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Impact of Fuel Station Traffic Intensity on Heavy Metal, Polycyclic Aromatic Hydrocarbon, and BTEX Contamination of Soil and Road Dust: Distribution, Source Apportionment, Ecological Status, and Human Health Risk Assessment in Abuja Municipal Area Council, Nigeria

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Abstract

Fuel stations represent complex environmental interfaces where petroleum handling and vehicular activity can contribute to contaminant accumulation in surrounding soils and road dust. This study investigated heavy metals, polycyclic aromatic hydrocarbons (PAHs), BTEX, source characteristics, ecological status, and human-health risks across high-traffic fuel stations (HTFS), low-traffic fuel stations (LTFS), and control sites in Abuja Municipal Area Council, Nigeria. Soil and road-dust samples were analysed for Ni, Cd, V, Pb, 16 priority PAHs and BTEX, followed by contamination indices, PAH diagnostic ratios, multivariate source assessment, deterministic risk estimation, and Monte Carlo simulation for adult and child receptors. Metal concentrations were low relative to adopted Nigerian petroleum-site intervention values, with Ni generally predominant. Σ PAHs ranged from 10.58–27.37 mg/kg in soil and 28.96–100.17 mg/kg in dust, with LTFS dust showing pronounced high-molecular-weight PAH enrichment (84.57%). BaP_{eq} reached 27.239 mg/kg in LTFS dust, corresponding to ingestion-based ILCRs of 1.17×10^{-4} for adults and 2.18×10^{-4} for children. Adult BTEX hazard indices remained below unity, whereas child HIs reached 2.49, principally associated with xylenes. Benzene-related cancer risks were also higher in children, reaching 5.60×10^{-4} . In contrast, metal contamination factors and potential ecological risk indices indicated low ecological concern. Probabilistic simulations generally produced low ingestion-related risk distributions, although upper-tail child risks persisted in selected dust scenarios. Overall, PAHs and BTEX represented more important screening-level health concerns than the investigated metals, highlighting the need for improved dust, fuel-emission, and environmental monitoring controls around fuel stations.

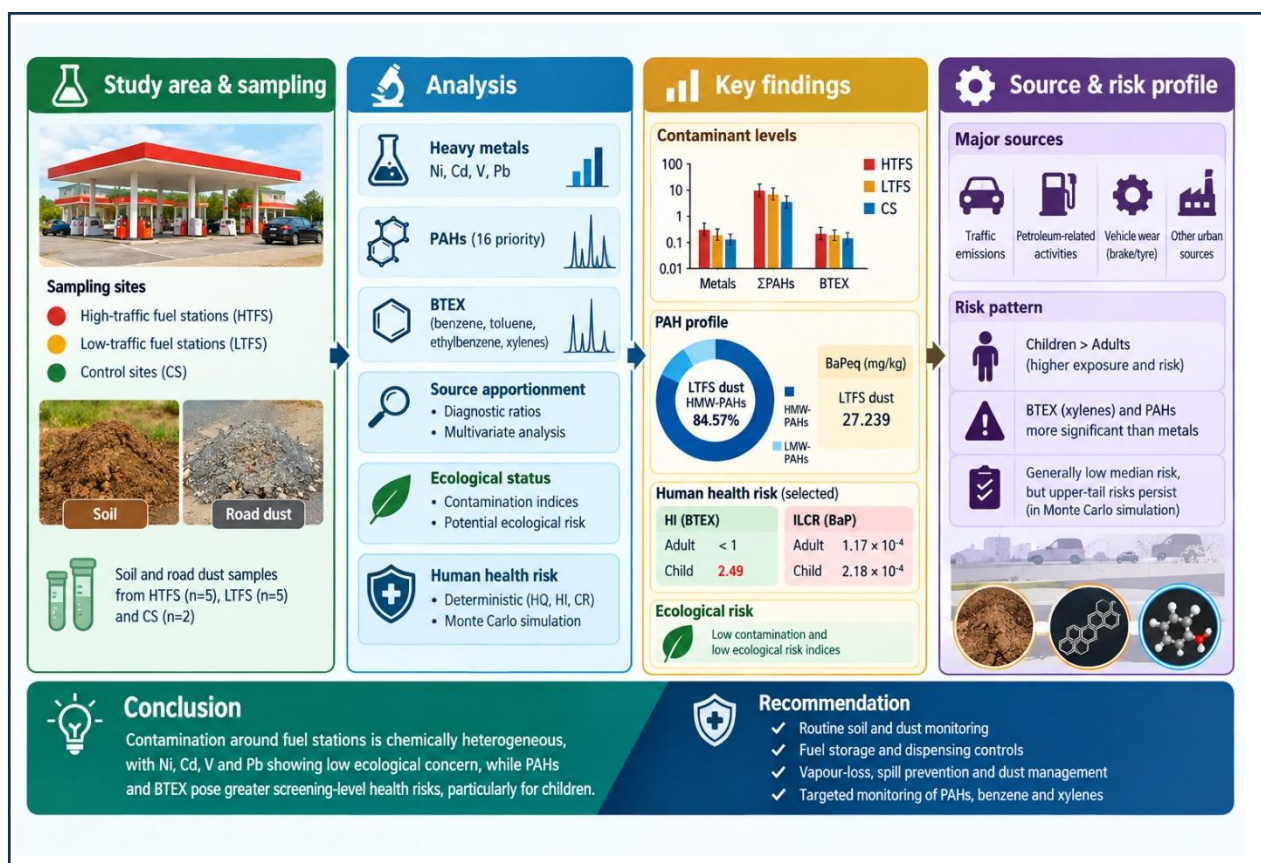
Keywords: Fuel stations; Road dust; PAHs; BTEX; Human-health risk; Abuja

Highlights

1. Integrated assessment linked fuel-station activity with metals, PAHs, BTEX and health risks.

2. Metal contamination and ecological-risk indices indicated consistently low concern.
3. LTFS road dust showed pronounced high-molecular-weight PAH enrichment.
4. Child BTEX hazard indices exceeded unity in selected soil and dust scenarios.
5. Monte Carlo analysis identified persistent upper-tail risks despite generally low median exposure.

Graphical Abstract



1. Introduction

Fuel stations represent important point sources of petroleum-associated contaminants within rapidly urbanising environments because fuel storage, transportation, dispensing and vehicle activity occur in close proximity to roads, commercial establishments and residential areas (Zheng et al., 2025; Opara & Pavshintsev, 2026). Accidental spills, leakage from underground storage

systems, fuel handling losses, lubricants, evaporative emissions and stormwater transport can introduce petroleum-derived contaminants into surrounding environmental compartments (Molua et al., 2024). At the same time, fuel stations are commonly located along busy transportation corridors where vehicle exhaust, brake and tyre wear, lubricating-oil losses and resuspended road materials provide additional sources of potentially toxic substances (John et al., 2026; Koley et al., 2026). Consequently, contamination measured around fuel stations cannot necessarily be attributed to petroleum-retailing activities alone.

Soil and road dust are particularly important environmental matrices for investigating this source complexity (Sager, 2020; Sun et al., 2023). Soil acts as a relatively persistent sink for metals and hydrophobic organic contaminants, whereas road dust represents a dynamic reservoir that receives atmospheric deposition and mechanical wear products and can subsequently be redistributed by wind, vehicle movement and stormwater runoff (Sager, 2020; Chen et al., 2023; Kolawole et al., 2026). Their close association with human activity also creates multiple exposure pathways, including incidental ingestion, dermal contact and inhalation of resuspended particles (Zhang et al., 2021). Studies from Nigeria have demonstrated significant contamination of roadside soils and dust by potentially toxic elements and PAHs, with traffic, combustion, industrial activities and petroleum-related processes contributing to observed contaminant patterns (George et al., 2024; Enuneku et al., 2021; Akinsete, & Olatimehin, 2024). Also, in Ibadan, roadside soils from high-traffic environments contained measurable PAHs and elevated Cd, Ni and Pb, while diagnostic ratios indicated mixed pyrogenic and petrogenic sources (Akinsete, & Olatimehin, 2024). Recent study done in Abuja have likewise identified traffic-related activities as important contributors to potentially toxic elements in urban dust (Ameen et al., 2026).

Heavy metals such as nickel (Ni), cadmium (Cd), vanadium (V) and lead (Pb) are of particular environmental concern because they are persistent and may accumulate in soil and dust over time (Chowdhury & Rahman, 2024). Their occurrence around petroleum facilities may reflect petroleum-associated inputs, fuel combustion, vehicular wear, lubricants and other urban activities. George (2024) who investigated of oil-impacted soils around petroleum-product retailing stations in Nigeria, demonstrated anthropogenic enrichment and identified vehicular emissions, atmospheric deposition and petroleum-related activities as plausible contributors. The environmental significance of these elements extends beyond their concentrations because their

persistence and potential mobility can contribute to long-term ecological exposure (Edo et al., 2024).

Polycyclic aromatic hydrocarbons (PAHs) and BTEX (benzene, toluene, ethylbenzene and xylenes) are important contaminants in fuel-station and traffic environments. PAHs originate from petroleum residues and incomplete combustion of gasoline and diesel, with high-molecular-weight compounds strongly associated with particulate matter; several, including benzo[a]pyrene, are carcinogenic or potentially carcinogenic (Akporthor et al., 2022). BTEX are associated with gasoline, fuel evaporation, dispensing and vehicular emissions, with benzene of particular concern because of its carcinogenicity (Muda et al., 2024; Das et al., 2025).

Source attribution around fuel stations is challenging because these facilities commonly coexist with major roads, vehicle-repair activities and other anthropogenic sources. Thus, elevated concentrations alone cannot establish contaminant origin. Integrating concentration patterns with traffic intensity, spatial distribution, correlations, multivariate analyses and PAH diagnostic ratios provides stronger evidence for distinguishing fuel-station, traffic-related and background contributions. Comparing high-traffic fuel stations (HTFS), low-traffic fuel stations (LTFS) and control sites (CS) can further clarify the influence of traffic and petroleum-retailing activities relative to urban background conditions.

Contamination assessment should also consider ecological and human-health implications (Ahmad et al., 2021). Indices such as contamination factor, geo-accumulation index, enrichment factor and potential ecological risk index provide complementary measures of contamination and ecological concern (Udo et al., 2026), while human-health risk assessment can evaluate exposure through ingestion, dermal contact and inhalation of contaminated soil and road dust (Rehman et al., 2025). Children may experience greater risks because of their exposure behaviour and lower body mass (Mama et al., 2020). Probabilistic approaches such as Monte Carlo simulation can further capture variability and uncertainty in exposure and risk estimates (Shahin, 2026; Zakari-Jiya et al., 2026).

Despite increasing evidence of contamination around Nigerian fuel stations and urban roads (Ulakpa et al., 2022; Adedeji et al., 2022; George et al., 2024; Anigilaje et al., 2024), integrated assessments simultaneously examining traffic intensity, heavy metals, PAHs, BTEX, source characteristics, ecological risk and human-health implications in both soil and road dust remain

limited, particularly in Abuja. This study therefore evaluates these contaminants across HTFS, LTFS and control locations to better characterise contamination patterns, potential sources and associated ecological and human-health risks.

