological constants used in the computational simulation are summarized in Table 1.

Table 1. Exposure parameters and toxicological constants used in the HHRA.

Parameter	Adults	Children	Reference
Ingestion rate (mg day ⁻¹)	100	200	[38,39]
Inhalation rate (m ³ day ⁻¹)	20	6.3	[38,39]
Exposure frequency (days year ⁻¹)	365	365	[38,40]
Exposure duration (years)	55	6	[38,40]
Body weight (kg)	70	25	[38,40]
Averaging time, non-cancer (days)	ED × 365	ED × 365	[38]
Particle emission factor (m ³ kg ⁻¹)	1.36 × 10 ⁹	1.36 × 10 ⁹	[38]

Non-carcinogenic risk was quantified as the hazard quotient (THQ) for each pathway and the hazard index (HI) as the sum across pathways:

$$THQ_{Ing} = \frac{ADI_{Ing}}{RfD} \quad (3)$$

$$THQ_{Inh} = \frac{ADI_{Inh}}{RfD} \quad (4)$$

$$HI = \sum_i^n THQ_i \quad (5)$$

where RfD is the reference dose. HI < 1 indicates that adverse health effects are unlikely even for sensitive populations; HI > 1 indicates potential for chronic non-carcinogenic health effects [38,40,41]. Incremental lifetime cancer risk (ILCR) for carcinogenic metals (As, Pb, Cd) was estimated as:

$$ILCR = ADI \times SF \quad (6)$$

where SF is the cancer slope factor (mg kg⁻¹ day⁻¹). ILCR < 10⁻⁶ is considered negligible; 10⁻⁶–10⁻⁴ falls within the acceptable regulatory range; ILCR > 10⁻⁴ is considered to require mitigation [38]. Iron (Fe) was included in the non-carcinogenic risk assessment because it was present at elevated concentrations relative to control soils and its HI contribution warrants reporting; it was excluded from carcinogenic assessment as no slope factor exists for Fe in USEPA databases. All reference doses and slope factors used are listed in Table 2.

Table 2. Reference doses and Slope Factors used in computation.

Metal Element	RfD Ingestion (mg/kg·day)	RfD Inhalation (mg/kg·day)	SF Ingestion (mg/kg·day) ⁻¹	SF Inhalation (mg/kg·day) ⁻¹
As	3.00E-04	3.01E-04	1.50E+00	1.51E-01
Pb	3.50E-03	3.25E-03	8.50E-03	—
Cd	1.00E-03	5.70E-05	3.80E-01	6.30E+00
Fe	7.00E-01	7.00E-01	—	—
Zn	3.00E-01	3.00E-01	—	—

2.5 Monte Carlo Simulation

Monte Carlo simulation (MCS) was implemented in Python 3.8.8 (Spyder IDE) using the NumPy and SciPy libraries. A total of 10,000 iterations were used, consistent with convergence criteria established in prior studies [15, 42]; pilot runs confirmed that key statistics (mean, 95th percentile) stabilized beyond 5,000 iterations, validating the selected iteration count.

Input variable distributions were assigned based on data characteristics and literature precedent [29–31]. HM concentrations were modelled as log-normal distributions ($\text{mean} = \ln(C_{\text{measured}})$, $\sigma = 0.5$), reflecting the typically right-skewed nature of environmental concentration data. Exposure rates were modelled as triangular distributions (minimum = 50% of nominal, mode = nominal, maximum = 150% of nominal), capturing plausible variation without requiring full distributional data. Output distributions for ADI proxy values were characterized by mean, median, 5th and 95th percentiles, and standard deviation.

3. Results

3.1 Soil Physicochemical Properties and Heavy Metal Concentrations

Selected physicochemical properties of dumpsite and control site soils are presented in the supplementary information in (Table S1). The dumpsite soils showed higher pH, organic carbon and nutrient concentrations compared to control site soils. Soil texture was classified as silt-clay loam (Dump-site) and silt loam (Control site), with bulk densities of 1.33 and 1.42 Mg m⁻³, respectively.

The Table 3 summarizes the Heavy metal concentrations for dumpsite and Control site. Dumpsite concentrations exceeded control site values for all metals, with the order Fe > Pb > Cd > As > Zn. The mean level of cadmium at the Dump-site (32.2 mg kg⁻¹) and control site (19.0 mg kg⁻¹) markedly exceeded the FAO/WHO permissible limit while the remaining metals (As, Pb, Fe, Zn) were within limits but still above the Control site levels. Pb at the Dump-site (40.2 mg kg⁻¹) was below the FAO/WHO limit of 50 mg kg⁻¹ but above the Control site value. Iron dominated the metal profile at both sites (719.7 and 320.3 mg kg⁻¹ respectively). No measured values were below the respective instrument detection limits.

