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### Spatial and Seasonal Patterns of Pb, Cd, Ni, Cr, Cu, and Zn in Water and Soil Across Contrasting Land-Use Types: Relationships with Physicochemical Parameters and Total Metal Load

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#### Abstract

Heavy metal contamination remains a persistent environmental concern because metals are non-biodegradable, may accumulate in soil and water, and can be redistributed by runoff, dust deposition, drainage, and resuspension. Objective: This study assessed spatial and temporal patterns of Pb, Cd, Ni, Cr, Cu, and Zn in water and soil across contrasting land-use categories and examined their relationships with physicochemical variables. Methods: The dataset included six sites representing semi-rural reference, residential, hospital/urban service, traffic, industrial/cement, and agricultural/suburban settings. Monthly date stamps covered October 2025 to March 2026, with three replicates per site, month, and matrix, yielding 216 observations. Metals were determined after acid digestion by atomic absorption spectrophotometry. Descriptive statistics, Shapiro-Wilk tests, Kruskal-Wallis tests, Benjamini-Hochberg-adjusted Dunn comparisons, and Spearman correlations were applied separately for water and soil. The industrial/cement site had the highest mean total measured metal load in water and soil, followed by the traffic site, whereas the semi-rural reference site had the lowest burden. Mean total load was  $0.619 \pm 0.223$  mg/L in water and  $334.19 \pm 99.02$  mg/kg dry weight in soil. Site-level differences were significant in both matrices, with a stronger spatial effect in soil. Water showed significant month/season grouping effects, but this temporal pattern was exploratory because the dataset covered only six months and spring was represented by one month. EC correlated strongly with total load in water and soil, while turbidity correlated moderately with water metal burden. Conclusion: The dataset supports a clear land-use gradient, with industrial/cement and traffic settings representing the main monitoring hotspots. The findings should be interpreted as total recoverable metal enrichment, not as direct toxicity, bioavailability, or source-apportionment evidence.

#### Keywords

Trace Metals;  
Water Quality;  
Soil Contamination;  
Land-Use Gradient;  
Electrical Conductivity;  
Cement Industry;  
Traffic Emissions.



## 1. Introduction

Heavy metal contamination in environmental matrices remains one of the most persistent forms of pollution because metals are not biologically degraded and may accumulate in water, soil, sediment, and biota over long periods [1-3]. Elements such as Pb, Cd, Ni, Cr, Cu, and Zn differ in toxicity, essentiality, mobility, and environmental behavior; however, enrichment above natural or local background levels can reduce microbial activity, inhibit plant growth, alter aquatic quality, and increase bioaccumulation concern [1,2]. In urban and peri-urban environments, metal enrichment usually reflects combined natural and anthropogenic inputs rather than a single source.

Water and soil are environmentally linked but functionally distinct contamination compartments. Water can reflect short-term contamination pulses caused by runoff, drainage, suspended particles, dilution, evaporation, and resuspension, whereas soil commonly acts as an integrative archive of longer-term deposition and accumulation [3-6]. This distinction is important because water concentrations are expressed in mg/L, while soil concentrations are expressed in mg/kg dry weight. Therefore, the two matrices should be analyzed separately and interpreted according to their own environmental behavior rather than directly pooled.

Land use is a major determinant of heavy metal loading. Industrial zones and cement-production areas may release metal-bearing dust through raw material handling, combustion, grinding, transport, and particulate deposition [7-9]. Traffic corridors and intersections may accumulate metals through non-exhaust emissions, including brake wear, tire wear, road-surface abrasion, lubricant residues, and road-dust resuspension [10-12]. Residential districts, hospital/urban service areas, agricultural/suburban areas, and semi-rural reference sites represent different combinations of diffuse domestic activity, service traffic, drainage influence, irrigation return flow, soil resuspension, and background conditions.

Physicochemical variables are essential for interpreting metal distribution. pH affects metal

solubility, adsorption, precipitation, and mobility, particularly for Cd, Zn, and Pb under changing acidity conditions [13,14]. Electrical conductivity (EC) reflects dissolved ionic strength and may indicate salinity, mineralization, drainage influence, or anthropogenic solute inputs [4,15]. Turbidity can represent suspended particulate matter that may transport particle-bound metals in water, while soil moisture can influence leaching, retention, redox conditions, and dust mobilization. Integrating physicochemical variables enables mechanistic interpretation of metal mobility, bioavailability, and source-receptor relationships beyond simple concentration reporting.

The present study addresses a monitoring gap by jointly assessing Pb, Cd, Ni, Cr, Cu, Zn, and total measured metal load in water and soil across six contrasting land-use categories over multiple months. The aims were to: (i) characterize metal concentrations in water and soil; (ii) compare spatial patterns among semi-rural reference, residential, hospital/urban service, traffic, industrial/cement, and agricultural/suburban sites; (iii) evaluate temporal group differences; and (iv) examine associations between total measured metal burden and pH, EC, turbidity, and soil moisture. Because no isotope tracers, receptor models, mineralogical fingerprints, or direct source markers were included, source interpretation was framed as land-use-consistent association rather than definitive source apportionment.

## 2. Materials and Methods

### 2.1 Study design

This study was designed as a spatially and temporally resolved observational environmental assessment of heavy metal contamination in water and soil across contrasting land-use categories. Separate analyses were conducted for water and soil because the matrices differ fundamentally in units, concentration scale, retention mechanisms, and environmental responsiveness.

2.2 Study area and site classification

Six sites were included, each representing a distinct land-use category (Table 1). This land-use classification was used as an exposure-related

grouping variable because each site type reflects different probable source structures, transport pathways, and environmental loading mechanisms.

