

Ecological and geochemical monitoring of the soil–surface water system in the impact zone of the thermoelectric power station

Taras Rychak^{1*}, Liudmyla Arkhypova¹, Natalya Rychak²,
Andriy Kotsyubynsky¹, Iryna Smyk¹, Diana Lynnyk¹

¹ Ivano-Frankivsk National Technical University of Oil and Gas 76019, 15 Karpatska, Ivano-Frankivsk, Ukraine

² V.N. Karazin Kharkiv National University 61022, 4 Svobody sq., Kharkiv, Ukraine

* Corresponding author's e-mail: taras_rychak@ukr.net

ABSTRACT

This study examined spatial patterns of chemical elements in soils within the potential impact zone of the Burshtyn Thermal Power Plant and compared the soil results with non-synchronous published surface-water data. Instrument screening reported 30 components; quantitative interpretation used ten variables with complete results for 19 sites, while Pb was quantified at 12 sites. Soil conditions were assessed using concentration factors, AI-normalized enrichment factors, the geoaccumulation index, Spearman correlations with Benjamini–Hochberg false-discovery-rate control, and exploratory principal component analysis (PCA). All enrichment factors for Cr, Ni, Cu, Zn, and quantified Pb were below 2. Pb values of 43–77 mg/kg were above the cited 32 mg/kg comparison value, although this screening result requires confirmation by complete analytical QA/QC. Mean Zn was 2.44 times the regional background. After FDR adjustment, significant associations remained for Fe–K, Fe–Ni, Fe–Cr, K–Ni, and K–Cr. No significant monotonic relationship was detected between distance and enrichment factors across all sites or within the six downwind sites. The first three PCs explained 80.1% of total variance. A reduced eight-variable PCA retained the Fe–Cr–Ni pattern and the Ca–Si contrast, whereas the Cu–Zn pattern was less stable. The water-to-soil ratio was redefined as an empirical cross-compartment aqueous-representation ratio (Rw/s, kg/L), rather than a leaching coefficient. Ca and Sr had the highest ratios, Mg and Zn intermediate ratios, and Fe, Mn, Cr, and Cu the lowest. Because soil samples were collected in 2026 and water data originated from earlier studies, the comparison does not demonstrate leaching or soil-to-water transfer. The multielement structure is consistent with a substantial lithogenic or mineral-matrix contribution, while localized anthropogenic contributions to Pb and Zn cannot be excluded.

Keywords: Burshtyn Thermal Power Plant, soils, surface waters, chemical elements, enrichment factor, principal component analysis, cross-compartment comparison.

INTRODUCTION

The operation of coal-fired thermal power plants (TPPs) generates gaseous emissions, fine particulate matter, fly ash, slag, and flue-gas cleaning products. During coal combustion, some chemical elements volatilize, condense onto the surfaces of fine particles, or become concentrated in solid residues. Analyses of fly ash and wet flue-gas desulfurization gypsum from coal-fired power plants have revealed the accumulation of

Cr, Pb, Cd, Cu, Zn, and As in combustion and flue-gas cleaning products. Most of the metals examined were retained in the ash, making ash disposal sites secondary sources of long-term environmental pressure (Fu et al., 2019). In Ukraine, the problem is aggravated by the large volumes of accumulated ash and slag waste, low utilization rates, wind erosion of exposed ash-disposal surfaces, and the potential seepage of contaminated water into adjacent soils and aquifers (Popov et al., 2021).

Soils within the zone of influence of a TPP receive atmospheric deposition, coal dust particles, and material transported by wind from ash disposal sites. Element accumulation in topsoil depends on the distance from the source, stack height, prevailing wind directions, topography, land use, and soil properties. In the vicinity of Serbia's largest coal-fired power plant, spatial differences were identified in the concentrations of Cd, Co, Cr, Cu, Fe, Mn, Ni, Pb, and Zn, and their distribution was found to depend on the characteristics of the soil environment (Ćujić et al., 2016). A subsequent soil-profile study conducted in the same area showed that soil texture, organic matter content, acidity, and mineralogical composition substantially influenced the accumulation of potentially toxic elements and the outcomes of source identification (Tanić et al., 2018). The spatial structure of soil contamination should therefore be interpreted in conjunction with the natural characteristics of the study area rather than solely in relation to distance from an industrial facility.

Field investigations around power plants located under different environmental conditions have identified varying groups of priority pollutants. In soils surrounding a large coal-fired TPP in Brazil, Cu, Pb, Zn, Ni, Cr, Fe, Mn, Cd, As, and Se were investigated, and their concentrations exhibited spatial heterogeneity depending on the locations of sampling sites relative to the power plant (da Silva Júnior et al., 2019). In China, the concentrations of As, Cd, Cr, Cu, Hg, Mn, Pb, and Zn varied with prevailing wind direction and distance from the emission source, while Cd, Hg, and As made the greatest contributions to the calculated potential ecological risk (Hu et al., 2021). These findings demonstrate that sampling networks should be designed with consideration of source configuration, meteorological conditions, and potential contaminant transport pathways.

Multivariate statistical methods can identify groups of elements exhibiting similar spatial patterns; however, statistical similarity does not necessarily indicate a common source. In a study of soils surrounding typical coal-fired power plants, positive matrix factorization was used to differentiate the contributions of coal combustion, industrial activities, traffic-related emissions, and natural processes (J. Jia et al., 2023). In a coal and power production area in western China, the assessment of As, Cd, Cr, Hg, Pb, Se, Zn, Co, Cu, Fe, Mn, and Ni combined the geoaccumulation index, the Nemerow pollution index, cluster

analysis, principal component analysis, and spatial interpolation. The authors identified contributions from natural sources, mine water, atmospheric emissions, and dust originating from coal and coal combustion products (Y. Jia et al., 2023). These results support the combined use of geochemical indices, spatial analysis, and multivariate statistical techniques.

The total concentration of an element characterizes its overall accumulation but does not fully reflect its environmental availability or mobility. In chernozem soils surrounding a coal-fired power plant, Cu, Pb, and Zn were distributed among several geochemical fractions that differed in stability and potential mobility (Burachevskaya et al., 2019). Long-term observations of soils near a coal-fired power plant showed that organic matter and the clay fraction contributed to metal sorption and could reduce metal mobility. At the same time, the accumulation of Pb, Zn, Cu, Ni, Mn, Cd, and Cr relative to the regional background indicated the long-term input of pollutants (Kravchenko et al., 2025). In soils surrounding two power plants in northwestern Türkiye, the concentrations of Cd, Cr, Mn, Ni, and Zn exceeded the global mean soil values adopted by the authors, whereas Co, Cu, and Pb remained below these values. Differences among elements and sampling locations demonstrated the limitations of assessments based on a single background benchmark (Parlak et al., 2024).

The transfer of elements from solid waste and soils into the aqueous phase is controlled by their chemical forms, pH, redox conditions, liquid-to-solid ratio, and the properties of the soil sorption complex. Experiments involving fly ash from five coal-fired power plants showed that the leaching of trace elements varied substantially with pH and the extraction procedure applied. Consequently, similar total concentrations of an element in different types of ash do not necessarily result in comparable rates of release into water (Zhao et al., 2020). Long-term column experiments using ash and compacted soil revealed different migration depths for Pb, Ni, Cd, and Cr. Their transport rates depended on soil compaction, water saturation, and the hydraulic conductivity of the material (Xu et al., 2025). These experimental findings demonstrate the need to distinguish indicators of element accumulation from indicators of potential aqueous migration.