181 **Table 3. Heavy metal concentrations (mg kg⁻¹) at study sites and comparison with reference ranges.**

Location	Site	As (mg/kg)	Pb (mg/kg)	Cd (mg/kg)	Fe (mg/kg)	Zn (mg/kg)
Control site	CS1	0.37	29.8	17.8	302.4	0.19
	CS2	0.39	30.6	18.6	315.7	0.20
	CS3	0.40	31.1	19.0	320.3	0.21
	CS4	0.41	31.6	19.4	326.1	0.22
	CS5	0.43	32.4	20.2	337.0	0.23
Mean		0.40	31.1	19.0	320.3	0.21
Dump-site	DS1	0.41	38.4	29.8	681.5	0.39
	DS2	0.43	39.6	31.1	704.2	0.41
	DS3	0.44	40.2	32.2	719.7	0.43
	DS4	0.45	40.8	33.1	735.6	0.45
	DS5	0.47	42.2	33.8	757.5	0.47
Mean		0.44	40.2	32.2	719.7	0.43

182 Descriptive statistics were used to summarize the concentrations of arsenic (As), lead (Pb), cadmium (Cd), iron (Fe),
183 and zinc (Zn) for the control and dump-site zones in Table 4 and 5. For each metal, the mean, standard deviation (SD),
184 range, and sample size (n = 5 per zone) were calculated. Differences in mean metal concentrations between the control
185 and dump-site zones were assessed using Welch's independent-samples t-test, which was selected because it does not
186 assume equal variances between the two groups. Statistical significance was evaluated at $\alpha = 0.05$, with two-tailed p-
187 values reported.

Table 4. Statistical Analysis for both Control-site and Dump-site respectively

Metal	Control site (n=5), Mean ± SD	Range	Dump-site (n=5), Mean ± SD	Range	Welch's t	df	p-value
As (mg/kg)	0.400 ± 0.022	0.37–0.43	0.440 ± 0.022	0.41–0.47	–2.83	8.00	0.022
Pb (mg/kg)	31.100 ± 0.985	29.80–32.40	40.240 ± 1.410	38.40–42.20	–11.88	7.15	<0.001
Cd (mg/kg)	19.000 ± 0.894	17.80–20.20	32.000 ± 1.592	29.80–33.80	–15.92	6.30	<0.001
Fe (mg/kg)	320.300 ± 12.788	302.40–337.00	719.700 ± 29.074	681.50–757.50	–28.12	5.49	<0.001
Zn (mg/kg)	0.210 ± 0.016	0.19–0.23	0.430 ± 0.032	0.39–0.47	–13.91	5.88	<0.001

Table 5. Concentration Comparison with regulatory standards other research values

Location/Source	As (mg/kg)	Pb (mg/kg)	Cd (mg/kg)	Fe (mg/kg)	Zn (mg/kg)
Control site	0.40	31.1	19.0	320.3	0.21
Dump-site	0.44	40.2	32.2	719.7	0.43
[54]	—	105.0 ± 7.1	39.0 ± 1.4	202.0 ± 1.4	0.7 ± 0.4
[32]	-	31.94 ± 12.99	18.70±8.97	-	584±200
[52]	-	79.75	3.0	-	265.75
FAO/WHO limit	20	50	3	100–1000	50–300
NESREA	20	164	3	-	421

3.2 Average Daily Intake

ADI values for ingestion and inhalation pathways are presented in supplementary information (Table S2) and comparison of these values between children and adult are seen in Figure 2 and 3. Ingestion ADI values were consistently higher than inhalation ADI values for both adults and children across all metals except for only Cd in adults, confirming ingestion as the dominant exposure pathway. Children showed higher ingestion ADI values than adults for all metals due to their greater ingestion rates and lower body weight. Fe and Pb contributed the highest ADI by ingestion.