Table 1. Site classification and dataset-provided land-use descriptors.

| Site | Site type                      | Land use         | Field description                         | Relative factor |
|------|--------------------------------|------------------|-------------------------------------------|-----------------|
| S1   | Semi-rural reference area      | Low-background   | Reference/low traffic                     | 1.00            |
| S2   | Residential district           | Residential      | Urban domestic activity                   | 1.18            |
| S3   | Hospital/urban service area    | Healthcare-urban | Hospital and service traffic              | 1.28            |
| S4   | Traffic intersection           | Traffic          | Dense vehicle activity and road dust      | 1.55            |
| S5   | Industrial/cement factory area | Industrial       | Industrial dust and fuel combustion       | 2.10            |
| S6   | Agricultural/suburban area     | Agricultural     | Soil resuspension and irrigation drainage | 1.30            |

2.3 Sampling strategy and temporal grouping

Water and soil samples were recorded monthly using dataset date stamps from 1 October 2025 to 1 March 2026. These date stamps were retained as the recorded monitoring window. For each site, matrix, and month, three replicate samples were recorded. The dataset therefore comprised six sites, two matrices, six months, and three replicates per site/month/matrix, yielding 216 observations: 108 water samples and 108 soil samples.

Seasonal labels followed the dataset structure: autumn covered October-November, winter covered December-February, and spring covered March. Because the temporal grouping was unbalanced and covered only six months, seasonal results were interpreted as exploratory month-grouped patterns rather than as a complete climatological seasonal cycle.

2.4 Sample preparation and digestion

Water samples were analyzed for total recoverable metal concentrations. Prior to analysis, samples were preserved with nitric acid, acidified to approximately pH <2, and subjected to acid-based digestion to release dissolved and particle-associated metal fractions. After digestion, samples were cooled, filtered where necessary, and transferred quantitatively to volumetric flasks. Final volume was completed with deionized water before instrumental determination.

Soil samples were air-dried, homogenized, and sieved to remove coarse debris. A known dry mass was digested using aqua regia prepared from concentrated nitric acid and hydrochloric acid. Digestion was performed under controlled heating until acid-recoverable metals were extracted. The digest was

cooled, filtered where necessary, and transferred quantitatively to a volumetric flask. Final volume was completed using deionized water. Procedural blanks were processed using the same digestion and dilution procedure [18-21].

The analytical approach therefore measured total recoverable or acid-extractable metals, not dissolved-only, exchangeable, carbonate-bound, Fe/Mn oxide-bound, organic-bound, residual, or bioavailable fractions. This distinction is central to interpretation because total recoverable concentrations may overestimate mobile or bioavailable pools.

## 2.5 Instrumental determination of heavy metals

Pb, Cd, Ni, Cr, Cu, and Zn were determined by atomic absorption spectrophotometry. Flame atomic absorption spectrophotometry was used for routine quantification within the linear calibration range, while graphite furnace atomic absorption spectrophotometry was used where low-level detection was analytically required, particularly for trace-level Pb and Cd in water. Concentrations were reported as mg/L for water and mg/kg dry weight for soil [22,23].

The analytical wavelengths followed standard AAS operating conditions: Pb at 217.0 nm, Cd at 228.8 nm, Ni at 232.0 nm, Cr at 357.9 nm, Cu at 324.8 nm, and Zn at 213.9 nm. Calibration curves were prepared using certified standard solutions, and results were accepted only when calibration and quality-control criteria were satisfied.

## 2.6 Analytical variables

The primary variables were Pb, Cd, Ni, Cr, Cu, and Zn. The dataset also included total metal load, calculated as the summed concentration of the six measured

metals within each sample. In this manuscript, total metal load is interpreted as total measured metal burden rather than total toxicity, because summed concentration does not account for metal-specific toxicity, chemical speciation, bioavailability, exposure pathway, or ecological sensitivity.

Physicochemical variables included pH, EC, water turbidity, and soil moisture. Turbidity was interpreted as a water-specific variable, while soil moisture was interpreted as a soil-specific variable.

## 2.7 Quality assurance and quality control

Quality assurance and quality control included procedural blanks, limits of detection (LOD), limits of quantification (LOQ), calibration standards, duplicate measurements when available, and spike-recovery assessment. The dataset included LOD, LOQ, blank result, and spike-recovery values for each metal and matrix (Table 2). For water, LODs ranged from 0.0001 to 0.010 mg/L, LOQs from 0.0003 to 0.0333 mg/L, and spike recovery from 89.9% to 104.3%. For soil, LODs ranged from 0.03 to 1.50 mg/kg dry weight, LOQs from 0.10 to 5.00 mg/kg dry weight, and spike recovery from 88.2% to 101.2%.

Blank values were low relative to measured concentrations, and recorded minimum concentrations exceeded the corresponding LOQs. However, the dataset did not include metal-specific duplicate digestion precision, matrix-spike frequency, or %RSD by batch. Therefore, the available QC data support general analytical reliability but do not fully document batch-level reproducibility.

Table 2. Analytical quality-control metrics available in the dataset.

| Matrix | Metal | LOD    | LOQ    | Unit             | Blank  | Spike recovery (%) |
|--------|-------|--------|--------|------------------|--------|--------------------|
| Water  | Cd    | 0.0001 | 0.0003 | mg/L             | 0.0001 | 89.9               |
| Water  | Ni    | 0.002  | 0.0067 | mg/L             | 0.0012 | 104.3              |
| Water  | Cr    | 0.0015 | 0.005  | mg/L             | 0.0009 | 96.7               |
| Water  | Cu    | 0.003  | 0.01   | mg/L             | 0.0008 | 98.4               |
| Water  | Zn    | 0.01   | 0.0333 | mg/L             | 0.0043 | 95.7               |
| Soil   | Pb    | 0.5    | 1.67   | mg/kg dry weight | 0.1    | 101.2              |
| Soil   | Cd    | 0.03   | 0.1    | mg/kg dry weight | 0.01   | 98.2               |
| Soil   | Ni    | 0.6    | 2      | mg/kg dry weight | 0.01   | 94.3               |
| Soil   | Cr    | 0.7    | 2.33   | mg/kg dry weight | 0.39   | 95.2               |
| Soil   | Cu    | 0.4    | 1.33   | mg/kg dry weight | 0.05   | 98.8               |
| Soil   | Zn    | 1.5    | 5      | mg/kg dry weight | 0.64   | 88.2               |

