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Probabilistic Human Health Risk Assessment of Potentially Hazardous Elements in Urban Park Soils. Implications for Risk-Based Environmental Management: A Case Study in the Municipality of Caldas, Colombia

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Abstract

Urban recreational soils can retain potentially hazardous elements (PHEs), but concentration contrasts alone do not establish their sources or health implications. This study characterized As, Cd, Ni, Pb, Cr, Zn, and Cu in surface soils from five urban parks and one rural reference site in Caldas, Colombia. One composite of 30 increments was collected per site, and the <75 µm fraction was analyzed by ICP-OES following acid digestion. Descriptive statistics, exploratory principal component analysis and hierarchical clustering, P5-relative geochemical indices, and 10,000-iteration Monte Carlo simulations were used to evaluate between-site patterns and conditional recreational exposure through ingestion and dermal contact. P2 exhibited the highest As, Cd, and Ni concentrations, whereas P4 had the highest Pb, Cr, Zn, and Cu concentrations, including 99.71 mg kg⁻¹ Pb and 189.04 mg kg⁻¹ Zn. Arsenic showed the greatest between-site variability among urban composites (CV = 125.6%). The first two principal components explained 87.19% of the variance, while the integrated geochemical indices identified P2 and P4 as the largest contrasts with P5. For children at P2, the combined hazard index had a median of 0.7062 and a 95th percentile of 1.4857, with 22.63% of simulated values exceeding 1. Quantified cancer risk was restricted to arsenic under explicit chemical-form and bioavailability assumptions. In the equal-contact urban scenario, ingestion P95 cancer-risk estimates for both age groups lay between 10⁻⁵ and 10⁻⁴. These findings support prioritizing confirmatory sampling and exposure refinement, particularly at P2. Interpretation is limited by one composite per site, externally transferred

exposure distributions, and the use of P5 as an operational comparator rather than a validated geochemical background.

Keywords: Urban soils, Heavy metals, Geochemical indices, Territorial planning.

1. Introduction

Urban soils receive potentially hazardous elements (PHEs) from multiple natural and human-related processes, and their concentrations can vary markedly among locations. Studies of roadside soils and industrial urban soils illustrate the influence of surrounding activities while also emphasizing the need to distinguish concentration patterns from evidence of specific sources (Werkenthin et al., 2014; Shen et al., 2024). Recreational areas warrant attention because repeated contact with soil can contribute to exposure through incidental ingestion and dermal contact. Assessments of urban recreational soils have therefore considered both elemental concentrations and exposure-related factors when interpreting potential health implications (Yang et al., 2018).

Geochemical indices and human-health risk models address complementary questions. Concentration ratios, geoaccumulation indices, and toxicity-weighted indices summarize contrasts relative to selected reference concentrations, but their interpretation depends on the suitability of those references and does not directly establish biological effects (Kowalska et al., 2018; Wieczorek and Baran, 2022). Exploratory multivariate analysis can describe associations among elements and similarities among sites, although source attribution requires additional evidence. Monte Carlo simulation provides a framework for propagating specified exposure distributions into conditional risk estimates; it does not compensate for limited environmental sampling or establish population disease frequencies (US EPA, 2001). The application of these methodologies to mining-affected soils in Colombia provides relevant methodological precedents and highlights the importance of integrating geochemical characterization and human-health risk assessment to evaluate the potential health risks associated with PHEs in environmental matrices (López et al., 2025; González-Valoys et al., 2025).

In Caldas, Antioquia, recreational sites located near roads and industrial activities provide a setting for examining how measured concentration contrasts translate into conditional screening priorities. Although the potential health implications of PHEs in urban recreational environments have been recognized, site-specific assessments remain necessary to characterize spatial variability and identify locations that warrant further investigation. This need is particularly relevant in municipalities where recreational areas coexist with roads and industrial activities, as

differences in land-use context may be reflected in contrasting elemental concentrations and exposure conditions. In addition, the interpretation of potential health risks requires more than identifying elevated concentrations or comparing sites with a reference location; it also requires evaluating plausible exposure pathways and accounting for variability in exposure-related parameters. Integrating these components can provide a more informative basis for environmental screening and help identify priorities for subsequent sampling and risk assessment.

This study aimed to integrate geochemical characterization and probabilistic recreational exposure assessment to support prioritization of further investigation. Specifically, it sought to (1) quantify As, Cd, Ni, Pb, Cr, Zn, and Cu concentrations and describe between-site variation across five urban parks and one rural reference site; (2) explore element co-variation and similarities among the six site composites using principal component analysis and hierarchical clustering; (3) compare the urban composites with P5 as an operational local reference, without treating it as a validated geochemical background; and (4) estimate conditional non-carcinogenic hazard and quantified arsenic cancer risk for child and adult receptors through ingestion and dermal contact using 10,000-iteration Monte Carlo simulations. An additional equal-contact urban scenario was used to complement, rather than replace, the site-specific assessment. The novelty of this study lies in the integration of complementary geochemical and probabilistic exposure assessment approaches within a local-scale investigation of recreational environments in Caldas, Antioquia, Colombia. The combined use of concentration-based indices, principal component analysis, hierarchical clustering, and Monte Carlo simulation enables the study to examine elemental contrasts, explore patterns of co-variation among sites, and characterize the variability of conditional health-risk estimates within a single analytical framework.

2. Materials and Methods

2.1. Study Area

The municipality of Caldas is located in Antioquia, Colombia, and forms part of the southernmost section of the Aburrá Valley within the Medellín Metropolitan Area (Fig.1). The municipality have an urban area of 1.85 km² and a rural area of 131.55 km². Regarding life zones, approximately 60% of the rural territory retains forest cover and semi-natural areas. The municipality is characterized by an intermontane valley relief surrounded by mountainous terrain with very steep slopes. The municipality includes the area known as Alto de San Miguel, a protected area and a key hydrological zone in the southeastern part of the municipality, where the Medellín River originates. The latter is recognized as an environmental determinant of the Medellín Metropolitan Area. The municipality generally records an average annual precipitation ranging from 2.000 to 2.500 mm.

The territorial dynamics are characterized by the presence of high-traffic road corridors, primarily used by heavy vehicles, such as tractor-trailers, public transportation vehicles, including buses, and private passenger vehicles. The municipality is crossed by two major roads that serve as regional and national transportation corridors. Regarding economic activities, the municipality is characterized by metalworking, ceramic and tableware manufacturing, and wood processing, as well as commercial activities and small-scale agriculture, primarily subsistence farming.

The municipality has experienced significant urban growth, resulting in the development of densely populated residential settlements. For example, in 2005, the population density of the municipality was approximately 150–155 inhabitants km^{-2} , increasing to approximately 670 inhabitants km^{-2} by 2018. In addition, the urban and rural populations have been estimated to have grown at annual rates of 3.56% and 1.58%, respectively. For 2026, the municipality reports a total population of 89,591 inhabitants, distributed across the following age groups: 13,880 inhabitants aged 0–14 years, representing the child population and 5,751 inhabitants aged 15–19 years, 52,899 inhabitants aged 20–59 years, and 17,141 inhabitants aged 60 years and over, representing the adult population; 78.99% of the population resides in the municipal urban area, while 21.01% lives in the rural area.

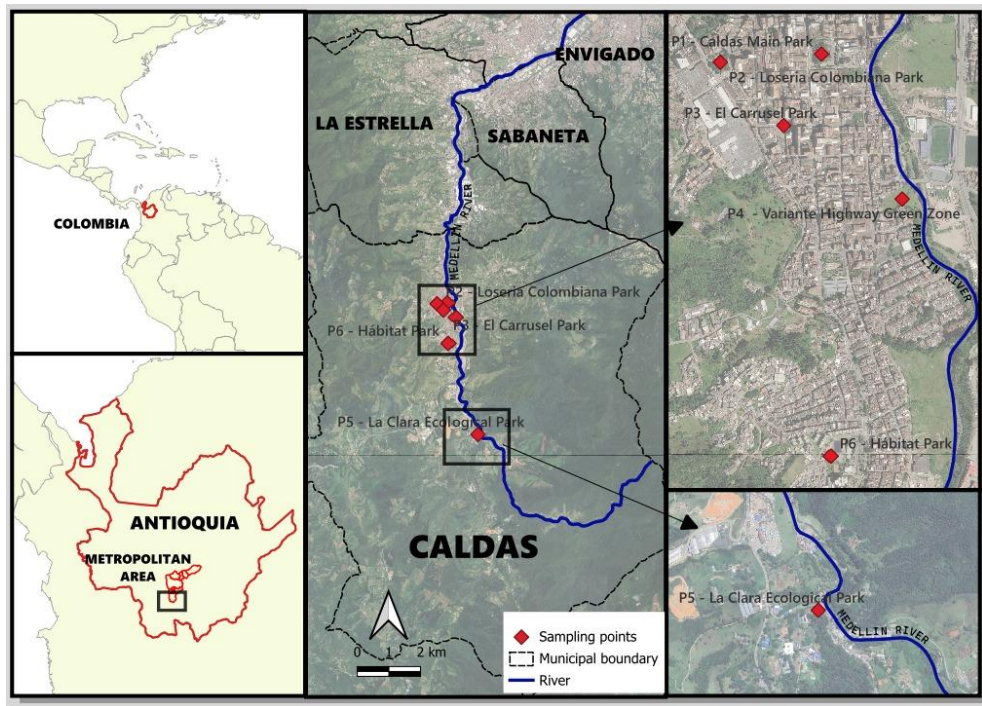


Fig. 1. Location of the study area and sampling sites in Caldas, Antioquia, Colombia. P1–P4 and P6 correspond to urban parks, whereas P5 is located within a rural natural reserve and serves as an operational local reference.