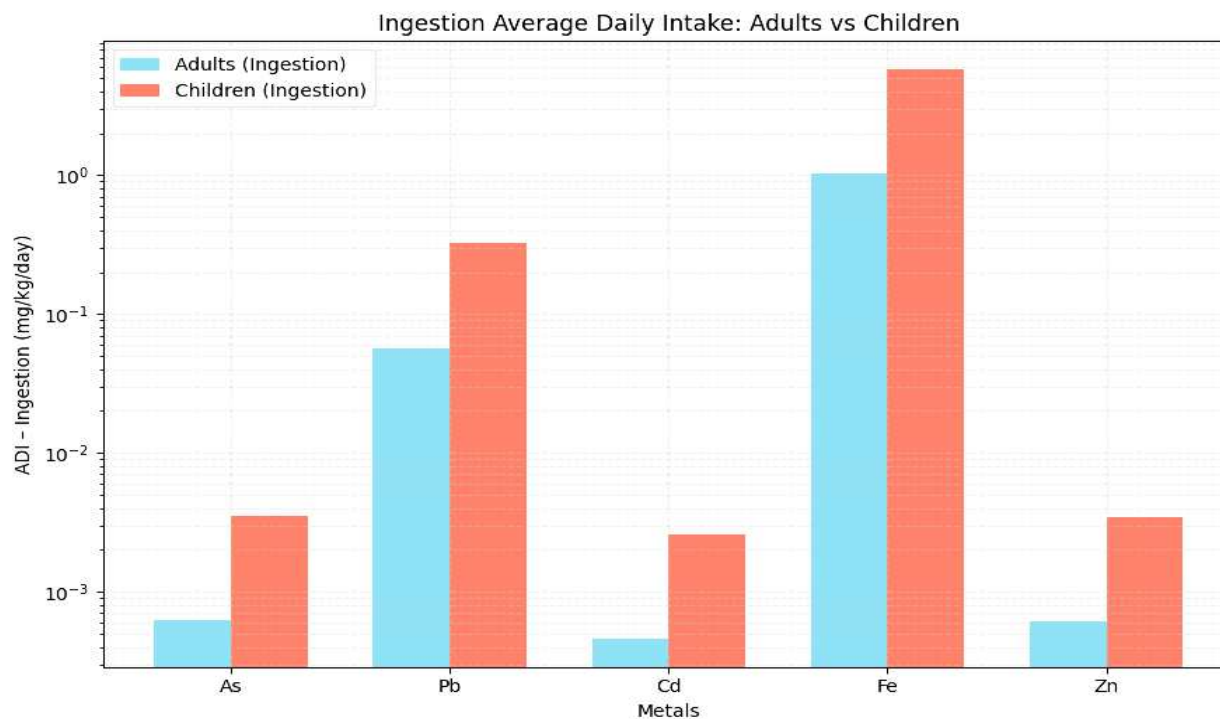


Figure 2. ADI comparison of Adult against Children for ingestion

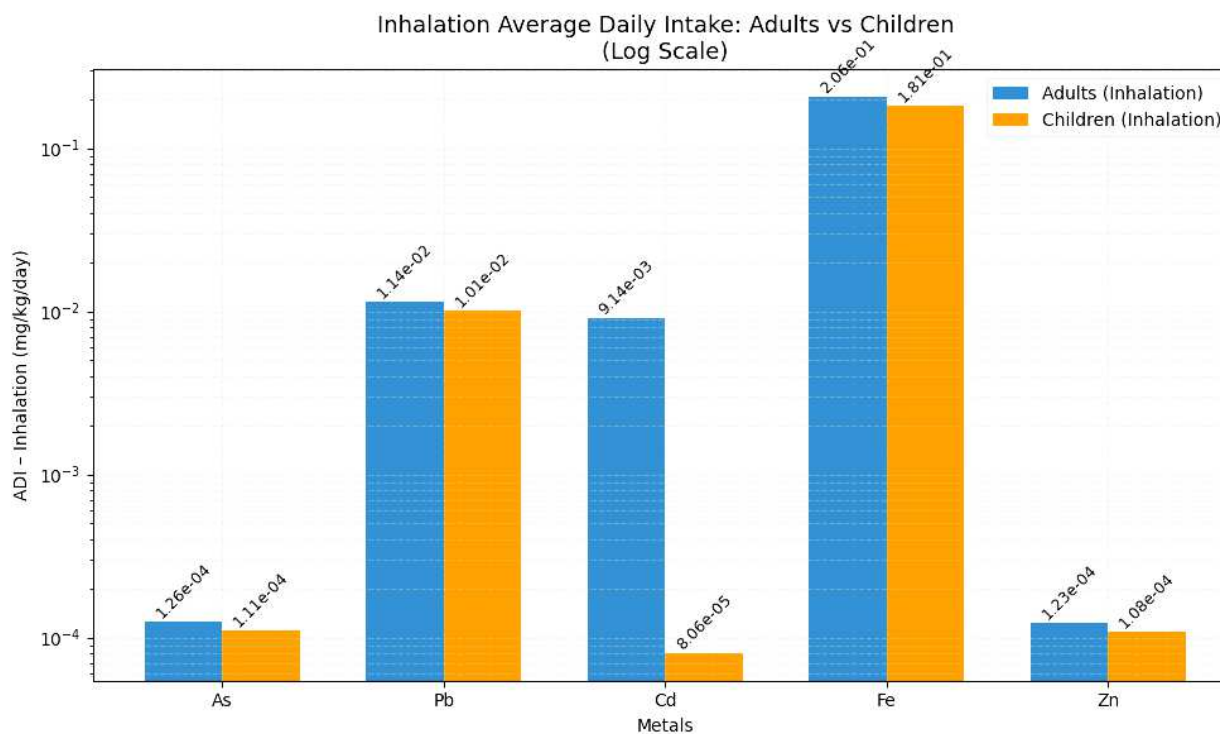


Figure 3. ADI Inhalation comparison for adult vs children

3.3 Non-Carcinogenic and Carcinogenic Risk Assessment

THQ, HI, and ILCR values are presented in Table 6. HI values exceeded unity(>1) for As, Pb, Cd, and Fe in both adults and children, indicating potential non-carcinogenic health risks from these metals except for Zn. Cd showed the highest HI in adults (62.1), attributable to the combined ingestion and inhalation THQ contributions and the low

RfD for Cd via inhalation. For children, Fe contributed the second-highest HI (8.48) after As (1.21×10^1). Zn HI values were below unity for both age groups (adults: 2.46×10^{-3} ; children: 1.18×10^{-2}). The Figure 4 visualizes the HI threshold for both age groups in comparison.

ILCR values for As, Pb, and Cd indicated medium-high to extremely high carcinogenic risk grades (Level V–VII) for adults and low-medium to extremely high (Level III–VII) for children [53]. Children showed higher ILCR values than adults for As and Pb, reflecting