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Distributions and risk assessment of heavy metals in solid waste in lead-zinc mining areas and across the soil, water body, sediment and agricultural product ecosystem in their surrounding areas

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ABSTRACT

To identify the root causes of heavy metal contamination in soils as well as prevent and control such contamination from its sources, this study explored the accumulation patterns and ecological risks of heavy metals like Cd and Pb in solid waste in mining areas and across the water body, sediment, soil and agricultural product ecosystem surrounding the mining areas. Focusing on the residual solid waste samples in lead-zinc deposits in a certain area of Guizhou Province, along with samples of topsoils, irrigation water, river sediments, and crops from surrounding areas. This study analyzed the distributions of eight heavy metals, i.e., Cd, As, Cr, Hg, Pb, Zn, Cu, and Ni, in the samples through field surveys and sample tests. Furthermore, this study assessed the contamination levels and ecological risks of heavy metals in soils, sediments, and agricultural products using methods such as the single-factor index, Nemerow composite index, and potential ecological risk assessment. The results indicate that heavy metals in the solid waste samples all exhibited concentrations exceeding their risk screening values, with 60% greater than their risk intervention values. The soils and sediments demonstrate slight and moderate comprehensive ecological risks of heavy metals. The single-factor potential ecological risks of heavy metals in both the soil and sediment samples decreased in the order of Hg, Cd, Pb, As, Cu, Zn, Cr, and Ni, suggesting the same sources of heavy metals in the soils and sediments. Most of the agricultural product samples exhibited over-limit concentrations of heavy metals dominated by Cd, Pb, Ni, and Cr, excluding Hg and As. The agricultural product assessment using the Nemerow composite index reveals that 35% of the agricultural product samples reached the heavy metal contamination level, implying that the agricultural products from farmland around the solid waste dumps have been contaminated with heavy metals. The eight heavy metals in the soil, sediment, and agricultural product samples manifested high coefficients of variation (CVs), indicating pronounced spatial variability. This suggests that their concentrations in soils, sediments, and agricultural products are significantly influenced by human mining activities. Additionally, the agricultural products exhibit strong transport and accumulation capacities for Cd, Cu, and Zn.

1. Introduction

Heavy metal contamination is considered one of the most serious anthropogenic environmental contamination types due to the toxicity and persistence of heavy metals in environments. Heavy metal contamination in soils has evolved into the most prominent environmental contamination in China. A contamination survey report revealed that heavy metals in soils across China show a total over-limit ratio of 16.1%, with an over-limit ratio of Cd of 7% among all survey points, suggesting a threat to food security (Yu T et al., 2021).

Researchers from China and abroad have conducted

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extensive studies on heavy metal contamination in various metal mining areas and their surrounding soils, highlighting the contamination situation, source identification, morphological analysis, and contamination assessment of heavy metals in soils of individual mining areas. Zheng XK et al. (2002) conducted a detailed analysis and elucidation of the current status and source distributions of heavy metal contamination in soils worldwide. Xu YN et al. (2007) and Zhang JH et al. (2013) investigated the distributions of heavy metals in stream sediments and soils in the Xiaoqinling gold mining area, revealing that stream sediments and soils in the area are subjected to varying degrees of heavy metal contamination. Wang WY et al. (2008) explored the distributions of heavy metals in soils and the ability of Ageratina adenophora to absorb heavy metals in the Shuiyindong gold deposit in Guizhou. Li HY et al. (2009) investigated and assessed five heavy metals, i.e., Zn, Cd, Pb, Cu, and As, in soils and plants surrounding indigenous zinc smelting zones in Bijie, Guizhou Province. They found that soils in these zones were subjected to moderate to severe heavy metal contamination, with Cd identified as the dominant contaminant, and that vegetables near these zones have been severely contaminated. Regions in the upper reaches of the Pearl River, including Yunnan and Guizhou, hold China's significant mineral bases. Soils in these regions suffer from severe Cd contamination and slight As and Cr contamination due to the original high background values of heavy metals and the impacts of waste gas, wastewater, and industrial residue from mineral exploitation (Yao B et al., 2020). Chen Y et al. (2012) and Gao YX et al. (2012) analyzed the heavy metal forms in soils around the gold mine in the upper reaches of the Miyun Reservoir.

Relevant studies have also been conducted abroad. Timofeev I et al. (2018) assessed the impacts of toxic heavy metals in soils around the Baikal tungsten-molybdenum mine, revealing that the heavy metal concentrations like Cd, Cu, As, Pb, and Mo around the tailings pond far exceed their background values. Kandziora-Ciupa et al. (2022) delved into the bioconcentration capacity of the leaves of Vaccinium myrtillus for heavy metals using methods such as the single-factor index, Nemerow composite index, and potential ecological risk index to assess the characteristics of leaves this plant and their physiological responses to heavy metal stress. They concluded that Cd poses the highest ecological threat in the area and that the shapes and volumes of Vaccinium myrtillus leaves changed considerably under heavy metal stress. This indicates that heavy metals significantly affect the leaves of this plant, which is enriched in large quantities of heavy metals. Similarly, using various contamination indices, Mahvi AH et al. (2022) analyzed heavy metal contamination in soils of different land use types in Iran, revealing that all soil samples were enriched in heavy metals, among which high-concentration Hg and low-concentration Cd exhibited high and moderate ecological risks, respectively.

Heavy metals can enter food chains through the water-soil-crop ecosystem, posing a threat to the health and lives of animals and humans (Zhong LY et al., 2011). In the mining and resource utilization of metal mines, the resulting heavy metal contamination in soils is characterized by various contamination elements, wide anomaly extension, and high contamination levels. Heavy metals are prone to accumulate in plants and then in animals and human bodies via food chains, inducing diseases such as cancerous lesions, along with significant ecological hazards (Bao LR et al., 2020). For instance, Pb poisoning harms the nervous, hematopoietic, and digestive systems of humans, while Cd poisoning can cause itai-itai disease (Xu YN et al., 2013; Zhu DN et al., 2021).

Presently, only simple surveys of cultivated land quality have been conducted in the study area, involving over-limit concentrations of heavy metals including Cd, Cr, and As in soils across cultivated lands, along with the over-limit Pb and Hg concentrations in soils locally. However, there is a lack of studies on the migration characteristics and influential factors of heavy metals in the solid waste - soil - water body - sediment - agricultural product system in the study area. This study aims to identify the root causes of heavy metal contamination in soils of the study area. By combining field surveys with sample tests, this study investigated the distribution patterns and potential risks of heavy metals in residual solid waste within lead-zinc deposits in a certain area of Guizhou Province and those across the soil - water body – sediment - agricultural product ecosystem affected by the solid waste. This study will provide a theoretical basis for future risk control of heavy metal contamination across the ecosystem, holding great significance for effective heavy metal contamination management and remediation in small river basins, as well as the protection of human health.

2. Overview of the study area

2.1. Topography and landforms

The study area is situated in the middle section of the eastern slope zone on the Yunnan-Guizhou Plateau. In this area, the Sancha River flows across the central part of Puding County, cutting the county into southern and northern portions. The northern portion, representing about one-third of the total county area, falls along the southern margin of the Wumeng Mountains and exhibits high mountains and steep slopes, with main peaks measuring greater than 1700 m in elevation. The southern portion belongs to the western end of the Miaoling Mountains, exhibiting a relatively flat terrain. The entire study area is higher in the east, north, and west, followed by the south, with the center the lowest in elevation. The highest point (elevation: 1846 m) occurs in the Doupeng Mountain in the northeast, and the lowest point (elevation: 1042 m) is observed in the Yangjiazhai Valley at the exit of the Sancha River, manifesting a relative elevation difference of 804 m.

