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ORIGINAL PAPER

# Investigating the environmental capacity of soil heavy metals and its determinants in agro-pastoral regions of the qinghai-tibetan plateau

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**Abstract** Environmental capacity (EC) serves as the basis for environmental planning and management, as a key indicator for assessing environmental risk and quality, and as a foundation for achieving sustainable development. Studies on EC typically address agricultural or urban rather than pastoral areas, with few examining agro-pastoral areas. The EC of the Tibetan Plateau is particularly important, considering its importance as an agricultural area and ecological reserve. To address this gap, the Qingshizui area in Menyuan County, a typical agro–pastoral area on the Tibetan Plateau, was selected to quantify soil EC and its spatial distribution. In terms of the dynamic and static annual soil EC for this region, the heavy metals were ranked as follows, in ascending order: Cd, Hg, Co, As, Sb, Ni, Cu, Pb, Cr, and Zn. Most of the areas with high residual EC were in the west. For the 10 heavy metals, residual EC was significantly affected by geological background. For all the heavy metals except Zn and Hg, residual EC was significantly affected by soil type. The heavy metal elements in the agro-pastoral area's soil are mildly enriched, suggesting minimal human impact. The composite EC index of this soil is 0.98, indicating an intermediate EC and low health risk. This study underscores that integrating agriculture and pastoralism can optimize land use and mitigate ecological pressures associated with these practices when done separately. Our research provides valuable insights for resource optimization, environmental conservation, and enhancing the welfare of farmers and herders in the Qinghai-Tibet region.

**Keywords** Environmental capacity · Tibetan Plateau · Agro-pastoral combination area · Trend prediction

## Introduction

Soil is a vital resource possessing self-purification capabilities (Mai et al., 2024; Wu, 2023). However, due to its unique ecological function, soil becomes the primary repository and destination for environmental heavy metal (HM) pollutants (Zhao et al., 2024). Heavy metals in soil are characterized by long latency periods, significant hazardousness, pronounced geographical variability, and challenges in detection (Liu et al., 2024a; Wang et al., 2024b; Yang et al., 2024). The accumulation and migration of these heavy metals in the soil environment could pose threats to the safety of human food and drinking water, disrupt the

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regional ecological equilibrium and environmental health, and influence the quality of the human habitat (Wang et al., 2024a). Soil environmental capacity (EC), a critical metric for the soil's contaminant bearing capability, is defined as the maximum amount of pollution that the soil can withstand within a specified space and timeframe, in adherence to regional soil environmental quality standards, while preventing damage to the natural and human living environments (Chen et al., 2001; Tian et al., 2022; Wu, 2017). Assessing the soil's total potential contaminant load is crucial for the prevention and management of local soil heavy metal contamination (Wang et al., 2023).

The Tibetan Plateau, with its unique natural conditions such as high altitude, cold climate, scarce precipitation, and sparse population distribution, significantly limits the development of farming activities (Guan et al., 2018; Yao et al., 2012). In contrast, its vast grasslands and abundant sunshine provide excellent conditions for animal husbandry, which has become the dominant economic activity in the region (Chai et al., 2022). Since the Industrial Revolution, the global temperature has generally risen, and the current global average surface temperature has surpassed that of 90% of the Holocene epoch (Marcott et al., 2013). This continuous warming trend is expected to enhance the richness and stability of pasture resources, thereby promoting the prosperity of animal husbandry. In the Tibetan Plateau, animal husbandry not only carries profound traditional pastoral culture but also gains more importance due to the imperfect market mechanisms (Tang et al., 2022). The implementation of the grassland transfer system has significantly improved the income levels of herdsmen, achieving dual growth in livestock income and non-livestock income by encouraging production investment and promoting non-pastoral employment opportunities. Historical data show that before the reform and opening, the growth of animal husbandry mainly depended on the expansion of livestock numbers, highlighting the traditional path of economic growth in the Tibetan Plateau's animal husbandry (Su, 2011). Given the harsh natural conditions of the Tibetan Plateau posing significant challenges to agriculture, leading to an imbalance between agricultural inputs and outputs, animal husbandry exhibits more evident economic advantages and development potential in this region. This study focuses on an area of the Tibetan Plateau characterized by a combination of agriculture and animal husbandry, aiming to provide a new perspective to address the lack of prior research on the EC of the Tibetan Plateau. By deeply analyzing the agro-pastoralism model in this region, we aim to explore how to achieve sustainable development of animal husbandry and promote positive interactions between agriculture and animal husbandry while respecting natural ecology and traditional cultures.

Owing to its high altitude, the ecosystems of the Tibetan Plateau are less capable of self-repair and are susceptible to damage by external factors, potentially leading to irreversible environmental and economic losses (Sai et al., 2022). Although the HM content of Tibetan Plateau ecosystems is relatively low under natural conditions (Wang et al., 2014), the impact of farming and grazing on soil quality can lead to localized HM contamination of soils. Human activities and geological processes are the main factors leading to HM accumulation (Liu et al., 2024b; Wang et al., 2022). Human activities include agriculture, industry, and transport, whereas geological action may involve natural processes such as rock weathering and soil erosion. EC research is highly important to mitigation of HM contamination in the Qinghai–Tibet region, to verify the feasibility of agro-pastoralism as a viable practice, and to provide a theoretical reference for future research.

Soil contamination by HMs has been widely studied on the Tibetan Plateau. Contamination by As and Cd, and the ecological risks it poses, is a serious problem in the "One River and Three Streams" watershed in Gannan, and the risk is highest in the watershed; further, Tibet's "One River and Two Tributaries" watershed exhibits high levels of Cd contamination (Du et al., 2021). On the Tibetan Plateau in the 1970s, Cd and Sb levels were 2.13 and 1.52 times higher in the topsoil than in the background soil (Yang et al., 2020). In Lhasa, Cu, Cr, Ni, Zn, Pb, Cd, As, and Hg levels were 1.41, 1.02, 0.86, 1.08, 1.09, 1.25, 0.90, and 0.87 times higher, respectively, in green spaces than in the city (Li et al., 2023b). In Ze Ku County, Qinghai Province, a typical alpine agricultural region in China, in surface soil (0–20 cm) samples, As exhibited the largest coefficient of variation (CV) and was more affected by anthropogenic influences than by other factors, whereas Cd, Pb, Co, Cr, Cu, Ni, Zn, and Hg had smaller CVs, and were less affected by human activity than by other factors (Li et al., 2023a).

Most studies on this topic have examined urban or agricultural areas; relatively few have addressed pastoral areas, and even fewer have examined agro-pastoral areas. The EC of agro-pastoral regions is therefore poorly understood, and EC should be examined in the agro-pastoral regions on the Tibetan Plateau.