higher exposure due to age-specific parameters. It is important to note that these estimates are based on total metal concentrations and conservative exposure assumptions; actual risks may be lower if only a fraction of metals is bioavailable. These results should therefore be regarded as indicative upper-bound estimates pending site-specific bioaccessibility data.

Table 6. Health risk assessment results.

Metal	THQ ing Adults	THQ inh Adults	HI Adults	ILCR Adults	Risk Grade Adults	THQ ing Children	THQ inh Children	HI Children	ILCR Children	Risk Grade Children
As	2.1E+0	4.18E−1	2.52E+0	9.62E−4	Level VI	1.17E+1	3.68E−1	1.21E+1	5.30E−3	Level VII
Pb	1.64E+1	3.53E+0	1.99E+1	4.88E−4	Level V	9.19E−1	3.12E+0	4.04E+1	2.73E−3	Level III
Cd	4.60E+1	1.61E+1	6.21E+1	7.54E−2	Level VII	2.56E+0	1.41E+0	3.97E+0	1.48E−3	Level VII
Fe	1.47E+0	2.94E−1	1.76E+0	—	—	8.23E+0	2.59E−1	8.49E+0	—	—
Zn	2.05E−3	4.10E−4	2.46E−3	—	—	1.15E−2	3.61E−4	1.19E−2	—	—

Risk grades: Level I $< 10^{-6}$; Level II 10^{-6} – 10^{-5} ; Level III 10^{-5} – 5×10^{-5} ; Level IV 5×10^{-5} – 10^{-4} ; Level V 10^{-4} – 5×10^{-4} ; Level VI 5×10^{-4} – 10^{-3} ; Level VII $\geq 10^{-3}$ [53].