The study area exhibits extensive carbonate rocks and intensively-developed karst, resulting in predominant karst landforms, which account for 84.3% of Puding County. The landforms in the study area can be divided into mountains, hills, and intermontane basins. Among them, mountains are primarily distributed in the northeast, north, and southwest, representing 34.8% of the total area; hills principally occur in the east and west, accounting for 49.6% of the total area, and intermontane basins are mainly distributed in the south and the central part, covering 15.6% of the total area.

2.2. Meteorology and hydrology

The study area, located at the intersection of cold and warm air currents, exhibits a mild and humid climate, frequent rainy days, and moderate sunshine, suggesting a typical plateau-type humid subtropical monsoon climate. However, its significant elevation differences and complex terrains lead to complex and diverse microclimates. Highland mountains and deeply incised valleys in the study area are characterized by pronounced vertical climate variations and differences in precipitation.

The study area is located in the west-central part of Guizhou Province, lying at the watershed between the Yangtze River and Pearl River systems. It is a typical karst landform concentration area in the world. Surface rivers in the study area all belong to the Yangtze River system, while most underground rivers are integral parts of the Yangtze River system except two, which flow into the Pearl River system. Rivers in the study area are characterized by pronounced rainfall sources from mountainous areas, remarkable flow changes, and dramatic differences between flood and dry seasons. During the flood season, floodwater rises and falls sharply, whereas some creeks frequently dry up in the dry season. Water bodies in the study area generally flow from west to east into the Sancha River, and eventually into the Yelang Lake Reservoir.

2.3. Geology

The study area resides along the southeastern margin of the Yangtze block and, tectonically, in the northern Guizhou uplift zone of the Yangtze platform, the NW-trending Weining tectonic deformation zone of the Liupanshui fault depression, and the Guiyang complex tectonic deformation zone of the Zunyi fault uplift. Strata in this area are dominated by Permian and Triassic strata, with the oldest and youngest strata exposed emerging as the Sinian-Cambrian Dengying Formation and the Cretaceous Maotai Formation, respectively. The Cambrian, Devonian, and Carboniferous strata are sporadically distributed in anticlinal cores, along with the possible presence of Emeishan basalts. The Jurassic Ziliujing Formation and the Cretaceous Maotai Formation are scattered in synclinal cores. The Quaternary strata exhibit an extremely sporadic distribution, small thicknesses, and complex genetic types, manifested as proluvium/eluvial loose sediments with various grain sizes. The Cambrian, Ordovician, Devonian, Carboniferous, Permian, and Triassic strata consist predominantly of marine carbonate deposits. In contrast, the Jurassic, Cretaceous, and Quaternary strata are identified as fluvial-lacustrine deposits in inland basins, intermontane basin deposits, and poly-genetic loose deposits in inland mountainous areas, respectively (Fig. 1).

The strata in the study area are generally dominated by sedimentary rocks, with a minor presence of igneous rocks. The sedimentary rocks primarily comprise carbonate rocks. The Cambrian to middle Late Triassic strata are dominated by marine carbonate rocks, intercalated with some marine clastics.

2.4. Mineral resources

The primary mineral resources exploited in the study area include coal mines and sand-gravel deposits, followed by lead-zinc deposits occurring principally in northern Jichangpo Town.

The lead-zinc deposits are primarily distributed near the core of the Wuzhishan anticline. Predominant ore-controlling structures in the study area include NNE- and NW-trending faults and the Wuzhishan anticline. Most lead-zinc deposits in

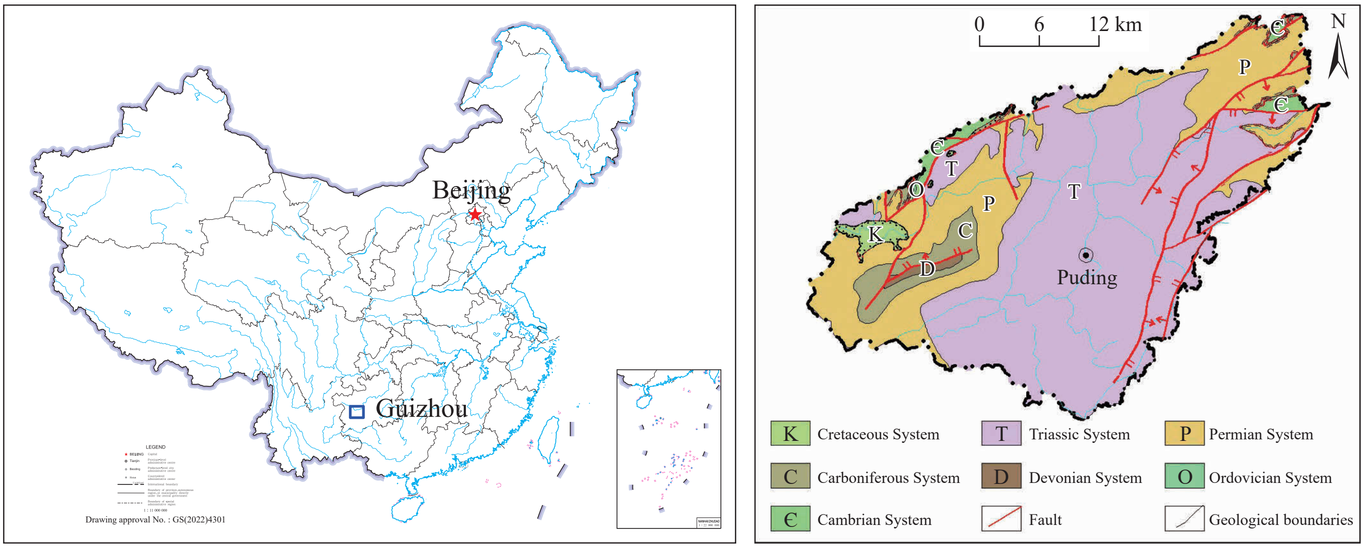

Fig. 1. Sketch map of regional geology.

the study area are found in the strong parts formed by the blocks intercalated by the abovementioned faults and the Wuzhishan anticline. The primary ore-bearing lithologies comprise the flint-bearing banded dolomites of the Sinian Dengying Formation and the nodular dolomites of the Cambrian Qingxudong Formation. Ore bodies in the study area are mainly stratiform and stratoid in shape, roughly aligning with the occurrence of ore-hosting strata. Additionally, the lead-zinc deposits are of the sedimentary exhalative type.

The mining of mineral resources in the study area has a long history. Despite its significant contributions to local development, the mining has left large quantities of solid waste, severely affecting and destroying ecosystems and causing many environmental contamination issues. Over decades, the mining of lead-zinc deposits in the study area has resulted in much lead-zinc solid waste. The solid waste, dominated by gangue and lead-zinc solid waste, left in the study area will become a 'chemical time bomb', posing serious potential hazards to the environment.

2.5. Eco-geology

The study area is characterized by complex geological structures, complex stratum lithologies, various landforms, and a fragile ecosystem. Its special natural geographical environment and geological tectonic setting lead to unique environmental and geologic issues.

On both sides of regional faults and in sections with secondary structures in the study area, rock masses are highly susceptible to fragmentation due to the compression from faults and folds, prone to cause geologic hazards. Relatively complete strata are exposed in the study area, where hard rocks interbedded with weak rocks and pure hard rocks are the most widely distributed. The zones of hard rocks interbedded with weak rocks are susceptible to geologic hazards like landslides and collapses. The pure hard rock zones, subjected to topographic cutting, frequently form steep rocky slopes, leading to geologic hazards like collapses along fracture and bedding surfaces. Besides, improper slope cutting in the engineering construction of roads and buildings is prone to form high and steep free surfaces, inducing potential geologic hazards like landslides and collapses.