## 2.8 Statistical analysis

All statistical analyses were conducted separately for water and soil. Descriptive statistics included mean, standard deviation, median, interquartile range, minimum, and maximum. Shapiro-Wilk tests were used to assess normality of metal and total-load distributions. Because the variables showed non-normal distributions, nonparametric methods were used for primary comparisons [16,17].

Site-level and temporal group differences were evaluated using the Kruskal-Wallis test. Where Kruskal-Wallis results were significant, Dunn post-hoc comparisons were interpreted with Benjamini-Hochberg false-discovery-rate adjustment. Spearman rank correlation was used to examine monotonic associations between total measured metal load and physicochemical variables. Exploratory regression models were not retained as primary evidence because the dataset-provided relative contamination factor is

a site-level derived index and may be circular if used to explain the same land-use gradient. Statistical significance was set at  $p < 0.05$ , and epsilon-squared was used as an effect-size estimate for Kruskal-Wallis tests.

## 2.9 Contextual comparison framework

Water concentrations were compared with WHO drinking-water guideline values for Pb, Cd, Ni, Cr, and Cu, with Zn interpreted cautiously because zinc guideline values are generally aesthetic or acceptability-based rather than health-based [24]. These comparisons provide screening context only; they do not establish drinking-water risk because the dataset does not define water-use purpose, exposure duration, treatment status, or consumption pathway.

For soil, national background values, a conservative reference element such as Fe, Al, Ti, or Mn, and land-use-specific regulatory standards were not provided. Therefore, classical enrichment factors could not be

calculated. Instead, site-to-reference ratios were calculated using the semi-rural reference site (S1) as a dataset-internal baseline. These ratios provide relative enrichment context but should not be interpreted as regulatory exceedance or ecological risk.

### 3. Results

#### 3.1 Overall descriptive patterns

The dataset contained 216 observations: 108 water samples and 108 soil samples. Mean total measured

metal load was  $0.619 \pm 0.223$  mg/L in water and  $334.19 \pm 99.02$  mg/kg dry weight in soil. Water and soil results are reported separately because their units and environmental meanings differ.

In water, Zn had the highest mean concentration (0.416 mg/L), followed by Cu (0.0995 mg/L), Ni (0.0451 mg/L), Cr (0.0319 mg/L), Pb (0.0236 mg/L), and Cd (0.0030 mg/L). In soil, Zn again had the highest mean concentration (131.42 mg/kg), followed by Cr (70.05 mg/kg), Pb (46.50 mg/kg), Ni (45.64 mg/kg), Cu (39.34 mg/kg), and Cd (1.24 mg/kg).

*Table 3. Descriptive statistics for metals and total measured metal load by matrix. Water concentrations are in mg/L; soil concentrations are in mg/kg dry weight.*

| Matrix | Variable         | Unit     | N   | Mean   | SD     | Median | Q1     | Q3     | Min    | Max    |
|--------|------------------|----------|-----|--------|--------|--------|--------|--------|--------|--------|
| Water  | Pb               | mg/L     | 108 | 0.0236 | 0.0111 | 0.0209 | 0.0168 | 0.0266 | 0.0088 | 0.0620 |
| Water  | Cd               | mg/L     | 108 | 0.0030 | 0.0012 | 0.0027 | 0.0022 | 0.0036 | 0.0012 | 0.0084 |
| Water  | Ni               | mg/L     | 108 | 0.0451 | 0.0177 | 0.0415 | 0.0344 | 0.0508 | 0.0149 | 0.0947 |
| Water  | Cr               | mg/L     | 108 | 0.0319 | 0.0135 | 0.0289 | 0.0224 | 0.0384 | 0.0132 | 0.0920 |
| Water  | Cu               | mg/L     | 108 | 0.0995 | 0.0383 | 0.0880 | 0.0710 | 0.1202 | 0.0440 | 0.2390 |
| Water  | Zn               | mg/L     | 108 | 0.4158 | 0.1734 | 0.3810 | 0.3020 | 0.4710 | 0.1400 | 1.0800 |
| Water  | Total_Metal_Load | mg/L     | 108 | 0.6189 | 0.2231 | 0.5724 | 0.4603 | 0.6928 | 0.2930 | 1.4584 |
| Soil   | Pb               | mg/kg dw | 108 | 46.50  | 17.57  | 42.89  | 34.47  | 55.44  | 16.67  | 93.36  |
| Soil   | Cd               | mg/kg dw | 108 | 1.24   | 0.40   | 1.17   | 0.97   | 1.43   | 0.45   | 2.39   |
| Soil   | Ni               | mg/kg dw | 108 | 45.64  | 15.93  | 42.66  | 33.28  | 53.96  | 19.88  | 95.00  |
| Soil   | Cr               | mg/kg dw | 108 | 70.05  | 24.49  | 64.98  | 50.33  | 83.84  | 30.03  | 135.00 |
| Soil   | Cu               | mg/kg dw | 108 | 39.34  | 13.28  | 36.42  | 29.66  | 48.57  | 16.40  | 80.31  |
| Soil   | Zn               | mg/kg dw | 108 | 131.42 | 47.30  | 120.66 | 96.82  | 155.96 | 61.53  | 297.78 |
| Soil   | Total_Metal_Load | mg/kg dw | 108 | 334.19 | 99.02  | 317.41 | 258.46 | 385.42 | 181.27 | 647.08 |

### 3.2 Land-use differences

A clear land-use gradient was observed in both matrices (Table 4). In water, the highest mean total measured metal load occurred at the industrial/cement factory site, S5 (0.9619 mg/L), followed by the traffic intersection, S4 (0.6703 mg/L). Intermediate values were observed at the hospital/urban service area, S3 (0.5734 mg/L), and agricultural/suburban site, S6 (0.5603 mg/L). Lower values occurred at the residential district, S2 (0.5083 mg/L), and the semi-rural reference site, S1 (0.4391 mg/L).