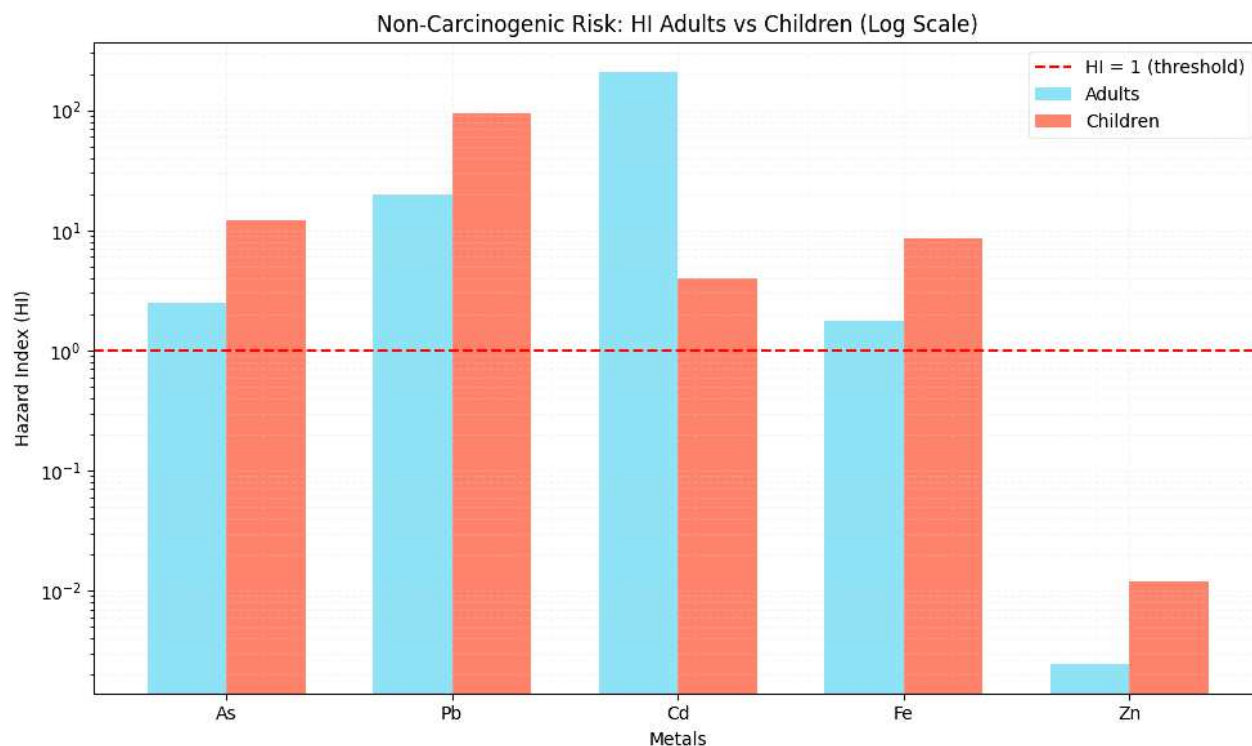


Figure. 4. Analysis of HI index results and comparison for adults against children

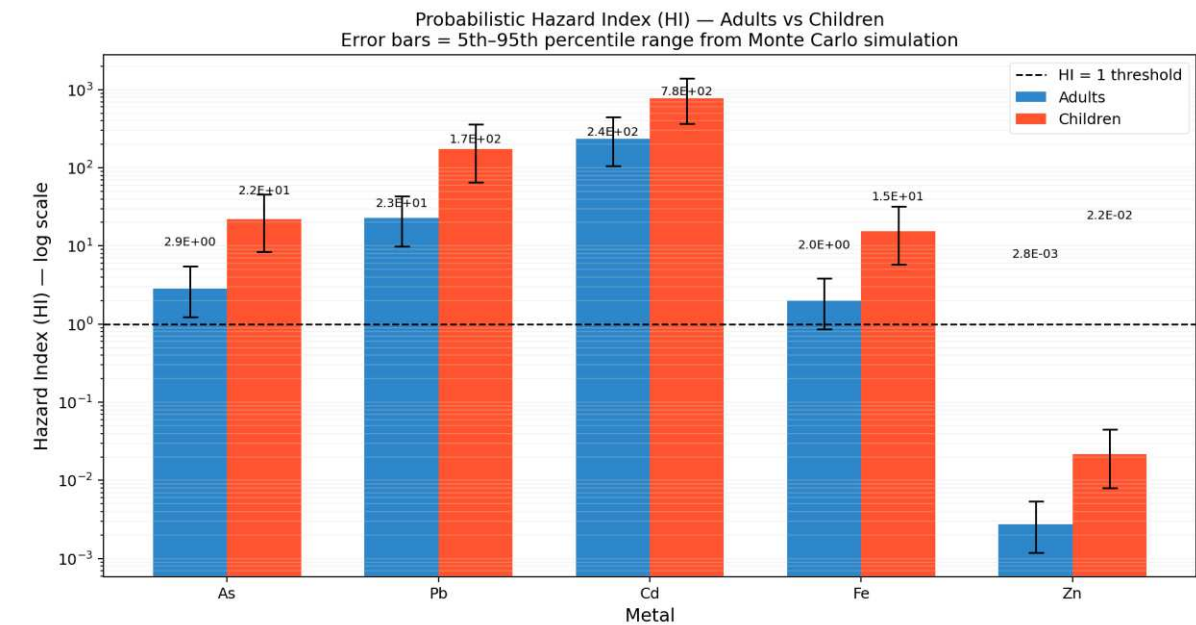
3.4 Monte Carlo Simulation Results

Monte Carlo simulation results for ADI-proxy distributions for As, Pb, and Cd are summarized in supplementary information (Table S3). Output distributions were right-skewed for all metals and both age groups, as evidenced by mean values consistently exceeding median values. Children exhibited higher mean and 95th-percentile values than adults across all metals, confirming greater vulnerability under probabilistic conditions. Standard deviations indicate substantial inter-individual variability, particularly for Pb and Cd in children. Furthermore, using the mean values of the ADI proxy distributions, Table 7 summarizes the probabilistic health risk assessment results. The Figure S3, S4, S5, S6, S7 and S8 (supplementary information) are the MCS plots for the probabilistic analysis of As, Pb and Cd for children and adults respectively. Furthermore, the data visualization of the HI index and ILCR for both age groups are illustrated in Figure 5 and 6 with markers and dotted line indicators for thresholds.