For this study, the six parks located within the municipality of Caldas were selected. Five parks are located in the urban area (P1, P2, P3, P4, and P6), while one is located in the rural area (P5) (Fig. 1). The parks are predominantly used for recreational purposes. The urban parks are situated near industrial facilities and/or high-traffic roads. Specifically, P1 is approximately 31 m from four major roads with high traffic volumes; P2 is approximately 45m from a ceramic tableware manufacturing facility and approximately 31 m from a major high-traffic road; P3 is approximately 16 m from major high-traffic roads; P4 is approximately 25 m from major high-traffic roads and approximately 57 m from several metalworking and automotive repair facilities.; and P6 is approximately 98 m from a major high-traffic road. P5 is located within a natural reserve, the nearest major road to P5 is approximately 2570 m away, there are no industrial activities within the natural reserve; only small businesses operated by local residents on their farms are present. These businesses are primarily focused on tourism-related activities and the sale of traditional foods.

2.2. Soil sampling

Surface soil was sampled at six recreational sites in Caldas: five urban parks (P1–P4 and P6) and one rural site used as an operational local reference (P5) (Fig. 1). At each site, 30 soil increments were collected from the 0–10 cm layer along a non-systematic zigzag sampling pattern distributed across vegetated areas and children’s play areas. The increments were pooled to obtain one composite sample per site. Thus, the site composite constituted the independent analytical unit, yielding six composites overall and five for the urban-only descriptive analysis. This design characterized between-site concentration differences but did not provide independent estimates of within-site variability. The composite samples were air-dried, sieved to <2 mm, and stored until further preparation and chemical analysis.

2.3. PHE concentrations

For PHE determination, the soil samples were first sieved to <0.075 mm. This sieve size was selected because it isolates particles most relevant to dermal adherence and accidental ingestion, particularly among children (O’Shea et al., 2021). Then, 1.0 g of soil sample was subjected to acid digestion in an Ethos One Milestone helping Chemist microwave. In brief, 10 ml of HNO₃ and 3 ml of HCl were added. The temperature of the samples was raised to 175 °C in approximately 5.5 min and remained at 175 °C for an additional 4.5 min (Salazar et al., 2024). After digestion, the samples were diluted with ultrapure water (DW) to a final volume of 15 ml and filtered through a 0.45 µm membrane filter. The filtrates were analyzed for As, Cd, Ni, Pb, Cr, Zn, and Cu using an Agilent 5100 inductively

coupled plasma optical emission spectrometer (ICP-OES). The digestion and instrumental analysis procedures followed those described by Salazar et al. (2024) and López et al. (2024).

Quality assurance and quality control procedures included calibration with four certified multielement standard solutions (MilliporeSigma), with calibration curves yielding $R^2 > 0.9996$. Recoveries relative to the certified solution concentrations ranged from 94% to 101%. Duplicate reference samples yielded coefficients of variation between 0.6% and 2.8%. These results indicate satisfactory calibration linearity, recovery of the certified solution standards, and analytical repeatability. Certified standard solutions and ultrapure-water blanks were included in each analytical batch to monitor instrument performance and potential contamination. All reagents were of analytical grade. Glassware was soaked overnight in a 3% HNO_3 solution, and ultrapure water with a resistivity of $18.2 \text{ M}\Omega \cdot \text{cm}$ (Milli-Q) was used throughout the procedure. Element-specific limits of detection (LOD) and quantification (LOQ) were established as part of the analytical quality control procedure.

2.4. Data Analysis

Descriptive statistics were calculated for the five urban park composites (P1–P4 and P6), including the minimum, maximum, arithmetic mean, sample standard deviation and coefficient of variation. P5 was presented separately as an operational local reference. Boxplots summarize between-site variation among the five urban composites and do not represent within-park variability. All six site composites were retained for the exploratory multivariate analyses. The concentrations measured in each site composite are provided in Supplementary Table S1. Exploratory multivariate analyses were performed using the concentrations of the seven potentially harmful elements (PHEs) in the six independent site composites (P1–P6), including P5. Element concentrations were standardized to z-scores before principal component analysis (PCA). Hierarchical cluster analysis (HCA) was performed on the standardized concentration profiles using Euclidean distance and Ward's minimum-variance method, implemented as ward.D2. PCA scores, element loadings, and the HCA dendrogram were examined to describe similarities among sites and patterns of co-variation among elements, following the exploratory application of PCA and Ward clustering to soil element concentrations by López et al. (2024). Given the limited number of independent site composites, these analyses were used to describe geochemical associations rather than to identify pollution sources definitively or estimate their contributions.

Geochemical indices were used to compare concentrations at the five urban parks (P1–P4 and P6) with those measured in the P5 composite. P5 was selected as an operational local reference because an independently established local

geochemical background dataset was unavailable. The use of less-impacted reference sites in soil geochemical assessments has precedent in Daurov et al. (2026b), although that study used two reference sites. Because P5 was represented by one composite, the indices were interpreted as comparisons with P5 rather than estimates of enrichment above a validated natural background.

Geoaccumulation index (I_{geo}). For each element (i) and urban site (j), the P5-normalized geoaccumulation-type index was calculated using the formulation proposed by Müller (1969):

$$I_{geo,j} = \log_2 \left(\frac{C_{ij}}{1.5 \times B_i} \right) \quad (1)$$

Where C_{ij} is the measured concentration of element i at urban site j , and B_i is the concentration of the same element measured in the P5 composite (mg kg^{-1}). P5 was selected as an operational local reference because an independently established local geochemical background dataset was unavailable. The use of less-impacted reference sites to calculate soil geochemical indices has precedent in Daurov et al. (2026b), although that study used two reference sites. The factor 1.5 is retained from the conventional I_{geo} equation; it does not quantify uncertainty in the P5 measurement. Because P5 is represented by one composite, the indices are interpreted as comparisons with P5 rather than estimates of enrichment above a validated natural background.

The contamination factor (CF) was calculated as:

$$CF_{ij} = \frac{C_{ij}}{B_i} \quad (2)$$

Thus, CF_{ij} values below, equal to, or above 1 indicate concentrations lower than, equal to, or higher than the corresponding concentration in P5.

The Nemerow pollution index (P_N) summarized the seven element-specific contamination factors at each urban site:

$$P_{N,j} = \sqrt{\frac{CF_{mean,j}^2 + CF_{max,j}^2}{2}} \quad (3)$$

Where $CF_{mean,j}$ is the arithmetic mean and $CF_{max,j}$ is the maximum of all CF values at site j. PN was used to compare the overall magnitude of concentrations relative to P5.

The element-specific potential ecological risk factor (E_r) and the potential ecological risk index (PERI) were calculated as:

$$E_{r,i,j} = T_{r,i} \times CF_{i,j} \quad (4)$$

$$PERI_j = \sum_{i=1}^n E_{r,i,j} \quad (5)$$

Here, n represents the seven analyzed elements, and $T_{r,i}$ denotes the conventional element-specific toxic-response factor: As = 10, Cd = 30, Ni = 5, Pb = 5, Cr = 2, Zn = 1, and Cu = 5. The index structure follows Håkanson (1980), while the selected factors are consistent with the soil application reported by Ahogle et al. (2023). E_r and PERI are interpreted as toxicity-weighted comparisons with the P5 composite. P5 was excluded as a target site because it served as the denominator. Conventional contamination and ecological-risk classes were not assigned.

Human health risk was assessed separately for children and adults at each of the six sampled sites (P1–P6) through incidental soil ingestion and dermal contact. Exposure doses were calculated using the US EPA framework and its application to Colombian soils (US EPA, 1989, 2004). The assessment represented repeated recreational contact with soil under the exposure assumptions documented in Supplementary Table S2. Inhalation and other exposure media were outside the quantified assessment.

Exposure frequency was represented by a triangular distribution with a minimum of 26, a mode of 120, and a maximum of 260 contact days year⁻¹, adopted from the recreational exposure scenario reported by González-Valoys et al. (2025). This distribution was transferred as an external scenario assumption; it did not represent measured park attendance in Caldas. Its expected value is approximately 135 contact days year⁻¹, equivalent to 2.6 contact days per week. Exposure duration was fixed at 6 years for children and 20 years for adults, and the assumed lifetime was 70 years. Dermal exposure assumed one soil-contact event per contact day. The distributions, fixed values, units, and parameter-specific sources are provided in Supplementary Table S2.

The chronic daily intake through incidental soil ingestion, CDI_{ing} , was calculated separately for each element, site, and age group using the soil-ingestion exposure equation (US EPA, 1989; López et al., 2025):

$$CDI_{ing,i,j,a} = \frac{C_{ij} \times IR_{ing,a} \times EF_{j,a} \times ED_a}{BW_a \times AT_a} \times 10^{-6} \quad (6)$$

In Equation (6), C is the measured soil concentration ($mg\ kg^{-1}$), IR_{ing} is the soil ingestion rate ($mg\ day^{-1}$), EF is contact frequency ($days\ year^{-1}$), ED is exposure duration (years), BW is body weight (kg), and AT_{nc} is the non-carcinogenic averaging time ($ED \times 365\ days$). The factor 10^{-6} converts soil mass from milligrams to kilograms. Age-specific distributions and fixed exposure values are documented with their parameter-specific sources in Table S2 (US EPA, 1989, 2004).