We therefore chose the Qingshizui area (Menyuan County, Qinghai Province) as the research location to examine the EC of the soil and its drivers in the agro-pastoral areas of the Qinghai-Tibetan Plateau. Soil EC was systematically analyzed via methods including spatial analysis, material-balance linear modeling, enrichment factor analysis, and using the EC index. The research objectives were 1) to establish local soil background values based on soil measurements, to understand the baseline condition of the soil; 2) to use models to calculate the environmental capacity of heavy metals in soil, predict future changes, and evaluate risks through environmental capacity indices and enrichment factor analysis to enable health risk warnings and advance identification and proactive response to potential environmental risks; and 3) to make proposals for avoiding and managing soil contamination in this agro-pastoral region.

## Materials and methods

### Study region

The study region (Fig. 1) is in western Menyuan County, Qinghai Province (101°00′ – 102°30′ E and 36°50′–37°50′ N), in the North Qilian ophiolite-mixed rock belt north of the Deep Great Fracture on the southern slope of the Darshan Mountains. The stratigraphic outcrops cover the Paleoproterozoic, Mesoproterozoic, Ordovician, Silurian, Permian, Triassic, Jurassic, Neoproterozoic, and Quaternary periods. The soils are primarily alpine meadow, mountain meadow, and black calcium soils, with alpine cold desert, grey-brown, and chestnut calcium soils distributed in the peripheral areas. The study region, a typical agro-pastoral area, primarily comprises dry land, forest land, grassland, and marshland, with 5193 ha of arable land and 31,200 ha of usable grassland.

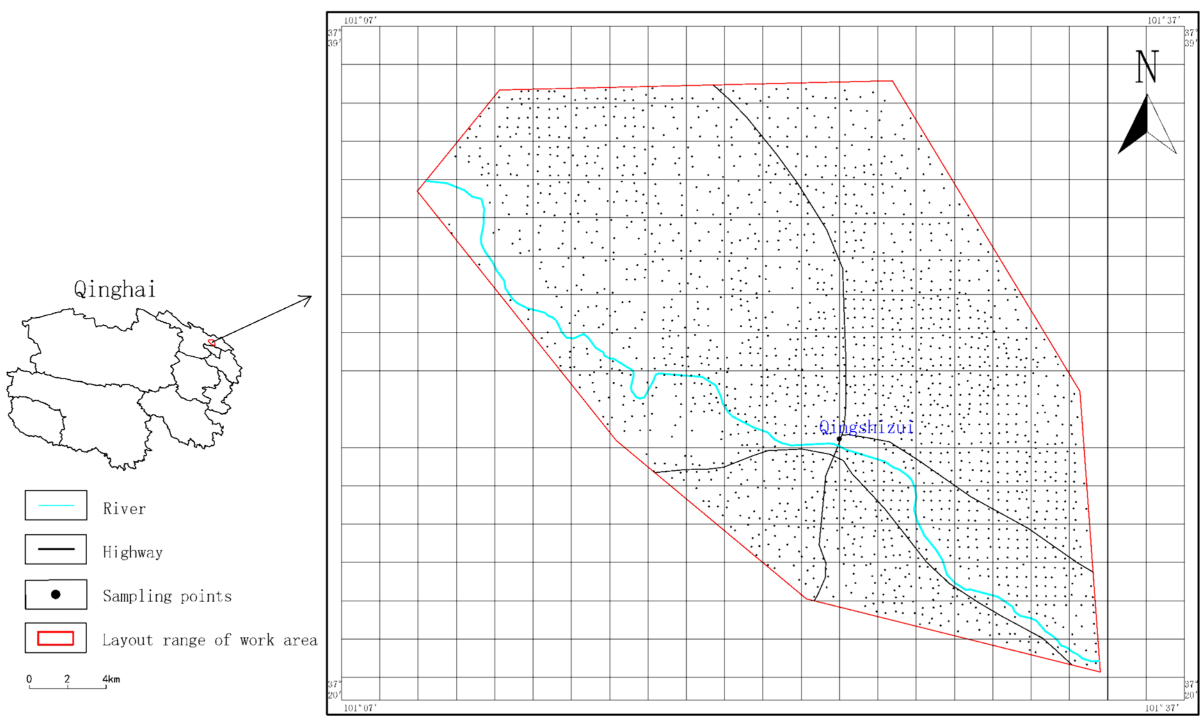


Fig. 1 Location of the study area

## Soil data collection and analysis

According to the Specification for Geochemical Evaluation of Land Quality (DZ/T0295-2016), the field-point locations were determined using a 1:50,000 topographic map, considering the uniformity and representativeness of the points, with a point density of 4–16 points/km$^2$ and sampling depth of 0–20 cm. In total, 3346 samples were taken.

Soil and sediment samples were collected, decontaminated, and bagged. Soil was dried on sample racks in the shade, away from direct sunlight. Soil was kneaded frequently during drying to prevent clumping. After drying, the samples were beaten with a mallet to the natural grain size and passed through a 10-mesh nylon sieve; the soil that passed through the sieve was mixed well and placed in a sample bag. Ensure that each soil sample weighs at least 200 g for analysis at the Qinghai Provincial Geological Mineral Testing Application Center.

Ten HMs were selected as indicators for the evaluation of the soil EC: As, Cd, Cr, Cu, Hg, Co, Ni, Pb, Sb, and Zn. Inductively coupled plasma mass spectrometry (ICP-MS) was used to determine Cd, Cu, and Pb levels, X-ray fluorescence spectrometry (XRF) to determine Cr, Ni, Zn, and Co levels, and atomic fluorescence spectrometry (AFS) to determine As, Hg, and Sb levels. For quality control, 12 pieces of national Grade I soil standard material GSS-17 to GSS-28 were inserted every 300 samples to ensure accuracy; the pass rate was 100% for all elements. Precision was ensured by inserting four pieces of the national Grade I standards every 100 samples into the blanks set aside in advance (GSS-8, GSS-9, GSS-11, GSS-12/GSS GSS-8, GSS-9, GSS-11, GSS-12/GSS-7, GSS-8, GSS-12, GSS-13). The samples were analyzed together, and the logarithmic difference between the single determination value of the four standards and the standard recommended value for each element (Δlgc) was calculated; the qualification rate was 100% for all elements. The method detection limits (MDLs) for As, Cd, Cr, Cu, Hg, Co, Ni, Pb, Sb, and Zn were 0.2, $2.1 \times 10^{-2}$, 3.0, 1.0, $5.0 \times 10^{-4}$, 1.0, 1.0, $8.5 \times 10^{-1}$, $4.0 \times 10^{-2}$ and 1.0 μg/g, respectively.