Primary mining activities in the study area include the mining of coal mines, sand-gravel deposits, and lead-zinc deposits. Intensive mining engineering has severely damaged the geologic environment in the study area. The open-pit mining of sand-gravel deposits has caused severe damage to topography, landforms, and land resources, possibly driving slope formation and collapses. Especially in the coal mining area, numerous geologic hazards such as geofractures, landslides, and collapses have occurred. Furthermore, large quantities of accumulated waste resulting from mining activities are prone to form landslides or debris flows, posing a serious threat to the lives and property safety of local residents and severely restricting local economic development. Troubleshooting indicates that the solid waste dumps of heavy nonferrous metals in the study area are characterized by a few types, small volumes, and concentrated distributions. This suggests that the resulting contamination of surrounding cultivated land and rivers can be prevented and controlled. The study area hosts 31 solid waste dumps in lead-zinc deposits and two pyrite solid waste dumps, with the former primarily distributed in Jichangpo Town, Puding County.

3. Materials and methods

3.1. Sampling

Based on field surveys, this study sampled the existing 31 lead-zinc solid waste dumps, as well as soils, irrigation water, sediments, and crops in surrounding areas that are potentially affected by the solid waste dumps. The sampling sites were arranged according to the following principles and methods:

As per the *Technical Specifications on Identification for Hazardous Waste* (HJ298-2019) and relevant guidelines, at least five sampling sites or eight samples were arranged for each type of solid waste dumps (the numbers can be adjusted in the case where one town contains the same type of solid waste). One sample was taken from each sampling.

For irrigation water and sediments, at least five samples were collected 5 km along the lower reaches or at the intersection of rivers (for rivers measuring less than 5 km in length, the sample number can be adjusted, but at least three samples should be taken in total). The irrigation water should be sampled along the perpendicular line of the sampling sites of sediments.

Soils were primarily sampled on both sides of ponds and rivers that might be affected by leachates, with at least four sampling sites arranged. Centering on the actual sampling sites determined in the field, subsampling sites were pinpointed based on the shapes of the blocks to be sampled. Specifically, subsampling sites were arranged in an S-shaped pattern for a rectangular block to be sampled, and an X- or chessboard-shaped pattern for an approximately square block. Furthermore, corresponding subsampling sites should be arranged in the same block, determined at a distance of about 20 m from the center of the actual sampling site.

Agricultural products were sampled in the areas subjected to soil sampling where agricultural products like rice and maize were planted and the contamination sources of solid waste posed serious contamination risks to cultivated land. The sampling sites were arranged in blocks where: (1) waste entered cultivated land directly; (2) leachates entered cultivated land; (3) contaminated surface water poured into cultivated land in the flood season; (4) river sediments that might be contaminated entered cultivated land.

Finally, this study collected 72 samples from residual solid waste, 77 soil samples from soils in cultivated land in the lower reaches of rivers that might be affected by solid waste, 63 irrigation water samples, 63 river sediment samples, and 23 samples of agricultural products (Fig. 2).

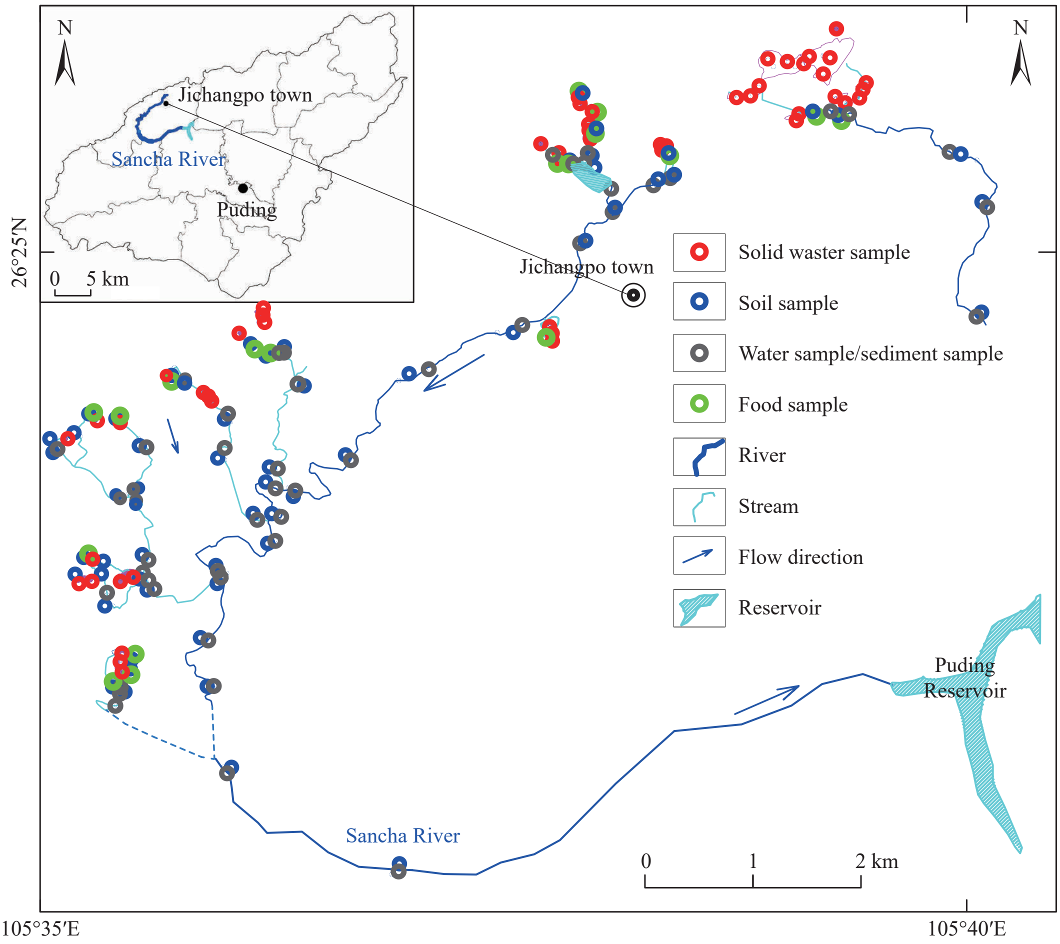

Fig. 2. Distribution of sampling sites.

3.2. Sample preparation and processing

The solid waste, soil, and sediment samples collected in the field were individually air-dried and mixed thoroughly into one sample measuring above 2 kg in weight using the quartering method. The prepared samples were crushed using a wooden stick and then passed through a 20-mesh sieve for preservation. Finally, the processed samples were sent to the laboratory for analysis and tests.

The irrigation water samples were processed according to corresponding testing methods. Specifically, a certain quantity of irrigation water samples was used to create new samples through digestion using an electric heating board. Then, the new samples measuring a constant volume were prepared for testing.

The samples of agricultural products were air-dried and then threshed, shelled, and chopped using a stainless steel knife. Subsequently, they were ground using an agate ball mill and passed through 20–60-mesh nylon sieves. Then, they were placed into ground-glass stoppered flasks. According to the corresponding testing method, a certain quantity of the prepared samples underwent microwave digestion. Then, samples measuring a constant volume were prepared for testing.

3.3. Sample tests

Sample tests were conducted at the Guizhou Kezheng Environmental Safety Testing Technology Co., Ltd., and the items analyzed included eight heavy metals (i.e., Cd, As, Hg, Cr, Pb, Cu, Zn, and Ni) and pH.

Residual solid waste was tested for leaching toxicity and the total contaminant concentration. The leaching toxicity was tested using methods specified in the *Solid Waste - Extraction Procedure for Leaching Toxicity - Sulphuric Acid & Nitric Acid Method* (HJ/T 299-2007) and *Solid Waste - Extraction Procedure for Leaching Toxicity - Horizontal Vibration Method* (HJ 557-2010).