The dermal absorbed dose from soil contact (DAD_{derm} , $mg\ kg^{-1}\ day^{-1}$) was calculated separately for each element, site, and age group using the soil-contact equation in US EPA Part E, which López et al. (2024) applied to Colombian soils under the designation CDI_{derm} (US EPA, 2004; López et al., 2025):

$$DAD_{derm,i,j,a} = \frac{C_{ij} \times SA_a \times AF_a \times ABS_{d,i} \times EF_{j,a} \times EV_a \times ED}{BW_a \times AT_a} \times 10^{-6} \quad (7)$$

In Equation (7), SA is the exposed skin area (cm^2), AF is soil adherence ($mg\ cm^{-2}$ per event), $ABS_{d,i}$ is the soil dermal absorption fraction, and EV is the number of contact events per contact day, fixed at 1. The remaining variables are defined above. Element-specific absorption assumptions and toxicity adjustments are documented in Table S3 (US EPA, 2004; Salazar et al., 2024). For non-carcinogenic exposure, $AT_{nc} = ED \times 365$; therefore, ED cancels algebraically in both exposure equations.

Non-carcinogenic hazard quotients (HQs) were calculated separately for each element and exposure pathway by dividing the corresponding daily dose by a pathway-compatible reference dose (US EPA, 2004; López et al., 2025). For ingestion, CDI_{ing} was divided by the oral reference dose (RfD_{oral}). For dermal contact, DAD_{derm} was divided by an absorbed-dose reference value (RfD_{abs}), derived from the oral reference dose using the gastrointestinal absorption fraction when this adjustment was supported for the element (US EPA, 2004). The element- and pathway-specific hazard quotients were calculated as follows:

$$HQ_{ing,i,j,a} = \frac{CDI_{ing,i,j,a}}{RfD_{oral,i}} \quad (8)$$

$$HQ_{derm,i,j,a} = \frac{DAD_{derm,i,j,a}}{RfD_{abs,i}} \quad (9)$$

The combined hazard index (HI) was calculated by summing the applicable element-specific hazard quotients for ingestion and dermal contact, separately for each sampling site and age group (US EPA, 2004):

$$HI_{j,a} = \sum_i (HQ_{ing,i,j,a} + HQ_{derm,i,j,a}) \quad (10)$$

Here, *i* denotes the element, *j* the sampling site, and *a* the age group. Pathway-specific indices were calculated as:

$$HI_{ing,j,a} = \sum_i (HQ_{ing,i,j,a}) \quad (11)$$

$$HI_{derm,j,a} = \sum_i (HQ_{derm,i,j,a}) \quad (12)$$

Non-carcinogenic screening quantified ingestion of As, Cd, Ni, Zn, and Cu. Oral reference values for Ni and Zn were 0.02 and 0.3 mg kg⁻¹ day⁻¹, respectively (US EPA, 1991, 2005). The Cu screening value was 0.04 mg kg⁻¹ day⁻¹, as documented by EPA RSL guidance using a HEAST-derived value (US EPA, 2026). For Cd, the ATSDR chronic oral MRL of 1×10⁻⁴ mg kg⁻¹ day⁻¹ was adopted following EPA recommendations (US EPA, 2021, 2026). Measured Ni was assigned to a soluble-salt screening scenario, and oral relative bioavailability was set to 1; chemical speciation and relative bioavailability were not measured.

Dermal HQs were quantified for As and Cd using soil ABS_d values of 0.03 and 0.001 (US EPA, 2004). Screening was extended to Ni and Zn using ABS_d = 0.001 as an explicit transfer assumption from Salazar et al. (2024, Supplementary Table S2), rather than a generic EPA default. Absorbed-dose reference values used GIABS = 0.04 for Ni and no oral toxicity adjustment for Zn, implemented as a factor of 1 (US EPA, 2004, Exhibit 4-1). Dermal Cu HQ was not included in the primary HI because the oral benchmark is primarily based on gastrointestinal effects, and its applicability to systemic toxicity following dermal absorption was not established for this assessment. This represents a limitation of the adopted route-to-route extrapolation, rather than an absence of published dermal Cu calculations or evidence of negligible dermal hazard (US EPA, 2004; Taylor et al., 2023).

Pb was excluded from conventional HQ aggregation because EPA provides no consensus oral RfD, and total Cr was excluded because oxidation-state data were unavailable (US EPA, 2026). HI was interpreted as a preliminary screening aggregate across the quantified combinations, rather than an organ-specific index (US EPA, 1989). Exclusions denote unquantified contributions, not negligible risk. These systemic dermal HQs do not estimate local skin irritation or sensitization.

Carcinogenic doses were averaged over the assumed lifetime using $AT_c = LT \times 365$ days, where LT is the assumed lifetime in years, fixed at 70. Lifetime average daily doses for ingestion and dermal contact were calculated separately for each element, sampling site, and age group using the following equations (US EPA, 1989, 2004):

$$LADD_{ing,i,j,a} = \frac{C_{i,j} \times IR_{ing,a} \times EF_a \times ED_a \times RBA_i}{BW_a \times AT_c} \times 10^{-6} \quad (13)$$

$$LADD_{derm,i,j,a} = \frac{C_{i,j} \times SA_a \times AF_a \times ABS_{d,i} \times EF_{j,a} \times EV_a \times ED_a}{BW_a \times AT_c} \times 10^{-6} \quad (14)$$

Here, LADD is the lifetime average daily dose ($\text{mg kg}^{-1} \text{ day}^{-1}$), and subscripts i, j, and a identify the element, sampling site, and age group, respectively. RBA is relative oral bioavailability, fixed at 1 for the quantified arsenic scenario; EV is fixed at one soil-contact event per contact day. The remaining exposure variables are defined above and documented in Table S2. Ingestion doses represent administered doses adjusted for relative bioavailability, whereas dermal doses represent absorbed doses (US EPA, 1989, 2004).

Pathway-specific incremental lifetime cancer risks were calculated by multiplying the lifetime average daily dose by the corresponding oral or absorbed-dose slope factor (US EPA, 1989, 2004):

$$CR_{ing,i,j,a} = LADD_{ing,i,j,a} \times SF_{oral,i} \quad (15)$$

$$CR_{derm,i,j,a} = LADD_{derm,i,j,a} \times SF_{abs,i} \quad (16)$$

Where $SF_{oral,i}$ and $SF_{abs,i}$ are expressed in $(\text{mg kg}^{-1} \text{ day}^{-1})^{-1}$. When supported by the toxicity assessment, the absorbed-dose slope factor was calculated as $SF_{abs,i} = SF_{oral,i} / GIABS_i$, following the oral-to-dermal adjustment framework (US

EPA, 2004). $GIABS_i$ is the gastrointestinal absorption factor used for this toxicity adjustment; a factor of 1 denotes no adjustment where recommended and does not represent absorption measured in the Caldas soils. This factor is distinct from relative oral bioavailability. Cancer risk is dimensionless and represents a conditional model estimate of additional lifetime cancer risk.

Cancer risk was quantified only for arsenic through ingestion and dermal contact, using the inorganic-As toxicity assessment (US EPA, 2025). All measured As was assigned to the inorganic fraction, and oral relative bioavailability was fixed at 1 as explicit screening assumptions; neither speciation nor relative bioavailability was measured. Other elements were not included without applicable toxicity inputs and compatible chemical-form information. In particular, total Cr was not treated as Cr(VI) (US EPA, 2026). Unquantified contributions were not interpreted as zero risk. The combined result is therefore termed quantified arsenic cancer risk.

Pathway-specific and combined cancer risks were calculated over the set of quantified carcinogenic elements, ICR, which contained As only in this assessment:

$$CR_{ing,j,a} = \sum_i^{ICR} CR_{ing,i,j,a} \quad (17)$$

$$CR_{derm,j,a} = \sum_i^{ICR} CR_{derm,i,j,a} \quad (18)$$

$$CR_{j,a} = CR_{ing,j,a} + CR_{derm,j,a} \quad (19)$$

For inorganic As, the oral reference dose was $6.0 \times 10^{-5} \text{ mg kg}^{-1} \text{ day}^{-1}$ and the oral cancer slope factor was 32 ($\text{mg kg}^{-1} \text{ day}^{-1}$)⁻¹, following the 2025 US EPA IRIS assessment (US EPA, 2025). Table S3 documents the toxicity values, absorption adjustments, chemical-form assumptions, and pathway eligibility. Child and adult scenarios represent separate exposure windows and were not added to construct a combined lifetime exposure history.