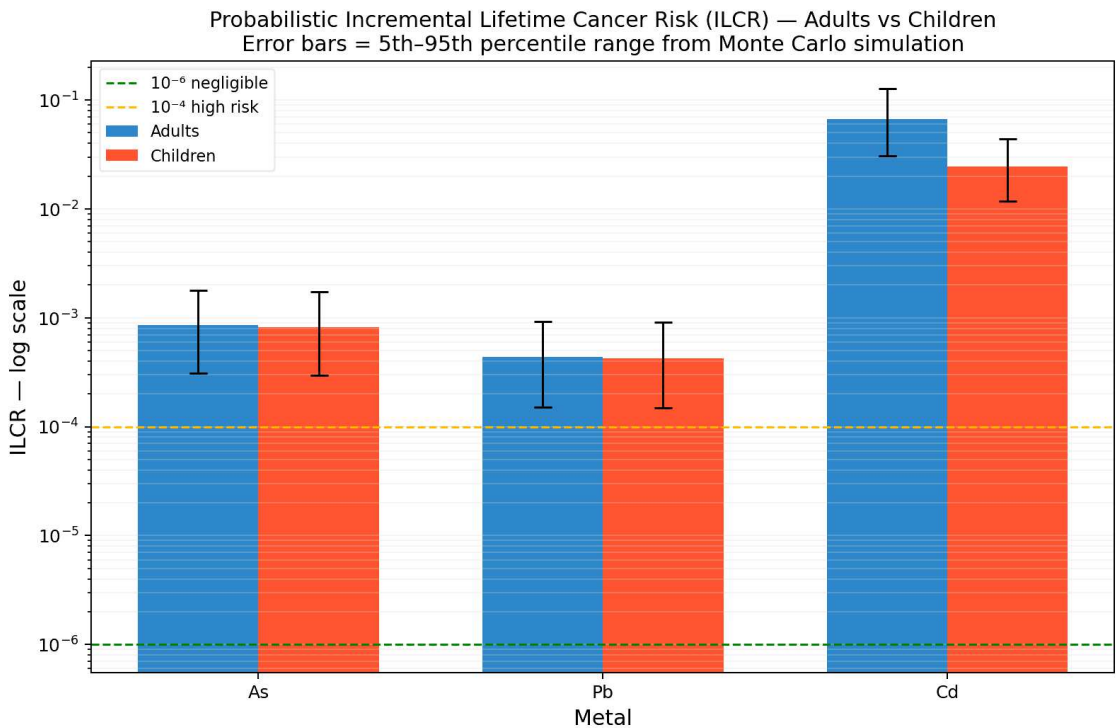
Table 7. Probabilistic Health risk assessment results using simulated mean ADI

Metal	THQ ing Adults	THQ inh Adults	HI Adults	ILCR Adults	Risk Grade Adults	THQ ing Children	THQ inh Children	HI Children	ILCR Children	Risk Grade Children
As	2.38E+00	4.71E-01	2.85E+00	8.60E-04	Level VI (high)	2.13E+00	8.13E-01	2.21E+00	8.25E-04	Level VI (high)
Pb	1.87E+01	4.01E+00	2.27E+00	4.36E-04	Level V (medium-high)	1.67E+00	6.85E+00	1.74E+00	4.27E-04	Level V (medium-high)
Cd	5.22E+01	1.83E+02	2.35E+00	6.73E-02	Level VII (extremely high)	4.63E+00	3.13E+00	7.75E+00	2.47E-02	Level VII (extremely high)
Fe	1.66E+00	3.31E-01	1.99E+00	-	-	1.49E+00	5.70E-01	1.54E+00	-	-
Zn	2.30E-03	4.62E-04	2.76E-03	-	-	2.09E-02	7.90E-04	2.17E-02	-	-

234 Risk grades: Level I $< 10^{-6}$; Level II 10^{-6} – 10^{-5} ; Level III 10^{-5} – 5×10^{-5} ; Level IV 5×10^{-5} – 10^{-4} ; Level V 10^{-4} –
 235 5×10^{-4} ; Level VI 5×10^{-4} – 10^{-3} ; Level VII $\geq 10^{-3}$ [53].