The present study therefore investigated the impact of fuel-station traffic intensity on the occurrence, spatial distribution, source characteristics, ecological status and potential human-health risks of heavy metals, PAHs and BTEX in soil and road dust around fuel stations in Abuja Municipal Area Council (AMAC), Nigeria. Specifically, the study aimed to: (i) determine the concentrations and distribution of selected heavy metals, 16 priority PAHs and BTEX in soil and road dust; (ii) compare contaminant burdens among high-traffic, low-traffic and control locations; (iii) evaluate the relationships between contaminant concentrations and traffic intensity; (iv) characterise contamination using contamination factor, geo-accumulation, enrichment and ecological-risk indices; (v) quantify deterministic and probabilistic non-carcinogenic and carcinogenic risks for adult and child receptors.

2. Materials and Methods

2.1 Study area

The study was conducted in Abuja Municipal Area Council (AMAC), Federal Capital Territory, Nigeria (Fig. 1). AMAC is characterised by rapid urbanisation, extensive road networks, residential and commercial development, substantial vehicular activity and numerous petroleum-product retailing stations. Fuel stations are distributed across roads with contrasting traffic environments and are frequently located near commercial facilities, residential areas, vehicle-repair activities and other anthropogenic sources. This environmental setting provides an appropriate natural gradient for evaluating the influence of traffic intensity on contaminant accumulation in soil and road dust.

The study area and sampling locations were mapped using geographical coordinates obtained during field surveys. The environmental setting surrounding each fuel station was evaluated within a 50-m radius, consistent with the contextual source-assessment framework used in the accompanying Abuja investigation.

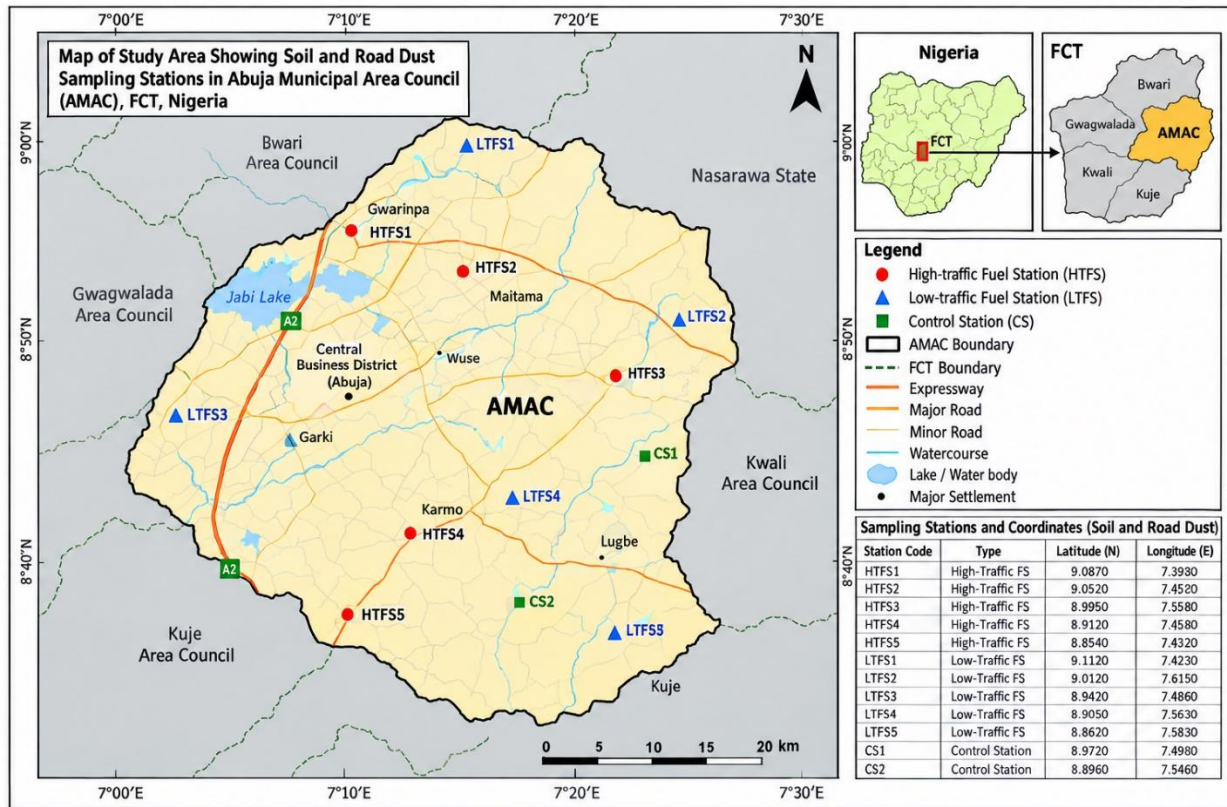


Fig. 1: Map of Study Area Showing Sampling Stations in Abuja Municipal Area Council (AMAC), FCT, Nigeria.

2.2 Study design and sampling strategy

A comparative cross-sectional environmental monitoring design was employed. Sampling locations were categorised into HTFS, LTFS and CS. The classification was based on road environment, observed vehicle activity, station throughput and traffic characteristics. HTFS were generally located along major or heavily trafficked roads and exhibited greater vehicle activity, whereas LTFS were located along relatively quieter roads with lower station activity. Control locations were selected away from direct fuel-station influence to provide an estimate of local background conditions.

Traffic intensity was characterised using direct field observations and, where available, station vehicle-throughput records. The classification represented fuel-station activity and traffic environment rather than being interpreted as a direct measurement of total road traffic. This

distinction is important because station throughput does not necessarily equal the total number of vehicles using the adjoining road.

Soil and road-dust samples were collected from each of the 12 designated sampling stations. Soil was collected from the upper 0–5 cm layer using acid-cleaned, non-contaminating equipment, while road dust was collected from representative depositional surfaces adjacent to the fuel stations and roadways using clean polyethylene or stainless-steel tools. Approximately 500 g of soil and 250 g of road dust were collected at each station to provide sufficient material for homogenisation, physicochemical characterisation and subsequent contaminant analysis. At each station, five spatially distributed increments were collected for each matrix and composited to obtain a representative station sample. The resulting composite samples were transferred into pre-cleaned, appropriately labelled containers, assigned unique identification codes, and transported to the laboratory for preparation and analysis.

2.3 Field characterisation and environmental setting

A structured field assessment was conducted at each sampling location to document variables relevant to contaminant occurrence and source interpretation. These included traffic intensity, road classification, observed vehicle activity, proximity to fuel pumps, proximity to underground storage-tank infrastructure where observable, evidence of petroleum spills, petroleum odour, fuel offloading activities, drainage characteristics and surrounding land use.

Potential secondary sources within the 50-m assessment zone were also documented, including vehicle-repair workshops, car-wash facilities, motor parks, other fuel stations, lubricant-related activities, waste-disposal areas, commercial developments, industrial activities and CNG/gas facilities. These observations were treated as contextual evidence and were not interpreted individually as proof of contaminant origin, consistent with the integrated source-apportionment approach of the Abuja water study.

2.4 Soil and road-dust sample preparation

Samples designated for metal and organic-contaminant analyses were air-dried at room temperature in a clean, dust-free environment. Visible foreign materials, including stones, plant fragments and other non-representative debris, were carefully removed prior to homogenisation.

The dried samples were gently disaggregated using cleaned equipment and sieved through 2-mm and 63- μm pre-cleaned sieves for soil and road dust, respectively. The sieved fractions were thoroughly homogenised to obtain representative analytical material. Homogenised samples intended for organic-contaminant analysis were stored in clean amber-glass containers, whereas those for metal analysis were stored in acid-cleaned, contamination-resistant containers. All prepared samples were kept under appropriate laboratory storage conditions until subsequent chemical extraction, digestion and instrumental analysis.

2.5 Heavy-metal analysis

Ni, Cd, V and Pb were selected as target metals based on their relevance to petroleum-associated and traffic-related contamination and consistency with the analytical framework of the accompanying Abuja study.

For metal determination, an accurately weighed portion of homogenised soil or road dust was subjected to acid digestion using analytical-grade acids under controlled laboratory conditions. Following digestion, samples were cooled, filtered where necessary and diluted to a known final volume using ultrapure water.

Metal concentrations were determined using atomic absorption spectrophotometry (AAS; Shimadzu AA-7000, Shimadzu Corporation, Japan), using the appropriate atomisation mode for each analyte. Multi-point calibration standards prepared from certified stock solutions were used for quantitative determination. Calibration acceptance criteria were established before sample analysis, and continuing calibration verification was performed throughout each analytical batch.

Results were expressed as mg kg^{-1} dry weight. Where applicable, concentrations below the analytical reporting limit were treated according to the laboratory's validated reporting procedure rather than being interpreted as true zero concentrations.

2.6 PAH extraction and analysis

The 16 USEPA priority PAHs were investigated: naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benzo[a]anthracene, chrysene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[a]pyrene, indeno[1,2,3-cd]pyrene, dibenzo[a,h]anthracene and benzo[g,h,i]perylene.

A known mass of homogenised soil or road dust was extracted using an appropriate validated organic-solvent extraction procedure. The extracts were concentrated under controlled conditions and subjected to appropriate clean-up to minimise matrix interference before instrumental analysis. The extraction and clean-up procedures were performed using solvent blanks, laboratory blanks and matrix-spiked samples to evaluate background contamination and analytical recovery.

PAHs were quantified by gas chromatography coupled with mass spectrometry (GC–MS), using the Agilent 7890B GC coupled with an Agilent 5977A MSD or the validated laboratory-equivalent instrumentation. Identification was based on retention time and characteristic mass spectral ions, with confirmation against authentic analytical standards. Quantification was performed using multi-point calibration curves prepared from certified PAH standards.

Total PAH concentration ($\Sigma 16\text{PAHs}$) was calculated as:

$$\Sigma \text{PAHs} = \sum_{i=1}^{16} C_i$$

where C_i is the concentration of individual PAH i .

Carcinogenic PAHs were additionally grouped for carcinogenic-risk assessment, and benzo[a]pyrene-equivalent concentrations were calculated using established relative potency/toxicity equivalency factors.

2.7 BTEX analysis

The target BTEX compounds comprised benzene, toluene, ethylbenzene and xylenes. Where measured in the analytical dataset, additional volatile/semi-volatile aromatic compounds such as chlorobenzene, styrene, p-cymene and dichlorobenzenes were reported separately and were not included in the BTEX sum.

Because BTEX compounds are volatile, soil and road-dust samples were collected and handled to minimise volatilisation losses. Samples were stored in tightly sealed containers and transported under cooled conditions before extraction. A validated solvent-extraction/headspace analytical procedure appropriate for solid environmental matrices was employed.

BTEX and other target volatile aromatic compounds were analysed by GC–MS. Compound identification was based on retention time and characteristic mass spectra, while quantification was performed using multi-point calibration with authenticated reference standards.

Total BTEX was calculated as:

$$\Sigma BTEX = C_{benzene} + C_{toluene} + C_{ethylbenzene} + C_{xylenes}$$

All concentrations were expressed on a dry-weight basis (mg kg⁻¹).

2.8 Quality assurance and quality control

Comprehensive QA/QC procedures were applied during sampling, sample preparation, extraction, digestion and instrumental analysis. Sampling equipment was cleaned between locations to minimise cross-contamination. Laboratory procedural blanks, field blanks where applicable, calibration standards, continuing calibration verification standards, matrix spikes, certified reference materials and replicate samples were incorporated into analytical batches.