## Data processing methods

### *Establishment of soil geochemical background values and risk reference values*

The soil geochemical background values (the background values in environments with high anthropogenic impacts) must be established for the agro-pastoral areas of the Tibetan Plateau, which are highly affected by anthropogenic disturbances. These values will support the further evaluation of local soil pollution levels, provide suggestions for the development of agro-pastoralism, and promote local economic development. Soil levels of the 10 HMs (As, Cd, Cr, Cu, Hg, Co, Ni, Pb, Sb, and Zn) in the research region varied greatly in the surface soil, owing to the influence of farming and grazing. Outliers beyond three standard deviations from the mean were eliminated to obtain the background dataset, and the mean was recalculated as the background value.

Determination of risk reference values is an important step in the study of soil EC. The risk reference values used here for As, Cd, Cr, Cu, Co, Hg, Ni, Pb, and Zn are for soils with pH 6.5–7.5 and are based on the risk screening standards in the Risk Control Standard for Soil Pollution on Agricultural Land (GB15618-2018). The risk reference values for Sb are based on the risk screening values for Sb (metallic) in the U.S. National Environmental Protection Agency's Regional Screening Levels (RSL). These reference values were designed to assess the risk of the contaminant to the soil ecosystem; this risk is considered low

**Table 1** Background and reference values of the heavy metals in the soil of the study area

| Element | Background Value (mg/kg) | Reference Value (mg/kg) |
|---|---|---|
| As | 14.67 | 30 |
| Cd | 0.19 | 0.3 |
| Cr | 81.02 | 200 |
| Cu | 30.52 | 100 |
| Hg | 0.03 | 2.4 |
| Co | 14.92 | 23 |
| Ni | 36.87 | 100 |
| Pb | 25.82 | 120 |
| Sb | 1.09 | 31 |
| Zn | 86.1 | 250 |

if the contaminant levels are equal to or lower than the reference values. Table 1 presents the background and risk reference values for the soil environment.

### *Spatial analysis*

The inverse distance weighting (IDW) method, based on the first law of geography, states that the similarity of attributes between objects decreases with increasing distance (Zhang & Duan, 2023). It assumes that the sample point closest to the interpolated point contributes the most to the value at the interpolated point, and that this contribution is inversely proportional to the distance (Jia et al., 2016). The use of linear, unbiased, and optimally estimated interpolation is common for evaluating the EC of soil for HMs and the EC index of soil HMs. This method does not require data transformation, and is calculated as follows:

$$Z\left(x_0\right) = \sum_{i=1}^{n} \varphi_i \times Z(x_i) \quad (1)$$

where i is the sample identifier, $Z(x_i)$ is the observed value, $Z(x_0)$ is the linear predicted value, n is the total number of measurement samples, and $\varphi_i$ is the optimum weight value for achieving unbiased prediction with the least variation.

### *Material-balance linear modeling*

(1) Static EC.

After establishing the soil background and risk reference values, the static EC of the soil was calculated as follows:

$$Q_{so} = M \times 10^{-6} \times \left(C_i - C_{bi}\right) \quad (2)$$

where $Q_{so}$ (kg/hm$^2$) is the static EC of the soil; M ($2.25 \times 10^6$ kg/hm$^2$) is the weight of the soil per hectare in the arable stratum; $C_i$ (mg/kg) is the risk reference value of each pollutant; and $C_{bi}$ (mg/kg) is the soil background value of each pollutant.

The soil's capacity to contain each pollutant, namely its static EC, is calculated using Eq. 2 (Wu et al., 2023). Although the parameters for calculating static EC are simple and easy to apply, static EC is less applicable to pollutants that are easily decomposed in the soil. Equation 2 can be used to calculate the static EC of the soil for each of the HM contaminants (Du et al., 2007).

(2) Residual EC.

Soil residual EC (REC), the discrepancy between the maximum potential soil burden and the current contamination level (Tian et al., 2022), can be calculated as follows using the soil measurements and risk reference values:

$$Q_i = M \times 10^{-6} \times \left(C_i - C_p\right) \quad (3)$$

where $Q_i$ (kg/hm$^2$) is the soil REC; M ($2.25 \times 10^6$ kg/hm$^2$) is the soil weight per hectare in the arable stratum; $C_i$ (mg/kg) is the risk reference value for the pollutant; and $C_p$ (mg/kg) is the measured HM level in soil.

(3) Dynamic EC.

Owing to naturally occurring background levels of elements in the soil, the input channels of each element are diverse and their input is continuous, and they are lost via crop absorption, runoff, and leakage. A dynamic equilibrium thus exists between the elements within the soil, causing the level of each element that the soil can contain to vary relative to the soil environmental quality standards (Ma et al., 2016). To elucidate this equilibrium, we established a linear model of the material balance to estimate the annual dynamic capacity of the soil:

$$Q_{in} = M \times \left(C_i - C_p \times K^n\right) \times \frac{1-K}{K \times (1-K^n)} \times 10^{-6} \quad (4)$$

where $Q_{in}$ (mg/kg) is the annual dynamic capacity of HM element i in the soil; M ($2.25 \times 10^6$ kg/hm$^2$) is the weight of the soil every acre in the arable stratum; $C_i$ (mg/kg) is the risk reference value for the pollutant; $C_p$ (mg/kg) is the actual measured value of HMs in the soil; K is the residual level of HMs in soil, set to 0.9 (Li, 2022; Wu, 2017); and n is the number of years in the study period. The dynamic EC of the soil in the agro-pastoral area should be set for three time periods (20, 30, and 50 years), to ensure that the quality of the soil environment will be effectively protected and controlled for the next 20, 30, and 50 years. Using time periods that are either too short or too long could lead to inaccurate control of EC, with consequences for the health and sustainable development of the entire ecosystem.

(4) EC index.

In China, the EC index method is the most used approach for evaluating soil environmental quality. For each element, the ratio of its soil REC to its static EC is known as the individual EC index; the EC indices are averaged to generate a composite EC index, as follows:

$$P(i)={}^{Q_i}/_{Q_{so}} \tag{5}$$

$$PI = \frac{1}{n}\sum_{i=1}^{n} P_i \tag{6}$$

where P(i) represents the individual EC index of the soil HM element i; PI is the composite EC index; and n is the number of elements. Table 2 presents the grading criteria for soil EC for each HM.

(5) Enrichment factors.

The enrichment factor (EF), which is essential for determining the amount of HM pollution and its enrichment in the soil, quantifies the influence of human activity on HM levels in the soil (Lin et al., 2024). EF is calculated as follows:

$$EF = \frac{C_x/C_r}{B_x/B_r} \tag{7}$$

where $C_x$ (mg/kg) is the actual measured value of the HM in the soil, $C_r$ (mg/kg) is the background value of the HM in the soil, $B_x$ (mg/kg) is the actual measured value of the reference element in the soil, and $B_r$ (mg/kg) is the background value of the reference element in the soil. Reference elements are typically those that are commonly found in the Earth's crust, less affected by human activity, and chemically stable. For Si, the CV in agro-pastoral areas is low, at only 7.66%. Therefore, Si was selected as the reference element here. Table 3 lists the enrichment factors corresponding to the enrichment-grade classification criteria.