The same ranking was broadly reproduced in soil. S5 showed the highest mean total measured metal load (495.92 mg/kg), followed by S4 (387.04 mg/kg). S3 and S6 showed intermediate burdens, while S2 and S1 showed lower burdens. Kruskal-Wallis's testing confirmed significant site-level differences for total load in both water ( $H = 57.48$ ,  $p < 0.001$ , epsilon-squared = 0.515) and soil ( $H = 80.91$ ,  $p < 0.001$ , epsilon-squared = 0.744). The stronger effect size in soil indicates a more pronounced spatial signal in this matrix.

Table 4. Mean metal concentrations and total measured metal load by site and matrix.

| Matrix | Site | Site type                      | N  | Pb     | Cd     | Ni     | Cr     | Cu     | Zn     | Total load |
|--------|------|--------------------------------|----|--------|--------|--------|--------|--------|--------|------------|
| Water  | S1   | Semi-rural reference area      | 18 | 0.0159 | 0.0022 | 0.0311 | 0.0212 | 0.0746 | 0.2940 | 0.4391     |
| Water  | S2   | Residential district           | 18 | 0.0184 | 0.0023 | 0.0367 | 0.0262 | 0.0899 | 0.3347 | 0.5083     |
| Water  | S3   | Hospital/urban service area    | 18 | 0.0207 | 0.0028 | 0.0422 | 0.0299 | 0.0898 | 0.3879 | 0.5734     |
| Water  | S4   | Traffic intersection           | 18 | 0.0263 | 0.0035 | 0.0526 | 0.0366 | 0.1042 | 0.4472 | 0.6703     |
| Water  | S5   | Industrial/cement factory area | 18 | 0.0382 | 0.0044 | 0.0645 | 0.0485 | 0.1523 | 0.6539 | 0.9619     |
| Water  | S6   | Agricultural/suburban area     | 18 | 0.0221 | 0.0027 | 0.0431 | 0.0290 | 0.0861 | 0.3771 | 0.5603     |
| Soil   | S1   | Semi-rural reference area      | 18 | 30.24  | 0.92   | 32.40  | 47.01  | 28.01  | 90.21  | 228.79     |
| Soil   | S2   | Residential district           | 18 | 39.20  | 1.02   | 35.76  | 57.54  | 31.37  | 108.02 | 272.90     |
| Soil   | S3   | Hospital/urban service area    | 18 | 42.61  | 1.16   | 43.71  | 65.87  | 37.96  | 122.50 | 313.82     |
| Soil   | S4   | Traffic intersection           | 18 | 57.36  | 1.37   | 49.98  | 83.62  | 45.21  | 149.50 | 387.04     |
| Soil   | S5   | Industrial/cement factory area | 18 | 68.63  | 1.79   | 69.43  | 105.53 | 58.41  | 192.13 | 495.92     |
| Soil   | S6   | Agricultural/suburban area     | 18 | 40.96  | 1.18   | 42.54  | 60.71  | 35.10  | 126.19 | 306.67     |

### 3.3 Contextual comparison with guideline and reference values

Water guideline screening showed that Pb had the clearest exceedance pattern relative to the WHO guideline value of 0.010 mg/L, with 106 of 108 water observations above this value (98.1%). Cd exceeded

0.003 mg/L in 41 samples (38.0%). Ni and Cr exceeded their screening values in smaller proportions of samples, while Cu and Zn remained below the selected guideline or acceptability values in all samples (Table 5). These comparisons provide concentration context only and do not constitute a human-health risk assessment.

*Table 5. Screening comparison of water metal concentrations with WHO guideline or acceptability values. This table is not a drinking-water risk assessment.*

| Metal | Reference value (mg/L) | Mean   | Median | Max    | Samples above value | % above | Mean/reference |
|-------|------------------------|--------|--------|--------|---------------------|---------|----------------|
| Pb    | 0.01                   | 0.0236 | 0.0209 | 0.0620 | 106/108             | 98.1    | 2.36           |
| Cd    | 0.003                  | 0.0030 | 0.0027 | 0.0084 | 41/108              | 38.0    | 1.00           |
| Ni    | 0.07                   | 0.0451 | 0.0415 | 0.0947 | 10/108              | 9.3     | 0.64           |
| Cr    | 0.05                   | 0.0319 | 0.0289 | 0.0920 | 11/108              | 10.2    | 0.64           |
| Cu    | 2                      | 0.0995 | 0.0880 | 0.2390 | 0/108               | 0.0     | 0.05           |
| Zn    | 3                      | 0.4158 | 0.3810 | 1.0800 | 0/108               | 0.0     | 0.14           |

For soil, no measured conservative reference element was available; therefore, classical enrichment factors could not be calculated. Site-to-reference ratios showed that total soil metal burden at S5 was more than twice the semi-rural reference value, while S4 showed a clear secondary elevation. To strengthen contamination contextualization without overstating source attribution, a reference-normalized pollution-

load index (PLI<sub>ref</sub>) was calculated from the geometric mean of metal-specific site/S1 ratios. PLI<sub>ref</sub> increased from 1.00 at S1 to 2.13 at S5 and 1.66 at S4, with lower-to-intermediate values at S2, S3, and S6 (Table 7). This index is an internal screening metric and should not be interpreted as a regulatory pollution-load index based on regional geochemical background.