Calibration curves were accepted only when the predefined linearity criteria were achieved. Analytical accuracy was evaluated using recovery tests and certified reference materials, while precision was assessed using replicate analyses and expressed as relative standard deviation (%RSD). Because matrix-specific numerical detection and quantification limits were not included in the analytical results supplied by the laboratory, dataset-derived estimates were calculated for the present study. The estimated limit of detection (LOD) was defined as three times the standard deviation (3 × SD), while the estimated limit of quantification (LOQ) was defined as ten times the standard deviation (10 × SD) of the five lowest positive concentrations for each analyte. Where fewer than five positive observations were available, all available positive observations were used. The estimated LOD and LOQ values for heavy metals, PAHs and BTEX in soil and dust are presented in Table S1.

2.9 Statistical analysis

Descriptive statistics were calculated for each contaminant and environmental matrix, including detection frequency, minimum, maximum, arithmetic mean, median, standard deviation, coefficient of variation and selected percentiles. Concentration distributions were evaluated using histograms, boxplots, Q–Q plots and appropriate normality tests. Where distributions were strongly skewed, logarithmic transformation was considered for multivariate analyses.

Differences in contaminant concentrations between soil and road dust were evaluated using independent-samples tests when distributional and variance assumptions were satisfied. Where assumptions were violated, non-parametric alternatives were used.

Differences among HTFS, LTFS and control locations were evaluated using one-way analysis of variance (ANOVA) when assumptions were met. Where normality or homogeneity assumptions were not satisfied, the Kruskal–Wallis test was used. Significant omnibus tests were followed by appropriate post-hoc comparisons. Statistical significance was set at $p < 0.05$, and effect sizes and confidence intervals were reported where appropriate.

Spearman's rank correlation coefficient was used to evaluate associations among metals, PAHs, BTEX and relevant field variables. Strong positive correlations were interpreted as evidence of potentially shared sources, common transport pathways or similar environmental behaviour, but not as definitive evidence of a common source.

2.10 PAH molecular-weight distribution and diagnostic ratios

PAHs were grouped into low-molecular-weight (LMW; 2–3 rings), medium-molecular-weight (MMW; 4 rings) and high-molecular-weight (HMW; 5–6 rings) classes. The percentage contribution of each class to $\Sigma 16$ PAHs was calculated.

The relative dominance of LMW compounds was considered potentially consistent with petroleum-related inputs, whereas greater representation of HMW PAHs was interpreted as indicative of combustion-related contributions. Because PAHs undergo differential degradation, volatilisation and environmental transformation, molecular-weight patterns were treated as supporting evidence rather than definitive source fingerprints.

Selected diagnostic ratios, including Ant/(Ant+Phe), Flu/(Flu+Pyr), BaA/(BaA+Chr) and, where analytically appropriate, IcdP/(IcdP+BghiP), were calculated only when both components were detected above their respective reporting limits. Diagnostic ratios were interpreted together with PCA, HCA, correlations, traffic classification and field observations. This integrated interpretation is preferable to relying on individual PAH ratios because environmental transformation can modify source signatures.

2.11 Soil and road-dust contamination assessment

The contamination status of soil and road dust was evaluated using complementary single- and multi-element indices.

Contamination factor

The contamination factor for each metal was calculated as:

$$CF_i = \frac{C_i}{B_i}$$

where C_i is the measured concentration of metal i and B_i is its corresponding geochemical background/reference concentration (Ni= 68, Cd= 0.3, V= 130 and Pb= 20 mg/kg; Turekian & Wedepohl average shale).

Geo-accumulation index

The geo-accumulation index was calculated as:

$$I_{geo} = \log_2 \left(\frac{C_i}{1.5B_i} \right)$$

where C_i is the measured concentration and B_i is the background concentration. The factor 1.5 accounts for natural lithogenic variability.

Enrichment factor

Enrichment factor was calculated using a conservative reference element:

$$EF = \frac{(C_i/C_{ref})_{sample}}{(B_i/B_{ref})_{background}}$$

where C_i and C_{ref} are the concentrations of the target metal and reference element in the sample, respectively, while B_i and B_{ref} represent their corresponding background concentrations.

The interpretation of EF was used to distinguish predominantly crustal contributions from anthropogenic enrichment.

Ecological risk factor and potential ecological risk index

The individual ecological risk factor was calculated as:

$$E_r^i = T_r^i \times CF_i$$

and the overall potential ecological risk index as:

$$PERI = \sum E_r^i$$

where T_r^i represents the toxic-response coefficient for metal i (Ni = 5, Cd = 30, V = 2 and Pb = 5; USEPA, 2026a).

The ecological-risk framework was applied separately to HTFS, LTFS and control/reference locations to determine whether increasing traffic intensity corresponded with increasing ecological risk. This approach is consistent with recent Nigerian road-dust studies that combine contamination and ecological-risk indices with source-apportionment techniques (Ameen et al., 2026; Owolabi et al., 2025).

3. Human health risk assessment

Human-health risk assessment was conducted within a source–pathway–receptor framework following USEPA principles. Separate assessments were undertaken for adults and children because of differences in body weight, exposure frequency, ingestion rates and behavioural characteristics.

Potential exposure pathways considered for soil and road dust were:

1. incidental ingestion;
2. dermal contact; and
3. inhalation of resuspended particulate matter.

The quantitative assessment was based on measured contaminant concentrations and established toxicity values. Estimated risks represent potential exposure under the specified assumptions and should not be interpreted as evidence of observed disease.

3.1 Heavy metals risk assessment

Chronic daily intake (CDI) through ingestion

CDI through ingestion was calculated using

$$CDI_{ing} = \frac{C \times IR \times EF}{BW \times 365}$$

where C is concentration in mg/kg, IR is soil/dust ingestion rate in kg/day, EF is exposure frequency, and BW is body weight.

Dermal exposure

Dermal absorbed dose was estimated as

$$D_{dermal} = \frac{C \times SA \times AF \times ABS \times 10^{-6} \times EF}{BW \times 365}$$

where SA is exposed skin surface area, AF is soil/dust adherence factor, ABS is dermal absorption fraction and CF is the appropriate unit-conversion factor.

Inhalation exposure

Inhalation exposure was estimated using a particulate emission factor of $1.36 \times 10^9 \text{ m}^3/\text{kg}$

$$CDI_{inh} = \frac{C \times IR_{air}}{PEF \times BW}$$

where IR_{air} is the inhalation rate and PEF is the particulate emission factor.

Exposure assumptions are stated in table S2.

Non-carcinogenic risk

Non-carcinogenic risk was assessed using the hazard quotient for each contaminant and pathway, calculated as

$$HQ = \frac{CDI_{ing}}{RfD}$$

where RfD is the contaminant-specific reference dose (Ni=0.020, Cd=0.001, V=0.007 mg/kg; USEPA, 1989; 1991).

The combined hazard index was calculated as

$$HI = \sum H Q_i$$

across contaminants and relevant exposure pathways. An HQ or HI below 1 was interpreted as indicating that the estimated exposure was below the corresponding reference dose under the specified assumptions, whereas values ≥ 1 were considered indicative of potential non-carcinogenic concern.

Carcinogenic risk through ingestion

Carcinogenic risk was calculated based on the lifetime average daily dose (LADD)

$$LADD = \frac{C \times IR \times EF \times ED}{BW \times AT_c}$$

where $AT_c = 70 \times 365 = 25,550$ days.

Cancer risk was also calculated as

$$CR = LADD \times SF$$

Where SF is oral slope factor (Ni= 0.84 mg/kg-day⁻¹ and Cd = 6.1 mg/kg-day)⁻¹ been used as screening values. Current EPA IRIS does not provide an oral cancer slope factor for Ni soluble salts; therefore, the Ni cancer estimates below is labelled as screening estimates based on the historical slope factor, not current IRIS-derived cancer risks. For V and Pb, current oral slope factor has not been established for the forms represented.

3.2 PAH carcinogenic risk

For carcinogenic PAHs, benzo[a]pyrene-equivalent concentration (BaPeq) was calculated as

$$BaP_{eq} = \sum_{i=1}^7 C_i \times RPF_i$$

where C_i is the concentration of each carcinogenic PAH and RPF_i is its toxicity-equivalency factor relative to benzo[a]pyrene. The RPFs are BaP = 1.0, BaA = 0.1, chrysene = 0.01, BbF = 0.1, BkF = 0.01, DBA = 1.0, and IcdP = 0.1 (USEPA, 2026b).

Also, incremental lifetime cancer risk (ILCR) was estimated using the applicable cancer slope factor

$$ILCR = LADD \times CSF$$

where LADD is lifetime average daily ingestion dose and CSF is the contaminant-specific cancer slope factor ($7.3 \text{ (mg/kg-day)}^{-1}$).

LADD was calculated as

$$LADD_{ing} = \frac{C_{BaPeq} \times IR \times 10^{-6} \times EF \times ED}{BW \times AT}$$

where

C_{BaPeq} = BaPeq concentration (mg/kg)

AT = averaging time = 70×365 days

10^{-6} converts mg of dust to kg of dust.

3.3 BTEX-related risk

The three xylene isomers were sum up to get total xylenes, consistent with EPA's use of the 0.20 mg/kg-day oral RfD for xylene mixtures (USEPA, 2026c). For benzene, the EPA IRIS oral slope factor of $5.5 \times 10^{-2} \text{ (mg/kg-day)}^{-1}$ was used (USEPA, n.d.). The noncancer RfDs used were toluene

= 0.08, ethylbenzene = 0.10, and total xylenes = 0.20 mg/kg-day (U.S. Department of the Navy, 2006).

Cancer risk was calculated as

$$CR = LADD \times SF$$

with lifetime averaging time of 70 years and the specified adult/child exposure durations of 20 and 6 years, respectively.

3.4 Monte Carlo probabilistic risk assessment

Monte Carlo simulation was conducted to characterise variability and uncertainty in estimated health risks. Separate simulations were performed for adults and children and stratified by environmental matrix and traffic class.

Measured contaminant concentrations were represented using empirical or fitted probability distributions selected on the basis of the observed dataset and goodness-of-fit diagnostics. Where sample numbers were insufficient to justify parametric distributions, bootstrap resampling of the observed concentrations was used, consistent with the approach adopted in the accompanying Abuja investigation.

Exposure parameters such as body weight, ingestion rate, exposure frequency and exposure duration were represented using appropriate probability distributions based on available exposure-factor information. A minimum of 10,000 iterations was used for each receptor–matrix–traffic scenario, with convergence assessed before final reporting.

Simulation outputs were summarised using the mean, median, 5th percentile and 95th percentile. The probability of exceeding the screening thresholds for $HI > 1$ and cancer-risk benchmarks was also calculated. Sensitivity analysis using rank-correlation coefficients was conducted to identify the contaminant concentrations and exposure parameters contributing most strongly to risk variability.