Elements can enter surface waters through seepage, surface runoff, erosion-mediated particle transport, and the remobilization of contaminated

bottom sediments. Geochemical and isotopic investigations of water near coal ash disposal facilities in the southeastern United States identified evidence of leakage from ash impoundments into adjacent surface water and groundwater systems (Harkness et al., 2016). A study of Sutton Lake in North Carolina documented the accumulation of As, Se, Cu, Sb, Ni, Cd, V, and Tl in bottom sediments as a result of accidental and previously unaccounted releases of coal ash (Vengosh et al., 2019). Bottom sediments therefore function as long-term archives of contamination capable of preserving evidence of coal ash inputs for decades.

Sediment analyses from five lakes located near coal-fired power plants in North Carolina revealed long-term inputs of coal ash associated with stormwater runoff, flooding, and water discharge from waste storage facilities. The accumulated solid particles posed a chronic risk to aquatic ecosystems even when element concentrations in the water column were not persistently elevated (Wang et al., 2022). Field observations near a large coastal ash disposal facility showed that flooding, erosion, and sediment resuspension following intense rainfall were associated with an almost two-fold increase in the concentrations of major and trace elements in surface water. Their distribution was simultaneously influenced by coprecipitation with Al and Fe, salinity, acidity, and seasonal changes in the hydrological regime (Anderson et al., 2025). Thus, concentrations measured in water primarily reflect the current hydrological conditions, whereas soils and bottom sediments record longer-term accumulation processes.

Distinguishing the anthropogenic contribution of a TPP from natural geochemical heterogeneity remains a major methodological challenge. An elevated element concentration relative to the global average does not demonstrate its industrial origin unless lithology, soil-profile distribution, and local background values are considered. In the Most coal basin of the Czech Republic, the enrichment of Cd, Hg, and Zn in upper soil horizons was associated with coal combustion, whereas the highest concentrations of As and Pb were attributed to geogenic sources. Geological information, soil-profile data, X-ray fluorescence spectrometry, inductively coupled plasma mass spectrometry, and ratios between potentially toxic and lithogenic elements were used to differentiate among sources (Adamec et al., 2024). A subsequent study of the same basin using principal component analysis and regression models

confirmed the dominant role of natural factors in the distribution of As, Pb, Cu, Sb, and Be, while coal combustion had a more pronounced influence on Cd and Zn (Matys Grygar et al., 2025). Geochemical indices, correlation patterns, and principal components should therefore be interpreted as a combined body of analytical evidence rather than as independent proof of element origin.

Previous studies of the Burshtyn TPP have examined individual components of terrestrial and aquatic ecosystems. Cd, Zn, Ni, Cu, Pb, Mn, and Fe were determined in substrates collected from its ash disposal sites, revealing localized contamination with Pb, Cu, and Cd. Analyses of tree leaves also demonstrated interspecific differences in the accumulation of the investigated metals (Semak and Mylenka, 2023). An environmental and toxicological assessment of the Burshtyn cooling reservoir revealed spatial differences in water quality. Some sampling sites met Class I water quality criteria, whereas shoreline areas within the city of Burshtyn and locations exposed to traffic pressure showed evidence of slight contamination and chronic toxicity (Rychak and Arkhypova, 2024).

An analysis of hydrochemical data collected between 1993 and 2024 covered the Burshtyn cooling reservoir, the Hnyla Lypa River, and the Dniester River as an interconnected natural–anthropogenic hydrological system. Long-term changes in Ca and Mg concentrations and statistically significant relationships between parameters measured in different parts of the system were identified, while heavy metals in surface waters were predominantly detected at trace concentrations (Rychak et al., 2025). A study of bottom sediments from seven sampling sites within the Hnyla Lypa River–Burshtyn Reservoir–Dniester River system indicated predominantly low or moderate ecotoxicological risk, a mainly natural biogeochemical cycle of Zn, and a localized anthropogenic contribution to Cd accumulation (Rychak, 2026). Existing studies have characterized ash disposal sites, vegetation, water, and bottom sediments; however, they have not provided a spatially coordinated multi-element analysis of soils within the adjacent catchment or compared their elemental composition with that of surface waters.

The cross-compartment relationship between elemental accumulation in soils and occurrence in surface water within the Burshtyn TPP area remains insufficiently characterized. Soil concentrations reflect accumulation over a longer period,

whereas water concentrations vary with hydrological conditions and sampling time. A comparison of the two matrices can therefore identify differences in relative aqueous representation, but cannot establish leaching, transport pathways, or transfer from a particular soil site without synchronous paired sampling.

The aim of this study was to assess the spatial distribution of chemical elements in soils within the potential impact zone of the Burshtyn TPP and to conduct a screening-level cross-compartment comparison with previously published surface-water data. The objectives were to: (1) characterize major and trace elements detected in the soil survey; (2) evaluate trace-element accumulation using concentration factors, enrichment factors, and the geoaccumulation index; (3) examine inter-element associations using multiplicity-adjusted correlation analysis and exploratory PCA; (4) test whether AI-normalized enrichment varied monotonically with distance or between upwind and downwind sectors; and (5) compare soil and water concentrations using empirical aqueous-representation indices.

MATERIALS AND METHODS

Study area and sampling design

Field investigations were conducted in spring 2026 in the area surrounding the Burshtyn thermal power plant (TPP) and within the catchments of the Burshtyn cooling reservoir, the Hnyla Lypa River, and the Dniester River. The spatial design accounted for the TPP stack height ($H = 250$ m), distance from the source, prevailing air-mass transport directions, topography, surface-runoff pathways, land use, and the presence of local traffic-related sources. Network design and field procedures were aligned with ISO 18400-102:2017, ISO 18400-104:2018, and ISO 18400-203:2018 concerning sampling techniques, spatial sampling strategies, and the investigation of potentially contaminated sites (International Organization for Standardization [ISO], 2017a, 2018a, 2018b).

The network comprised 19 sites assigned to five spatial categories. Six downwind sites were located 0.5–17.5 km from the TPP, six upwind sites 0.5–15.0 km away, two oblique sites

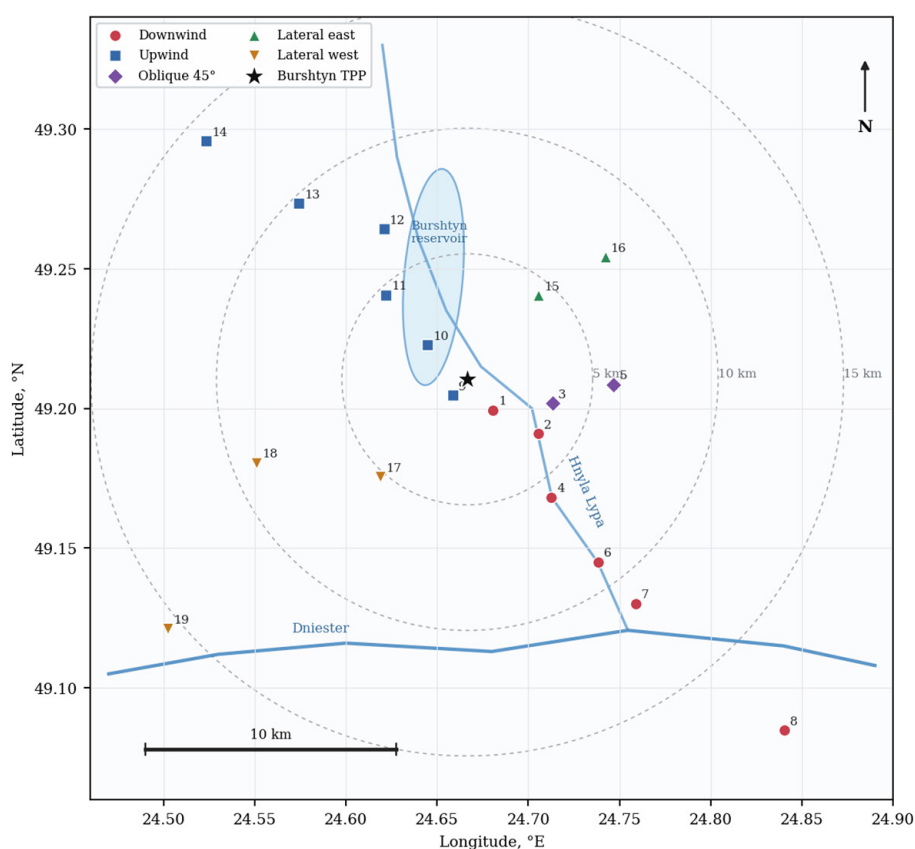


Figure 1. Georeferenced sampling network around the Burshtyn TPP with spatial categories and 5, 10, and 15 km distance rings. Source: sampling coordinates from the field records; TPP marker at 49.21038° N, 24.66654° E. River courses and the reservoir outline are generalized for orientation

positioned approximately 45° from the main transport axis at 2.0 and 5.2 km, two eastern lateral sites at 5.0 and 7.5 km, and three western lateral sites at 5.0, 10.0, and 15.0 km. The categories were used for exploratory comparisons and did not represent measured plume boundaries.