*Table 6. Site-to-reference ratios using S1 as the internal reference site. Ratios are descriptive and are not classical enrichment factors.*

| Site | Site type                      | Water total/S1 | Soil total/S1 | Water Pb/S1 | Water Cd/S1 | Soil Pb/S1 | Soil Cd/S1 |
|------|--------------------------------|----------------|---------------|-------------|-------------|------------|------------|
| S1   | Semi-rural reference area      | 1.00           | 1.00          | 1.00        | 1.00        | 1.00       | 1.00       |
| S2   | Residential district           | 1.16           | 1.19          | 1.15        | 1.05        | 1.30       | 1.11       |
| S3   | Hospital/urban service area    | 1.31           | 1.37          | 1.30        | 1.27        | 1.41       | 1.26       |
| S4   | Traffic intersection           | 1.53           | 1.69          | 1.65        | 1.56        | 1.90       | 1.49       |
| S5   | Industrial/cement factory area | 2.19           | 2.17          | 2.40        | 1.96        | 2.27       | 1.95       |
| S6   | Agricultural/suburban area     | 1.28           | 1.34          | 1.39        | 1.23        | 1.35       | 1.28       |

Table 7. Reference-normalized soil contamination factors and pollution-load index (PLIref) using S1 as the internal reference site.

| Site | Site type                      | Pb   | Cd   | Ni   | Cr   | Cu   | Zn   | PLIref | Total ratio |
|------|--------------------------------|------|------|------|------|------|------|--------|-------------|
| S1   | Semi-rural reference area      | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00   | 1.00        |
| S2   | Residential district           | 1.30 | 1.11 | 1.10 | 1.22 | 1.12 | 1.20 | 1.17   | 1.19        |
| S3   | Hospital/urban service area    | 1.41 | 1.26 | 1.35 | 1.40 | 1.36 | 1.36 | 1.36   | 1.37        |
| S4   | Traffic intersection           | 1.90 | 1.49 | 1.54 | 1.78 | 1.61 | 1.66 | 1.66   | 1.69        |
| S5   | Industrial/cement factory area | 2.27 | 1.95 | 2.14 | 2.24 | 2.09 | 2.13 | 2.13   | 2.17        |
| S6   | Agricultural/suburban area     | 1.35 | 1.28 | 1.31 | 1.29 | 1.25 | 1.40 | 1.31   | 1.34        |

*Note. Values are site/S1 ratios calculated from mean soil concentrations. PLIref is the geometric mean of the six metal-specific ratios. This is a reference-normalized screening index, not a classical enrichment factor based on a conservative reference element.*

### 3.4 Temporal variation

Temporal grouping effects were more evident in water than in soil (Table 8). Water total load differed by season label ( $H = 10.57$ ,  $p = 0.005$ ) and by sampling month ( $H = 15.92$ ,  $p = 0.007$ ). Mean water total load peaked in December 2025 (0.7226 mg/L), remained elevated in January 2026 (0.6663 mg/L), and declined by March 2026 (0.4929 mg/L).

Soil showed a descriptive winter elevation, with the highest monthly mean in December 2025 (386.59 mg/kg), but the season-label effect did not reach conventional significance ( $H = 5.25$ ,  $p = 0.072$ ). Because the dataset covered only six months and spring was represented by March only, temporal results should be interpreted as exploratory month-grouped patterns rather than full seasonal cycles.

Table 8. Monthly mean total measured metal load by matrix.

| Matrix | Month    | Season label | N  | Mean total load | SD     |
|--------|----------|--------------|----|-----------------|--------|
| Water  | Oct 2025 | Autumn       | 18 | 0.6379          | 0.2183 |
| Water  | Nov 2025 | Autumn       | 18 | 0.6516          | 0.2074 |
| Water  | Dec 2025 | Winter       | 18 | 0.7227          | 0.3242 |
| Water  | Jan 2026 | Winter       | 18 | 0.6663          | 0.1698 |
| Water  | Feb 2026 | Winter       | 18 | 0.5417          | 0.1457 |
| Water  | Mar 2026 | Spring       | 18 | 0.4929          | 0.1706 |
| Soil   | Oct 2025 | Autumn       | 18 | 322.52          | 95.73  |
| Soil   | Nov 2025 | Autumn       | 18 | 336.64          | 96.47  |
| Soil   | Dec 2025 | Winter       | 18 | 386.58          | 112.71 |
| Soil   | Jan 2026 | Winter       | 18 | 356.26          | 99.62  |
| Soil   | Feb 2026 | Winter       | 18 | 310.81          | 81.47  |
| Soil   | Mar 2026 | Spring       | 18 | 292.34          | 89.30  |

3.5 Relationships with physicochemical variables

Water pH was near-neutral to slightly alkaline, with a mean of  $7.53 \pm 0.22$ . Mean EC was  $1903.86 \pm 555.09 \mu\text{S}/\text{cm}$ , and turbidity averaged  $12.55 \pm 5.66 \text{ NTU}$ . Soil pH was also slightly alkaline, with a mean of  $7.74 \pm 0.19$ . Soil EC averaged  $1353.23 \pm 411.77 \mu\text{S}/\text{cm}$ , and soil moisture averaged  $7.31 \pm 2.21\%$ .