After the solid waste samples underwent microwave digestion, the Hg and As elements in them were tested based on the *Solid Waste - Determination of Mercury, Arsenic, Selenium, Bismuth, Antimony. Microwave Dissolution/Atomic Fluorescence Spectrometry* (HJ 702-2014), and elements Cd, Cr, Pb, Cu, Zn, and Ni in these samples were tested as per the *Solid Waste - Determination of 22 Metal Elements - Inductively Coupled Plasma Optical Emission Spectrometry* (HJ 781-2016).

For soil and sediment samples, relevant elements in them were tested principally based on the *Soil and Sediment - Determination of Mercury, Arsenic, Selenium, Bismuth, Antimony - Microwave Dissolution/Atomic Fluorescence Spectrometry* (HJ 680-2013) and *Soil and Sediment - Determination of Copper, Zinc, Lead, Nickel and Chromium - Flame Atomic Absorption Spectrophotometry* (HJ 491-2019).

Besides, the tests of soil and sediment samples also followed the corresponding indices and analytical methods specified in the *Soil Environmental Quality - Risk Control Standard for Soil Contamination of Agricultural Land (Trial)* (GB 15618-2018).

The irrigation water quality was tested following the *Environmental Quality Standards for Surface Water* (GB 3838-2002) and analytical methods specified in the *Water Quality - Determination of Mercury, Arsenic, Selenium, Bismuth and Antimony - Atomic Fluorescence Spectrometry* (HJ 694-2014), and the *Water Quality - Determination of Copper, Zinc, Lead and Cadmium - Atomic Absorption Spectrometry* (GB/T 7475-1987).

The agricultural product samples were tested using the corresponding indices and analytical methods specified in the *National Food Safety Standard - Determination of Multi-Elements in Food* (GB 5009-12-2017).

In this study, primary instruments for analysis and tests included an atomic adsorption spectrophotometer, a flame atomic adsorption spectrometer, an atomic fluorescence spectrometer, a microwave digestion system, a graphite heating plate, a horizontal oscillator, a soil pulverizer, and an electric blast drying oven.

3.4. Analytical testing quality control

During the sampling process, samples were randomly selected as blank and duplicate samples, with the over 5% of various samples subjected to quality control. In the calibration of analytical instruments, the correlation coefficients (R) of their calibration curves exceeded 0.999. For each batch of samples, at least two blank samples were analyzed and tested, with the test results below the detection limits of the used methods. The duplicate samples were tested based on testing parameters and their mass concentration levels, with relative deviations of the test results ranging from 10% to 30%.

3.5. Assessment methods

Many methods are available to assess the risks of heavy metals in soils both in China and abroad. Among them, primary total amount index methods include the single-factor index, Nemerow composite index, geoaccumulation index, and potential ecological risk index (Wang Y et al., 2018; Chu S et al., 2018). Currently, the indices employed in China include the Nemerow composite index, enrichment factor, geoaccumulation index, and potential ecological risk index (Yang L et al., 2022).

The arithmetic or geometric mean of the single-factor index somewhat weakens the weights of heavy metals in the comprehensive assessment, rendering the single-factor index only applicable to the assessment of specific areas contaminated with single factors. However, the single-factor index serves as the basis of environmental quality grading and comprehensive assessment (Sun HY et al., 2023). The Nemerow composite index is an internationally widely accepted comprehensive assessment method for heavy metal contamination (Guo JG et al., 2021). The potential ecological risk index integrates heavy metal concentrations with the biotoxicity differences and additive effects of multiple elements, thus capable of comprehensively reflecting the potential impacts of multiple heavy metals on ecosystems (Qin SC et al., 2018).

The enrichment factor utilizes reference elements to normalize contamination elements, reducing the impacts of ambient media and sample collection and processing on element concentrations. However, the instability of reference elements increases the uncertainty of background values in the assessment of regional soil environmental quality (Zhang XZ et al., 2006). The geoaccumulation index underlines the assessment of heavy metal concentrations by comparison with background values, principally reflecting the enrichment degrees of exogenous heavy metals. In contrast, the potential ecological risk index considers the biotoxicity effects of different heavy metals besides their concentrations, thus yielding more practical results (Xu ZC et al., 2009).

Therefore, this study employed the single-factor index, Nemerow composite index, and potential ecological risk index to assess heavy metal contamination.

The single-factor index and Nemerow composite index have been widely applied in the geochemical assessment of heavy metals in soils (Nemerow NL, 1974; Xi CZ et al., 2023; Dong QY et al., 2023; Table 1). The single- factor index can be calculated using the following equation (Equ. 1):

$$P_i = C_i / S_i \tag{1}$$

where P_i is the contaminant index of heavy metal i, C_i is the contaminant test value of heavy metal i, and S_i is the risk screening value of heavy metal i.

The risk screening values of soil contaminants were determined by refereeing to the risk screening values for soil contamination of agricultural land specified in the *Soil Environmental Quality - Risk Control Standard for Soil Contamination of Agricultural Land* (Ministry of Ecology and Environment, 2018).

The Nemerow composite index can be calculated using the following equation (Equ. 2):

$$P_n = \sqrt{\frac{(\mathrm{P^2imax} + \mathrm{P^2iave})}{2}} \tag{2}$$

where P_n is the composite contamination index, and P_{imax} and P_{iave} represent the maximum of a single-factor index and the arithmetic mean of several single-factor indices, respectively.

Table 1. Grading criteria for single-factor index and Nemerow composite index.

Grade	Single factor index (P_i)	Nemerow composite index (P_n)	Contamination level
0	<0.7	<0.7	Non-contamination
1	0.7–1.0	0.7–1.0	Warning values
2	1.0–2.0	1.0–2.0	Mild
3	2.0–3.0	2.0–3.0	Moderate
4	>3.0	>3.0	Severe

The potential risk index, proposed by Swedish scholar Hakanson L (1980), considers the synergistic effects of the types, concentrations, environmental background values, and toxicity levels of heavy metals. This index has been extensively used to assess the risks of heavy metals in soils and sediments and relevant environmental quality (Yu X et al. 2015; Dai JR et al. 2018; Bao LR et al. 2020; Li XY et al. 2022; Li JF et al. 2023; Sun HY et al. 2023; Table 2). The equations for the potential risk index are as follows (Cao SW, 2022; Equ. 3–5):

$$C_f^i = C^i / C_n^i \tag{3}$$

$$E_r^i = T_r^i \times C_f^i \tag{4}$$

$$RI = \sum E_r^i \tag{5}$$

where C_f^i is the contamination coefficient of an individual metal, C^i is the measured concentration of heavy metal i in the test sample, C_n^i is the background value of heavy metal i, T_r^i is the toxicity coefficient of various heavy metals, E_r^i is the potential ecological risk coefficient of an individual metal, and RI is the potential comprehensive risk index of various heavy metals. In this study, C_n^i was set at the background values of elements in soils in Guizhou Province (Zn: 104.21 mg/kg; Cr: 98.98 mg/kg; Ni: 39.30 mg/kg; Cd: 0.40 mg/kg; Hg: 0.13 mg/kg; As: 13.48 mg/kg; Pb: 33.57 mg/kg; Cu: 34.50 mg/kg; Cai DW et al. 2020; Jiang PH et al. 2023).

Based on available studies, the toxicity coefficients (T_r^i) of heavy metals were set as follows: Zn=1<Cr=2<Cu=Ni=Pb=5<As=10<Cd=30<Hg=40 (Hakanson L, 1980; Chen JS et al., 1999; Xu ZQ et al., 2008; Sun HY et al., 2021; Tu CL et al., 2023; Huang Q et al., 2024).

4. Results and analysis

4.1. Assessment of heavy metal contamination in solid waste

Heavy metals in the solid waste samples were assessed based on the risk screening and intervention values specified in GB 15618-2018.