Simulated Meteoblue archive data for 2024–2025 were used only to support field-season stratification of sites (Meteoblue, 2025). Because the sampled 5–20 cm mineral soil represents processes extending beyond this two-year meteorological window, the modeled winds were not treated as a long-term climatology or as evidence of historical deposition pathways. Directional labels are therefore descriptive categories for the sampling design (Figure 1). Composite samples were collected from the 5–20 cm mineral-soil interval using an envelope pattern. This interval was selected

to characterize the mixed mineral horizon and to reduce the influence of transient litter and loose surface particles. Excluding 0–5 cm reduces sensitivity to the most recent atmospheric deposition; consequently, the present design cannot resolve recent surface inputs or vertical enrichment. Geographic coordinates, surface cover, land use, topography, and proximity to roads and railways were recorded. Packaging, transport, and storage followed ISO 18400-105:2017 (ISO, 2017b). Site locations and settings are summarized in Table 1.

Sample preparation and EDXRF analysis

Analytical measurements were performed in 2026 at the Department of Ecology, Ivano-Frankivsk National Technical University of Oil and Gas. The samples were air-dried,

Table 1. Soil sampling sites within the potential impact zone of the Burshtyn TPP

No.	Code	Sector	Distance, km	Coordinates	Sampling-site setting
1	1.1	Downwind	0.5	49.199382, 24.680643	Open unvegetated soil; level surface; Dniester loess terrace.
2	1.2	Downwind	2.5	49.190885, 24.705663	Agricultural land near Zadnistrianske; open level site; Dniester loess terrace.
3	1.3	Oblique (45°)	2.0	49.201878, 24.713216	Agricultural land near Bovshiv; unpaved road; Hnyla Lypa floodplain.
4	1.4	Downwind	5.0	49.168218, 24.712530	Agricultural land; shelterbelt within 50 m; R-03 road about 100 m; railway about 70 m; Hnyla Lypa floodplain.
5	1.5	Oblique (45°)	5.2	49.208607, 24.746862	Agricultural land; unpaved road; level site; right bank of the Narayivka River.
6	1.6	Downwind	7.5	49.145060, 24.738140	Agricultural land; unpaved road; broad left-bank floodplain of the Hnyla Lypa.
7	1.7	Downwind	10.0	49.130235, 24.758828	Agricultural land near Tustan; road about 100 m; railway about 150 m; Dniester floodplain.
8	1.8	Downwind	17.5	49.084891, 24.840281	Agricultural land near Lany; unpaved road; elevation about 260 m.
9	2.9	Upwind	0.5	49.204707, 24.658492	Open terrain; shelterbelt; road more than 100 m; railway about 50 m; Dniester loess terrace.
10	2.10	Upwind	2.5	49.222677, 24.644523	Grass-covered site; shelterbelts; road about 70 m; railway about 100 m; Dniester loess terrace.
11	2.11	Upwind	5.0	49.240584, 24.621842	Rural housing; orchards; road about 70 m; ravine-gully relief; elevation 140–160 m.
12	2.12	Upwind	7.5	49.264226, 24.620780	Gardens and orchards; rural housing; unpaved road; right bank of the Hnyla Lypa; elevation 240–260 m.
13	2.13	Upwind	10.0	49.273355, 24.573991	Agricultural land; unpaved road; forest about 300 m; watershed between the Hnyla Lypa and Okhaba.
14	2.14	Upwind	15.0	49.295675, 24.523361	Agricultural land near Chahriv; unpaved road; rural housing; right bank of the Okhaba.
15	3.15	Lateral, east	5.0	49.240565, 24.705666	Agricultural land near Kuriv; unpaved road; shelterbelt; Hnyla Lypa–Narayivka watershed; elevation about 300 m.
16	3.16	Lateral, east	7.5	49.254262, 24.742077	Agricultural land near Zhalybory; unpaved roads; forest about 300 m; left bank of the Uizdskyi Stream.
17	4.17	Lateral, west	5.0	49.175572, 24.618610	Agricultural land near Nimshyn; unpaved road; shelterbelt; left bank of the Dniester; elevation 220–240 m.
18	4.18	Lateral, west	10.0	49.180566, 24.550761	Agricultural land near Kolodiiv; right-bank floodplain of the Syvka; elevation 220–240 m.
19	4.19	Lateral, west	15.0	49.121251, 24.502459	Agricultural land near Voinyliv; right-bank floodplain of the Syvka; elevation about 240 m.

homogenized, and milled to a fine powder. The general pretreatment sequence was consistent with ISO 11464:2006, which covers drying, crushing, sieving, dividing, and milling of soil samples for physicochemical analysis (ISO, 2006).

Elemental composition and the mass fractions (%) of chemical elements were determined using an EXPERT 4L precision benchtop analyzer (INAM, Kyiv, Ukraine). Measurements were performed by nondestructive energy-dispersive X-ray fluorescence spectrometry (EDXRF). The method is based on the excitation of atoms by primary X-rays and the energy-resolved detection of their characteristic fluorescence radiation. Elemental mass fractions were calculated using the instrument's fundamental-parameters algorithm. The configuration used in this study covered elements from sodium (Na, $Z = 11$) to uranium (U, $Z = 92$). Each sample was analyzed in two consecutive stages: a 15 s survey measurement followed by a 25 s refined measurement.

The analytical workflow and reporting were aligned with ISO 18227:2025 (ISO, 2025). The instrument software initially screened 30 reported components: Si, Al, Fe, Ca, K, Mg, O, Pb, Zn, Cu, Ni, Cr, Mn, As, Mo, V, S, P, Sr, Ba, Rb, Ti, Nb, Ga, Y, Sb, Ag, Te, Th, and Cl. This list describes screening coverage, not 30 quantitatively validated analytes. Oxygen lies outside the direct detection range and was treated as a software estimate. Quantitative statistical interpretation was restricted to directly measured elements meeting the stated detection-frequency and reproducibility criteria.

Major-element concentrations were expressed as dry-mass percentages and trace elements as mg/kg. Each sample was measured in replicate. Complete quantitative series were available for Si, Al, K, Fe, Mg, Ca, Cr, Ni, Cu, and Zn. The software returned quantitative Pb values for 12 samples; the other seven were recorded as missing rather than zero and were excluded from multivariate and spatial tests. Until the laboratory quality-control record is completed with the certified reference material, recoveries, precision, LOD, LOQ, and uncertainty, the Pb comparison is interpreted as a screening result requiring confirmatory analysis.