## Statistical analysis

The geographical distribution of the sampling points was mapped using ArcGIS 10.8. The spatial distribution of the EC was described using IDW interpolation. The data were analyzed using ANOVA in SPSS 27, to examine the distribution characteristics and to identify significant differences and the drivers of EC, considering geological background, land-use type, and soil type. A confidence level of 0.05 was applied. Origin 2022 was used for data visualization.

# Results and discussion

## Soil HM content

Table 4 presents the levels the 10 HMs in the 3346 soil samples. The surface soil contained As (15.21 mg/kg), Cd (0.20 mg/kg), Cr (84.23 mg/kg), Cu (31.16 mg/kg), Hg (0.04 mg/kg), Co (14.90 mg/kg), Ni (37.95 mg/kg), Pb (25.81 mg/kg), Sb (1.10 mg/kg), and Zn (85.50 mg/kg). These values do not, on average, exceed the risk reference values. The research region is in the Menyuan Basin, a vast catchment area with fracture zones the east and west and significant development of meso-basic volcanic rocks, the

**Table 2** Classification standards for soil heavy metal environmental capacity

| Environment capacity level | Environment capacity index | Risk level |
|---|---|---|
| High capacity | $P > 1$ | No risk |
| Medium capacity | $0.7 < P \leq 1$ | Mild risk |
| Low capacity | $0.3 < P \leq 0.7$ | Moderate risk |
| Alert capacity | $0 < P \leq 0.3$ | Severe risk |
| Overload capacity | $P \leq 0$ | Extreme risk |

**Table 3** Enrichment level classification criteria

| EF | Enrichment level |
|---|---|
| $EF \leq 1$ | No enrichment |
| $1 < EF \leq 2$ | Mild enrichment |
| $2 < EF \leq 5$ | Moderate enrichment |
| $5 < EF \leq 20$ | Significant enrichment |
| $20 < EF \leq 40$ | Strong enrichment |
| $EF > 40$ | Extreme enrichment |

**Table 4** Descriptive statistics for heavy metal content (mg/kg)

| Element | Minimum | Maximum | Mean | Standard deviation | Coefficient of variation |
|---|---|---|---|---|---|
| As | 1.75 | 91.79 | 15.21 | 4.43 | 0.29 |
| Cd | 0.07 | 0.66 | 0.20 | 0.05 | 0.26 |
| Cr | 20.40 | 516.00 | 84.23 | 18.70 | 0.22 |
| Cu | 11.60 | 73.60 | 31.16 | 7.47 | 0.24 |
| Hg | 0.00 | 3.23 | 0.04 | 0.08 | 2.04 |
| Co | 4.82 | 26.30 | 14.90 | 2.42 | 0.16 |
| Ni | 10.90 | 102.00 | 37.95 | 9.38 | 0.25 |
| Pb | 10.80 | 69.80 | 25.81 | 3.37 | 0.13 |
| Sb | 0.10 | 1.99 | 1.10 | 0.25 | 0.22 |
| Zn | 39.40 | 145.00 | 85.50 | 11.10 | 0.13 |

primary mineral-bearing sections in the area. Owing to the pooling effect of flowing water, levels of polymetallic elements such as Cr, Ni, Hg, Cu, and Pb were high, with some surface enrichment. CV reflects the variability and homogeneity of the HMs in the soil, partially capturing the impact of human activity on soil HM levels. A higher CV indicates greater differences in HM levels between sites and greater diffusion of these elements (Du et al., 2007). Owing to the relatively high level of development in the study region, Hg exhibited a CV of 2.04, indicating high variability and high dispersion that may be related to the local agro-pastoralism. For Pb, Co, Cr, Cu, Sb, Ni, and Zn, the CV was low, at $\leq 0.25$, indicating that the distribution of these elements is affected primarily by natural factors. For As and Cd, the CV was intermediate ($> 0.25$ and $< 0.5$), potentially reflecting the influence of local agro-pastoral activity.

## Static EC

Static EC was highest for Zn, followed by Cr, Pb, Cu, Ni, Sb, As, Co, Hg, and Cd, at 368.78, 267.71, 211.91, 156.33, 142.04, 67.30, 34.49, 18.18, 5.33, and 0.25 kg/hm$^2$, respectively (Eq. 2). Static EC varied among the elements, being relatively high for Zn, Cr, Pb, Cu, and Ni, and therefore posing a low risk. In the agro-pastoral areas, the average levels of these elements were similar to the background soil values, and their CVs were low, indicating that their levels in the soil are influenced primarily by natural factors and less by anthropogenic factors. The static ECs of As, Co, Hg, and Cd were relatively low, indicating that they pose a high risk. These findings reveal that the levels of Cd in the soil of the study region exceed the safety standards, indicating that it poses an ecological risk in the area.

## Distribution of REC

IDW interpolation was used to obtain the distribution of REC (Eqs. 1 and 3; Fig. 2; Table 5). For As, the mean REC was 35.16 kg/hm$^2$. The REC for As was highest in the montane meadow soils and lowest in black calcareous soils, and was low in the eastern and northern regions and high in the southwestern region. For Cu, the mean REC was 159.20 kg/hm$^2$. The REC for Cu was highest in the montane meadow soils and lowest in black calcareous soils, and was high throughout the southwest, with low REC values concentrated in the eastern and northern regions. For Pb, the mean REC was 212.05 kg/hm$^2$. The REC for Pb was highest in marshland soils and lowest in alpine meadow soils; it was lowest in the center of the study area, with high levels in the western and southern regions, although its REC was distributed more evenly than that of the other elements. For Co, the mean REC was 19.85 kg/hm$^2$; the Co REC was highest in the montane meadow soils and lowest in the black calcareous soils. For Co, most of the areas with low a REC were located in the east, whereas those with high REC were mostly located in the southwestern region. For Cr, the mean REC was 265.91 kg/hm$^2$; the Cr REC was highest in the mountain soils and lowest in the alpine meadow soils, respectively. Cr REC values were lowest in the center of the study area, and most of the points with high Cr REC were in the southwest. For Ni, the mean REC was 143.77 kg/hm$^2$; Ni REC was highest in the montane meadow soils and lowest in black calcareous soils. Most of the points with low Ni REC were located in the eastern part of the study area, and most of the points with high Ni REC in the southwestern areas. For Zn, the mean REC was 371.38 kg/hm$^2$; Zn REC was highest in the mountain soils and lowest in the alpine meadow soils, respectively. Zn REC was predominantly low in the central part of the study area, while it was high in some parts of the southwestern area. For Cd, Hg, and Sb, the REC values were 0.21, 5.32, and 67.49 kg/hm$^2$, respectively; the RECs of