4. Results

4.1 Concentration of Contaminants in Dust and Soil

Heavy Metal Concentrations in Dust and Sand

The mean concentrations of heavy metals in road dust collected from HTFS and control sites CS are presented in Fig. 2a. Ni recorded the highest mean concentration in both HTFS (0.1254 mg/kg) and CS (0.1190 mg/kg), followed by V, Cd, and Pb. Cd exhibited a higher mean concentration at the control sites (0.0640 mg/kg) than at the high-traffic fuel stations (0.0490 mg/kg). Similarly, V concentrations were slightly higher in CS (0.0585 mg/kg) than in HTFS (0.0518 mg/kg), while Pb concentrations were 0.0370 mg/kg and 0.0330 mg/kg in CS and HTFS, respectively.

Heavy metals in sand collected from HTFS and control sites are shown in Fig. 2b. Ni was the most abundant heavy metal, with mean concentrations of 0.1442 mg/kg in HTFS and 0.1240 mg/kg in CS. Pb was the second most abundant metal, averaging 0.0594 mg/kg at HTFS and 0.0515 mg/kg at CS. V concentrations were comparable between HTFS (0.0342 mg/kg) and CS (0.0340 mg/kg), while Cd showed mean concentrations of 0.0364 mg/kg and 0.0295 mg/kg, respectively.

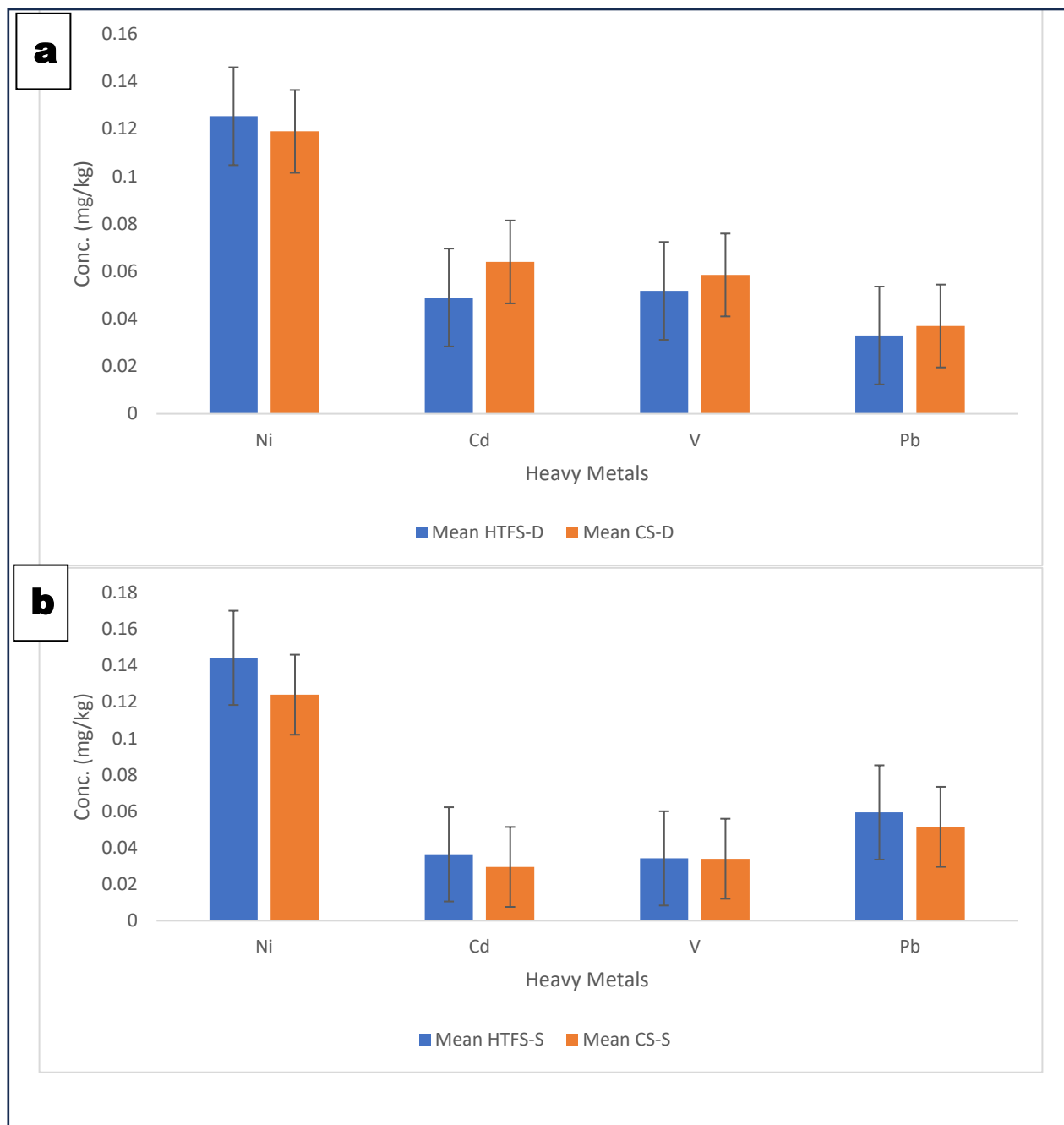


Fig. 2: Heavy metal concentration in samples. a= dust; b= Sand

PAH Concentrations in Dust and Sand

The mean concentrations of the 17 priority PAHs in dust collected from HTFS, LTFS, and control sites are presented in Fig. 3a. Distinct variations in PAH composition and abundance were observed among the three sampling locations, indicating differences in contamination sources and traffic intensity.

At the HTFS, phenanthrene was the predominant PAH, with a mean concentration of 9.651 mg/kg, followed by acenaphthene (2.605 mg/kg), acenaphthylene (2.455 mg/kg), fluorene (2.148 mg/kg), benzo(e)anthracene (1.977 mg/kg), anthracene (1.965 mg/kg), chrysene (1.784 mg/kg), and benzo(a,h,i)perylene (1.622 mg/kg). Moderate concentrations were recorded for pyrene (0.952 mg/kg), benzo(a)anthracene (0.508 mg/kg), benzo(b)fluoranthene (0.746 mg/kg), benzo(k)fluoranthene (0.589 mg/kg), indeno(1,2,3-c,d)pyrene (0.652 mg/kg), naphthalene (0.395 mg/kg), benzo(a)pyrene (0.109 mg/kg), and dibenzo(a,h)anthracene (0.073 mg/kg). Fluoranthene occurred at a relatively low concentration of 0.733 mg/kg.

Dust from the LTFS exhibited a markedly different PAH profile. The highest mean concentrations were observed for dibenzo(a,h)anthracene (21.549 mg/kg), benzo(b)fluoranthene (18.932 mg/kg), indeno(1,2,3-c,d)pyrene (15.244 mg/kg), and benzo(e)anthracene (13.983 mg/kg). Benzo(a,h,i)perylene also showed a relatively high concentration (7.392 mg/kg), followed by phenanthrene (5.475 mg/kg), benzo(k)fluoranthene (5.438 mg/kg), fluoranthene (3.252 mg/kg), fluorene (3.018 mg/kg), and benzo(a)pyrene (2.182 mg/kg). Lower concentrations were recorded for chrysene (1.369 mg/kg), acenaphthene (0.868 mg/kg), anthracene (0.527 mg/kg), pyrene (0.455 mg/kg), benzo(a)anthracene (0.231 mg/kg), naphthalene (0.163 mg/kg), and acenaphthylene (0.097 mg/kg).

In the control dust samples, phenanthrene remained the dominant compound with a mean concentration of 11.566 mg/kg, followed by fluorene (5.137 mg/kg), acenaphthene (4.390 mg/kg), anthracene (3.681 mg/kg), and acenaphthylene (3.285 mg/kg). Most of the high-molecular-weight PAHs occurred at considerably lower concentrations than in LTFS-D, including benzo(a,h,i)perylene (0.203 mg/kg), chrysene (0.171 mg/kg), indeno(1,2,3-c,d)pyrene (0.163 mg/kg), benzo(a)pyrene (0.136 mg/kg), benzo(a)anthracene (0.107 mg/kg), benzo(e)anthracene

(0.050 mg/kg), benzo(k)fluoranthene (0.007 mg/kg), dibenzo(a,h)anthracene (0.006 mg/kg), and benzo(b)fluoranthene (0.00007 mg/kg). Pyrene (0.016 mg/kg) and fluoranthene (0.124 mg/kg) were detected at comparatively low concentrations.

Sand samples from HTFS contained higher concentrations of most PAHs than those from LTFS and the control sites (Fig. 3b), indicating greater hydrocarbon contamination associated with intensive vehicular and fuel-dispensing activities.

Among the measured compounds, phenanthrene recorded the highest mean concentration in HTFS (11.852 mg/kg), followed by fluorene (7.173 mg/kg), anthracene (3.174 mg/kg), and acenaphthene (2.118 mg/kg). Moderate concentrations were observed for fluoranthene (1.262 mg/kg), acenaphthylene (0.749 mg/kg), benzo(a)anthracene (0.398 mg/kg), and chrysene (0.326 mg/kg), whereas benzo(b)fluoranthene (0.00035 mg/kg), benzo(e)anthracene (0.00551 mg/kg), benzo(k)fluoranthene (0.0183 mg/kg), benzo(a)pyrene (0.0205 mg/kg), dibenzo(a,h)anthracene (0.0445 mg/kg), and indeno(1,2,3-c,d)pyrene (0.0768 mg/kg) occurred at relatively low concentrations. Benzo(a,h,i)perylene was not detected in HTFS.

At the LTFS, phenanthrene remained the predominant PAH with a mean concentration of 3.008 mg/kg, followed by fluorene (2.074 mg/kg) and anthracene (1.701 mg/kg). Acenaphthene (0.645 mg/kg), fluoranthene (0.730 mg/kg), benzo(e)anthracene (0.544 mg/kg), indeno(1,2,3-c,d)pyrene (0.307 mg/kg), and pyrene (0.255 mg/kg) were also detected. The highest concentration among all compounds at LTFS-S was recorded for benzo(a,h,i)perylene (5.363 mg/kg), exceeding the corresponding concentrations at HTFS and the control sites.

Control sand contained generally lower PAH concentrations than HTFS. Phenanthrene was again the dominant compound with a mean concentration of 6.051 mg/kg, followed by anthracene (1.884 mg/kg), fluoranthene (1.082 mg/kg), acenaphthene (0.835 mg/kg), and fluorene (0.721 mg/kg). Several high-molecular-weight PAHs, including benzo(a)anthracene, chrysene, benzo(a)pyrene, benzo(a,h,i)perylene, benzo(b)fluoranthene, benzo(k)fluoranthene, dibenzo(a,h)anthracene, and indeno(1,2,3-c,d)pyrene, were not detected in the control samples.

BTEX concentrations in dust and sand

BTEX and related aromatic compounds were detected in dust from the HTFS, LTFS, and CS, with clear differences in their concentration profiles (Fig. 4a). At the HTFS, the predominant

compounds were o-xylene (26.60 mg/kg), styrene (11.42 mg/kg), benzene (7.46 mg/kg), ethylbenzene (3.35 mg/kg), and chlorobenzene (3.19 mg/kg). At the LTFS, higher and more diverse concentrations were observed, dominated by p-cymene (27.56 mg/kg), o-xylene (14.12 mg/kg), m-xylene (13.37 mg/kg), chlorobenzene (10.98 mg/kg), and benzene (9.30 mg/kg). In contrast, concentrations at the CS were generally low, with chlorobenzene (3.07 mg/kg), benzene (0.94 mg/kg), and o-xylene (0.48 mg/kg) being the major detected compounds.