Geochemical indices

Concentration factors (CF), Al-normalized enrichment factors (EF), and the geoaccumulation index (I_{geo}) were used as element-specific screening metrics for Cr, Ni, Cu, and Zn; Pb-specific values were calculated only for the 12 samples with quantitative Pb results. No Pb values were imputed. PLI and NPI were excluded because a consistent five-element set was unavailable at all 19 sites. The equations retained in the analysis are shown in Table 2 (Jahan et al., 2024; Nadem et al., 2025).

Al was selected as the EF reference element because it had low spatial variability ($CV = 5.9\%$) and represented a major lithogenic matrix component. Regional background concentrations were taken from published datasets for Ukrainian soils and the Hnyla Lypa basin (Fatieiev and Pashchenko, 2003; Prikhodko, 2009). A single regional background was applied only as a screening reference. The sites include different topographic and soil settings, and texture, organic carbon, carbonate content, parent material, and mineralogy were not measured consistently at every site. Al normalization reduces, but does not eliminate, this source of uncertainty. Comparisons with Ukrainian regulatory values (Ministry of Health of Ukraine, 2020; Cabinet of Ministers of Ukraine, 2021) were therefore treated cautiously and were not used alone to assign element origin.

Statistical analysis

Arithmetic mean, median, minimum, maximum, standard deviation, and CV were calculated. Spatial variability was classified as very low at $CV < 10\%$, low at 10–15%, moderate at 15–25%, high at 25–50%, and very high at $CV > 50\%$. Inter-element relationships were evaluated using two-sided Spearman rank correlations. To control multiplicity across the 55 pairwise tests, p values were adjusted with the Benjamini–Hochberg false-discovery-rate procedure; $q < 0.05$ was used for inferential interpretation (Benjamini and Hochberg, 1995). Unadjusted associations were described only as exploratory patterns.

Table 2. Geochemical indices used for soil assessment

Index	Equation	Interpretation
CF	$CF_i = C_i / B_i$	Ratio of the measured concentration to the background concentration.
EF	$EF_i = (C_i/CAI)_{\text{sample}} / (B_i/BAI)_{\text{background}}$	Normalized enrichment; Al was used as the reference element.
I_{geo}	$I_{geo_i} = \log_2[C_i / (1.5 \times B_i)]$	Background-normalized index with a factor of 1.5 for natural variability.

PCA of z-standardized data was used as an exploratory visualization, not as a confirmatory source-apportionment model. The primary matrix comprised Al, Si, Fe, Ca, K, Ti, Cr, Ni, Cu, and Zn (19 observations and 10 variables). Components with eigenvalues greater than 1 were retained and interpreted together with explained variance. Sensitivity to variable selection was assessed with a reduced eight-variable PCA comprising Al, Si, Fe, Ca, Cr, Ni, Cu, and Zn; K was omitted because of its near-collinearity with Fe, and Ti was omitted to reduce dimensionality. Only patterns retained in the reduced model were considered comparatively stable.

Cross-compartment comparison with published surface-water data

Soil samples were collected in spring 2026, whereas the surface-water values were obtained from previously published hydrochemical datasets for the Burshtyn cooling reservoir, the Hnyla Lypa River, and the Dniester River (Rychak and Arkhypova, 2024; Rychak et al., 2025). The datasets were neither synchronous nor paired by sampling site. Their comparison was therefore restricted to screening-level differences in cross-compartment representation and was not used to infer leaching, transport, or soil-to-water transfer. An empirical cross-compartment aqueous-representation ratio, denoted Rw/s , was calculated as follows:

$$Rw/s = \frac{C_{wi}}{C_{si}} \quad (1)$$

where: C_{wi} is the concentration of element i in surface water (mg/L), and C_{si} is its concentration in catchment soil (mg/kg). Soil percentages were converted to mg/kg before calculation. The dimensional unit of C_{wi}/C_{si} is kg/L.

Rw/s was used only to rank the concentration of each element in water relative to its concentration in the compared soil dataset. A second literature-derived index, denoted A_i , was retained to show how normalization by total mineralization changes the cross-compartment ranking (Sherstyuk, 2015; Dolin et al., 2018):

$$A_i = \frac{m_i \times 100}{a \times n_i} \quad (2)$$

where: m_i is the concentration of element i in surface water; a is total water mineralization;

and n_i is the mean content of the corresponding element in soils of the study area, expressed in compatible mass units.

The source methodology classifies $A_i > 10^5$ as class I, $10^3 < A_i < 10^5$ as class II, and $A_i < 10^3$ as class III. In the present study these classes describe values of the empirical index only; they are not evidence of actual migration intensity. Soil and water pH, redox conditions, and soil properties were not direct variables in either equation.

Both indices compare concentrations measured in different matrices and at different times. They do not represent laboratory leaching tests, desorption kinetics, transport rates, mass flux, or transfer from an individual soil site to a receiving water body.

RESULTS

The survey characterized soil elemental heterogeneity at 19 sites assigned to five spatial categories: six downwind, six upwind, two oblique, two eastern lateral, and three western lateral sites. Distances ranged from 0.5 to 17.5 km. Because the 0–5 cm layer was not sampled, the results represent the 5–20 cm mineral horizon and should not be interpreted as a direct measure of recent atmospheric deposition.

The instrument software screened 30 reported components. Quantitative interpretation used complete 19-site series for Si, Al, K, Fe, Mg, Ca, Cr, Ni, Cu, and Zn; Pb had 12 quantitative values and was excluded from PCA and spatial tests. Oxygen was a software estimate and was not analyzed statistically. The Results therefore distinguish initial screening coverage from the smaller quantitatively interpreted dataset.

Trace-element maxima occurred at different locations. Cr reached 159 mg/kg at downwind site 8, 17.5 km from the plant, and the highest quantified Pb value, 77 mg/kg, occurred at downwind site 1, 0.5 km away. Ni and Cu maxima were recorded at the two oblique sites, whereas Zn reached 206 mg/kg at upwind site 9. Maxima alone do not constitute a formal gradient test; they indicate only that no common visual ordering was evident across the measured elements.

Maximum concentrations exceeded the selected regional background by factors of 2.12 for Cr, 1.38 for Ni, 2.05 for Cu, 4.12 for Zn, and 7.70 for Pb. Direct numerical comparison with the cited

32 mg/kg total-content value was made for Pb because its analytical basis corresponded to total EDXRF content. In the absence of complete method-specific QA/QC, this comparison is screening-level and does not by itself establish regulatory non-compliance. Cr, Ni, Cu, and Zn values were treated as background or comparison benchmarks rather than regulatory exceedances (Table 3).

Among the 12 samples with quantitative Pb results, concentrations ranged from 43 to 77 mg/kg and were numerically above the cited total-content value of 32 mg/kg by factors of 1.34–2.41. The mean was 59.5 mg/kg, or 5.95 times the regional background. These findings apply only to the quantified subset and require confirmation after the Pb LOD, LOQ, recovery, precision, and uncertainty are documented. No inference was made for the seven samples without a reportable value. Mean Cr, Cu, and Zn concentrations were 1.44, 1.49, and 2.44 times background, whereas mean Ni was 0.92 of background. At the same time, comparison of absolute background ratios with normalized EF values revealed different evaluative emphases. Mean Zn was 2.44 times the regional background and mean Pb was 5.95 times the background among quantified samples, whereas their mean EF values were 0.986 and 1.022, and every site-specific EF remained below 2. At first sight, these results could appear inconsistent. However, the background ratio represented absolute concentration, while EF incorporated the concurrent variation of the trace element and Al. A high absolute background ratio was therefore not treated as equivalent to a high normalized-enrichment class; both measures were retained as complementary descriptors.

Table 3. Regional background and comparison values of element concentrations in soil, mg/kg

Element	Regional background concentration	Comparison value
Mn	850	1500
Cr	75	100
Co	8	50
Ni	40	85
Cu	20	55
Zn	50	100
Cd	0.5	3
Pb	10	32
Sr	300	1000

Source: compiled by the authors from Fatieiev and Pashchenko (2003), Prihodko (2009), Ministry of Health of Ukraine (2020), and the study results.