236 Figure. 5: Probabilistic Hazard Index (HI) - Adult vs Children.



237 Figure.6. Probabilistic Incremental lifetime cancer risk (ILCR) - Adults and Children

4. Discussion

4.1 Concentration of heavy metals in soil samples

The descriptive and inferential statistical results are presented in Table 4. Arsenic concentrations increased from 0.40 ± 0.022 mg/kg at the control site to 0.44 ± 0.022 mg/kg at the dump site, representing a 10.0% increase and a statistically significant difference between the zones ($p = 0.022$). Lead concentrations increased from 31.10 ± 0.985 mg/kg to 40.24 ± 1.410 mg/kg, with a significant difference between zones ($p < 0.001$). Similarly, cadmium increased from 19.00 ± 0.894 mg/kg at the control site to 32.00 ± 1.592 mg/kg at the dump site ($p < 0.001$). Iron showed the largest absolute difference, increasing from 320.30 ± 12.788 mg/kg to 719.70 ± 29.074 mg/kg, with a statistically significant difference ($p < 0.001$). Zinc concentrations also increased from 0.21 ± 0.016 mg/kg to 0.43 ± 0.032 mg/kg, representing an approximately 104.8% increase ($p < 0.001$). Overall, the dump-site zone exhibited significantly higher concentrations of all five investigated metals than the control zone.

Heavy metal concentrations at the dump-site exceeded control site values for every metal analyzed, a pattern consistent with anthropogenic input from waste disposal activities. The dominance of Fe corroborates findings from comparable Nigerian dumpsite studies [32–35], where Fe accumulation is attributed to scrap metal deposition, ferrous waste, and corrosion products. The National Environmental Standards and Regulations Enforcement Agency (NESREA) in conjunction with the Food and Agricultural Organization/World Health Organization (FAO/WHO) stipulated the permissible limits for As, Cd, Pb, and Zn as 20 mg/kg, 3 mg/kg, 164 mg/kg and 421 mg/kg, respectively [32,54,55]. Of greater concern is the marked elevation of Cd above FAO/WHO and NESREA permissible limits, which is plausibly linked to industrial emissions, smelting operations, and the sewage sludge disposal previously documented in the study area. Concentration ratios alone cannot establish source attribution with certainty; however, the consistent and statistically meaningful difference between dump-site and control site values across multiple metals lends support to an anthropogenic origin. Rigorous confirmation would require formal source apportionment through principal component analysis (PCA) or positive matrix factorization (PMF), combined with enrichment factor (EF) assessment. This is identified as a priority for future work.

4.2 Average Daily Intake and Non-Carcinogenic risk estimates

Ingestion clearly drove exposure, producing much higher ADI values than inhalation in line with prior soil HM studies [24, 25, 29,52] and unsurprising given how much soil children in particular tend to ingest. Fe's ADI values were high simply because Fe concentrations at the site were high. We kept it in the non-carcinogenic assessment because its HI > 1 , even though it carries far less toxicological weight than As, Pb, or Cd. Dermal exposure was considered a secondary pathway because metal uptake through intact skin from soil is generally limited. Heavy metals are commonly bound to soil particles, reducing desorption and bioavailability, while ionic metal species show low permeability across the skin barrier. US EPA guidance indicates that dermal risks from metals in soil are generally minor relative to direct soil ingestion, with reported dermal risks ranging from less than 1% to 11% of oral risks and averaging approximately 2.5% [56]. Hence, this study did not take into account the dermal effect.

Non-carcinogenic HI values are greater than one ($HI > 1$) for As, Pb, Cd, and Fe in both age groups, indicating potential for chronic health effects from cumulative HM exposure at this site. The extremely high HI for Cd in adults (62.1) calls for careful interpretation, while the calculation is consistent with the USEPA framework and the measured concentration, the very low RfD for Cd via inhalation (5.7×10^{-5} mg kg⁻¹ day⁻¹) amplifies the inhalation THQ contribution substantially. This result is mathematically consistent but reflects the conservative nature of the assessment rather than necessarily indicating imminent clinical toxicity. Comparable elevated HI values have been reported at similar dumpsite studies in Nigeria and sub-Saharan Africa [32–35, 44–48]. In similar research [52] reported HI values as ‘unacceptable’ on the Kuje dump-site. According to a World Health Organization report, children are a vulnerable population to health risks via ingestion because they drink more water, consume more food, and breathe more air in proportion to their weight. Children's immune, digestive, reproductive, and nervous systems are still growing. At the early part of development, exposure to toxic elements causes irreversible damage [52].