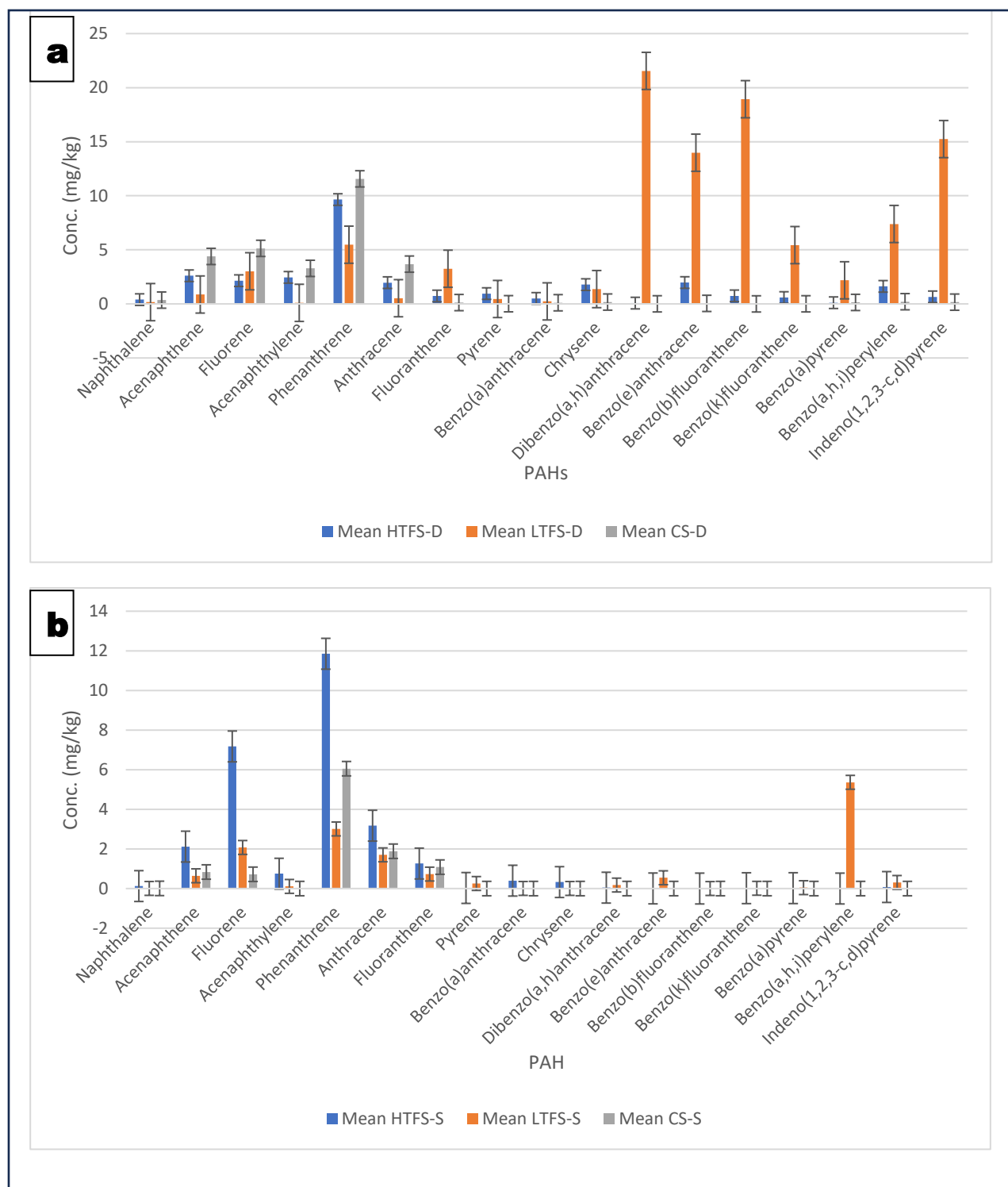


Fig. 3: PAHs concentrations in samples. a= dust; b= sand

For sand, at the HTFS, the predominant compounds were p-cymene (11.47 mg/kg), ethylbenzene (7.14 mg/kg), toluene (6.23 mg/kg), benzene (5.49 mg/kg), m-xylene (3.13 mg/kg), and styrene (0.69 mg/kg) (Fig. 4b). In contrast, LTFS sand generally contained lower concentrations, with benzene (0.25 mg/kg), chlorobenzene (0.30 mg/kg), o-xylene (0.09 mg/kg), and ethylbenzene (0.07 mg/kg) being the principal detected compounds. The CS exhibited comparatively elevated concentrations of o-xylene (9.79 mg/kg) and benzene (7.54 mg/kg), together with lower levels of chlorobenzene (0.48 mg/kg), p-xylene (1.77 mg/kg), styrene (0.28 mg/kg), and ethylbenzene (0.13 mg/kg).

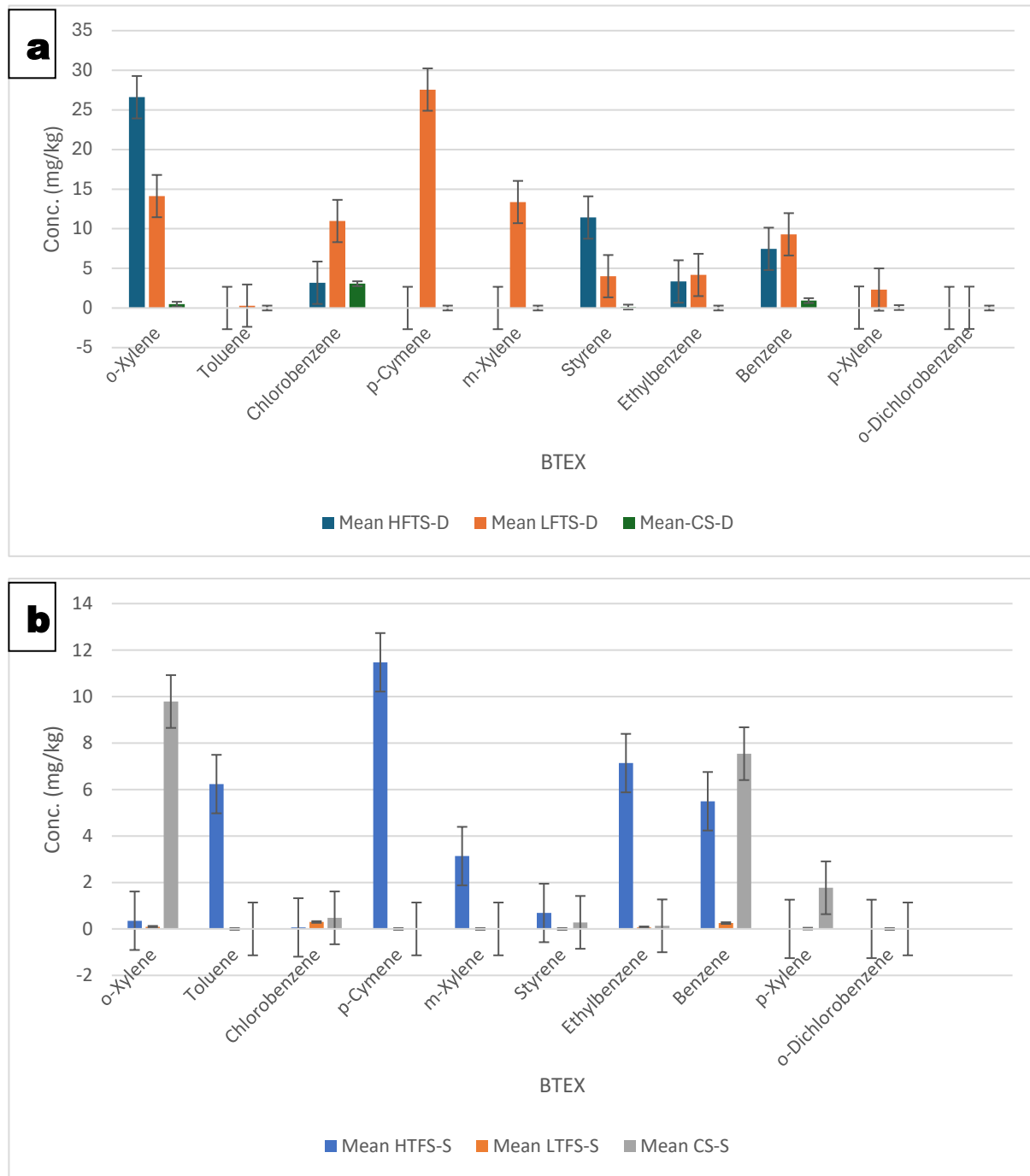


Fig. 4: BTEX concentration in samples. a= Dust; b= Sand

4.2 PAH molecular-weight distribution and diagnostic ratios in sand and dust

The total Σ PAH concentrations in sand were 27.37, 14.99, and 10.58 mg/kg at the HTFS, LTFS, and CS, respectively (Table 1a). LMW PAHs (2–3 rings) predominated at all sites, contributing 92.03% of Σ PAHs at HTFS, 50.32% at LTFS, and 89.77% at CS. MMW PAHs (4 rings) accounted for 7.36%, 6.65%, and 10.23%, respectively, whereas HMW PAHs (5–6 rings) contributed 0.61% at HTFS, 43.04% at LTFS, and 0% at CS. Thus, HTFS and CS were strongly dominated by LMW compounds, while LTFS exhibited a more mixed molecular-weight profile, largely due to the relatively high concentration of benzo(a,h,i)perylene.

The diagnostic ratios further indicated differences in the PAH profiles among sites. The Ant/(Ant+Phe) ratios were 0.211, 0.361, and 0.237 for HTFS, LTFS, and CS, respectively. Flu/(Flu+Pyr) ratios were 0.978, 0.741, and 1.000, while BaA/(BaA+Chr) ratios were 0.550 and 0.543 at HTFS and LTFS, respectively; the ratio was not calculable at CS because both compounds were below detection. The IcdP/(IcdP+BghiP) ratio was 1.000 at HTFS and 0.054 at LTFS, but was not calculable at CS because both compounds were not detected.

For dust, the total Σ PAH concentrations were 28.96, 100.17, and 29.39 mg/kg at the HTFS, LTFS, and CS, respectively (Table 1b). LMW PAHs (2–3 rings) accounted for 66.35% of Σ PAHs at HTFS, 10.13% at LTFS, and 96.66% at CS. MMW PAHs (4 rings) contributed 13.73%, 5.30%, and 1.42%, respectively, while HMW PAHs (5–6 rings) accounted for 19.92% at HTFS, 84.57% at LTFS, and 1.92% at CS. Thus, HTFS and CS dust were predominantly characterized by LMW PAHs, whereas LTFS showed a pronounced dominance of HMW compounds, particularly dibenzo(a,h)anthracene, benzo(e)anthracene, benzo(b)fluoranthene, and indeno(1,2,3-c,d)pyrene.