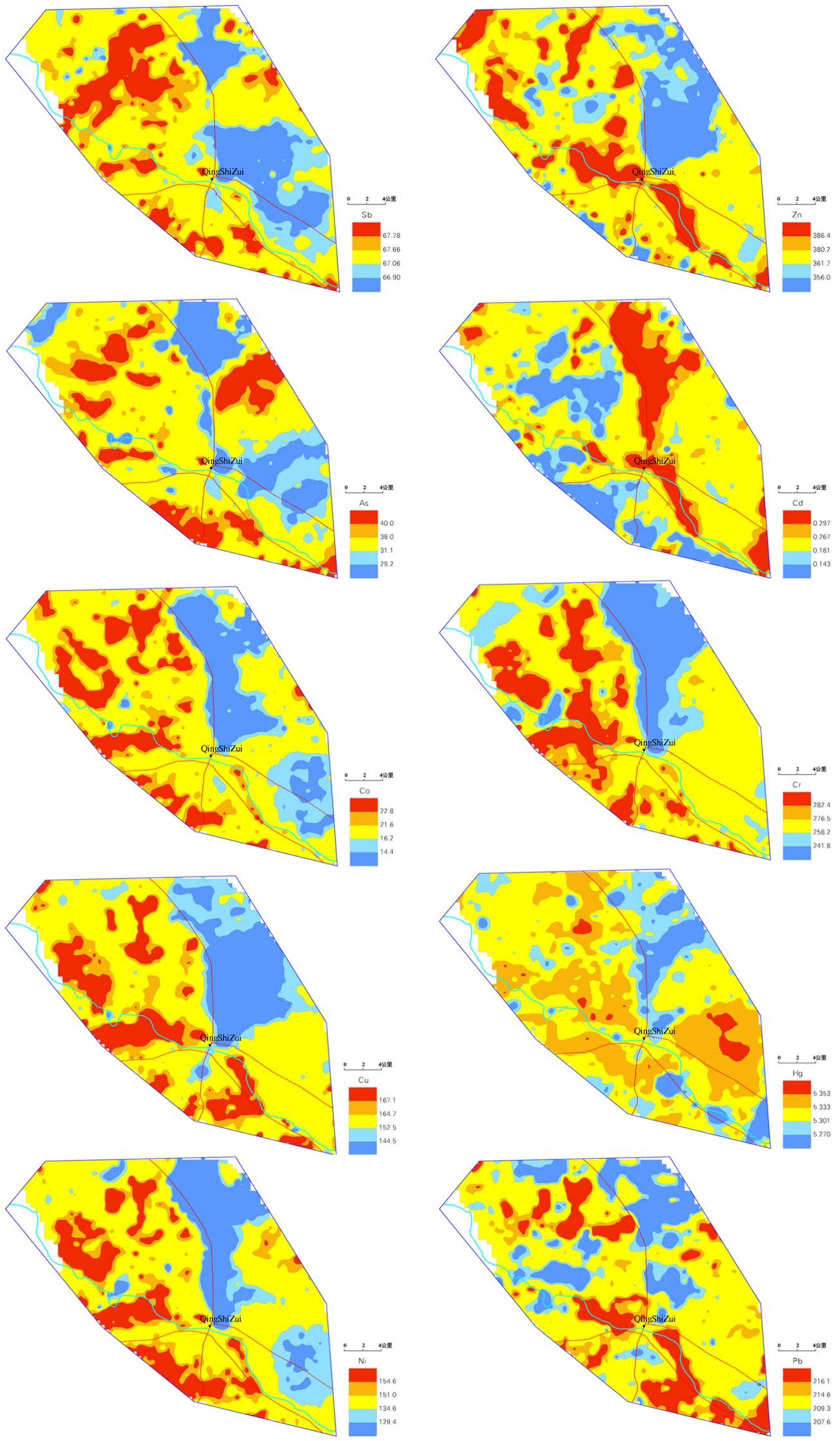


**Fig. 2** Residual environmental capacity of the soils in the study area to accommodate heavy metals

these HMs were more evenly spatially distributed than those of the other HMs.

The significance of geological background, land-use type, and soil type as drivers of REC was examined (Table 6). Geological background markedly affected REC for all 10 elements, indicating that the levels of most of the elements in this agro-pastoral area depend on to geological background. In the

**Table 5** Residual capacity (kg/hm$^2$) of the soils of the study area

| Agrotype | As | Sb | Hg | Cd | Cu | Pb | Co | Cr | Ni | Zn |
|---|---|---|---|---|---|---|---|---|---|---|
| Chernozem | 32.69 | 67.21 | 5.31 | 0.23 | 153.69 | 212.25 | 17.66 | 260.14 | 138.20 | 370.17 |
| Alpine meadow soil | 33.33 | 67.35 | 5.31 | 0.22 | 154.04 | 210.28 | 18.89 | 251.60 | 139.08 | 367.69 |
| Boggy soil | 36.44 | 67.74 | 5.32 | 0.22 | 162.46 | 214.30 | 21.10 | 272.34 | 143.89 | 373.42 |
| Mountain meadow soil | 38.16 | 67.67 | 5.32 | 0.18 | 166.62 | 211.36 | 21.76 | 279.55 | 153.90 | 374.25 |
| Mean value | 35.16 | 67.49 | 5.32 | 0.21 | 159.20 | 212.05 | 19.85 | 265.91 | 143.77 | 371.38 |

**Table 6** Significance of geological background, land-use type, and soil type as drivers of heavy metal environmental carrying capacity

| | Soil type | Land-use type | Geological background |
|---|---|---|---|
| As | 0.000 | 0.000 | 0.000 |
| Sb | 0.000 | 0.000 | 0.000 |
| Hg | 0.854 | 0.884 | 0.000 |
| Cd | 0.000 | 0.000 | 0.000 |
| Cu | 0.000 | 0.000 | 0.000 |
| Pb | 0.000 | 0.000 | 0.000 |
| Co | 0.000 | 0.000 | 0.000 |
| Cr | 0.000 | 0.000 | 0.000 |
| Ni | 0.000 | 0.000 | 0.000 |
| Zn | 0.052 | 0.000 | 0.001 |

Qinghai-Tibet area, the bedrock is gradually weathered under the combined influence of climatic conditions, soil type, and geological structure, causing the constant release and accumulation of HMs into the soil (Dung et al., 2013), thus continuously reducing the EC of the soil.