EC showed the strongest association with total measured metal load in both matrices (Table 9). In water, total load correlated strongly with EC ( $\rho =$

$0.725$ ,  $p < 0.001$ ) and moderately with turbidity ( $\rho = 0.448$ ,  $p < 0.001$ ). In soil, total load correlated strongly with EC ( $\rho = 0.738$ ,  $p < 0.001$ ). Soil moisture was not meaningfully associated with total load, and pH showed weak correlation with total recoverable metal load in both matrices. The correlation heatmap further visualizes the matrix-specific pattern: EC aligned positively with most metals and total load in both matrices, whereas turbidity was most informative in water and soil moisture showed no consistent relationship with soil total load (Figure 3).

Table 9. Spearman correlations between total measured metal load and physicochemical variables.

| Matrix | Variable                       | Spearman rho | p-value |
|--------|--------------------------------|--------------|---------|
| Water  | pH                             | -0.130       | 0.180   |
| Water  | EC ( $\mu\text{S}/\text{cm}$ ) | 0.725        | <0.001  |
| Water  | Turbidity (NTU)                | 0.448        | <0.001  |
| Soil   | pH                             | 0.096        | 0.322   |
| Soil   | EC ( $\mu\text{S}/\text{cm}$ ) | 0.738        | <0.001  |
| Soil   | Soil moisture (%)              | -0.072       | 0.458   |

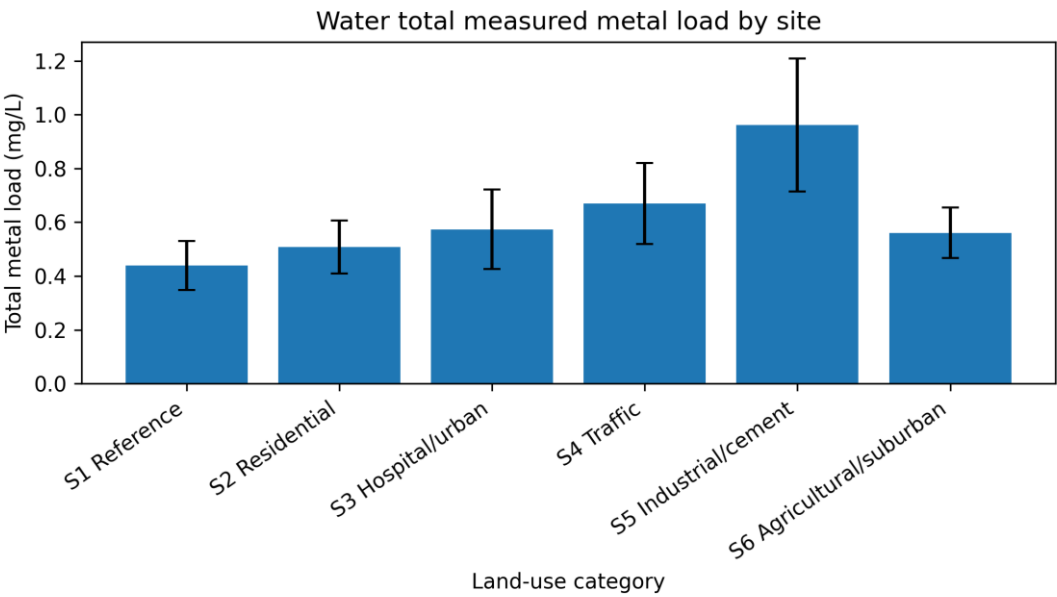


Figure 1. Mean total measured metal load in water by site. Error bars represent  $\pm 1 \text{ SD}$  ( $n = 18$  per site).

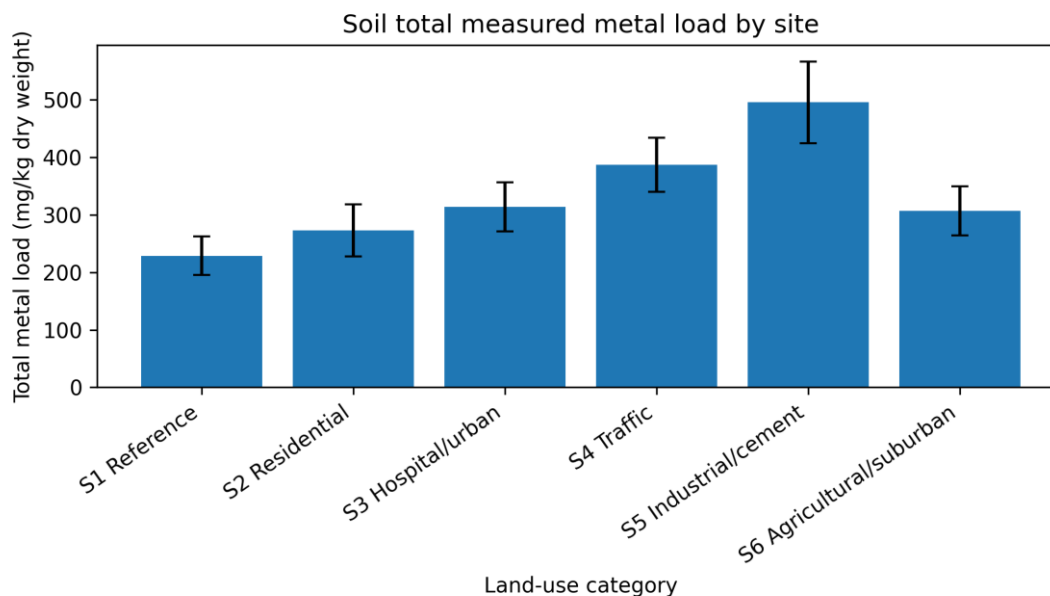


Figure 2. Mean total measured metal load in soil by site. Error bars represent  $\pm 1$  SD ( $n = 18$  per site).