In accordance with the preceding comparison, normalization to Al was used to evaluate trace-element concentrations relative to a stable component of the soil matrix. Al showed low spatial variability ($CV = 5.9\%$) and served as the reference element in the EF calculations. All site-specific EF values for Cr, Ni, Cu, Zn, and Pb remained below 2. Mean EF values were 0.842 for Cr, 0.970 for Ni, 0.940 for Cu, 0.986 for Zn, and 1.022 for Pb. The ranges were 0.438–1.237 for Cr, 0.747–1.121 for Ni, 0.628–1.237 for Cu, 0.692–1.938 for Zn, and 0.757–1.477 for Pb. Notably, Zn had the widest EF interval, consistent with the pronounced spatial variability of its absolute concentrations.

As can be seen from Table 4, site-specific EF values clarified the local accumulation structure. Site 8 simultaneously had the highest EF values for Cr (1.237) and Ni (1.121) but the lowest EF for Cu (0.628). The maximum Cu EF occurred at eastern lateral site 15 (1.237), the maximum Zn EF at upwind site 9 (1.938), and the maximum Pb EF at downwind site 1 (1.477). Minimum values for Cr and Zn were recorded at downwind site 6, for Ni at site 3, and for Pb at upwind site 11. On the one hand, individual sites were characterized by maxima of particular elements; on the other hand, the same sites could contain minima of other components. Taken together, these observations indicated different spatial configurations of the concentration fields.

Sector means did not identify one spatial category with the highest EF for every trace element. The eastern lateral sector had the highest mean EF for Zn (1.165), followed by upwind (1.131), western lateral (1.028), downwind (0.827), and oblique sites (0.784). Quantified Pb had the highest mean EF in the downwind category (1.184). Eastern lateral sites also had the highest mean EF for Cr (1.003) and Cu (1.064), whereas Ni was highest upwind (1.022). These descriptive means were based on small and unequal groups and were not interpreted as evidence of a directional source.

Distance–EF correlations were non-significant for every element across the full network and within the downwind subset (Table 5). The strongest downwind coefficient was observed for Cu ($\rho = -0.600$), but the adjusted value was $q = 0.957$ with only six observations. Upwind–downwind comparisons were likewise non-significant. The available evidence therefore supports only the statement that no clear monotonic enrichment gradient was apparent within the present

Table 4. Enrichment factors (EF) at the sampling sites

Site	Sector	EF Cr	EF Ni	EF Cu	EF Zn	EF Pb
1	Downwind	0.965	1.016	1.075	0.770	1.477
2	Downwind	0.692	1.103	0.817	0.732	1.197
3	Oblique (45°)	0.766	0.747	0.945	0.817	-
4	Downwind	0.602	0.912	0.924	0.753	-
5	Oblique (45°)	0.752	1.020	0.736	0.752	-
6	Downwind	0.438	1.025	0.834	0.692	0.811
7	Downwind	0.675	0.810	0.840	1.226	-
8	Downwind	1.237	1.121	0.628	0.787	1.251
9	Upwind	0.746	1.019	0.929	1.938	1.096
10	Upwind	0.928	0.934	1.124	1.612	-
11	Upwind	1.067	0.940	0.966	0.838	0.757
12	Upwind	0.492	1.088	1.160	0.843	-
13	Upwind	0.915	1.110	0.949	0.783	0.782
14	Upwind	0.978	1.043	0.837	0.772	0.820
15	Lateral, east	1.005	0.999	1.237	1.522	1.013
16	Lateral, east	1.002	0.956	0.890	0.808	1.017
17	Lateral, west	0.851	0.865	0.929	0.805	0.976
18	Lateral, west	1.037	0.945	1.104	1.110	1.073
19	Lateral, west	0.859	0.778	0.936	1.168	
Mean	-	0.842	0.970	0.940	0.986	1.022

Note: calculated by the authors from the EDXRF measurements of soil elemental composition.

Table 5. Exploratory spatial tests based on enrichment factors

Element	All sites, n	Distance–EF rho (q)	Downwind, n	Distance–EF rho (q)	Upwind vs downwind p
Cr	19	0.214 (0.473)	6	0.029 (0.957)	0.485
Ni	19	0.226 (0.473)	6	0.143 (0.957)	0.818
Cu	19	–0.295 (0.473)	6	–0.600 (0.957)	0.065
Zn	19	–0.059 (0.809)	6	0.429 (0.957)	0.065
Pb	12	–0.280 (0.473)	4	–0.400 (0.957)	0.114

Note: calculated from the site-specific EF values in Table 4. q values are Benjamini–Hochberg adjusted within each five-element distance-test family; upwind–downwind p values are two-sided Mann–Whitney tests.

sampling network; statistical power was limited by the small sector sizes.

For Cr, Ni, Cu, and Zn, mean Igeo values were negative, ranging from –0.78 for Cr to –0.54 for Ni. The highest individual Igeo value was 0.24 for Zn. Most values therefore fell in the uncontaminated class, with isolated results in the transition from uncontaminated to moderately contaminated. Pb was not included in this complete-site summary because seven values were missing.

To deepen the spatial assessment, the descriptive statistics in Table 6 were examined. The results distinguished stable soil-matrix components from elements with pronounced local variability.

Si had a mean of 34.26%, a median of 34.19%, and a range of 32.39–35.97%; the corresponding Al values were 4.25%, 4.25%, and 3.81–4.68%. The near identity of the means and medians, together with CV values of 3.0% for Si and 5.9% for Al, reflected their uniform distribution within the dataset. In comparison, K varied from 4.46 to 6.33% with a CV of 10.4%, whereas Fe varied from 2.45 to 4.11% with a CV of 16.4%. The gradual increase in variability from Si and Al to K and Fe therefore defined the structure of the major matrix components.

Particular attention should be given to Ca, which showed the strongest contrast among the

major elements. Its concentration ranged from 0.78 to 5.96%, and the maximum exceeded the minimum by a factor of 7.64. The mean of 2.13% was substantially higher than the median of 1.46%, showing the influence of sites with elevated Ca concentrations on the overall mean. The Ca coefficient of variation reached 79.2%, the highest value in the entire dataset. At the same time, Mg displayed lower but still pronounced heterogeneity, with a mean of 2.03%, a median of 1.97%, a range of 1.37–3.40%, and a CV of 23.6%. In contrast to the stable distributions of Si and Al, Ca and Mg were distributed much less uniformly across the study area.

A corresponding examination of trace elements showed that Zn had the widest absolute range, 82–206 mg/kg. Its mean was 121.8 mg/kg, the median was 113 mg/kg, and the CV was 31.9%. Cr ranged from 53 to 159 mg/kg, with a mean of 108.2 mg/kg and a CV of 28.0%. In contrast, Ni and Cu showed more compact distributions: 27–55 mg/kg for Ni, with a mean of 36.9 mg/kg and a CV of 16.4%; and 20–41 mg/kg for Cu, with a mean of 29.7 mg/kg and a CV of 18.1%. The 12 quantitative Pb determinations produced a range of 43–77 mg/kg, a mean of 59.5 mg/kg, and a CV of 18.4%. Overall, Zn and Cr formed the most spatially contrasting trace-element fields, whereas Ni, Cu, and Pb showed moderate coefficients of variation.