Monte Carlo simulations used 10,000 iterations for each sampling site and age group. Body weight was sampled from a normal distribution truncated above zero, while soil ingestion rate and contact frequency were sampled independently from the distributions in Table S2. Concentrations and other model inputs were held fixed. Within each age group, exposure draws were shared across elements, pathways, and sites. Applicable HQs and arsenic cancer risks

were summed within each iteration before aggregate summaries were calculated. Results were characterized primarily by P50, P95, and simulated fractions exceeding HI = 1 or cancer-risk reference levels of 10^{-6} , 10^{-5} , and 10^{-4} . ECDFs displayed both age groups with P50 and P95 markers. These outputs describe conditional model distributions, not confidence limits or population disease frequencies (US EPA, 2001).

An exploratory urban scenario assigned equal contact weights of 0.20 to P1–P4 and P6 and evaluated each element at its contact-weighted mean concentration; P5 was assessed separately. Element-specific HQ simulations were pooled with equal numbers of values per element within each age–pathway combination to construct descriptive boxplots (Fig. 8A). These mixtures combine between-element differences and simulated exposure variability and do not constitute HI. Pathway-specific HI was calculated by summing the applicable HQs within each iteration (Fig. 8B). Pathway-specific urban HI and quantified arsenic cancer-risk ECDFs are presented in Fig. 9.

Sensitivity to the expanded dermal scope was assessed by comparing the main As/Cd/Ni/Zn dermal model with a restricted As/Cd scope, retaining identical oral inputs, concentrations, and exposure draws. This comparison isolates the modeled contribution of Ni and Zn under the transferred $ABS_d = 0.001$ assumption (Salazar et al., 2024). It does

3. Results and Discussion

3.1 PHE concentrations and between-site variability

Table 1 summarizes the descriptive statistics for potentially hazardous element (PHE) concentrations in the $<75 \mu\text{m}$ soil fraction from the five urban parks and presents the P5 concentrations separately as an operational local reference. Individual concentrations for all six site composites are provided in Supplementary Table S1. Mean concentrations decreased in the order $\text{Zn} > \text{Pb} > \text{Cu} > \text{Ni} > \text{Cr} > \text{As} > \text{Cd}$, with respective values of 109.00, 58.29, 26.49, 13.84, 12.92, 4.57, and 0.79 mg kg^{-1} . These estimates describe the five sampled urban composites rather than the municipality-wide distribution of PHEs. Because each park was represented by one composite sample, the descriptive statistics characterize between-site differences and do not estimate within-site variability.

Table 1. Descriptive statistics for PHE concentrations at the five urban sites and measured concentrations at the P5 local reference site (mg kg^{-1}). Each site is represented by one composite sample.

Element	Min	Max	Mean	SD	P5
As	0.03	13.74	4.57	5.74	3.78

Element	Min	Max	Mean	SD	P5
Cd	0.25	1.31	0.79	0.42	0.13
Ni	5.56	29.42	13.84	10.05	11.78
Pb	17.95	99.71	58.29	32.50	19.89
Cr	7.14	18.78	12.92	4.19	9.04
Zn	74.69	189.04	109.00	47.24	68.52
Cu	8.56	41.17	26.49	13.92	12.08

Between-site variability differed markedly among elements. As exhibited the highest coefficient of variation (125.6%), followed by Ni (72.7%), Pb (55.8%), Cu (52.5%), Cd (52.3%), Zn (43.3%), and Cr (32.4%). Thus, Ni was among the most variable elements, whereas Cr displayed the lowest relative dispersion. These differences indicate element-specific concentration patterns across the sampled parks, but they do not independently establish the origin of the measured elements. Evidence from road-deposited sediments in Villavicencio, Colombia, shows that spatial metal distributions can reflect both land-use characteristics and physical surface conditions, illustrating the need to consider multiple controls when interpreting urban concentration contrasts (Trujillo-González et al., 2019).

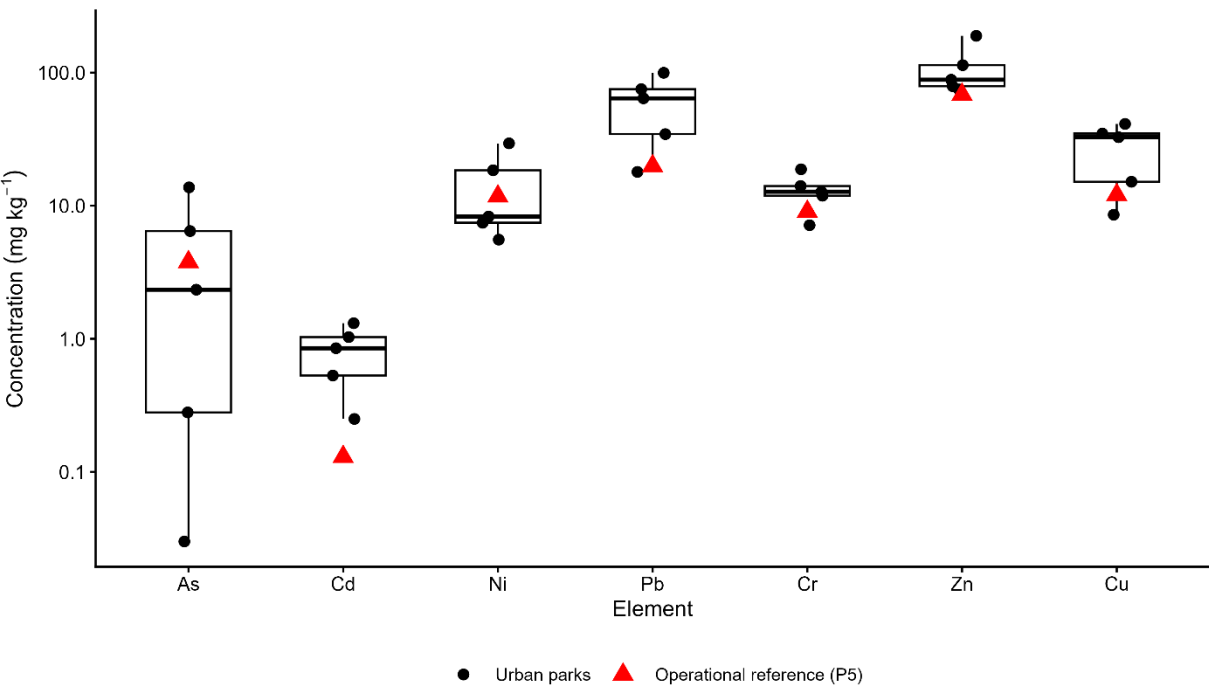


Fig. 2. PHE concentration distributions in the $<75 \mu\text{m}$ soil fraction from five urban park composites. Boxes represent the interquartile range, with the median indicated by the horizontal line. Whiskers extend to the most extreme

observations within 1.5 times the interquartile range. Black points show all five urban composites. Red triangles indicate P5 as an operational local reference, excluded from the boxplot summaries. The y-axis is logarithmic. Concentrations are expressed in mg kg^{-1} .

The site-specific concentration profiles complement the descriptive statistics by identifying the composites responsible for the observed contrasts (Fig. 3). P2 exhibited the highest concentrations of As, Cd, and Ni, whereas P4 exhibited the highest concentrations of Pb, Cr, Zn, and Cu. These contrasting profiles indicate that the largest concentrations were distributed across different parks rather than concentrated at a single site for all elements. The P5 measurements provide an operational comparator for the subsequent geochemical indices; however, differences relative to this composite should not be interpreted as enrichment above a validated natural background. Accordingly, the graphical comparison describes measured concentration contrasts without assigning pollution sources.

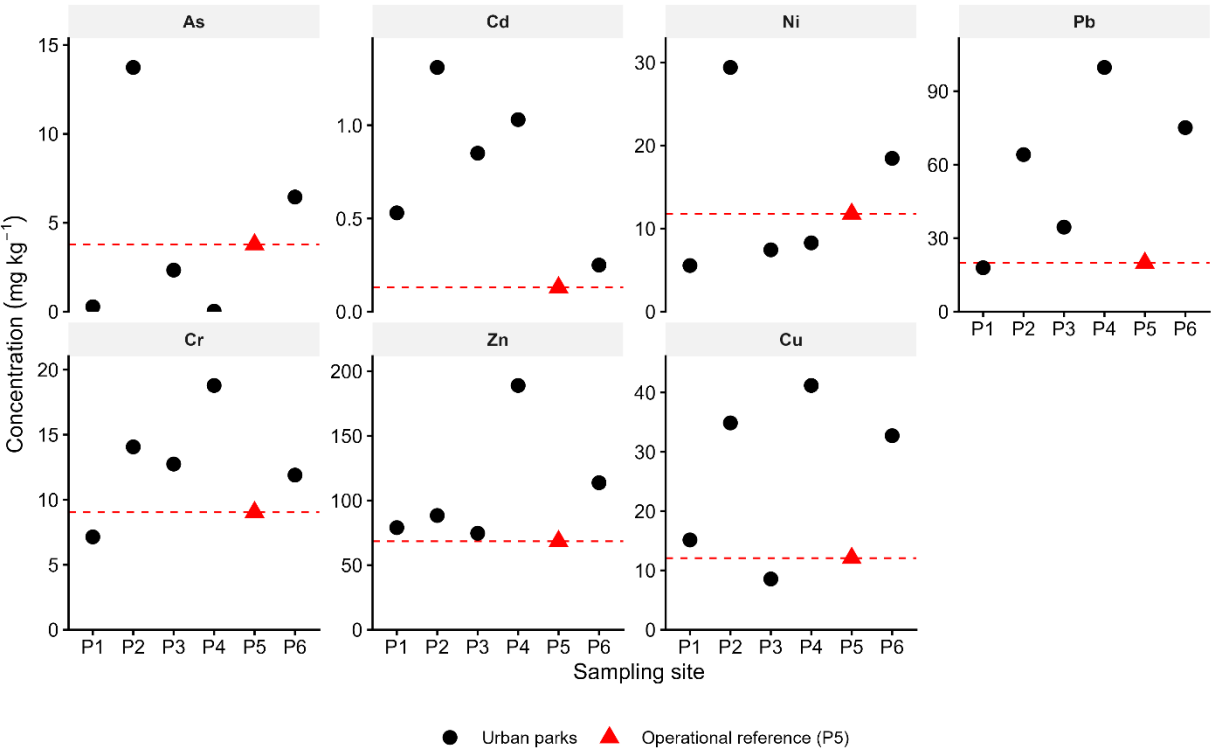


Fig. 3. Measured PHE concentrations at six sampling sites. Each point represents one composite sample comprising 30 soil increments. Black circles indicate urban parks, whereas red triangles indicate P5 as an operational local reference. Red dashed lines show the element-specific P5 concentrations. Panels use independent linear y-axis scales. Concentrations are expressed in mg kg^{-1} .