Levels of arsenic in the soil are significantly affected by the geological background, which in this case primarily comprises Paleoproterozoic hornblende rocks and Holocene moraines. The Paleoproterozoic hornblende rock group comprises primarily gray dioritic hornblende sandwiched between black cloud dioritic gneiss, quartz schist, with a small number of marble lenses that may contain As-rich minerals. The Holocene moraine primarily comprises breccias, clasts, and sands, contributing to the enrichment of As. This indicates that this region has a high background level of As. Levels of Hg are substantially affected by geological background. Because Hg is an incompatible element, its spatial distribution is the same as that of the volcanic rocks, indicating that this study area exhibits high background values of Hg. The high levels of Zn, Co, Cr, and Ni, ferrophilic elements, result from the presence of the basic volcanic rock in the area, indicating that their background levels are high in the study area.

Levels of As, Cd, and Ni were significantly affected by soil type, with mountain meadow soil exhibiting the greatest effects. Herders in the region choose productive and well-grown mountain meadow soils as four-season pastures, and overgrazing leads to pasture degradation. Stripping the turf layer away exposes the bedrock and increases the risk of wind and water erosion. This may allow HMs to enter the soil and increase HM activity in the soil, resulting in surface enrichment. Simultaneously, the HM contamination of the soil can arise from animal excretions, which may contain high levels of N, P, and other chemicals. The accumulation of these substances in soil can have long-term effects on soil quality and ecology (Aarons et al., 2009). Overgrazing can lead to a change in community structure from monoculture to complexity to monoculture, thereby affecting ecosystem stability (Li & Zhao, 2005). Overgrazing can damage the soil's microbial community, thus reducing the number and diversity of microorganisms and thereby weakening fixation of HMs and increasing their concentration in the soil (Holt, 1997). Reasonable control of grazing density and frequency, sealing of already degraded mountain meadow soils, replanting of grass seeds, and vegetation restoration are required. The use of manure as a fertilizer to promote crop growth can reduce cultivation costs, increase farmers' income, and promote the development of the local economy.

With the exception of Hg, the other HMs were significantly affected by land-use type. Based on the significance analysis, Cd levels were affected primarily by natural grassland, Cu and Zn levels by dry land, and Sb levels by marshland, suggesting that grazing affects the amount of Cd in the soil. Maintaining the livestock population below the maximum carrying capacity of natural grassland favors grassland vegetation recovery and the ecological protection (Qian et al., 2007). Therefore, it is necessary to limit the number of grazing livestock to avoid overgrazing that can lead to grassland degradation. Most of the arable land on the Tibetan Plateau comprises dryland and irrigated land, and farmers use chemical fertilizers, pesticides, and agricultural plastic films in the process of plowing, substantially increasing the N and P content of the river water within the irrigated portion of the arable land. Nonetheless, the HM content of the study area is still relatively low level (Zhang et al., 2019). To stabilize and enhance the function of the ecological safety barrier on the plateau, it is important to consider the ecological and environmental effects of arable land use (Zhang et al., 2019).

## Dynamic EC forecasting

Considering that HM levels in the soil are in dynamic equilibrium, it is essential to consider the soil's dynamic EC when evaluating EC. The dynamic EC of the 10 HMs in the agro-pastoral area was projected for periods of 20, 30, and 50 years (Eqs. 1 and 3; Fig. 3). Based on these projections, the soils' self-purification capacity in the agro-pastoral areas is expected to be limited, with the EC projected to decline for most of the HMs (Fig. 3). Therefore, it is imperative to develop strategies to prevent and control soil contamination. Even without anthropogenic contributions, the soil's EC only declines as a result of natural input. The rate of decline in EC varied among the HMs, with Hg and Sb declining at rates of 11.5% and 11.3%, respectively. Their ECs are expected to decline rapidly over the next 50 years, with that of Hg projected to reach to a minimum of 0.60 kg/hm$^2$ in 2074, indicating an extremely high risk of surpassing the EC. Furthermore, different soil types exhibit different ECs for different elements. In Yanqi County, the dynamic capacity of Cd in agricultural soils is declining at a rate of approximately 7.98% (Alli, 2020), whereas in agro-pastoral areas it is declining at ca. 3.06%. In pasture in Jinghe County, the dynamic EC of Co is declining at ca. 7.13% (Ma et al., 2016), whereas in the farming and grazing area, it is declining at ca. 4.48%. This is because, in farming areas, land use and chemical fertilizer and pesticide utilization are more concentrated, accelerating the decline in EC. In pastoral areas, economic activities rely primarily on animal husbandry, and the large numbers of livestock consume and trample the grassland resources, leading to environmental problems such as pasture degradation and soil erosion, thus accelerating the decline in dynamic EC. By integrating agriculture and pastoralism, agro-pastoral areas can achieve complementary and optimal use of resources and reduce the pressure on the environment caused by agriculture or pastoralism alone.

## Composited EC evaluation

The separate and composite EC indices for the 10 HMs were visualized (Eqs. 1 and 5; Fig. 4). The mean composite EC index of the soil in the agro-pastoral area was 0.98, indicating an intermediate EC and low associated health risks. Most of the points with low composite EC values were in the eastern portion of the study region, with EC being surpassed in some places. Relative to the other HMs, the soil ECs were lower for As, Cd, Cu, Cr, Ni, and Zn, at 0.96, 0.91, 0.99, 0.97, 0.98, and 0.98, respectively, indicating that they pose a higher risk. Cd exhibited the most substantial contamination and a large CV, suggesting that, in addition to the influence of the geological background, it may also be influenced by agro-pastoralism activities. Owing to human cultivation, fertilization, and overgrazing, the accumulation of these HMs in the soil has increased significantly, leading to local soil contamination. The low CV for Cu indicates that its levels were influenced primarily by geological background. Most of the points with high Cu were located in the southwest of the region. The presence of Cu mining sites near the southwestern part of the study region may be the primary cause of the low enrichment of Cu in the area.

Table 7 presents the EC indices of the 10 HMs by soil type. For each HM, the EC index varied slightly with soil type. PI was highest in the marshland soil (mean, 0.98) and lowest in the alpine meadow soil (mean, 0.95). Marshland soil contains sufficient water and exhibits strong moisture retention, favoring the growth of hydrophytes. This soil is rich in organic matter, highly fertile, and provides nutrients for wetland ecosystems. Marshland soils must be utilized with caution to avoid overexploitation, which can lead to wetland degradation.