As indicated by the summary statistics, the ascending CV sequence was Si (3.0%) < Al (5.9%) < K (10.4%) < Fe = Ni (16.4%) < Cu (18.1%) < Pb (18.4%) < Mg (23.6%) < Cr (28.0%) < Zn (31.9%) < Ca (79.2%). This order confirms that classification as a major or trace element did not

itself determine the degree of spatial heterogeneity. For example, Ca exceeded every trace element in relative variability, whereas Ni varied to the same extent as Fe. Furthermore, the close agreement between the mean and median for Si, Al, Cr, Ni, and Cu indicated little distributional asymmetry; the largest mean–median difference occurred for Ca.

Pb statistics were based on 12 quantitative determinations, compared with 19 observations for the other elements. The Pb mean and CV therefore describe only the quantified subset. They cannot be used to infer spatial uniformity or to characterize the seven sites without reportable Pb values.

Spearman coefficients were re-evaluated with Benjamini–Hochberg adjustment across 55 pairwise tests. Five positive associations remained significant at $q < 0.05$: Fe–K ($\rho = 0.93$, $q < 0.001$), Fe–Ni ($\rho = 0.69$, $q = 0.030$), Fe–Cr ($\rho = 0.65$, $q = 0.035$), K–Ni ($\rho = 0.65$, $q = 0.035$), and K–Cr ($\rho = 0.64$, $q = 0.035$). These results support a compact Fe–K–Cr–Ni correlation core within the sampled dataset.

Associations that had met the unadjusted $p < 0.05$ threshold did not remain significant after FDR control. These included Si–Ca ($\rho = -0.57$, $q = 0.099$), Si–Cu ($\rho = -0.51$, $q = 0.179$), Fe–Cu and K–Cu ($\rho = 0.50$, $q = 0.179$), Cu–Zn ($\rho = 0.49$, $q = 0.183$), and Cr–Ni ($\rho = 0.47$, $q = 0.212$). They are retained in Table 7 as descriptive effect sizes but are not treated as statistically confirmed relationships.

Pb and Mg did not participate in any FDR-significant pair. Pb coefficients ranged from -0.21 with Ca to 0.21 with Cr and were based on 12 observations. Mg coefficients ranged from

Table 6. Descriptive statistics of major- and trace-element concentrations in soil

Element	Unit	N	Mean	Median	Min	Max	CV, %	Variability
Si	%	19	34.26	34.19	32.39	35.97	3.0	Very low
Al	%	19	4.25	4.25	3.81	4.68	5.9	Very low
K	%	19	5.24	5.11	4.46	6.33	10.4	Low
Fe	%	19	2.92	2.74	2.45	4.11	16.4	Moderate
Mg	%	19	2.03	1.97	1.37	3.4	23.6	Moderate
Ca	%	19	2.13	1.46	0.78	5.96	79.2	Very high
Cr	mg/kg	19	108.2	109	53	159	28.0	High
Ni	mg/kg	19	36.9	37	27	55	16.4	Moderate
Cu	mg/kg	19	29.7	29	20	41	18.1	Moderate
Zn	mg/kg	19	121.8	113	82	206	31.9	High
Pb	mg/kg	12	59.5	57	43	77	18.4	Moderate

Note: calculated by the authors from the laboratory analysis of soil samples.

Table 7. Spearman rank-correlation matrix

Element	Al	Si	Fe	Ca	K	Mg	Cr	Ni	Cu	Zn	Pb
Al	1.00										
Si	0.34	1.00									
Fe	0.10	-0.14	1.00								
Ca	-0.22	-0.57	-0.43	1.00							
K	0.14	-0.01	0.93*	0.12	1.00						
Mg	-0.07	-0.23	0.20	-0.18	0.19	1.00					
Cr	-0.13	-0.02	0.65*	-0.39	0.64*	-0.16	1.00				
Ni	-0.26	-0.07	0.69*	-0.17	0.65*	0.00	0.47	1.00			
Cu	0.00	-0.51	0.50	0.22	0.50	-0.08	0.41	0.17	1.00		
Zn	0.40	-0.10	0.23	0.20	0.39	-0.08	0.20	-0.16	0.49	1.00	
Pb	0.06	-0.13	0.01	-0.21	0.17	0.02	0.21	-0.01	-0.09	-0.01	1.00

Note: calculated by the authors; * $q < 0.05$ after Benjamini–Hochberg adjustment across 55 pairwise tests.

–0.23 with Si to 0.20 with Fe. The absence of a significant association should be interpreted cautiously because the sample size, particularly for Pb, limits power.

Exploratory PCA of ten variables produced three components with eigenvalues greater than 1. PC1 had an eigenvalue of 4.14 and explained 41.4% of variance, PC2 had an eigenvalue of 2.31 and explained 23.1%, and PC3 had an eigenvalue of 1.56 and explained 15.6%. Cumulative variance was 80.1%. With only 19 observations for ten variables, these components are used to visualize covariance structure and not to establish sources (Table 8).

The primary loading matrix showed Fe and K (0.47 each), Ti (0.39), Ni (0.38), and Cr (0.33) on PC1. PC2 contrasted Ca (0.53) with Si (–0.53) and Al (–0.51). PC3 had positive loadings for Zn (0.62) and Cu (0.54). Given the small sample-to-variable ratio, these patterns describe the present

dataset and do not identify lithogenic or anthropogenic sources independently.

The PC1–PC2 biplot (Figure 2) visualizes the first two components, which together explain 64.5%

Table 9. Loadings of the first three principal components

Element	PC1	PC2	PC3
Al	0.03	-0.51	0.18
Si	-0.12	-0.53	-0.34
Fe	0.47	0.04	0.03
Ca	-0.21	0.53	0.19
K	0.47	-0.12	0.01
Ti	0.39	0.18	-0.25
Cr	0.33	-0.10	-0.02
Ni	0.38	0.11	-0.29
Cu	0.29	0.13	0.54
Zn	0.06	-0.30	0.62

Note: calculated by the authors from the PCA results.

Table 8. Eigenvalues and explained variance of the principal components

Component	Eigenvalue	Explained variance, %	Cumulative variance, %
PC1	4.14	41.4	41.4
PC2	2.31	23.1	64.5
PC3	1.56	15.6	80.1
PC4	0.74	7.4	87.5
PC5	0.44	4.4	91.9
PC6	0.29	2.9	94.8
PC7	0.22	2.2	97.0
PC8	0.17	1.7	98.7
PC9	0.1	1.0	99.7
PC10	0.03	0.3	100.0

Note: calculated by the authors from standardized soil-element data.

of variance. Fe, K, Ti, Cr, and Ni point toward positive PC1, whereas Ca is opposed to Si and Al on PC2. Cu and Zn cannot be fully evaluated in this plane because their largest loadings occur on PC3.

The additional PC1–PC3 loading plot (Figure 3) displays the Cu and Zn loadings that are obscured in the PC1–PC2 plane. A reduced PCA using Al, Si, Fe, Ca, Cr, Ni, Cu, and Zn explained 79.7% of variance in its first three components (33.7%, 25.6%, and 20.5%). The Fe–Cr–Ni direction and the Ca–Si contrast persisted. In the reduced third component, Zn remained prominent, but Cu no longer had a

comparably large loading. The Cu–Zn interpretation is therefore less stable than the Fe–Cr–Ni and Ca–Si patterns and is treated as tentative.

The cross-compartment comparison used published water data and the 2026 soil survey. The largest Rw/s ranges were reported for Ca (0.25–0.77 kg/L) and Sr (0.42–0.76 kg/L). These values indicate high water concentration relative to the compared soil concentration under the empirical ratio; they do not measure leaching.