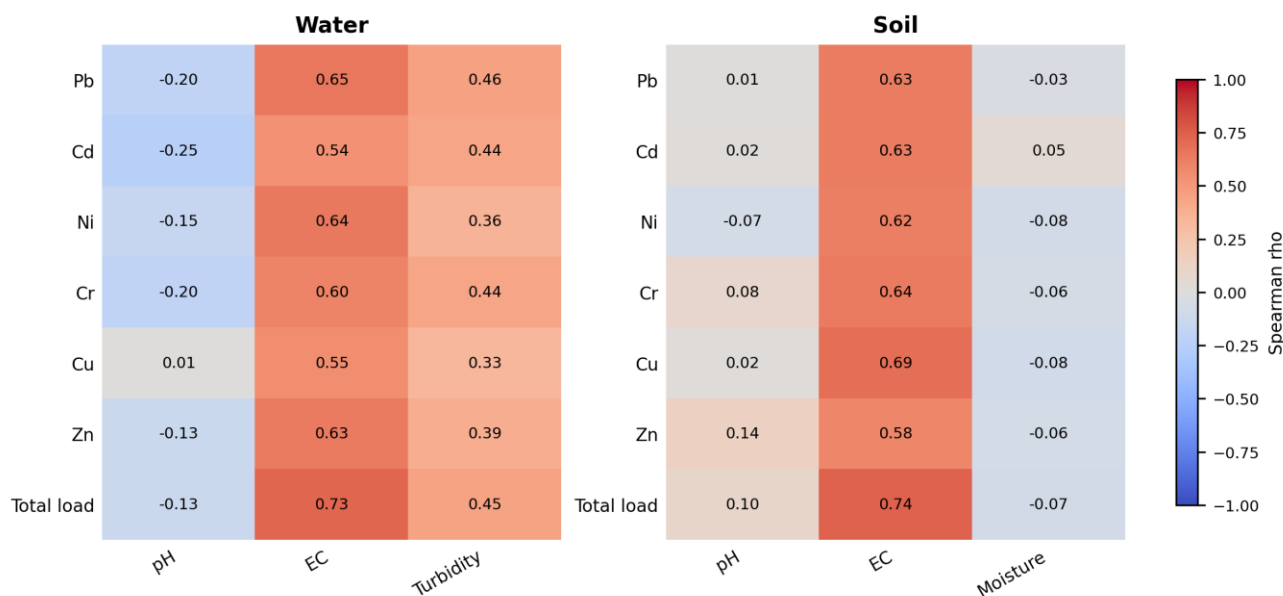


Figure 3. Spearman correlation heatmap showing associations between metals, total measured metal load, and physicochemical variables in water and soil. Values represent Spearman rho; water turbidity applies only to water, and soil moisture applies only to soil.

#### 4. Discussion

The main finding of this study is a consistent land-use-related gradient in total recoverable metal burden across both water and soil. The industrial/cement factory site showed the highest total measured metal load in both matrices, followed by the traffic intersection, whereas the semi-rural reference site showed the lowest concentrations. This pattern is consistent with, but does not prove, the contribution

of industrial/cement-related particulates and traffic-associated road dust to local metal enrichment [7-12]. The elevated burden at the industrial/cement site is consistent with cement-related emissions, but direct source confirmation was not performed. Cement-related activities can generate alkaline mineral dust and metal-bearing particulates during raw material processing, combustion, grinding, transport, and deposition [7-9]. Once released, these particles may

accumulate in topsoil through dry deposition or enter water through runoff, drainage, and suspended particulate transport. Nevertheless, geogenic background, construction debris, historical deposition, and mixed urban inputs cannot be excluded without source markers.

The traffic intersection showed the second-highest total measured metal load in both matrices. This pattern is consistent with evidence that non-exhaust vehicle emissions contribute metals to road-deposited sediments through brake wear, tire wear, pavement abrasion, and road-dust resuspension [10-12]. Cu and Zn are commonly associated with brake and tire-related sources, while Pb, Ni, and Cr may reflect mixed traffic, pavement, industrial-background, and geogenic contributions. The present data support traffic-site enrichment relative to the reference and residential sites but do not provide direct source apportionment.

Soil showed a stronger spatial effect than water, as reflected by the larger site-level epsilon-squared value. This is environmentally plausible because soil can retain metals through adsorption, precipitation, complexation, and incorporation into fine particles, mineral surfaces, and organic matter [13-15]. Water, by contrast, is more responsive to short-term hydrological and particulate processes, including runoff, turbidity changes, dilution, and seasonal redistribution. This matrix-specific behavior explains why soil more clearly preserved spatial differences, whereas water showed stronger temporal responsiveness.

The temporal pattern should be interpreted cautiously. Water total load peaked in December and remained elevated in January before declining toward March. This may reflect runoff, drainage connectivity, suspended particulate transport, or changes in dilution and ionic strength. However, the monitoring window covered only six months, and the seasonal grouping was unbalanced. Therefore, the water signal is best interpreted as a short-term temporal pattern rather than a complete seasonal cycle. The soil signal was descriptive and did not reach conventional significance by season label.

The strong association between EC and total measured metal load should be interpreted as co-variation rather than proof of causation. EC reflects dissolved ionic strength and can increase alongside salinity, mineralization, drainage inputs, or anthropogenic solute loads [4]. In aquatic systems, ionic strength can influence metal partitioning between dissolved and particle-bound forms, while in soils it may reflect soluble salts and contamination-associated mineral inputs. The positive association between turbidity and water total load supports the role of suspended particles as potential metal carriers.

pH showed weak correlation with total recoverable metal load in this dataset. This does not mean pH is environmentally unimportant. Rather, acid digestion extracts both pH-sensitive fractions and pH-insensitive residual or mineral-bound fractions; therefore, total recoverable concentrations may not track ambient pH as strongly as dissolved or exchangeable fractions would. The observed pH range was slightly alkaline and relatively narrow, which may also reduce observable statistical associations. Future work should compare filtered dissolved metals at ambient pH with total recoverable concentrations to evaluate pH-dependent mobility.