The diagnostic ratios also showed distinct PAH profiles among the sites. The Ant/(Ant+Phe) ratios were 0.169, 0.088, and 0.241 for HTFS, LTFS, and CS, respectively, while Flu/(Flu+Pyr) ratios were 0.435, 0.877, and 0.885, respectively. The BaA/(BaA+Chr) ratios were 0.222, 0.144, and 0.384 for HTFS, LTFS, and CS, respectively. The particularly high Flu/(Flu+Pyr) ratio at LTFS, together with its strong HMW contribution, indicates a markedly different PAH profile from the predominantly LMW-dominated HTFS and CS dust.

Table 1: PAH molecular-weight distribution and diagnostic ratios in sand and dust

Site	Σ PAHs (mg/kg)	LM W (%)	MM W (%)	HM W (%)	Ant/(Ant+Phe)	Flu/(Flu+Pyr)	BaA/(BaA+Chr)	IcdP/(IcdP+BghiP)
(a) Sand								
HTFS	27.37	92.0 3	7.36	0.61	0.211	0.978	0.550	1.000
LTFS	14.99	50.3 2	6.65	43.0 4	0.361	0.741	0.543	0.054
CS	10.58	89.7 7	10.2 3	0.00	0.237	1.000	NC	NC
(b) Dust								
HTFS	28.96	66.3 5	13.7 3	19.9 2	0.169	0.435	0.222	NC
LTFS	100.1 7	10.1 3	5.30	84.5 7	0.088	0.877	0.144	NC
CS	29.39	96.6 6	1.42	1.92	0.241	0.885	0.384	NC

NC = not calculable because the required PAH compounds were not detected above the reporting limit.

LMW = low molecular weight (2–3 rings); MMW = medium molecular weight (4 rings); HMW = high molecular weight (5–6 rings).

4.3 Contamination and ecological-risk indices of heavy metals in sand and dust

The contamination indices indicated very low contamination by Ni, Cd, V and Pb in both road dust and soil relative to the adopted average-shale background values (Table 2). All CF values were substantially below 1, with the highest value observed for Cd in control-site road dust (CF = 0.213), followed by Cd in HTFS-D (0.163). The Igeo values were consistently negative, ranging from –12.486 for V in CS-S to –2.814 for Cd in CS-D, indicating that all four metals were within the unpolluted/lowest contamination category based on the Igeo framework.

The ecological-risk assessment similarly indicated low potential ecological risk at all locations. PERI values ranged from 2.97 in CS-S to 6.42 in CS-D, all substantially below the threshold of 150 for low ecological risk. Cd contributed most strongly to the calculated ecological risk because of its higher toxic-response coefficient ($T_r = 30$), although its individual risk values remained low (2.95–6.40). Ni, V and Pb contributed negligibly to the overall risk. Notably, the control road dust exhibited a slightly higher PERI (6.42) than HTFS-D (4.92), whereas HTFS-S (3.67) was higher than CS-S (2.97).

Table 2: Contamination and ecological-risk indices of heavy metals in sand and dust

Site	Metal	Dust			Sand		
		CF	Igeo	E_r^i	CF	Igeo	E_r^i
HTFS	Ni	0.00184	−9.668	0.0092	0.00212	−9.466	0.0106
	Cd	0.16333	−3.199	4.9000	0.12133	−3.628	3.6400
	V	0.00040	−11.878	0.0008	0.00026	−12.477	0.0005
	Pb	0.00165	−9.828	0.0083	0.00297	−8.980	0.0149
	$\Sigma CF/PERI$	0.16723		4.9183	0.12669		3.6660
	(ΣE_r^i)			(Low)			(Low)
CS	Ni	0.00175	−9.743	0.0088	0.00182	−9.684	0.0091
	Cd	0.21333	−2.814	6.4000	0.09833	−3.931	2.9500
	V	0.00045	−11.703	0.0009	0.00026	−12.486	0.0005
	Pb	0.00185	−9.663	0.0093	0.00258	−9.186	0.0129
	$\Sigma CF/PERI$	0.21738		6.4189	0.10299		2.9725
	(ΣE_r^i)			(Low)			(Low)

The conventional interpretation is $CF < 1$ = low contamination, while $E_r^i < 40$ indicates low individual ecological risk and $PERI < 150$ indicates low overall potential ecological risk.

4.4 Human Health Risk Assessment

4.4.1 Heavy metals risk

Chronic daily intake through ingestion of heavy metals in soil and dust

Estimated chronic exposure through incidental ingestion was substantially greater than exposure through dermal contact and inhalation of resuspended particles (Table 3). Across all sampling categories, ingestion-derived HQs for Ni, Cd and V were below 1 for both adults and children, with the highest individual HQ observed for Cd in children at the CS-D site (8.18×10^{-4}). Corresponding hazard indices for Ni, Cd and V ranged from 5.56×10^{-5} to 1.07×10^{-4} for adults and from 5.19×10^{-4} to 1.00×10^{-3} for children.

Dermal absorbed dose and inhalation exposure of heavy metals in soil and dust

Dermal exposure estimates were consistently low across all sampling locations and metals (Table 4). The estimated total dermal absorbed doses ranged from 2.28×10^{-8} to 2.78×10^{-8} mg/kg-day in adults and from 1.59×10^{-7} to 1.88×10^{-7} mg/kg-day in children. Children had higher estimated dermal doses than adults, reflecting their lower body weight relative to the assumed skin-contact exposure.

Inhalation exposure to metal-bearing resuspended particles was substantially lower than both ingestion and dermal exposure. Estimated total inhalation doses ranged from 4.94×10^{-11} to 5.82×10^{-11} mg/kg-day in adults and 9.86×10^{-11} to 1.04×10^{-10} mg/kg-day in children. Overall, the results indicate that dermal contact and inhalation contributed relatively little to total heavy-metal exposure compared with incidental ingestion, with inhalation representing the least important pathway under the assumed exposure conditions.

Carcinogenic risk through ingestion of heavy metals in soil and dust

Lifetime average daily doses were estimated for Ni and Cd because these metals had applicable screening-level oral cancer slope factors (Table 5). The resulting combined carcinogenic risks ranged from 1.67×10^{-7} to 2.88×10^{-7} for adults and from 3.11×10^{-7} to 5.37×10^{-7} for children. The highest estimated risk occurred among children at CS-D (5.37×10^{-7}), followed by HTFS-D (4.43×10^{-7}).

Table 3. Ingestion CDI and HQ for adults and children through ingestion of heavy metals in soil and dust

Sample	Metal	CDI (mg/kg- day)	HQ	CDI (mg/kg- day)	HQ
		Adult		Children	
HTFS-D	Ni	1.72×10^{-7}	8.59×10^{-6}	1.60×10^{-6}	8.02×10^{-5}
	Cd	6.71×10^{-8}	6.71×10^{-5}	6.26×10^{-7}	6.26×10^{-4}
	V	7.09×10^{-8}	1.01×10^{-5}	6.62×10^{-7}	9.46×10^{-5}
	Pb	4.52×10^{-8}	—	5.08×10^{-7}	—
	HI		8.58×10^{-5}		8.01×10^{-4}
CS-D	Ni	1.64×10^{-7}	8.15×10^{-6}	1.52×10^{-6}	7.61×10^{-5}
	Cd	8.77×10^{-8}	8.77×10^{-5}	8.18×10^{-7}	8.18×10^{-4}
	V	8.01×10^{-8}	1.14×10^{-5}	7.48×10^{-7}	1.07×10^{-4}
	Pb	5.07×10^{-8}	—	5.70×10^{-7}	—
	HI		1.07×10^{-4}		1.00×10^{-3}
HTFS-S	Ni	1.98×10^{-7}	9.88×10^{-6}	1.84×10^{-6}	9.22×10^{-5}
	Cd	4.99×10^{-8}	4.99×10^{-5}	4.65×10^{-7}	4.65×10^{-4}
	V	4.67×10^{-8}	6.69×10^{-6}	4.37×10^{-7}	6.25×10^{-5}
	Pb	8.14×10^{-8}	—	7.92×10^{-7}	—
	HI		6.64×10^{-5}		6.20×10^{-4}
CS-S	Ni	1.70×10^{-7}	8.49×10^{-6}	1.59×10^{-6}	7.93×10^{-5}
	Cd	4.04×10^{-8}	4.04×10^{-5}	3.77×10^{-7}	3.77×10^{-4}
	V	4.66×10^{-8}	6.65×10^{-6}	4.35×10^{-7}	6.21×10^{-5}
	Pb	7.06×10^{-8}	—	6.87×10^{-7}	—
	HI		5.56×10^{-5}		5.19×10^{-4}

Note: HTFS-S = high-traffic fuel station soil; HTFS-D = high-traffic fuel station dust; CS-S = control site soil; CS-D = control site dust.

Table 4: Dermal absorbed dose and Inhalation exposure of heavy metals in samples

Sample	Dermal absorbed dose		Inhalation exposure	
	Adult dose*	total	Child dose*	total
HTFS-D	2.56×10^{-8}		1.72×10^{-7}	
CS-D	2.28×10^{-8}		1.66×10^{-7}	
HTFS-S	2.78×10^{-8}		1.88×10^{-7}	
CS-S	2.43×10^{-8}		1.59×10^{-7}	
			Adult total CDI	Child total CDI
HTFS-D			5.42×10^{-11}	9.86×10^{-11}
CS-D			5.82×10^{-11}	1.04×10^{-10}
HTFS-S			5.09×10^{-11}	1.02×10^{-10}
CS-S			4.94×10^{-11}	9.90×10^{-11}

*Sum of the estimated absorbed doses for Ni, Cd, V and Pb, in mg/kg-day.

Table 5: Carcinogenic risk through ingestion of heavy metals in soil and dust

Age groups	Sample	Ni LADD	Ni CR	Cd LADD	Cd CR	Combined Ni+Cd CR
Adult	HTFS-D	7.36×10^{-8}	6.18×10^{-8}	2.88×10^{-8}	1.75×10^{-7}	2.37×10^{-7}
	CS-D	6.99×10^{-8}	5.87×10^{-8}	3.76×10^{-8}	2.29×10^{-7}	2.88×10^{-7}
	HTFS-S	8.47×10^{-8}	7.11×10^{-8}	2.14×10^{-8}	1.30×10^{-7}	2.01×10^{-7}
	CS-S	7.28×10^{-8}	6.12×10^{-8}	1.73×10^{-8}	1.06×10^{-7}	1.67×10^{-7}
Child	HTFS-D	1.37×10^{-7}	1.15×10^{-7}	5.37×10^{-8}	3.28×10^{-7}	4.43×10^{-7}
	CS-D	1.30×10^{-7}	1.10×10^{-7}	7.01×10^{-8}	4.28×10^{-7}	5.37×10^{-7}
	HTFS-S	1.58×10^{-7}	1.33×10^{-7}	3.99×10^{-8}	2.43×10^{-7}	3.76×10^{-7}
	CS-S	1.36×10^{-7}	1.14×10^{-7}	3.23×10^{-8}	1.97×10^{-7}	3.11×10^{-7}

EPA's RSL framework defines cancer risk as the probability of an individual developing cancer over a lifetime and uses a target risk of 10^{-6} for its generic screening tables.

4.4.2 PAH carcinogenic risk

The carcinogenic PAH assessment (Table 6) showed marked variation in BaP-equivalent concentrations and associated cancer risks among the sampling groups. BaPeq concentrations ranged from 0.116 to 27.239 mg/kg, with the highest concentration observed in LFTS-D, followed by HTFS-D (0.39

7 mg/kg), LTFS-S (0.253 mg/kg), CS-D (0.171 mg/kg), and HTFS-S (0.116 mg/kg). CS-S recorded no detectable carcinogenic PAH contribution.