Rw/s ranged from 0.006 to 0.008 kg/L for Mg and from 0.001 to 0.003 kg/L for Zn, while Fe,

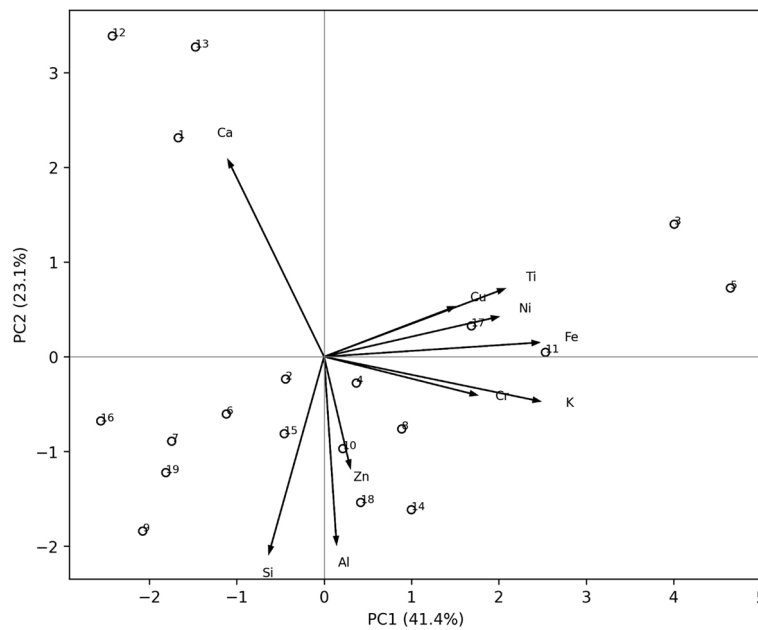


Figure 2. Exploratory PCA biplot of soil elemental composition on PC1 and PC2; points 1–19 are sampling sites and arrows are variable loadings. Source: constructed by the authors from the PCA results

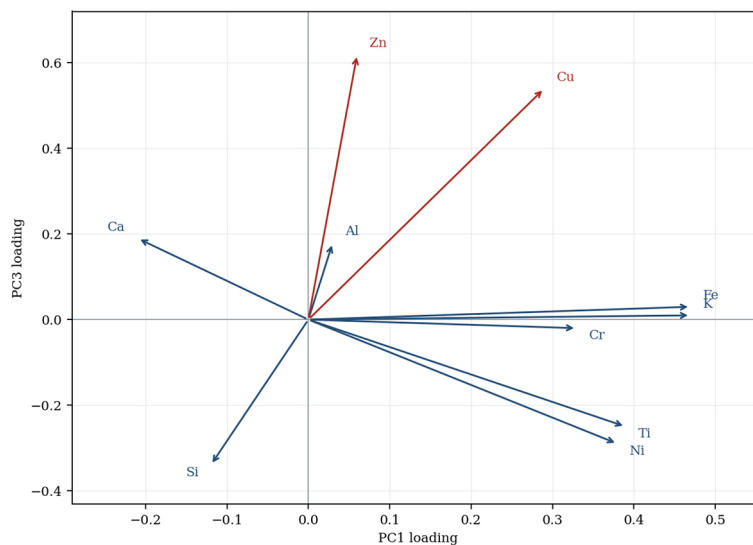


Figure 3. Exploratory PCA loading plot for PC1 and PC3; Cu and Zn are highlighted because their largest loadings occur on PC3. Source: constructed from the loading matrix in Table 9

Mn, Cr, and Cu were below 0.0005 kg/L. The ratio therefore separated Ca and Sr, Mg and Zn, and the low-ratio group of Fe, Mn, Cr, and Cu. Because the underlying datasets were not synchronous, only the ordering of cross-compartment representation was interpreted. A_i produced a broadly similar ranking but differed for Cu. Ca and Sr fell in class I; Mg, Zn, and Cu in class II; and Fe, Mn, and Cr in class III. Cu therefore had a low Rw/s but a class II A_i value. The discrepancy reflects the different normalization used by the two indices and cannot be interpreted as evidence of actual Cu transport (Table 10).

Soil accumulation and cross-compartment aqueous representation did not follow the same order. Zn had a mean above regional background and the widest trace-element EF range, but an intermediate Rw/s . Cr had high spatial variability and a low Rw/s . Ca combined the highest soil CV with the highest ratio, whereas Fe belonged to the lowest-ratio group. These results are a comparative screen across matrices rather than evidence of transfer pathways. Overall, the measured soils were spatially heterogeneous, and no clear monotonic EF–distance gradient was apparent within the present network. All site-specific EF values were below 2. The exploratory PCA indicated an Fe–K–Cr–Ni core and a Ca contrast with Si and Al; the Cu–Zn pattern was less stable in sensitivity analysis. Ca and Sr had the highest empirical aqueous-representation ratios, while Fe, Mn, and Cr had the lowest.

DISCUSSION

Element-specific concentrations, EF, I_{geo} , multiplicity-adjusted correlations, and exploratory PCA distinguish low Al-normalized enrichment from localized concentration deviations. All

EF values were below 2, while the 12 quantified Pb results were above the cited comparison value and Zn showed the greatest trace-element variability. The Pb observation remains screening-level until complete QA/QC is documented.

Local maxima at different distances and directions resemble the spatial heterogeneity reported around other coal-fired power plants (Čujić et al., 2016). Formal tests of EF against distance were not significant in either the full network or the six downwind sites, and upwind–downwind comparisons were also non-significant. With such small directional groups, the defensible statement is that no clear monotonic gradient was apparent in this sampling network. Resuspension, relief, land use, and mineral-matrix heterogeneity remain hypotheses requiring direct measurement.

The contrast with more heavily contaminated settings is also informative. Around a coal-fired power plant in China, Hu et al. (2021) reported moderate to heavy soil pollution and a comprehensive potential ecological risk index of 155.09–293.04, dominated by Cd, Hg, and As. The low mean indices obtained for the Burshtyn area cannot be directly contrasted with those values as evidence of lower risk from the complete contaminant mixture because Cd, Hg, and As were not included in the complete analytical dataset of the present study. This limitation is material for coal-energy sites: Fu et al. (2019) identified Cr, Pb, Cd, Cu, Zn, and As in fly ash and flue-gas-desulfurization gypsum, with 88.9–98.7% of the investigated metals retained in ash. Accordingly, the finding of low multi-element enrichment applies to the measured Cr, Ni, Cu, Zn, and Pb and cannot be extended to elements that were not analyzed.

A relevant local context is provided by research conducted directly on the ash and slag dumps of

Table 10. Empirical indices of cross-compartment aqueous representation

Element	Rw/s , kg/L	Rw/s group	A_i	A_i class	Cross-compartment interpretation
Ca	0.25–0.77	High	7.18×10^5 – $2.21 \times 10^{6*}$	I	Highest relative aqueous representation
Sr	0.42–0.76	High	7.18×10^5 – $2.21 \times 10^{6*}$	I	Highest relative aqueous representation
Mg	0.006–0.008	Moderate	1.9×10^4	II	Intermediate relative representation
Zn	0.001–0.003	Moderate	4.7×10^3 – 8.5×10^3	II	Intermediate relative representation
Cu	<0.0005	Low	1.4×10^3	II	Low Rw/s ; class II A_i
Fe	<0.0005	Low	46.5–657.1	III	Low relative representation
Mn	<0.0005	Low	142.9–285.7	III	Low relative representation
Cr	<0.0005	Low	206.2–259.7	III	Low relative representation

Note: calculated by the authors from the soil survey and previously published, non-synchronous surface-water datasets; a common A_i range is reported for Ca and Sr.

the Burshtyn TPP. Semak and Mylenka (2023) identified contamination of technogenic substrates mainly by Pb, Cu, and Cd and reported element-specific accumulation by the leaves of woody species. The difference between their findings and the low mean EF values across the 30-km zone is not contradictory because the studies address different spatial scales and environmental matrices. Ash-dump material represents the primary waste-accumulation zone, whereas natural catchment soils reflect subsequent dispersion, mixing with lithogenic material, and retention by the soil matrix. Taken together, the two studies demonstrate the need to distinguish the condition of the ash-disposal site from that of the wider surrounding landscape. At the same time, the absence of Cd from the current soil dataset prevents evaluation of whether the cadmium component identified at the ash dump extends beyond its boundaries.