Table 2. Grading criteria for the assessment of potential ecological risks.

E_r^i	Hazard level of single-factor ecological risk	RI	Hazard level of comprehensive ecological risk
<40	Low	<150	Low
40–80	Moderate	150–300	Moderate
80–160	High	300–600	High
160-320	Very high	600–1200	Very high
≥320	Extreme high	≥1200	Extreme high

The test results of the total concentrations of various heavy metals in solid waste (Table 3) indicate that solid waste exhibits a weak acidity, with average, maximum, and minimum pH values of 6.67, 9.04, and 3.48, respectively. The statistics of heavy metal concentrations reveal that the solid waste features average Cd, As, Hg, Cr, Pb, Cu, Zn, and Ni concentrations of 20.25 mg/kg, 32.67 mg/kg, 4.07 mg/kg, 59.78 mg/kg, 175.29 mg/kg, 175.22 mg/kg, 4221.70 mg/kg, and 288.06 mg/kg, respectively.

All 72 solid waste samples showed heavy metal concentrations surpassing the risk screening values. Among them, 43 samples showed heavy metal concentrations greater than the risk intervention values. Fig. 3 illustrates the proportions of solid waste samples with over-limit heavy metal concentrations. Specifically, samples with Cd and Hg concentrations exceeding the risk intervention values represented 60% and 11%, respectively, and those with Zn concentrations surpassing the risk screening value accounted for up to 75%—the highest proportion compared to other heavy metals. The maximum Cd, As, Hg, Cr, and Pb concentrations in the solid waste samples were 215.35, 1.33, 24.56, 0.51, and 2.42 times their risk intervention values, respectively. The maximum Cu, Zn, and Ni concentrations in them were 20.22, 170.80, and 41.34 times their risk screening values, respectively. These findings demonstrate relatively high heavy metal concentrations in solid waste, suggesting that these heavy metals have the potential to contaminate surrounding land.

4.2. Assessment of heavy metal contamination in soils

The test results of the total concentrations of various heavy metals in soils indicate the soils exhibit average, maximum, and minimum pH values of 7.17, 8.28, and 4.25, respectively. The soils display average Cd, As, Hg, Cr, Pb, Cu, Zn, and Ni concentrations of 0.63 mg/kg, 22.46 mg/kg, 0.92 mg/kg, 56.78 mg/kg, 128.80 mg/kg, 43.48 mg/kg, 288.49 mg/kg, and 37.56 mg/kg, respectively.

Based on the grading criteria or the single-factor index and the Nemerow composite index (Wang CY et al., 2021; Bao LR et al., 2020), the statistics of the assessment results of 77 soil samples derived using both indices are shown in Table 4. Compared to the risk screening values of heavy metal contamination in soils, the single-factor contamination indices of Cr, Ni, and Cu are all less than 1, below the warning level, suggesting minor contamination risks. Soil samples with over-limit As concentrations exhibited the highest proportion,

Table 3. Statistics of the heavy metal concentrations in solid waste (mg/kg).

Solid waste	Cd	As	Hg	Cr	Pb	Cu	Zn	Ni
Maximum	430.69	199.00	61.40	430.69	1209.02	4043.64	42700	4133.93
Minimum	0.19	4.01	0.12	0.49	5.75	1.00	7.59	0.58
Average	20.25	32.67	4.07	59.78	175.29	175.22	4221.70	288.06
Standard deviation	52.82	30.42	10.20	76.49	257.31	614.96	8633.42	842.67
CV	2.61	0.93	2.51	1.28	1.47	3.51	2.05	2.93

followed by those with over-limit Zn, Cd, Pb, and Hg concentrations sequentially, which were 39%, 32%, 31%, 29%, and 19%, respectively. The assessment results derived using the Nemerow composite index suggest that 58% of the soil samples reached the contamination level.

CVs allow for the elimination of the impacts of heavy metal concentrations across various dimensions (Zhan YZ et al., 2011). Therefore, they are apt at reflecting both average variations in heavy metal concentrations across sampling sites and the intensity of human activities. It is generally accepted that CVs of <15%, 15%–35%, and >35% represent low, moderate, and high variabilities, respectively (Guo YH et al., 2017; Li X et al., 2017; Chen ZY et al., 2023).

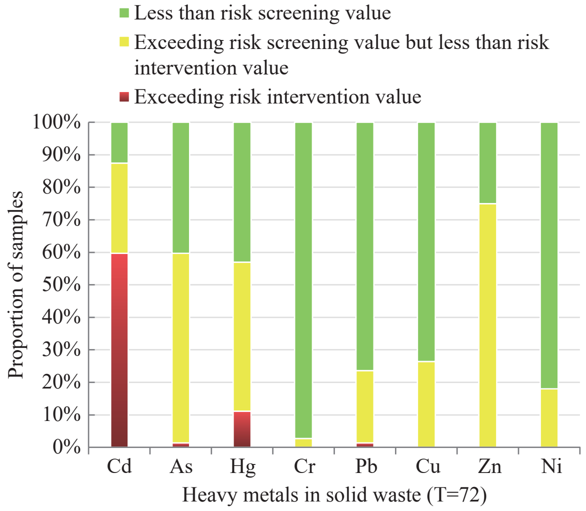

Fig. 3. Proportions of solid waste samples with over-limit heavy metal concentrations.

In soil samples, eight heavy metals except for Ni all showed CVs of greater than 35%, demonstrating significant spatial variability. This suggests that the Cd, As, Hg, Cr, Pb, Cu and Zn, concentrations in soils are heavily influenced by anthropogenic factors, especially mining activities. Hg in soil samples exhibited the maximum CV (1.15), indicating local Hg enrichment with local Hg contamination being 14.18 times the limit. This is likely due to the high mobility of Hg in the natural environment. This element can be found in a series of redox reactions and desorption in soils, accumulating in soils while being released into the atmosphere, thus exhibiting significant spatial variability (Jiang F et al., 2019).

The assessment results of soil samples reveal that soils in the study area have been contaminated with heavy metals, especially As, Pb, Zn, and Cd.

With the soil background values of Guizhou Province as reference values, this study calculated the potential ecological risk indices of heavy metals in soils of cultivated land in the study area (Table 5). The results indicate that Cr, Zn, Cu, and Ni in all soil samples exhibited potential ecological risk indices of less than 40, indicating low ecological risks. Element As exhibited a potential ecological risk index varying from 2.81 to 59.12, carrying low risks in most samples and moderate risks in two samples. Pb exhibited a potential ecological risk index ranging from 0 to 80.88, posing low risks in most samples, moderate risks in six samples, and high risks in only one sample. Cd manifested a potential ecological risk index varying from 0 to 256.5, posing low risks in 43 samples (55.8%), moderate risks in 23 samples, high risks in nine samples, and very high risks in two samples. Hg exhibited potential ecological risk indices varying from 19.38 to 2181.54, posing low risks in nine samples, moderate/high

Table 4. **Single-factor index and Nemerow composite index of heavy metals in soils.**

Soil	Single factor index P_i								Nemerow composite index P_n
	Cd	As	Hg	Cr	Pb	Cu	Zn	Ni	
Maximum	4.28	3.19	14.18	0.33	3.47	0.67	2.78	0.73	10.15
Minimum	0.03	0.15	0.00	0.06	0.08	0.02	0.12	0.06	0.13
Average	0.98	0.96	1.07	0.18	0.74	0.23	0.90	0.32	1.53
Standard deviation	0.90	0.69	1.22	0.07	0.71	0.17	0.72	0.11	0.89
CV	0.91	0.72	1.15	0.39	0.96	0.73	0.80	0.33	0.58
Number of samples with over-limit heavy metals	24	30	15	0	22	0	25	0	45
Proportion of samples with over-limit heavy metals	31%	39%	19%	0%	29%	0%	32%	0%	58%