The estimated adult ILCR ranged from 4.97×10^{-7} to 1.17×10^{-4} , while the corresponding child ILCR ranged from 9.27×10^{-7} to 2.18×10^{-4} . The LFTS-D group exhibited the highest cancer risk for both adults (1.17×10^{-4}) and children (2.18×10^{-4}), exceeding the 10^{-4} risk benchmark. Child ILCRs were consistently higher than adult values across the contaminated groups, reflecting the higher dust ingestion rate relative to body weight. Overall, the results indicate that low-traffic fuel-station dust (LFTS-D) represented the principal contributor to PAH-related carcinogenic risk among the assessed sampling groups.

Table 6: Benzo[a]pyrene-equivalent concentrations and carcinogenic risk associated with PAHs in soil/dust

Sample group	BaPeq (mg/kg)	Adult LADD (mg/kg-day)	Adult ILCR	Child LADD (mg/kg-day)	Child ILCR	Risk interpretation*
HFTS-D	0.396560	2.33×10^{-7}	1.70×10^{-6}	4.35×10^{-7}	3.17×10^{-6}	Low-acceptable
LFTS-D	27.239308	1.60×10^{-5}	1.17×10^{-4}	2.99×10^{-5}	2.18×10^{-4}	High
CS-D	0.170821	1.00×10^{-7}	7.32×10^{-7}	1.87×10^{-7}	1.37×10^{-6}	Low-acceptable
HTFS-S	0.115907	6.80×10^{-8}	4.97×10^{-7}	1.27×10^{-7}	9.27×10^{-7}	Low
LTFS-S	0.252827	1.48×10^{-7}	1.08×10^{-6}	2.77×10^{-7}	2.02×10^{-6}	Low-acceptable
CS-S	0.000000	0	0	0	0	No carcinogenic risk detected
Total/Range	—	—	4.97×10^{-7} – 1.17×10^{-4}	—	9.27×10^{-7} – 2.18×10^{-4}	—

*An ILCR of 10^{-6} represents approximately one additional theoretical cancer case per million exposed persons, while 10^{-4} represents approximately one additional case per 10,000 exposed persons.

4.4.3 BTEX-related risk

BTEX-related health risks were evaluated through soil/road-dust ingestion, with benzene assessed for carcinogenicity and toluene, ethylbenzene and xylenes assessed for non-carcinogenic effects. Adult BTEX HIs ranged from 0.0015 to 0.2332 and remained below unity across all traffic and environmental-media classes (Table S3). In contrast, child HIs ranged from 0.0157 to 2.4873, exceeding unity at HTFS-S (2.13), HTFS-D (2.13) and LTFS-D (2.49). Total xylenes were the dominant contributor to cumulative non-carcinogenic risk, particularly among children.

Benzene-associated cancer risks ranged from 4.74×10^{-6} to 1.75×10^{-4} for adults and from 1.52×10^{-5} to 5.60×10^{-4} for children, based on the USEPA oral slope factor of $5.5 \times 10^{-2} (\text{mg kg}^{-1} \text{ day}^{-1})^{-1}$ (Table S4). The highest estimated risk occurred for LTFS-D in both adults and children, while children consistently exhibited greater carcinogenic risk than adults.

4.4.4. Monte Carlo ingestion-risk

The probabilistic results represent age-specific exposure distributions rather than a single generic receptor (Table S5; Fig. 5). Ingestion-related non-cancer and cancer risks were low across the soil and road-dust scenarios. No principal scenario produced a meaningful probability of $\text{HI} > 1$ or $\text{TCR} > 1 \times 10^{-4}$. The LTFS child road-dust scenario produced median HI of approximately 3.33×10^{-5} and P95 HI of 4.67×10^{-4} ; the corresponding P95 TCR was approximately 9.39×10^{-7} . The ingestion pathway alone would therefore not indicate a substantial human-health concern.

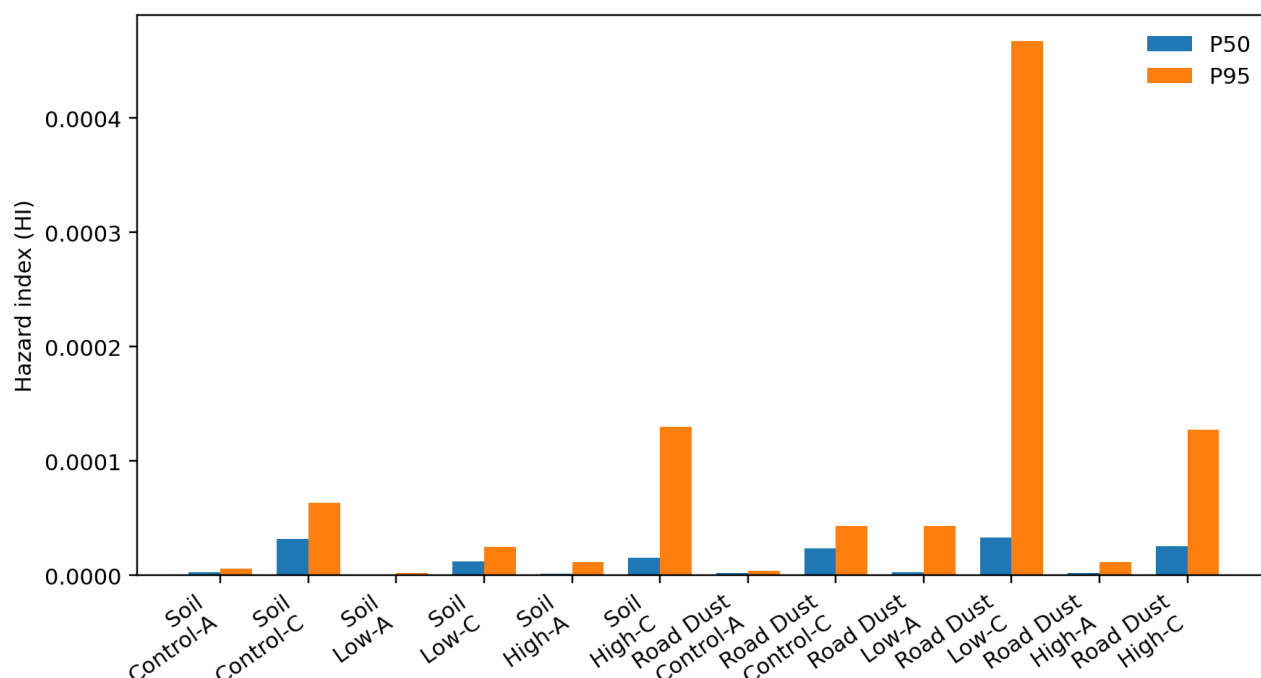


Figure 5: Median (P50) and upper-tail (P95) ingestion hazard index

5. Discussion

The mean concentrations of heavy metals in road dust and sand were generally low, although Ni was consistently the dominant metal. The generally higher concentrations of Ni, Cd and Pb in HTFS sand relative to controls suggest some contribution from traffic, fuel dispensing and associated particulate deposition (Olowoyo et al., 2022). The concentrations observed in the present study were substantially below the Nigerian EGASPIN intervention values reported for petroleum-impacted soil/sediment, which are approximately 210 mg/kg for Ni, 12 mg/kg for Cd and 530 mg/kg for Pb (DPR, 2002). The present maximum concentrations of Ni (0.1442 mg/kg), Cd (0.0490 mg/kg in dust) and Pb (0.0594 mg/kg in sand) represent only a small fraction of these intervention values, indicating no evidence of severe metal contamination based on this regulatory benchmark.

The relatively low concentrations are also consistent with some Nigerian studies, although considerable site-to-site variability has been reported. A study of soils around petroleum-product retailing stations in Ile-Ife found substantially higher metal concentrations, and evidence of

enrichment for several metals, while vehicular emissions, atmospheric deposition and weathering were identified as potential contributors (Adebiyi et al., 2022). Similarly, roadside soils and dust from Enugu contained Ni concentrations of up to 6.93 and 4.68 mg/kg, respectively, and Pb concentrations of up to 4.16 and 14.63 mg/kg, respectively, considerably exceeding the levels observed in the present study (Ichu et al., 2021; Nnamani & Ukamaka, 2025). Conversely, a study around petrol stations in Kogi State reported Cd concentrations of 0.0102–0.0743 mg/kg and Pb concentrations of 0.0002–0.0099 mg/kg (Kadili et al., 2021a), illustrating the substantial variability in metal burdens among Nigerian fuel-station environments.

The absence of marked enrichment in the present dust and sand samples may reflect relatively limited metal accumulation, differences in fuel-station operation, traffic intensity, substrate characteristics, atmospheric deposition and local background geology. Nevertheless, the consistent predominance of Ni and the tendency for higher metal concentrations in HTFS sand indicate that traffic and fuel-station activities may contribute to localized accumulation (Cuput et al., 2021).

In contrast, the PAH burden was more notable. The present Σ PAH concentrations in sand and particularly dust at LTFS were substantially higher than concentrations reported in several Nigerian roadside and vehicle-associated soils. For instance, Σ 11PAHs of approximately 0.263–0.380 mg/kg were reported in heavy-traffic roadside soils in Ibadan (Akinsete & Olatimehin, 2024), while Σ 16PAHs in southern Nigerian vehicle parks generally ranged from 0.066 to 5.322 mg/kg depending on season and soil depth (Emoyan et al., 2020). A recent investigation of petrol stations in Kogi State similarly reported much lower mean Σ 16PAHs of 2.55–3.45 μ g/kg (Kadili et al., 2021b). The markedly higher PAH concentrations observed in the present study, particularly in LTFS dust, therefore indicate a localized accumulation of petroleum- and combustion-derived hydrocarbons.

The predominance of high-molecular-weight PAHs in LTFS dust is of regulatory and health significance, as several are carcinogenic or potentially carcinogenic (Harasym & Nizio, 2026). NESREA regulations do not establish a single comprehensive soil-quality guideline covering all individual PAHs; therefore, international soil-quality benchmarks may be used for comparative assessment where appropriate (NESREA, n.d.). Kadili et al. (2021b; 2026) compared PAHs in soils near petrol-station in Nigeria against Dutch and Canadian soil criteria and found substantially lower concentrations than those observed in the present LTFS dust.

The BTEX results further indicate localized petroleum-related contamination. The relatively elevated benzene, ethylbenzene and xylene concentrations observed at HTFS and LTFS are consistent with emissions associated with fuel handling, evaporation and vehicular activity (Song et al., 2023). However, the comparatively high benzene and o-xylene concentrations at some control locations indicate that regional traffic or other urban sources may contribute to the observed burden (Dehhaghi et al., 2023). This spatial heterogeneity is consistent with evidence from Nigerian petroleum-retailing environments in which fuel handling, vehicular emissions and atmospheric deposition have been identified as important contributors to BTEX accumulation in soils and dust (Ganiyu et al., 2023; Godwin et al., 2026).