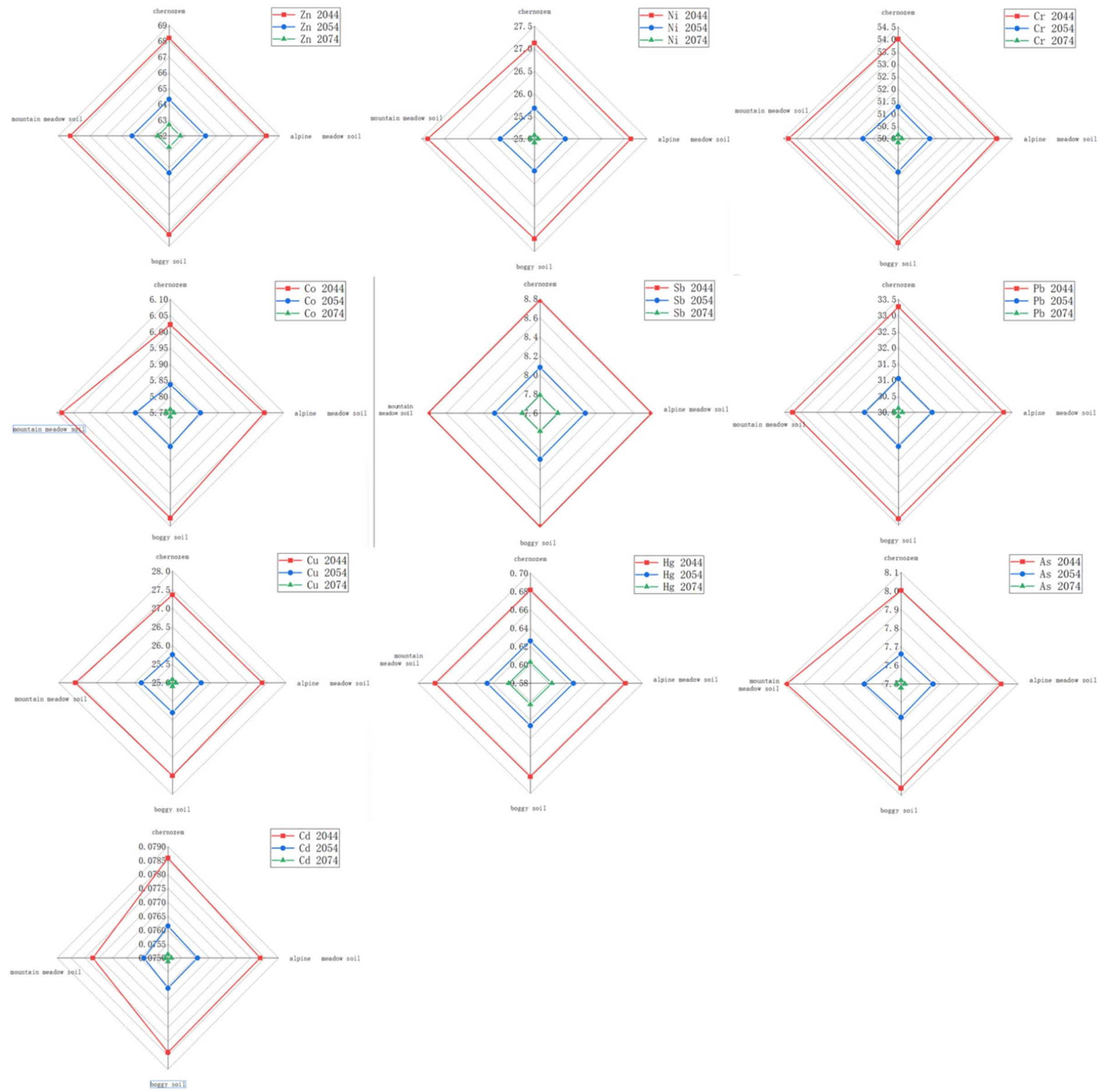


**Fig. 3** Dynamic environmental capacity of the soils of the study area

Alpine and mountain meadow soils are suitable for pasture use because of their high quality and lush pasture grasses. However, to ensure the sustainable pasture utilization, pasture management must be strengthened, and grazing intensity must be controlled to avoid the degradation due to overgrazing. When a pasture is degraded, its vegetation can be restored by appropriately replanting high-quality grass from seed. However, this restoration should be combined with agricultural techniques, such as increasing the application of organic fertilizers and applying appropriate amounts of chemical fertilizers, to promote forage grass growth and improve yields. Soil that exhibits sanding or wind erosion degradation should be immediately replanted with grass seeds, with the emphasis on conserving soil