Table 5. **Potential ecological risk indices of heavy metals in soils.**

Soil		Single-factor ecological risk index Eri								Comprehensive risk index RI
		Cd	As	Hg	Cr	Pb	Cu	Zn	Ni	
Maximum		256.5	59.12	2181.54	2.35	80.88	19.42	8.00	10.31	2236.11
Minimum		0.00	2.81	19.38	0.38	0.00	0.43	0.29	0.00	35.96
Average		46.30	16.66	208.45	1.15	18.93	6.30	2.29	2.67	300.05
Standard deviation		44.39	11.03	383.24	0.46	18.59	4.19	1.73	2.83	386.98
CV		0.96	0.66	1.84	0.40	0.98	0.66	0.75	1.06	1.29
Hazard level of ecological risks	Low	43	75	9	77	70	77	77	77	25
	Moderate	23	2	24	0	6	0	0	0	36
	High	9	0	25	0	1	0	0	0	8
	Very high	2	0	7	0	0	0	0	0	5
	Extremely high	0	0	12	0	0	0	0	0	3

risks in most samples (63.6%), very high risks in 17 samples, and extremely high risks in 12 samples. Therefore, Hg and Cd are identified as the primary elements posing potential ecological risks to soils in the study area, followed by As and Pb, which carry moderate to high risks in several samples and low risks in the remaining samples.

The single-factor indices of Cr, Cu, and Ni indicate minor contamination risks. This largely aligns with the potential ecological risk index-based assessment results where the potential ecological risk indices were all below 40, suggesting low ecological risks. Despite mild contamination overall, Hg exhibits a relatively high toxicity coefficient, thereby manifesting a high potential ecological risk index. In contrast, despite relatively serious contamination, Zn exhibits a relatively low potential ecological risk index due to its low toxicity coefficient. The assessment results derived using the single-factor potential ecological risk indices indicate that Hg posed the highest ecological risks, emerging as the dominant contributing factor to ecological risks, followed by Cd. The single-factor ecological risk indices decreased in the order of Hg, Cd, Pb, As, Cu, Zn, Ni, and Cr.

The soil samples exhibited potential comprehensive ecological risk indices (*RI*) of heavy metals ranging from 35.96 to 2236.11. Among them, 79.2% posed low to medium comprehensive ecological risks of heavy metals, while 20.8% of the samples carried high, very high, and extremely high risks.

4.3. Assessment of heavy metal contamination in river sediments

The test results indicate that river sediments in the study area exhibit average Cd, As, Hg, Cr, Pb, Cu, Zn, and Ni concentrations of 1.05 mg/kg, 21.25 mg/kg, 0.96 mg/kg, 50.29 mg/kg, 114.27 mg/kg, 39.74 mg/kg, 199.05 mg/kg, and 12.01 mg/kg, respectively.

The heavy metal contamination in river sediments was also assessed using the potential ecological risk index coined by Swedish scholar Hakanson L (1980).

The results indicate that Cu, Cr, Ni, and Zn in all 63 sediment samples showed potential ecological risk indices below 40, indicating low potential ecological risks. Elements As and Pb displayed average potential ecological risk indices of 15.65 and 16.77, respectively, carrying low risks in most samples and moderate and high risks in the rest. Hg manifested a potential ecological risk index ranging from 20.11 to 5969.23, posing low to extremely high risks dominated by moderate (33.3%) and high (34.9%) risks, with extremely high risks observed in six samples. Cd showed a potential ecological risk index ranging from 0.75 to 1425.25, posing low to extremely high risks dominated by low (36.5%) and moderate (38.1%) risks, with high to extremely high risks observed in 16 samples (Table 6). Overall, heavy metals in river sediments primarily exhibited low potential ecological risks. Among them, Hg exhibited the highest ecological risks, followed by Cd. The single-factor ecological risk indices decreased in the order of Hg, Cd, Pb, As, Cu, Zn, Cr, and Ni, roughly aligning with the assessment results of heavy metals in surrounding soils. The findings evidence that heavy metals in soils and river sediments originate from the same sources, primarily attributable to human mining activities.

All eight heavy metals in sediment samples showed CVs exceeding 35%, indicating significant spatial variability. This result also suggests that the concentrations of these heavy metals in sediments are heavily influenced by anthropogenic factors, primarily sourced from solid waste generated from lead-zinc mining activities.

The potential comprehensive ecological risk index of heavy metals in the sediment samples ranged from 62.07 to 6014.27, averaging 411.39. Hence, heavy metals pose low to moderate potential ecological risks primarily and high to extremely high risks partially.

4.4. Assessment of heavy metal contamination in irrigation water

After heavy metals enter water bodies, a small fraction of heavy metal ions will be suspended in water while the majority will be deposited in sediments. That is why the heavy metal concentrations in sediments reach up to 100, or even 1000, times those in water bodies (Bai J et al., 2011).

In this study, 63 irrigation water samples were taken from rivers, hilly ponds, and other surface water bodies around solid waste dumps. They showed pH values ranging from 6.33 to 7.54 and Cd, As, Hg, Pb, Cu, Zn, and Ni concentrations of 0.001 mg/kg, 0.0003–0.0013 mg/kg,

Table 6. Potential ecological risk indices of heavy metals in river sediments.

River sediments		Single-factor ecological risk index Eri								Comprehensive risk index *RI*
		Cd	As	Hg	Cr	Pb	Cu	Zn	Ni	
Maximum		425.25	143.18	5969.23	2.02	82.81	12.90	6.11	3.31	6014.27
Minimum		0.75	0.77	20.11	0.28	1.49	1.74	0.40	0.00	62.07
Average		77.62	15.65	291.87	1.01	16.77	5.74	1.89	0.82	411.39
Standard deviation		80.11	22.36	954.21	0.35	17.20	2.56	1.59	0.99	947.95
CV		1.03	1.43	3.27	0.36	1.03	0.45	0.84	1.20	2.30
Hazard level of ecological risks	Low	23	60	12	63	57	63	63	63	25
	Moderate	24	1	21	0	5	0	0	0	25
	High	8	2	22	0	1	0	0	0	7
	Very high	7	0	2	0	0	0	0	0	3
	Extremely high	1	0	6	0	0	0	0	0	3

0.04–0.26 mg/kg, 0.01 mg/kg, 0.05 mg/kg, 0.05 mg/kg, and 0.005 mg/kg, respectively. Compared to the irrigation water limits specified in the *Standard for Irrigation Water Quality* (GB 5084-2021), none of the samples exceeded the limits. In contrast, they exhibited Cd, As, Hg, Pb, Cu, Zn, and Ni concentrations 20 to 5000 times those in water bodies. This suggests that the irrigation water is uncontaminated or that most heavy metals in water bodies have been deposited in river sediments or transferred to suspended solids.

4.5. Assessment of heavy metal contamination in agricultural products

Eight heavy metals in 23 agricultural product samples from fields around solid waste dumps in the study area were tested and analyzed. The results indicate that these samples exhibited average Cd, As, Hg, Cr, Pb, Cu, Zn, and Ni concentrations of 0.14 mg/kg, 0.02 mg/kg, 0, 0.78 mg/kg, 0.22 mg/kg, 2.82 mg/kg, 17.58 mg/kg, and 0.40 mg/kg, respectively.

The heavy metal contamination in agricultural products was assessed using the single-factor index and Nemerow composite index (Zhou Y et al., 2020).

The maximum residue limits of Pb, Cd, Hg, As, Ni, and Cr in agricultural products (grains) were determined as per the *National Food Safety Standard - Maximum Levels of Contaminants in Foods* (GB 2762-2022). The maximum residue limits of Cu and Zn were determined according to the *Limits of Eight Elements in Cereals, Legume, Tubes, and Its Products* (NY 861-2004; Table 7).