P2, located approximately 45 m from a ceramic tableware manufacturing facility and 31 m from a major road, exhibited the highest concentrations of As (13.74 mg kg^{-1}), Cd (1.31 mg kg^{-1}), and Ni (29.42 mg kg^{-1}). The As and Cd concentrations were respectively 3.63 and 10.08 times those measured in P5. This spatial setting makes industrial and traffic-related inputs plausible hypotheses for further investigation, although proximity alone cannot distinguish their contributions from differences in soil materials or other local inputs. For comparison, Shen et al. (2024) investigated industrial urban soils in Shanghai and identified both natural and anthropogenic contributions using positive matrix factorization. Their findings illustrate the potential coexistence of multiple sources, but their source classifications cannot be transferred directly to the Caldas composites.

P4 exhibited the highest concentrations of Pb (99.71 mg kg^{-1}), Zn ($189.04 \text{ mg kg}^{-1}$), Cu (41.17 mg kg^{-1}), and Cr (18.78 mg kg^{-1}). Pb and Zn concentrations were respectively 5.01 and 2.76 times those measured in P5. Its proximity to a major road and to metalworking and automotive repair facilities provides a basis for considering traffic-related and industrial inputs as possible contributors. A review of European roadside soils documented elevated metal concentrations in surface layers close to roads, with patterns influenced by distance and soil depth (Werkenthin et al., 2014). Colombian evidence likewise associates urban metal distributions with land use and physical surface characteristics (Trujillo-González et al., 2019). These studies support the plausibility of such inputs, but the Caldas data do not resolve their individual contributions. P5 exhibited the lowest concentrations of Cd and Zn among the six composites, while its concentrations of the remaining elements overlapped the urban ranges, reinforcing its interpretation as an operational local reference.

3.2 Exploratory multivariate patterns in PHE concentrations

Principal component analysis (PCA) and hierarchical cluster analysis (HCA) were applied to the standardized concentrations of seven PHEs in the six site composites, including P5, to examine element co-variation and similarities among sampling sites (Figs. 4 and 5). Related multivariate approaches have been used to explore metal distributions in Colombian urban road-deposited sediments and mining-affected soils (Trujillo-González et al., 2019; López et al., 2024). Here, the analyses describe the measured concentration profiles rather than providing quantitative source apportionment.

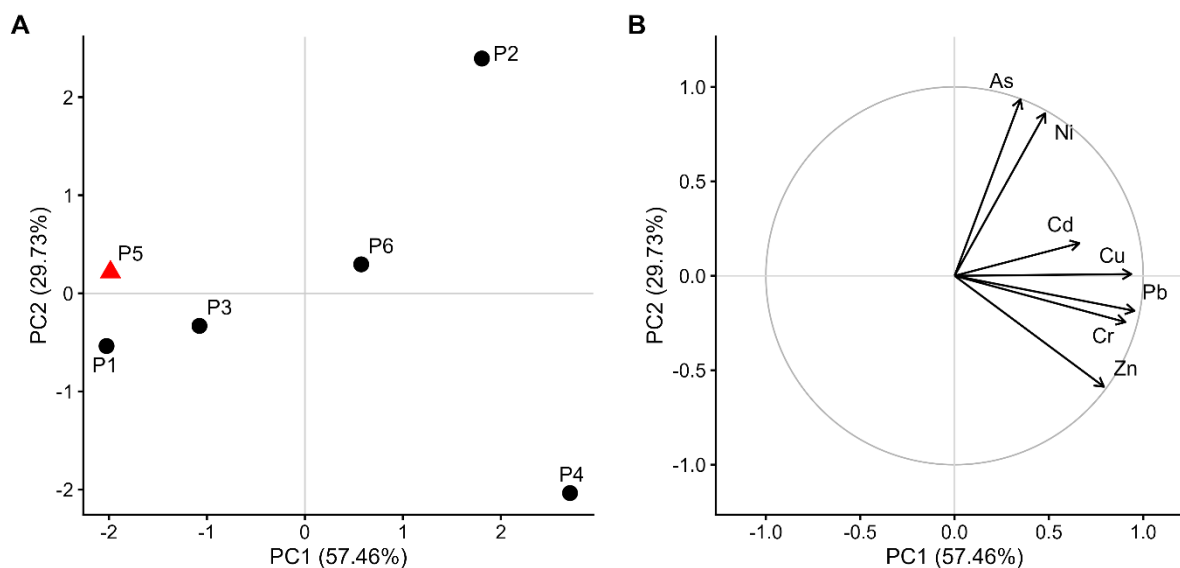


Fig. 4. Exploratory PCA of standardized PHE concentrations in six site composites from Caldas, Colombia. (A) PC1–PC2 site scores, with urban parks shown in black and P5 in red. (B) Correlations between standardized element concentrations and PC1–PC2, displayed within a unit correlation circle. Percentages indicate explained variance. P5 is included in the analysis as a site composite.

The first two principal components explained 87.19% of the total variance, with PC1 and PC2 accounting for 57.46% and 29.73%, respectively (Fig. 4). With the component signs oriented toward positive Pb on PC1 and positive As on PC2, PC1 showed positive correlations with Pb ($r = 0.95$), Cu ($r = 0.94$), Cr ($r = 0.91$), and Zn ($r = 0.79$), with a weaker association for Cd ($r = 0.66$). PC2 was positively associated with As ($r = 0.93$) and Ni ($r = 0.86$) and negatively associated with Zn ($r = -0.59$). P4 had the highest PC1 score and a negative PC2 score, whereas P2 had positive scores on both components, particularly PC2 (Fig. 4A). Thus, the two sites represented contrasting multielement profiles rather than a single homogeneous high-concentration pattern.

The Ward.D2 dendrogram provided a complementary description of site similarities (Fig. 5). P1 and P5 merged first, followed by the incorporation of P3, whereas P2 and P6 formed a separate pair that subsequently joined P4. An illustrative two-branch partition therefore grouped P1, P3, and P5 separately from P2, P4, and P6. These branches summarize similarities in standardized concentration profiles; they do not establish exposure categories or demonstrate that the sites within a branch share a common emission source.

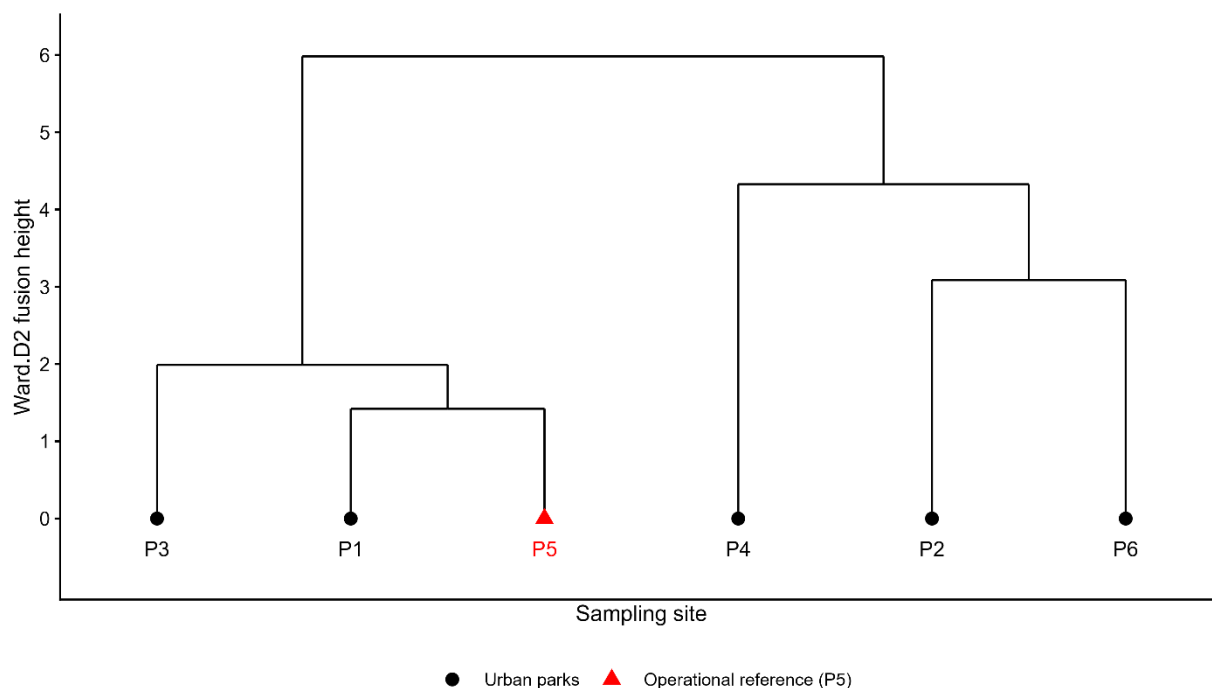


Fig. 5. Hierarchical clustering of six site composites using Euclidean distances and Ward's minimum-variance method (Ward.D2) on standardized PHE concentrations. Black labels and markers identify urban parks, whereas red identifies P5. Fusion heights describe statistical dissimilarity among concentration profiles. The dendrogram does not assign pollution sources or risk classifications.