4.3 Carcinogenic risk estimates

Carcinogenic risk estimates (ILCR) indicated medium-high to extremely high risk levels for As and Cd across both age groups [52]. Children exhibited consistently higher cancer risk for As and Pb than adults, reflecting age-specific differences in exposure. The ILCR value for Cd in adults was notably elevated (7.54×10^{-2}), similarly reflecting the

combined contribution of ingestion and inhalation pathways relative to a low slope factor denominator. These results are consistent with findings from comparable studies conducted in Nigeria [32,35] and internationally [3,21,24]. It should be noted that these values represent indicative upper-bound estimates derived from total metal concentrations and deterministic exposure assumptions. The probabilistic estimates presented in Table S1 offer a more nuanced characterization of risk distribution across the exposed population.

4.4 Monte Carlo simulation distribution

Monte Carlo simulation outputs corroborated the deterministic findings while providing additional insight into exposure variability. The right-skewed distributions presented in Figures S3–S8 (supplementary information) indicate that although most individuals experience risk levels near the median, a subset of the population is subject to substantially elevated exposure at the upper tail (95th percentile). This finding carries direct policy relevance, as risk management strategies should be designed to specifically address high-exposure subgroups rather than population averages. The comparatively larger standard deviations observed for Pb and Cd in children reflect the combined influence of concentration variability (log-normal distribution, $\sigma = 0.5$) and exposure rate variability (triangular distribution), consistent with variability patterns reported in prior MCS-based risk assessments conducted in comparable settings [29–31].

It is acknowledged that this study has limitations. The sample size ($n = 10$) limits spatial representativeness and constrains formal statistical comparisons between sites is insufficient to achieve adequate statistical power. Bioavailability of metals was not measured, total concentrations were used as a conservative proxy. The dermal pathway was excluded from this assessment due ingestion is consistently reported as the dominant exposure pathway for soil-based heavy metal contamination, with dermal contribution typically representing a minor fraction of total risk because of contact unavailability. Input distributions for MCS were assigned based on literature precedent rather than site-specific distributional data. Future studies should increase sample size for deterministic approach, incorporate bioaccessibility measurements, include the dermal pathway, and conduct formal sensitivity analysis to identify the key drivers of risk uncertainty.

5. Statistical Considerations

The Monte Carlo simulation provides probabilistic estimates of exposure distributions and implicitly accounts for parameter uncertainty, representing the primary quantitative uncertainty analysis in this study.

6. Conclusions and Recommendations

This study is a preliminary, site-specific risk estimate that characterized heavy metal contamination and associated human health risks at the Agwan-Tiv municipal dumpsite, Kuje, Abuja, Nigeria. Dumpsite soils showed elevated concentrations of As, Pb, Cd, Fe, and Zn relative to the control site, with Cd exceeding FAO/WHO and NESREA permissible limits. Risk assessment indicated potential non-carcinogenic health risks ($HI > 1$) for As, Pb, Cd, and Fe in both adults and children, and indicative upper-bound carcinogenic risks ranging from medium-high to extremely high for As, Pb, and Cd. Children were identified as the more vulnerable group, exhibiting higher ILCR and HI values for most metals. Monte Carlo simulation confirmed the deterministic findings and demonstrated that a subset of high-exposure individuals, particularly children, may face risks substantially above median estimates.

These results provide a scientifically grounded, indicative basis for risk management prioritization at this site.

The following actions are recommended based on these findings:

- Soil remediation at the dumpsite, prioritizing Cd and As hotspots.
- Regulation and improvement of waste disposal practices at the site.
- Community health awareness programmes targeted at families with young children residing in proximity to the dumpsite.
- Follow-up studies incorporating bioaccessibility measurements, larger sample sizes, formal source apportionment, and dermal pathway assessment.
- Periodic monitoring of HM concentrations to track temporal changes at the site.

7. CRediT authorship contribution statement

Imah MO: Conceptualization, Methodology, Data curation, Formal analysis, Visualization, Writing - original draft, Writing - review & editing. **Ofomola M:** Methodology, Project administration, Resources, Writing - review & editing, Supervision. **Akpojotor GE:** Project administration, Writing - review & editing, Supervision.

8.Declarations

The authors declare no competing interests.

9. Ethics approval and consent to participate

All ethical guidelines and research protocols were strictly followed.

Consent for publication

We hereby declare that this research work titled “*Human Health Risk Assessment of Heavy Metals in Municipal Dumpsite Soils Using Monte Carlo Simulation: A Case Study from Kuje, Nigeria*” is original work and has not been submitted in part or in full for any degree, diploma, or qualification at any other university or institution. We hereby declare a consent for it to be published

Consent to Participate

Not applicable

Availability of data and materials

All data generated or analysed during this study are included in this published article [and its supplementary information files].

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