Among 23 agricultural product samples, only three samples showed heavy metal concentrations within the limits, while the remaining 20 samples displayed over-limit heavy metal concentrations to varying degrees. The over-limit ratios of heavy metals in the 20 samples decreased in the order of Cd (30%), Pb (22%), Ni (17%), Cr (13%), Cu, and Zn. Hence, Cd, Pb, Ni, and Cr are identified as major over-limit factors (Table 8).

Hg and As exhibited single-factor indices of less than 1, below the warning level, suggesting minor contamination risks. The assessment results derived using the Nemerow composite index reveal that 35% of agricultural product samples reached the contamination level, evidencing that the agricultural products from the fields around the solid waste dumps have been contaminated.

Table 7. Limits of heavy metals in agricultural products (mg/kg).

Heavy metal	Cd	As	Hg	Cr	Pb	Ni	Cu	Zn
Agricultural products	0.1	0.5	0.02	1.0	0.2	1.0	10	50

Besides, all eight heavy metals in agricultural product samples showed CVs above 35%, indicating significant spatial variability. This also suggests that the concentrations of these heavy metals in agricultural products were heavily influenced by anthropogenic factors, especially mining activities.

5. Discussion

5.1. Solid waste

The correlation analysis results of Cd, Pb, and Zn in solid waste (Table 9) indicate an extremely significant positive correlation between Cd and Pb and significant positive correlations between Pb and Zn and between Cd and Zn. These findings are consistent with previous results stating that Cd, Pb, and Zn exhibit extremely strong correlations in bedrocks and that Cd is frequently associated with Zn due to their similar chemical properties (Yang L et al., 2022).

Additionally, the correlation analysis results reveal extremely significant positive correlations between Hg and Zn, Cr and Zn, and Cu and Ni and significant positive correlations between Cd and As, Cd and Hg, As and Cr, and As and Zn. This suggests that these elements are all sourced from rock and soil minerals or influenced by human mining activities.

5.2. River sediments

The significance of correlations between heavy metals can indicate whether they originate from the same or similar sources (Wang L et al, 2015; Jiang YF and Guo X, 2019).

The correlation analysis results of heavy metals in sediments (Table 10) reveal extremely significant positive correlations between Cd and As, Cd and Cr, Cd and Pb, Cd and Cu, As and Cu, Cr and Cu, Cr and Ni, and Cu and Ni, implying that they originate from similar sources or pose similar levels of contamination.

5.3. Soils

The correlation analysis results of Zn, Cd, and Pb in soils (Table 11) reveal extremely significant positive correlations

Table 8. Single-factor index and Nemerow composite index of heavy metals in agriculture products.

Agricultural product	Single factor index P_i								Nemerow composite index P_n
	Cd	As	Hg	Cr	Pb	Cu	Zn	Ni	
Maximum	7.50	0.35	0.90	8.24	14.20	1.47	1.06	3.16	10.27
Minimum	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.09
Average	1.46	0.05	0.04	0.81	1.14	0.33	0.34	0.42	1.77
Standard deviation	2.13	0.09	0.19	2.01	2.99	0.35	0.21	0.94	2.57
CV	1.46	1.67	4.58	2.47	2.62	1.07	0.63	2.27	1.45
Number of samples with over-limit heavy metals	7	0	0	3	5	2	1	4	8
Proportion of samples with over-limit heavy metals	30%	0%	0%	13%	22%	9%	4%	17%	35%

Table 9. Correlation coefficients of heavy metals in solid waste.

Solid waste	Cd	As	Hg	Cr	Pb	Cu	Zn	Ni
Cd	1	0.28*	0.30*	0.03	0.35**	−0.16	0.25*	−0.16
As		1	0.09	0.28*	0.08	0.10	0.13	0.02
Hg			1	0.18	0.23	−0.06	0.66**	−0.09
Cr				1	0.03	−0.01	0.58**	−0.08
Pb					1	0.01	0.24*	0.10
Cu						1	0.05	0.49**
Zn							1	−0.10
Ni								1

Note : * denotes significant correlation at a significance level of 0.05, and ** denotes significant correlation at a significance level of 0.01. n=70, r0.05=0.235, and r0.01=0.306.

between Zn and Cd, Zn and Pb, and Cd and Pb, with correlation coefficients of 0.49, 0.60, and 0.38, respectively (n=77, $r0.05$=0.224). This suggests that Zn, Cd, and Pb possibly exhibit the same sources or similar transport paths. Additionally, lead-zinc mining areas are frequently associated with Cd contamination.

Additionally, the correlation analysis results also indicate significant positive correlations between Cd and As, Cd and Hg, Cd and Cr, and Cd and Cu and extremely significant positive correlations between As and Pb, and Cu and Cr, implying that they possibly originate from similar sources, which are dominated by lead-zinc solid waste or the weathering of soil-forming parent rocks.

5.4. Agricultural products

After activation and dissolution by interstitial water, heavy metals in soils migrate from plant roots into plants. Then, they attach to plant cells and accumulate in roots, stems, leaves, flowers, and fruits. Different plant species, or even the same plant species, possess different abilities to absorb and transport heavy metals in various soils.

Wei SH et al. (2003) hold that the bioaccumulation factor (BAF) is a critical measure of plants' ability to accumulate heavy metals. A higher BAF suggests a higher accumulation capacity.

The BAF, defined as the ratio of an element's concentration in the overground portion of a plant to that in soils where the plant grows, represents the plant’s ability to accumulate the element (Yoon J et al., 2006). This measure is primarily employed to assess the relationships of heavy metal concentrations between soils/sediments and plant organs (Galal Tarek M, et al., 2017).

The equation for the BAF is as follows (Equ. 6):

$$\mathrm{BAF} = C_{plant}/C_{soil} \tag{6}$$

where C_{plant} is the heavy metal concentration in the overground portion or roots of a plant, and C_{soil} is the heavy metal concentration in soils.

Calculations indicate that the agricultural products exhibited average BAFs of Cd, As, Hg, Cr, Pb, Cu, Zn, and Ni of 0.72, 0.01, 0, 0.08, 0.01, 0.13, 0.12, and 0.05, respectively. This suggests that the crops have a high ability to transport and accumulate Cd, Cu, and Zn, aligning with the findings of Yang MX et al. (2015). Plants manifest higher accumulation capacities for Cu and Zn—essential elements for their growth—compared to non-essential elements. This is likely because plants possess a stronger ability to absorb their nutritional elements like Cu and Zn. Agricultural product samples exhibited a strong accumulation capacity for Cd, with the BAFs of Cd in partial samples exceeding 1. This is

Table 10. Correlation coefficients of heavy metals in river sediments.

Sediment	Cd	As	Hg	Cr	Pb	Cu	Zn	Ni	Cd
Cd	1	0.47**	0.23	0.39**	0.59 **	0.42**	0.18	0.01	1
As		1	0.03	0.22	0.18	0.47**	−0.13	−0.11	
Hg			1	−0.17	−0.12	−0.17	−0.13	0.06	
Cr				1	0.01	0.69**	−0.05	0.64 **	
Pb					1	0.09	0.23	−0.27	
Cu						1	−0.08	0.51**	
Zn							1	−0.11	
Ni								1	

Note: * denotes significant correlation at a significance level of 0.05, and ** denotes significant correlation at a significance level of 0.01. n=70, r0.05=0.248, and r0.01=0.322.

Table 11. Correlation coefficients of heavy metals in soils.