The observed patterns are compatible with multiple controls on urban PHE concentrations, but their origin remains unresolved. Trujillo-González et al. (2019) examined both land-use and physical surface influences in Colombian road-deposited sediments, while Shen et al. (2024) used positive matrix factorization to investigate source contributions in industrial urban soils. These studies provide context for considering multiple influences, without validating a source interpretation for the Caldas components. With six independent composites and no source-specific measurements, the present PCA and HCA support hypotheses for further investigation rather than definitive anthropogenic–geogenic separation.

3.3. Geochemical contrasts relative to the operational local reference

The geochemical indices provided complementary summaries of the concentration contrasts between the five urban park composites and P5. CF and the P5-normalized I_{geo} describe element-specific differences, whereas PN integrates the mean and maximum concentration ratios and PERI incorporates conventional toxic-response weights. The selection and interpretation of these indices depend on their formulation and reference concentrations, as discussed

by Prasad Ahirvar et al. (2023) and Kowalska et al. (2018). Accordingly, the results are presented as relative comparisons rather than classifications against a validated natural background (Table 2).

Table 2. Integrated geochemical indices relative to the P5 operational reference at five urban parks in Caldas, Colombia.

Site	CFmean	CFmax	Element with maximum CF	PN	PERI	Cd contribution (%)
P1	1.246	4.077	Cd	3.014	138.921	88.04
P2	3.595	10.077	Cd	7.565	386.093	78.30
P3	1.819	6.538	Cd	4.799	221.636	88.50
P4	3.127	7.923	Cd	6.023	290.310	81.88
P6	2.094	3.778	Pb	3.054	119.314	48.35

CFmean and CFmax represent the arithmetic mean and maximum of the seven element-specific concentration ratios, respectively. $PN = [(CFmean^2 + CFmax^2)/2]^{1/2}$; $PERI = \sum(T_{r,i} \times CF_i)$. The Cd contribution represents its percentage contribution to PERI. All indices are dimensionless and were calculated from one composite per site. P5 was used as the denominator and excluded as a target site. No conventional pollution or ecological-risk classes were assigned.

P2 exhibited the highest PN (7.565), followed by P4 (6.023), P3 (4.799), P6 (3.054), and P1 (3.014). The PERI ranking differed for the two lowest-ranked sites: $P2 (386.093) > P4 (290.310) > P3 (221.636) > P1 (138.921) > P6 (119.314)$. Thus, the integrated indices consistently identified P2 and P4 as the largest contrasts with P5, while the relative positions of P1 and P6 depended on whether conventional toxic-response weights were included (Fig. 6).

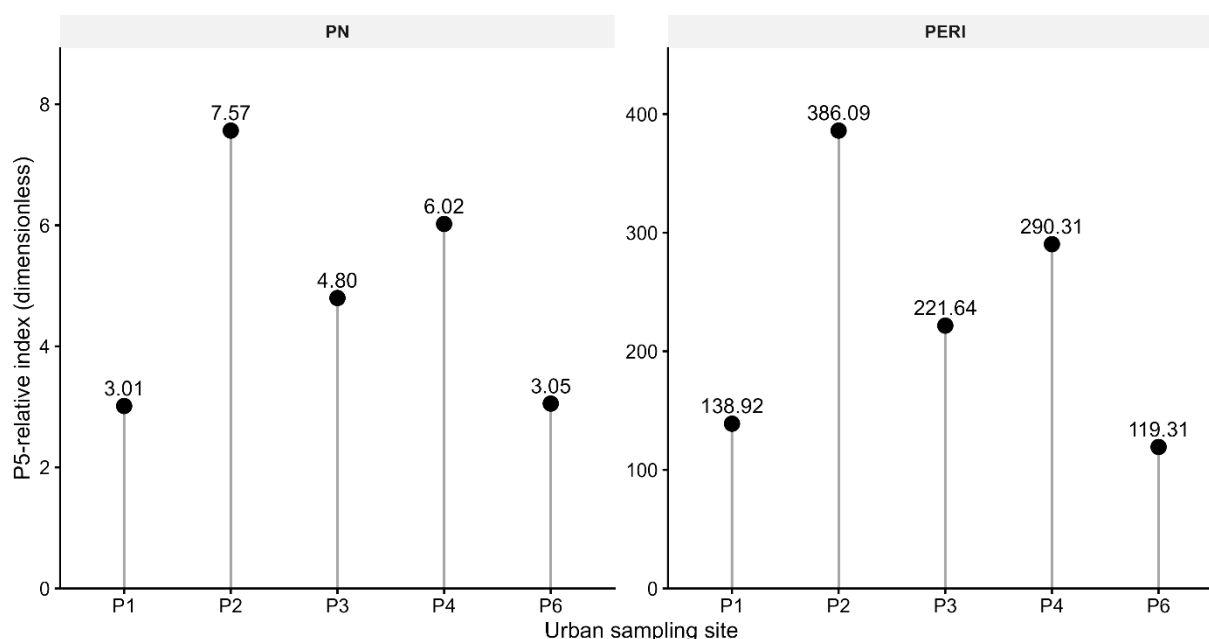


Fig. 6. Integrated geochemical indices relative to the P5 operational reference at five urban parks. Panels show PN and PERI on independent linear scales. Black points represent individual site composites, and labels show calculated values. P5 is used only as the denominator. No conventional pollution or ecological-risk thresholds are applied.

Element-specific comparisons clarified the patterns underlying the integrated indices. Cd exhibited the highest CF at P1, P2, P3, and P4, reaching 4.077, 10.077, 6.538, and 7.923, respectively, whereas Pb produced the highest CF at P6 (3.778). Therefore, the maximum-ratio term in PN was driven by Cd at four urban sites, despite the larger absolute concentrations of other elements. The P5-normalized Igeo provides a logarithmic representation of these ratios, with $I_{geo} = \log_2(CF) - \log_2(1.5)$. CF and Igeo consequently describe the same underlying concentration comparison rather than independent evidence of contamination.

Cd contributed the largest share of PERI at every urban site, accounting for 48.35–88.50% of the index. At P2, Cd contributed 78.30%, whereas As contributed 9.41%. This dominance reflects both the conventional Cd weight of 30 and its low concentration in the P5 denominator (0.13 mg kg^{-1}). Ahogle et al. (2023) likewise reported that Cd made the largest contribution to the cumulative index in peri-urban agricultural soils, although their sampling design and reference concentrations differed from those used here. The Caldas percentages describe the mathematical composition of PERI and should not be interpreted as measured contributions to ecosystem damage.

The site-by-element matrices provide a detailed view of the concentration contrasts with P5 (Fig. 7). P2 exhibited the largest CF values for As, Cd, and Ni, whereas P4 exhibited the largest values for Pb, Cr, Zn, and Cu. The Igeo panel

preserves these element-specific rankings while expressing the ratios on a logarithmic scale. These matrices summarize the sampled composites and do not represent spatial interpolation across the municipality.

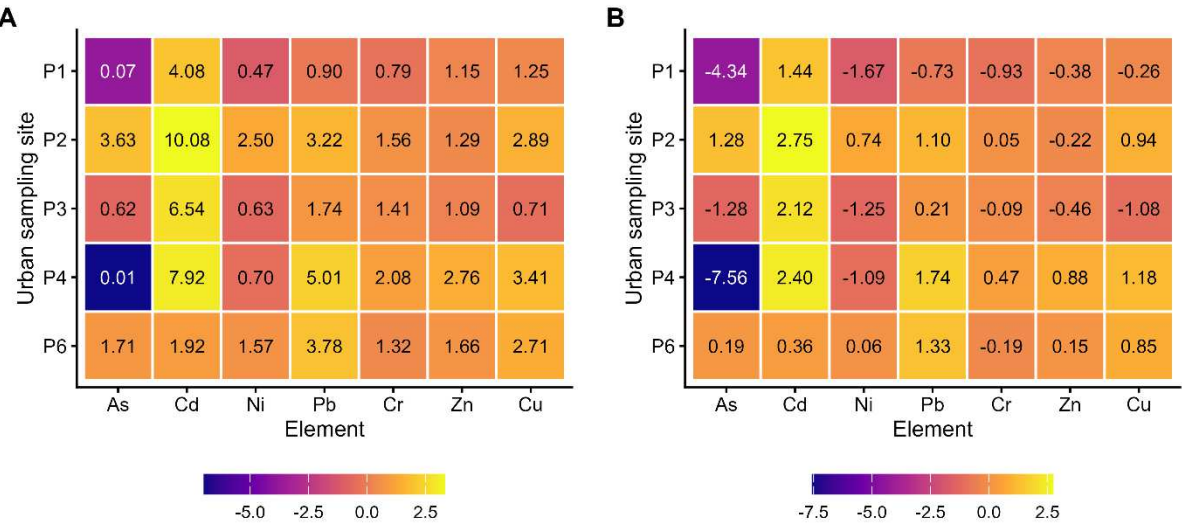


Fig. 7. Element-specific geochemical comparisons with P5 at five urban parks. (A) CF values, with cell shading based on $\log_2(\text{CF})$ and labels showing untransformed CF. (B) P5-normalized Igeo values. Colours represent the numerical values according to the separate colour scales shown below each panel. The matrices summarize sampled site composites and are not spatially interpolated maps or ecological-risk classifications.