Pb requires cautious interpretation. The 12 quantified values were numerically above the selected comparison value, but concentration alone cannot identify their source, and the missing QA/QC parameters preclude a definitive regulatory assessment. Studies in the Most Basin show that Pb anomalies may be geogenic while other elements better trace coal combustion (Adamec et al., 2024; Matys Grygar et al., 2025). In the Burshtyn dataset, low Al-normalized EF and the Fe–K–Cr–Ni pattern are consistent with a substantial mineral-matrix contribution. They do not demonstrate that natural control predominated or exclude localized anthropogenic Pb input.

The Zn distribution further illustrates the limits of unequivocal attribution. Its mean concentration was 2.44 times the regional background, its coefficient of variation was 31.9%, and its maximum EF approached the upper boundary of minimal enrichment; nevertheless, the concentration maximum occurred in the upwind sector. This configuration weakens an explanation based solely on direct deposition from the stack along the prevailing wind direction, although it does not exclude secondary dust transport or contributions from other local sources. Tanić et al. (2018) likewise showed that element-specific maxima around a power plant occurred at different distances and were related to soil properties. In soils near two coal-fired power plants in northwestern Turkey, Cd, Cr, Mn, Ni, and Zn exceeded the global soil averages adopted by the authors, whereas Co, Cu, and Pb were lower, and the inter-element correlations also varied in composition (Parlak et al.,

2024). This comparison confirms that the priority-metal profile is site-specific and cannot be transferred mechanically between power plants.

After FDR correction, the Fe–K, Fe–Ni, Fe–Cr, K–Ni, and K–Cr associations remained significant. This pattern and their PC1 loadings are consistent with shared mineral-matrix control, but cannot determine source. The Si–Ca and Cu–Zn correlations did not remain significant after correction, and the Cu–Zn PCA pattern weakened in the reduced analysis. Source attribution therefore requires parent-material, grain-size, mineralogical, ash, and coal data. PCA stability is limited by 19 observations for ten variables. Although the first three components explained 80.1% of variance, variable reduction altered the third-component interpretation. The stable features were the Fe–Cr–Ni direction and Ca–Si contrast; the Cu–Zn association was not stable enough for source inference. Larger sampling campaigns and resampling-based loading intervals are needed before the component structure can be generalized.

Rw/s and Ai demonstrate that ranking depends on the chosen cross-compartment normalization. Ca and Sr had the highest relative aqueous representation, Mg and Zn were intermediate, and Fe, Mn, Cr, and Cu had the lowest Rw/s values. Experimental studies show that actual leaching depends on pH, liquid-to-solid ratio, and material composition (Jankowski et al., 2006; Zhao et al., 2020). The present ratios do not test those mechanisms.

Cu was class II under Ai but below 0.0005 kg/L under Rw/s. The difference is methodological and should not be averaged into a single mobility class. Synchronous sampling, dissolved and particulate water fractions, sequential extraction, and controlled leaching tests are required to determine Cu release and transport.

High Ca and Sr ratios are not evidence of toxic contamination. Published water records and the 2026 soil survey have different temporal resolutions. Their comparison identifies relative cross-compartment representation only; causal soil-to-water linkage can be tested only through synchronized sampling in the same microcatchments.

A practical direction for strengthening the monitoring design is also supported by Smyk and Arkhypova (2025). In the upper Prut basin, the authors integrated mobile hydrochemical diagnostics with systematic state-monitoring data and mathematical modelling. Their concentration results are not used here for direct comparison because the natural setting and type of

anthropogenic pressure differ. The methodologically relevant point is the coordination of rapid measurements with long-term records. Applied to the Burshtyn TPP system, this principle would require seasonally synchronized monitoring of soils, surface waters, and suspended matter, together with documentation of hydrological conditions at each sampling event. Such a design would distinguish stable geochemical structure from short-term runoff- or flood-related variation.

The need for independent transport tracers is supported by studies of coal-ash disposal sites and aquatic ecosystems. Harkness et al. (2016) identified leakage from ash ponds using a combination of geochemical and isotopic evidence, while Wang et al. (2022) applied morphological, magnetic, elemental, and strontium-isotope indicators to reconstruct multi-decadal inputs of coal ash to lake sediments. A comparable multiproxy approach in the Burshtyn system could test whether local Pb and Zn maxima are related to ash particles, other anthropogenic sources, or natural mineral heterogeneity. Comparison of soils with bottom sediments would be particularly informative because a single water concentration reflects current hydrological conditions, whereas sediments integrate a longer record of material transport.

The study is limited by 19 samples from one field campaign, seven non-quantitative Pb results, and incomplete reporting of CRM performance, LOD, LOQ, precision, and uncertainty in the available analytical record. EDXRF described total elemental composition but not geochemical fractions. Cd, Hg, As, and Se were absent from the complete dataset. Texture, organic carbon, carbonates, mineralogy, and parent material were not measured consistently, and coal, fly ash, and ash-disposal material were not analyzed as end-members. Surface-water data were non-synchronous. These constraints limit regulatory interpretation, source attribution, and any inference of soil-to-water transfer.

The most defensible interpretation is low Al-normalized enrichment for the measured elements, with localized Pb and Zn deviations. The multielement structure is consistent with a substantial lithogenic or mineral-matrix contribution, while localized anthropogenic contributions cannot be excluded. Further work should document complete QA/QC, re-analyze Pb, add Cd, Hg, As, and Se, sample 0–5 cm and deeper profile

intervals, characterize parent material and ash, and synchronize soil, water, suspended-matter, and sediment sampling.

CONCLUSIONS

The study supports the following conclusions. The 2026 soil survey identified spatial heterogeneity within the potential impact zone of the Burshtyn TPP. Comparison with earlier surface-water datasets provided a screening-level view of cross-compartment representation, but did not measure leaching or transfer.

Al-normalized enrichment was low: all EF values for Cr, Ni, Cu, Zn, and quantified Pb were below 2. The 12 quantified Pb values ranged from 43 to 77 mg/kg and were above the cited 32 mg/kg comparison value; this finding requires confirmation with complete QA/QC and cannot be extended to seven missing results. Mean Zn was 2.44 times regional background and had a CV of 31.9%, supporting targeted follow-up of Pb and Zn. No clear monotonic spatial gradient was apparent. EF–distance correlations were non-significant across all sites and within the six downwind sites, and upwind–downwind contrasts were also non-significant. After FDR correction, only the Fe–K, Fe–Ni, Fe–Cr, K–Ni, and K–Cr correlations remained significant. Exploratory PCA and reduced-model sensitivity were consistent with a substantial mineral-matrix contribution, but did not demonstrate dominant natural control or exclude localized anthropogenic input. The corrected Rw/s unit is kg/L. Reported ratios were highest for Ca (0.25–0.77 kg/L) and Sr (0.42–0.76 kg/L), intermediate for Mg and Zn, and below 0.0005 kg/L for Fe, Mn, Cr, and Cu.

Because the soil and water data were not collected synchronously, these values describe empirical aqueous representation only. They are not leaching coefficients, transport rates, or mass fluxes. The conclusions are constrained by the small single-season soil dataset, incomplete Pb quantification and QA/QC reporting, a common regional background across heterogeneous soils, the absence of 0–5 cm samples and source end-members, the small PCA sample-to-variable ratio, and non-synchronous water data. Further monitoring should combine complete analytical QA/QC with profile-based soil sampling and synchronized analysis of surface water, suspended matter, and bottom sediments.

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