Soils	Cd	As	Hg	Cr	Pb	Cu	Zn
Cd	1	0.50**	0.27*	0.33**	0.38**	0.26*	0.49**
As		1	0.02	0.20	0.25*	0.14	0.17
Hg			1	−0.05	−0.03	0.02	−0.06
Cr				1	0.07	0.69**	−0.03
Pb					1	−0.14	0.60**
Cu						1	−0.26
Zn							1

Note : * denotes significant correlation at a significance level of 0.05, and ** denotes significant correlation at a significance level of 0.01. n=77, r0.05=0.224, and r0.01=0.292.

attributed to the high mobility of Cd in soils and the high ability of plants to absorb Cd. Cd exhibits enhanced solubility in slightly acidic soil, thus possessing strong migration and accumulation capacities.

Additionally, the correlation analysis results of Zn, Cd, and Cu in agricultural products from the study area (Fig. 4) reveal significant positive correlations between Zn and Cu, Cu and Cd, and Zn and Cd, with correlation coefficients of 0.93, 0.90, and 0.85, respectively (n=11, $r0.05$=0.602). This suggests that agricultural products have similar abilities to absorb Zn, Cd, and Cu or that Zn, Cd, and Zn share the same sources.

The analyses of the correlations of Zn, Cd, Cu, and Pb in agricultural products with those in soils reveals correlation coefficients of 0.14, 0.51, 0.15, and 0.07, respectively (n=11; $r0.05$=0.602), suggesting nonsignificant correlations. Hence, the adsorption of agricultural products for heavy metals is independent of the total heavy metal concentrations in soils. Generally, the amount of heavy metals adsorbed by plants depends on the concentrations of active heavy metals, such as water-soluble and ion-exchangeable heavy metals, in soils (Zhou GH et al., 2005).

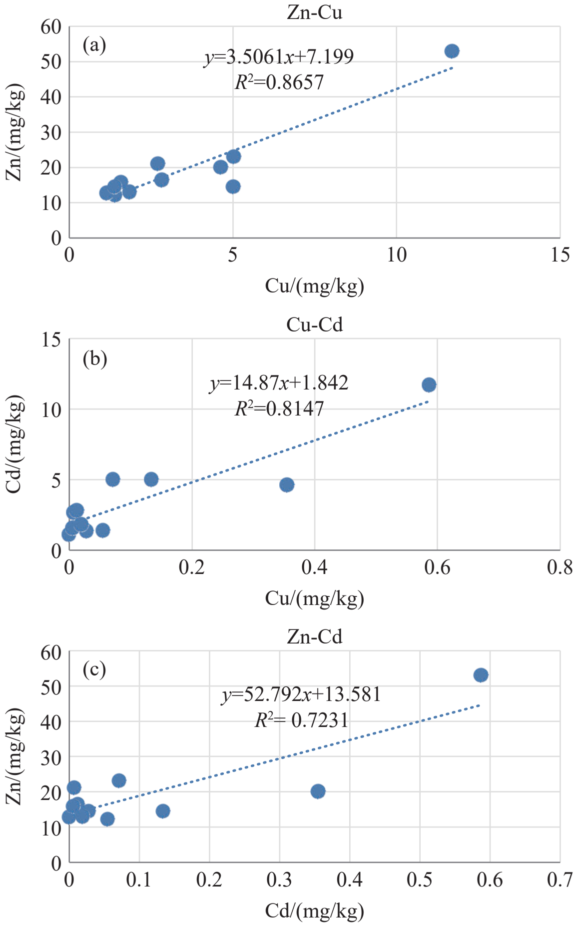

Fig. 4. Curves showing the correlations of Zn, Cd, and Cu in agriculture products.

5.5. Correlations of heavy metal concentrations in solid waste, soils, sediments, irrigation water, and agricultural products

The statistical analysis reveals that the respective Cd, Hg, Cu, Zn, and Ni concentrations in solid waste are around 10 times those in soils and sediments. In contrast, the respective As, Cr, and Pb concentrations in solid waste are close to those in soils and sediments, exhibiting the same orders of magnitude. The soils and sediments manifest similar Cd, As, Hg, Cr, Pb, Cu, Zn, and Ni concentrations, suggesting comparable abilities to absorb and accumulate the eight heavy metals (Table 12).

It is generally accepted that BAFs of an element of 0.01–0.1, 0.1–1, and 1–10 suggest weak, moderate, and strong adsorption capacities of plants for the element. The agricultural products in the study area exhibit BAFs of As, Cr, Pb, and Ni ranging from 0.01 to 0.1, indicating weak adsorption capacities for these elements. They display BAFs of Zn and Cu ranging from 0.1 to 1, suggesting strong adsorption capacities for both. Furthermore, they show a BAF of Cd ranging from 0.1 to 10, indicating moderate to strong adsorption ability to this element.

6. Conclusions

(i) The solid waste samples from the study area exhibited high heavy metal concentrations, part of which exceeded the risk screening values and intervention values. This indicates that the solid waste has the potential to contaminate surrounding land. The surrounding oils and sediments primarily carry low to moderate potential comprehensive ecological risks of heavy metals. In contrast, the irrigation water exhibits no over-limit heavy metal concentrations. Most agricultural products in surrounding fields show somewhat over-limit heavy metals dominated by Cd, Pb, Ni, and Cr, suggesting that agricultural products have been contaminated with heavy metals.

(ii) The analysis and assessment of heavy metals in soils and sediments within the study area suggest that Cr, Cu, and Ni roughly pose no contamination risk. The single-factor

Table 12. Average heavy metal concentrations in solid waste, soils, sediments, irrigation water, and agricultural products in the study area.

Heavy metal	Cd	As	Hg	Cr	Pb	Cu	Zn	Ni
Solid waste	20.25	32.67	4.07	59.78	175.29	175.22	4221.7	288.06
Soils	0.63	22.46	0.92	56.78	128.8	43.48	288.49	37.56
Sediment	1.05	21.25	0.96	50.29	114.27	39.74	199.05	12.01
Irrigation water	0.001	0.0013	0.15	0	0.01	0.05	0.05	0.005
Agricultural products	0.14	0.02	0	0.78	0.22	2.82	17.58	0.4

ecological risk indices of eight heavy metals in soils and sediments decrease in the order of Hg, Cd, Pb, As, Cu, Zn, Ni, and Cr, implying the same sources of heavy metals in soils and sediments.

(iii) The concentrations of the eight heavy metals in irrigation water surrounding the solid waste dumps fall below the basic limits specified for irrigation water quality. However, the Cd, As, Hg, Pb, Cu, Zn, and Ni concentrations in sediments are 20 to 5000 times those in irrigation water. This contrast suggests that the irrigation water is not contaminated or that the heavy metals in water bodies have largely migrated into river sediments.

(iv) For soils and sediments, the concentrations of the eight heavy metals manifest high CVs, implying significant spatial variability of these heavy metals. Furthermore, most heavy metals in soils and sediments exhibit significant positive correlations. These findings suggest that the concentrations of the eight heavy metals in soils and sediments have been significantly affected by anthropogenic factors and that these heavy metals all originate from the zinc-lead solid waste generated during mining.

(v) Significant positive correlations between Zn, Cd, and Cu can be observed in agricultural products from the study area, suggesting that the agricultural products possess similar abilities to absorb Zn, Cd, and Cu or that the three heavy metals share the same sources. Additionally, the agricultural products exhibit strong transport and accumulation capacities for these heavy metals.

CRediT authorship contribution statement

Zhi-qiang Wu conceptualized the entire paper and prepared the manuscript. Hai-ying Li participated in the conception and article verification. Liu-yan Lü and Guo-jun Liang participated in the investigation, sample collection and drawing illustrations. Ting-ting Wu and Jiang-xia Zhu supervised the field investigation and sample collection of the whole project. All authors contributed to the manuscript.

Declaration of competing interest

The authors declare no conflict to interest.

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