The integrated comparison identifies P2 and P4 as priorities for confirmatory sampling because they exhibit the largest overall contrasts with P5. However, the magnitude of these contrasts remains dependent on the reference concentrations. Daurov et al. (2026a) similarly distinguished local comparative reference values from regional natural background in an urban soil assessment. Furthermore, numerical indices do not directly measure biological effects; Wiczorek and Baran (2022) combined geochemical indices with biotests to evaluate soil contamination. Additional reference-site sampling, soil-property characterization, and biological or bioavailability measurements would strengthen the interpretation of the Caldas results.

3.4 Probabilistic Human Health Risk Assessment under Recreational Exposure Scenarios

The probabilistic assessment was interpreted as a conditional screening analysis for recreational contact with the sampled soils. For each sampling site, measured concentrations were held fixed, while selected exposure inputs were varied according to the distributions in Table S2. Children and adults were assessed as separate receptor groups, and only the ingestion and dermal pathways with documented toxicity inputs were quantified. The resulting distributions characterize variability under the specified assumptions; they are not estimates of disease incidence or of the

probability that an individual park visitor will develop an adverse effect. Probabilistic risk assessment provides a framework for representing exposure variability and uncertainty, but its interpretation remains conditional on the model inputs and conceptual model (U.S. EPA, 2001).

Figure 8 summarizes the equal-contact urban scenario by age group and exposure pathway, with P5 evaluated separately. Panel A presents descriptive mixtures of element-specific HQ simulations, whereas panel B presents pathway-specific HI calculated by summing the applicable HQs within each iteration. Ingestion includes As, Cd, Ni, Zn, and Cu, while dermal contact includes As, Cd, Ni, and Zn under the assumptions documented in Supplementary Table S3. The Ni and Zn dermal estimates use the absorption fraction adopted from Salazar et al. (2024).

A

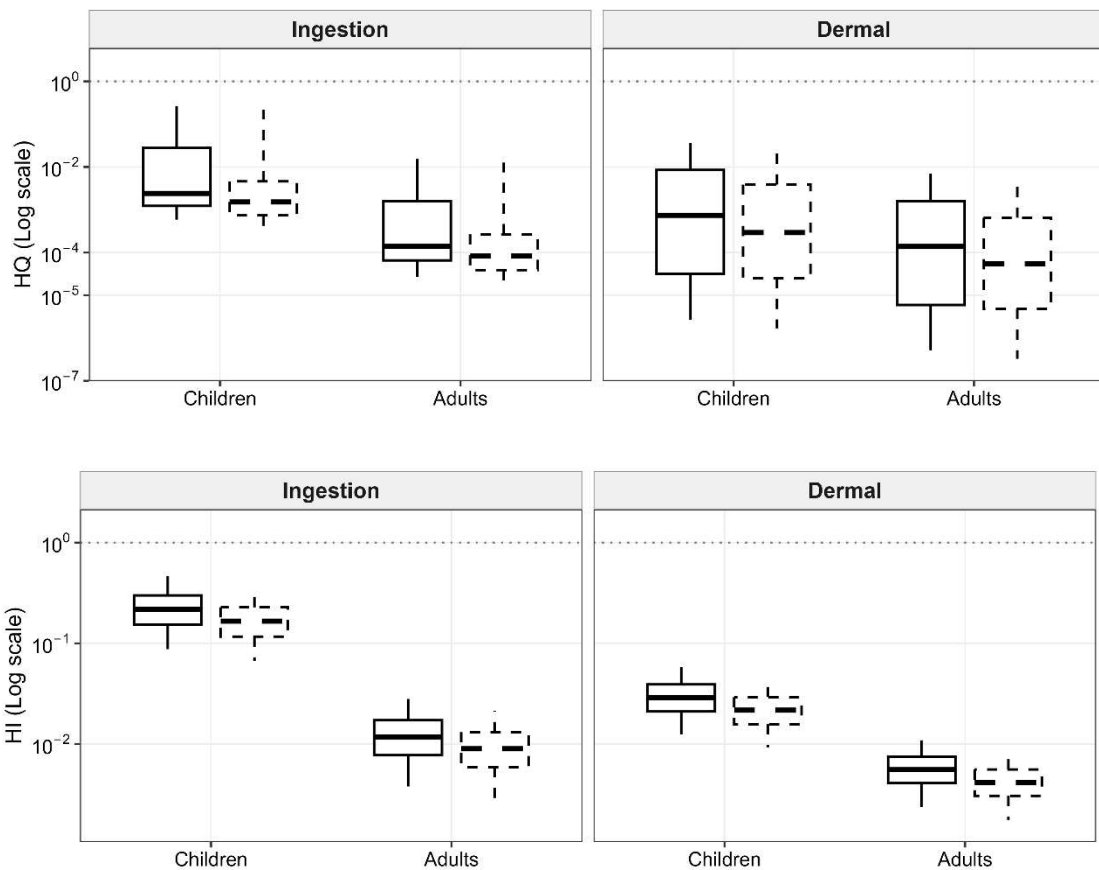


Fig. 8. Non-carcinogenic screening under an equal-contact scenario across five sampled urban parks in Caldas. (A) Descriptive element-pooled HQ distributions by age and pathway. Ingestion includes As, Cd, Ni, Zn, and Cu; dermal contact includes As, Cd, Ni, and Zn. (B) Pathway-specific HI calculated from iteration-level HQ sums. Within each pair, the urban scenario is shown on the left with a solid black outline and P5 on the right with a dashed black outline. P5 is excluded from the urban concentration. Boxes show P25–P75, median P50, and P05–P95 whiskers. Axes use Log scaling; dotted lines mark HQ or HI = 1. Ni/Zn dermal absorption is assumed (Salazar et al., 2024).

Under the equal-contact urban scenario, children exhibited higher pathway-specific HI distributions than adults, and ingestion produced higher HI values than dermal contact in both age groups (Fig. 8B). Median and P95 values were also higher for the urban scenario than for P5 within each age–pathway combination. These contrasts reflect the measured concentrations and the adopted exposure and toxicity inputs, rather than differences in observed health outcomes (US EPA, 1989, 2004). The element-pooled HQ distributions in Fig. 8A provide a descriptive comparison

across the included elements; because they are mixtures rather than sums, their medians should not be interpreted as aggregate hazard or used to quantify the relative contributions of ingestion and dermal contact.

Children exhibited higher HI medians and P95 values than adults at every sampling site (Table 3). P2 showed the highest child HI, with P50 = 0.7062 and P95 = 1.4857; 22.63% of simulated values exceeded HI = 1. At P6, the child P95 was 0.6795, although 0.74% of simulations exceeded 1, illustrating that a P95 below the screening reference does not exclude exceedances in the upper tail. Adult P95 values ranged from 0.0049 to 0.1083, with no simulated exceedances of HI = 1. These findings apply to the quantified subset in Table S3 and describe conditional screening outcomes rather than population disease frequencies (US EPA, 2001).

Table 3. Simulated combined non-carcinogenic hazard index for the quantified element–pathway subset, by sampling site and age group.

Site	Age	P05	P50	P90	P95	Pr(HI>1)
P1	Children	0.0130	0.0317	0.0570	0.0667	0.0000
P2	Children	0.2883	0.7062	1.2692	1.4857	0.2263
P3	Children	0.0572	0.1401	0.2517	0.2944	1.0000e-04
P4	Children	0.0154	0.0377	0.0677	0.0791	0.0000
P5	Children	0.0769	0.1884	0.3388	0.3965	7.0000e-04
P6	Children	0.1318	0.3228	0.5805	0.6795	0.0074
P1	Adults	8.9756e-04	0.0023	0.0042	0.0049	0.0000
P2	Adults	0.0195	0.0497	0.0925	0.1083	0.0000
P3	Adults	0.0039	0.0099	0.0184	0.0216	0.0000
P4	Adults	0.0011	0.0027	0.0050	0.0059	0.0000
P5	Adults	0.0052	0.0132	0.0246	0.0288	0.0000
P6	Adults	0.0089	0.0226	0.0422	0.0494	0.0000

HI is dimensionless and includes the element–pathway combinations documented in Supplementary Table S3. P05, P50, P90, and P95 denote simulated percentiles. $\text{Pr}(\text{HI} > 1)$ is the fraction of iterations with $\text{HI} > 1$, expressed on a 0–1 scale. Children and adults represent separate receptor scenarios. Exceedance fractions describe the specified simulations and do not represent observed frequencies of adverse health effects.