Zn was the largest numerical contributor to total measured metal load in both matrices. However, total load should not be equated with total toxicity because summed concentration is dominated by abundant metals and does not account for metal-specific toxicity, chemical speciation, bioavailability, exposure route, or ecological sensitivity. Cd and Pb may have greater toxicological relevance than their proportional contribution to total load suggests.

The guideline comparison provides useful screening context for water. Pb exceeded the WHO drinking-water guideline value in most water samples, while Cd exceeded its value in approximately one-third of samples. However, the dataset does not specify whether water was intended for drinking, irrigation, drainage, or environmental monitoring. Therefore, these comparisons should be interpreted as concentration-context screening rather than a formal health-risk assessment. A complete risk assessment

would require exposure pathways, ingestion rates, site use, dissolved fractions, and metal-specific hazard modeling. The reference-normalized PLI<sub>ref</sub> similarly provides a transparent soil screening context: S5 and S4 remained the dominant soil hotspots even when the six metals were integrated geometrically rather than summed arithmetically.

The environmental significance of the study lies in hotspot identification and monitoring prioritization. The industrial/cement and traffic sites consistently showed higher burdens than the reference site, while residential, hospital/urban service, and agricultural/suburban sites occupied intermediate positions. These findings support targeted follow-up monitoring but do not replace direct source apportionment, speciation analysis, or regulatory risk assessment.

## 5. Limitations

The primary limitation is that the analytical method measured total recoverable or acid-extractable metals rather than speciation, dissolved fractions, exchangeable fractions, or bioavailable pools. Total recoverable concentrations may overestimate mobile or biologically available metal fractions. Future studies should include 0.45  $\mu\text{m}$  filtration for water and sequential extraction protocols such as BCR or Tessier procedures, or single-extraction bioavailability tests such as DTPA or  $\text{CaCl}_2$  for soil [25,26].

Second, spatial association does not prove source attribution. The industrial/cement and traffic labels are useful for grouping, but no isotope tracers, mineralogical fingerprints, tire/brake markers, rare-earth patterns, or receptor modeling were used. Future studies should include PCA with source markers, positive matrix factorization, Pb isotope ratios, Zn isotope fingerprints, or distance-to-source gradients.

Third, temporal coverage was limited to six months and the seasonal grouping was unbalanced. Spring was represented by one month only, and no rainfall, wind, temperature, runoff, or event-based data were available. Year-round monthly sampling over multiple

years is required to distinguish true seasonal cycles from short-term monthly fluctuations.

Fourth, soil particle-size distribution, texture, clay content, organic matter, cation exchange capacity, Fe/Mn oxides, and mineralogy were not characterized. These properties strongly influence metal retention and could partly explain spatial variation. Future work should measure these properties and analyze fine fractions separately, particularly the  $<63 \mu\text{m}$  fraction.

Fifth, agricultural management history was not documented. Fertilizer type and application rate, irrigation water source, pesticide use, manure input, and soil amendment history were unavailable. Therefore, the intermediate burden at the agricultural/suburban site cannot be assigned to irrigation drainage, fertilizer-derived metals, livestock manure, or historical contamination.

Sixth, QC information was limited to LOD, LOQ, blank results, and spike recovery. Batch-level replicate digestion precision, matrix-spike frequency, and %RSD were not available. Future reports should include one procedural blank, one duplicate digestion, and one matrix spike per analytical batch, together with metal-specific precision.

Finally, the study did not perform ecological or human-health risk modeling. Guideline comparisons and site-to-reference ratios provide screening context but do not establish exposure risk, ecological risk, or regulatory classification.

## 6. Conclusion

This revised analysis demonstrates a consistent spatial gradient of total recoverable Pb, Cd, Ni, Cr, Cu, and Zn in water and soil across contrasting land-use categories. The industrial/cement factory site was the dominant hotspot, followed by the traffic intersection, while the semi-rural reference site showed the lowest burden. Soil showed a stronger spatial signal, consistent with cumulative deposition and retention, whereas water showed clearer short-term temporal responsiveness. Water guideline screening indicated that Pb, and to a lesser extent Cd, require particular attention, but these comparisons

are contextual and do not represent a formal health-risk assessment. Overall, the study is best framed as a preliminary but statistically robust spatial-temporal screening assessment that identifies monitoring priorities and justifies more detailed follow-up work using speciation, source tracing, and risk-based methods.

## 7. Recommendations

- Routine monitoring should prioritize industrial/cement and traffic-impacted sites while maintaining the semi-rural reference site as a background comparator.
- Future monitoring should include Pb, Cd, Ni, Cr, Cu, Zn, pH, EC, turbidity, soil moisture, rainfall, wind direction, runoff indicators, and land-use intensity metrics.
- Water monitoring should distinguish dissolved and particulate fractions using field filtration and parallel total recoverable analysis.
- Soil studies should include particle-size distribution, organic matter, CEC, Fe/Mn oxide content, and sequential extraction to evaluate mobility and bioavailability.
- Source-tracing studies should include conservative reference elements, enrichment factors, geoaccumulation indices, PCA/PMF, isotope ratios, and distance-to-source gradients.
- Future datasets should include Fe, Al, Ti, or another conservative reference element to calculate classical enrichment factors, geoaccumulation indices, and pollution-load indices against regional background values.
- Traffic and industrial emission control should include dust suppression, road-dust management, runoff control, and regular inspection around cement-related activity zones.
- Agricultural/suburban follow-up should document fertilizer, manure, pesticide, irrigation, and amendment histories and analyze these inputs where feasible.

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