The HHRA indicates that exposure to Ni, Cd and V in the investigated soil and road dust was generally low under the adopted exposure assumptions. Individual ingestion HQs and cumulative HIs remained below the commonly applied threshold of 1 for adults and children, indicating low non-carcinogenic risk. This is consistent with recent Nigerian roadside-dust studies, where metal-related HIs were generally below unity (Nduka et al., 2025). In Eastern Nigeria, Pb, Cd and Ni concentrations in roadside dust were generally low and the reported non-cancer risks were insignificant, although ingestion remained the dominant exposure pathway (Nduka et al., 2025).

The estimated metal concentrations were also considerably below petroleum-site intervention values used in Nigeria, supporting the low-risk interpretation. However, Cd warrants continued attention because it commonly contributes disproportionately to cumulative risk. Recent studies in Osogbo and Ado-Ekiti similarly identified Cd as an important contributor to roadside-dust health risk, with Cd exceeding the applicable guideline in Osogbo, Osun state (Aturamu et al., 2024) and Cd/Pb contributing substantially to risk in Ado-Ekiti (Owolabi et al., 2025). Children consistently showed greater exposure than adults, reflecting their higher intake relative to body weight and greater potential for contact with contaminated dust. This pattern agrees with recent Nigerian assessments identifying children as the more vulnerable receptor (Aghomi et al., 2026; Okoro et al., 2025).

The estimated Ni–Cd carcinogenic risks remained below the adopted screening benchmark, suggesting low carcinogenic potential under the specified assumptions. Nevertheless, these estimates should be interpreted as screening-level results because cancer-risk values are sensitive to the selected toxicity factors and exposure parameters (Vesković & Onjia, 2025). Pb was not

incorporated into the conventional HI or cancer-risk summation because a current conventional oral reference dose and cancer slope factor are not available for inorganic Pb (Dirani et al., 2026); therefore, its presence should be interpreted as a potential exposure concern requiring continued monitoring rather than as a quantified cancer probability.

In contrast to the metals, the PAH and BTEX results indicate greater potential concern. Nduka et al. (2025) also reported that PAHs in roadside dust from selected areas of Eastern Nigeria contributed appreciably to exposure and cancer risk, particularly through ingestion and inhalation of contaminated dust. Thus, the PAH profile, particularly the enrichment of carcinogenic HMW congeners at LTFS, warrants greater attention than the relatively low metal burden.

BTEX represented an additional exposure concern. Although non-carcinogenic BTEX risk was generally low for adults, child HIs exceeded unity at HTFS-S, HTFS-D and LTFS-D, driven mainly by xylenes and, in some scenarios, ethylbenzene. Benzene-associated cancer risks were also higher for children than adults and reached the upper end of commonly used risk-management ranges at LTFS-D. These findings are relevant because benzene is a known human carcinogen, with exposure associated with an increased risk of leukemia (USEPA, 2003), while occupational and environmental BTEX exposure remains a recognized concern around Nigerian petrol stations (Anigilaje et al., 2024; Abubakar et al., 2025; Bebetidoh et al., 2026).

For monte carlo assessment, upper-tail child cancer risk is important in selected road-dust scenarios. Road dust deserves particular attention because it can integrate contaminant deposition from vehicle emissions, fuel handling, surface spills and resuspended particles (Oluwalana-Sanusi et al., 2026). The non-monotonic traffic pattern observed in the present simulations reinforces the need for multivariate source analysis. Correlation analysis, principal component analysis, hierarchical clustering and PAH diagnostic ratios may help distinguish petroleum-related, combustion-related and mixed source signatures. However, any source conclusion should remain proportionate to the relatively small number of field observations.

Interpretation is constrained by the cross-sectional design, relatively small number of sampling locations, potential co-occurrence of traffic and other urban sources, and reliance on screening-level toxicity and exposure assumptions. The Monte Carlo assessment considered ingestion only; therefore, dermal and inhalation pathways, contaminant bioavailability, seasonal variability, and

temporal changes in fuel-station activity were not probabilistically characterised. Source attribution should consequently be regarded as indicative rather than definitive.

6. Conclusion

The findings demonstrate that contamination around fuel stations in AMAC is chemically heterogeneous, with Ni, Cd, V and Pb exhibiting low contamination and ecological-risk indices, while PAHs and BTEX presented comparatively greater screening-level health concerns. The pronounced HMW-PAH burden in LTFS dust and elevated child BTEX risks indicate that road dust and volatile petroleum-related contaminants warrant particular attention. Routine surveillance should therefore integrate soil and dust monitoring with fuel-storage and dispensing controls, vapour-loss and spill prevention, effective housekeeping and dust management, and targeted monitoring of PAHs, benzene and xylenes. Future studies should incorporate seasonal sampling, larger spatial coverage, direct traffic measurements, contaminant bioavailability and simultaneous dermal and inhalation exposure assessment to strengthen source attribution and risk characterisation.

Declarations

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Supplementary

Table S1: LOD and LOQ for heavy metals, PAHs, and BTEX.

Analyte	Soil (mg/kg)	LOD	Soil (mg/kg)	LOQ	Dust LOD (mg/kg)	Dust LOQ (mg/kg)
(a) Heavy metals						
Ni	0.0206		0.0687		0.1293	0.4309
Cd	0.0155		0.0518		0.0173	0.0577
V	0.0184		0.0615		0.0272	0.0908
Pb	0.0209		0.0698		0.0229	0.0765
(b) PAHs						
Naphthalene	0.878		2.926		0.791	2.637
Acenaphthene	2.633		8.775		4.318	14.395
Fluorene	1.482		4.939		2.536	8.455
Acenaphthylene	0.314		1.048		8.425	28.084
Phenanthrene	7.067		23.558		21.695	72.318
Anthracene	1.065		3.549		1.177	3.924
Fluoranthene	2.952		9.840		17.329	57.764
Pyrene	2.395		7.984		7.098	23.661
Benzo(a)anthracene	4.157		13.855		3.172	10.573
Chrysene	3.406		11.355		13.425	44.751
Dibenzo(a,h)anthracene	1.397		4.658		161.168	537.225
Benzo(e)anthracene	5.715		19.051		99.823	332.742
Benzo(b)fluoranthene	0.0165		0.0550		125.793	419.309

Benzo(k)fluoranthene	0.0541	0.1802	37.068	123.560
Benzo(a)pyrene	0.259	0.864	15.436	51.455
Benzo(a,h,i)perylene	36.337	121.122	52.149	173.830
Indeno(1,2,3-c,d)pyrene	2.445	8.149	112.034	373.448
(c) BTEX				
o-Xylene	15.434	51.446	82.502	275.008
Toluene	—	—	2.605	8.682
Chlorobenzene	1.355	4.518	16.961	56.536
p-Cymene	—	—	—	—
m-Xylene	—	—	—	—
Styrene	6.088	20.294	74.094	246.980
Ethylbenzene	0.728	2.428	22.323	74.410
Benzene	12.336	41.120	21.308	71.028
p-Xylene	7.379	24.596	17.117	57.057
o-Dichlorobenzene	—	—	—	—

Table S2: Human Health Risk Exposure assumptions

Parameter	Adult	Child	Basis
Body weight (BW)	80 kg	15 kg	USEPA-type default
Soil/dust ingestion rate (IR)	100 mg/day	200 mg/day	EPA soil/dust exposure factors
Exposure frequency (EF)	350 days/year	350 days/year	Conservative chronic exposure
Exposure duration (ED)	30 years	6 years	Screening assumption
Inhalation rate	20 m ³ /day	7.6 m ³ /day	EPA exposure-factor approach
Particulate emission factor (PEF)	1.36 × 10 ⁹ m ³ /kg	1.36 × 10 ⁹ m ³ /kg	USEPA screening value
Dermal surface area	5,700 cm ²	2,800 cm ²	EPA dermal guidance
Soil adherence factor	0.07 mg/cm ²	0.20 mg/cm ²	EPA dermal guidance
Cd dermal absorption	0.001	0.001	USEPA RAGS
Ni dermal absorption	0.03	0.03	USEPA-derived value
V/Pb dermal absorption	0.01	0.01	conservative generic inorganic-metal assumption

USEPA (1995; 2026a)

Table S3: BTEX non-carcinogenic risk

Medium/Traffic	Receptor	HQ-Toluene	HQ-Ethylbenzene	HQ-Xylenes	HI-BTEX
HTFS-S	Adult	0.0934	0.0855	0.0209	0.1998
HTFS-S	Child	0.9962	0.9124	0.2229	2.1315
LTFS-S	Adult	0.0000	0.0008	0.0006	0.0015
LTFS-S	Child	0.0000	0.0090	0.0067	0.0157
CS-S	Adult	0.0000	0.0016	0.0693	0.0708
CS-S	Child	0.0000	0.0169	0.7388	0.7557
HTFS-D	Adult	0.0000	0.0401	0.1597	0.1998
HTFS-D	Child	0.0000	0.4281	1.7034	2.1315
LTFS-D	Adult	0.0044	0.0500	0.1787	0.2332
LTFS-D	Child	0.0475	0.5334	1.9065	2.4873
CS-D	Adult	0.0000	0.0000	0.0032	0.0032
CS-D	Child	0.0000	0.0000	0.0340	0.0340

Table S4: Benzene carcinogenic risk

Medium/Traffic	Adult CR	Child CR
HTFS-S	1.03×10^{-4}	3.31×10^{-4}
LTFS-S	4.74×10^{-6}	1.52×10^{-5}
CS-S	1.42×10^{-4}	4.55×10^{-4}
HTFS-D	1.41×10^{-4}	4.50×10^{-4}
LTFS-D	1.75×10^{-4}	5.60×10^{-4}
CS-D	1.77×10^{-5}	5.65×10^{-5}

Table S5: Monte Carlo ingestion-risk

Medium	Traffic	Receptor	HI P50	HI P95	P(HI>1)	TCR P50	TCR P95	P(TCR>10 ⁻⁴)
Soil	Control	Adult	2.964E-06	5.684E-06	0.00%	1.86E-10	3.58E-10	0.00%
Soil	Control	Child	3.198E-05	6.355E-05	0.00%	1.44E-09	2.61E-09	0.00%
Soil	Low	Adult	1.133E-06	2.247E-06	0.00%	7.38E-11	5.14E-10	0.00%
Soil	Low	Child	1.208E-05	2.526E-05	0.00%	1.16E-09	8.96E-09	0.00%
Soil	High	Adult	1.436E-06	1.178E-05	0.00%	8.50E-11	6.93E-10	0.00%
Soil	High	Child	1.549E-05	1.297E-04	0.00%	1.36E-09	4.64E-09	0.00%
Road Dust	Control	Adult	2.206E-06	3.873E-06	0.00%	9.31E-11	1.92E-10	0.00%
Road Dust	Control	Child	2.364E-05	4.360E-05	0.00%	1.36E-09	3.02E-09	0.00%
Road Dust	Low	Adult	2.925E-06	4.305E-05	0.00%	1.21E-10	5.43E-08	0.00%
Road Dust	Low	Child	3.333E-05	4.670E-04	0.00%	2.04E-09	9.39E-07	0.00%
Road Dust	High	Adult	2.416E-06	1.183E-05	0.00%	2.28E-10	9.25E-10	0.00%
Road Dust	High	Child	2.558E-05	1.278E-04	0.00%	1.61E-09	1.34E-08	0.00%