To complement the site-specific combined HI summaries, Fig. 9 presents pathway-specific empirical cumulative distributions for the equal-contact urban scenario. Children and adults are overlaid within each panel, allowing comparison of the simulated distributions and their P50 and P95 values. Panels A–B show non-carcinogenic HI, whereas panels C–D show quantified arsenic cancer risk. These distributions describe the specified recreational exposure scenario and should not be interpreted as population disease frequencies (US EPA, 2001).

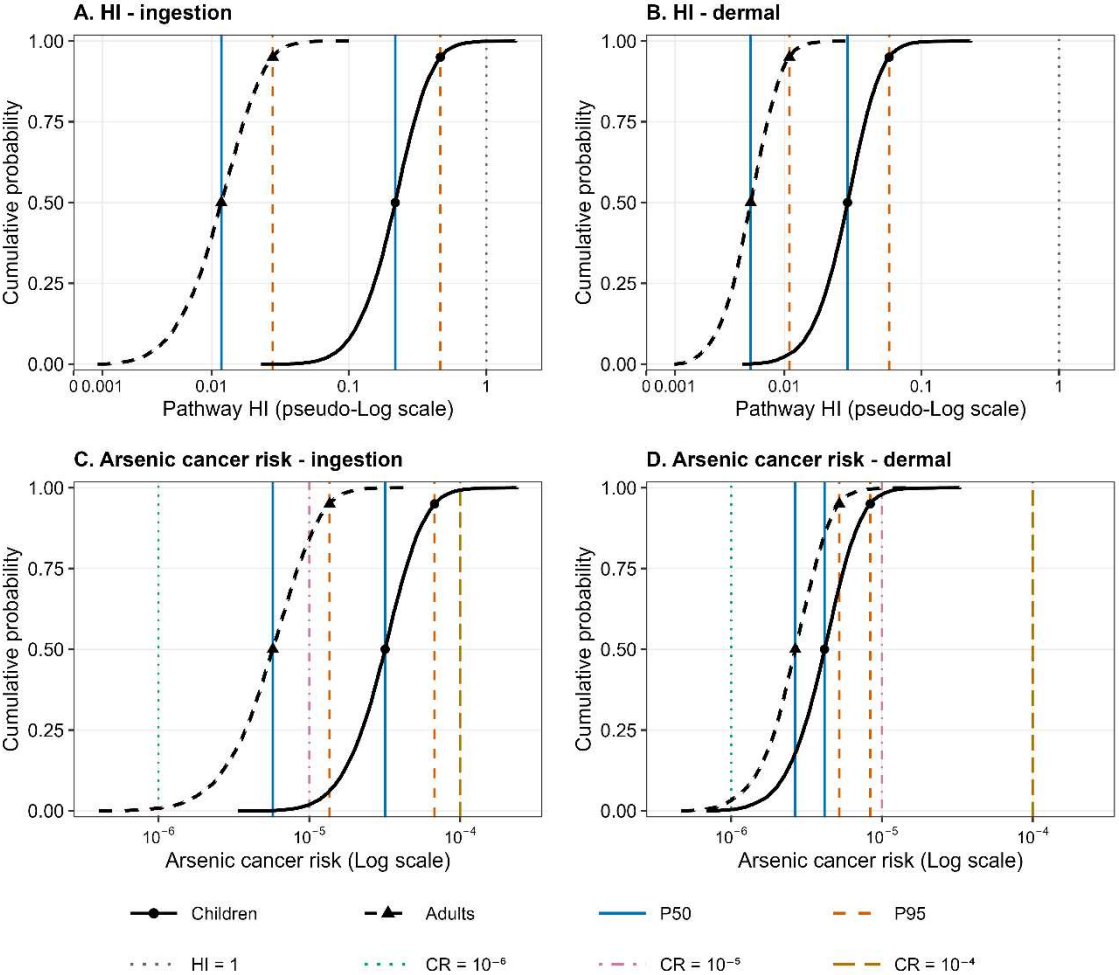


Fig. 9. Pathway-specific empirical cumulative distributions under the equal-contact urban recreational scenario. Panels show HI for ingestion (A) and dermal contact (B), and arsenic lifetime cancer risk for ingestion (C) and dermal

contact (D). Black solid curves and circles represent children; black dashed curves and triangles represent adults. Blue solid and orange dashed vertical lines indicate age-specific P50 and P95, respectively. The grey dotted line marks $HI = 1$; green, magenta, and ochre lines mark cancer-risk reference levels of 10^{-6} , 10^{-5} , and 10^{-4} . HI uses pseudo-Log scaling, whereas cancer risk uses Log scaling. The urban scenario assigns equal contact weights to P1–P4 and P6 and excludes P5. Cancer-risk reference levels provide a screening and risk-management context rather than Colombian regulatory limits (US EPA, 2026).

Under the equal-contact urban scenario, the arsenic cancer-risk distributions for children were shifted towards higher values than those for adults, while ingestion produced higher estimates than dermal contact within each age group (Fig. 9C–D). In the displayed distributions, ingestion P95 values for both age groups lie between 10^{-5} and 10^{-4} , whereas dermal P95 values lie between 10^{-6} and 10^{-5} . These age-group contrasts reflect the combined effects of the adopted exposure factors and separate exposure durations, rather than a comparison of otherwise identical lifetime exposure histories (US EPA, 1989, 2004).

The quantified cancer-risk estimates apply only to arsenic and depend on the screening assumptions that measured arsenic was inorganic and oral relative bioavailability was 1; neither assumption was verified experimentally in the sampled soils (US EPA, 2025). The reference levels of 10^{-6} and 10^{-4} delimit EPA’s cancer-risk management range, with 10^{-5} included as an intermediate reference. Accordingly, exceeding 10^{-6} alone does not establish exceedance of a universal maximum permissible risk, and the simulated fractions above these references should not be interpreted as observed cancer incidence (US EPA, 2001, 2026).

Site-specific estimates revealed arsenic cancer-risk contrasts that were less apparent in the equal-contact urban scenario. For children at P2, combined ingestion and dermal cancer risk had a median of 1.09×10^{-4} and a P95 of 2.29×10^{-4} , with 56.10% of simulated values exceeding 10^{-4} . At P6, the corresponding P95 was 1.07×10^{-4} , with 6.91% of simulations exceeding this reference. In contrast, the equal-contact urban scenario yielded a child P95 of 7.60×10^{-5} and an exceedance fraction of 1.24%. These results show why the urban scenario complements rather than replaces site-specific assessment. The exceedance fractions describe conditional model outputs, not observed cancer incidence.

These screening results support prioritizing confirmatory exposure assessment at P2, where the child combined HI distribution showed the largest simulated exceedance fraction (Table 3). Interpretation remains limited by the use of one composite per site, transferred exposure distributions, and assumed chemical forms and absorption factors. In particular, the Ni and Zn dermal estimates depend on the transferred ABSd assumption documented by Salazar et al. (2024). Additional sampling, local attendance information, and measurements of arsenic speciation and oral bioaccessibility would help refine the assessment. P5 provides a separate operational comparison and should not be interpreted as a health-based reference (US EPA, 2001, 2004).

4. Conclusions

Surface-soil analysis revealed contrasting PHE concentration profiles among the six sampled recreational sites in Caldas. Among the five urban parks, P2 had the highest As, Cd, and Ni concentrations, whereas P4 had the highest Pb, Cr, Zn, and Cu concentrations. PCA and hierarchical clustering described these multielement contrasts but did not establish their sources. The findings therefore characterize differences among the sampled site composites rather than within-park variability or municipality-wide soil conditions.

Geochemical indices consistently identified P2 and P4 as the sites with the largest integrated contrasts relative to P5. Cd dominated PERI because of both its conventional toxic-response weight and its concentration relative to the P5 denominator. These results support comparative prioritization of the sampled sites; they do not establish contamination classes against a validated natural background or demonstrate ecological damage.

Under the specified recreational exposure assumptions, probabilistic screening identified P2 as the priority for confirmatory human-health assessment. The equal-contact urban scenario complemented the site-specific assessment but did not replace it, because averaging concentrations across parks reduced the visibility of local contrasts. Cancer-risk estimates were restricted to arsenic and depended on the assumed inorganic fraction, relative oral bioavailability, and exposure parameters. Further sampling at P2 and P4, together with additional reference-site characterization, local soil-contact information, and arsenic speciation and oral bioaccessibility measurements, would help determine whether the observed contrasts warrant site-specific exposure-management measures.

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Competing Interests

The authors declare that they have no conflict of interest.

Data availability

Data will be made available on request.

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Credit author statement

Juan D. Correa: formal analysis, visualization, project administration, funding acquisition and writing – original draft. Andrea Tamayo: Supervision, project administration, funding acquisition, resources, review and editing. Juan P. Salazar: Supervision, project administration, funding acquisition, resources, review and editing. Julián E. López: Conceptualization, methodology, data curation, validation, formal analysis, visualization, project administration, funding acquisition and writing – original draft.

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