**Fig. 4** Map of the soils' environmental capacity to accommodate heavy metals

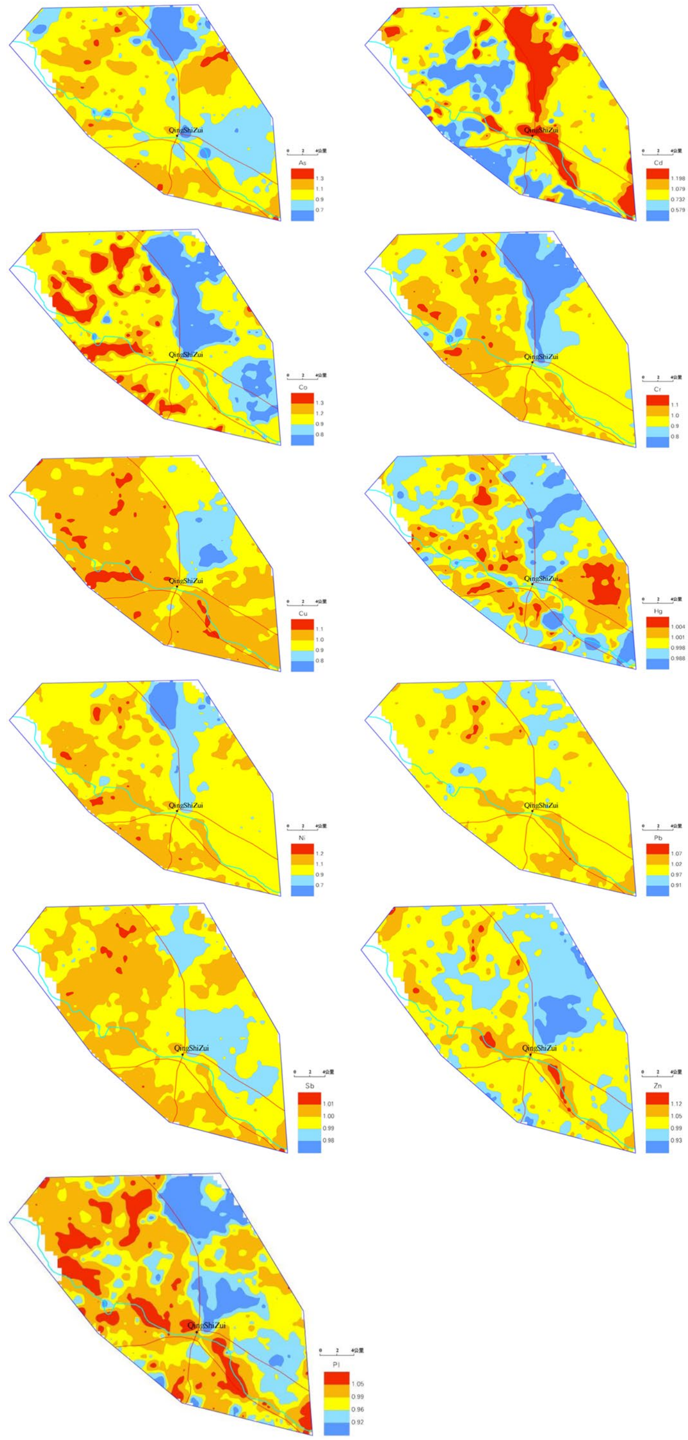

**Table 7** Soil environmental capacity of the study area

| Soil type | As | Sb | Hg | Cd | Cu | Pb | Co | Cr | Ni | Zn |
|---|---|---|---|---|---|---|---|---|---|---|
| Chernozem | 0.95 | 1.00 | 1.00 | 0.94 | 0.98 | 1.00 | 0.83 | 0.97 | 0.97 | 1.00 |
| Alpine meadow soil | 0.97 | 1.00 | 1.00 | 0.88 | 0.99 | 0.99 | 0.80 | 0.94 | 0.98 | 1.00 |
| Marshland soil | 1.06 | 1.01 | 1.00 | 0.89 | 1.04 | 1.01 | 0.75 | 1.02 | 1.01 | 1.01 |
| Mountain meadow soil | 1.11 | 1.01 | 1.00 | 0.71 | 1.07 | 1.00 | 0.73 | 1.04 | 1.08 | 1.01 |
| Mean value | 1.02 | 1.00 | 1.00 | 0.85 | 1.02 | 1.00 | 0.78 | 0.99 | 1.01 | 1.01 |

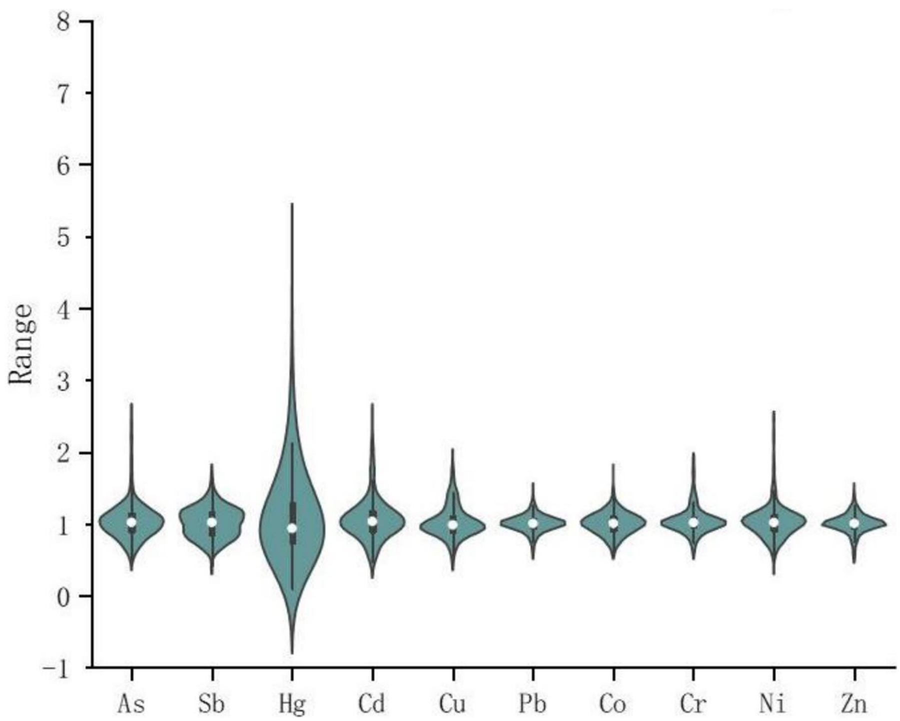


**Fig. 5** Violin and box plots of the heavy metal enrichment factors

and water. Cultivated black calcareous soils need to be scientifically fertilized, and water conservation should be strengthened to increase food production.

## Enrichment factor analysis

Based on Eq. (6), the enrichment factors of the ten HMs in the soil of the agro-pastoral area were calculated (Fig. 5). In the farming and grazing area, Hg (1.23) exhibited the highest enrichment, followed by Cd (1.07), Cr (1.05), As (1.05), Ni (1.04), Cu (1.03), Sb (1.01), Pb (1.01), Co (1.01), and Zn (1.00). Overall, all the HMs were somewhat enriched, indicating that most of the soils were less affected by anthropogenic activities than by other drivers. Hg was significantly affected only by geological background (Table 6). The research region is in the North Qilian metallogenic subzone, where magmatic activity is strong, and basal-ultramafic rocks tend to occur in clusters that are typically produced in association with volcanic rocks. This results in local high background levels of HMs in this area.

## Uncertainty analysis

Research on EC is crucial for elucidating the background levels of HMs in the soil and determining the risk reference values. Here, the local soil background values were established based on the location of the agro-pastoral area, which is somewhat accurate. The risk reference values for As, Cd, Cr, Cu, Co, Hg, Ni, Pb, and Zn are based on the risk screening values for soils with pH values of 6.5–7.5 in the Risk Control Standards for Soil Pollution on

Agricultural Land (GB15618-2018), and that for Sb was based on the risk screening value for antimony (metallic) in the U.S. National Environmental Protection Agency's Regional Screening Level (RSL). Considering that the Qinghai-Tibet region has many unique environmental characteristics, there may be some uncertainty in the risk reference values used.

Little is currently known about the dynamic EC of the soil for HMs in the Qinghai-Tibet area, and a unified local standard has not yet been developed. In the future, more data should be collected and advanced analytical methods applied to develop standards that are compatible with this particular geographic environment. Considering that HMs are in dynamic equilibrium in the soil, dynamic EC can better describe the cumulative dynamics of HMs in the soil. Here, the K value was taken as 0.9 to determine the dynamic EC of the soil.

There was substantial uncertainty in the projected dynamic EC, owing to differences in the residual levels of each HM in the soil. To assess dynamic EC more accurately in future, it will be necessary to incorporate Qinghai-Tibet region-specific processes of soil weathering, changes in soil properties, and HM leaching to generate K values for each HM that are consistent with the situation in this region. The dynamic EC formula, an exponential model, has been extensively accepted and applied; during practical application, the decline in EC gradually slows over the projected period. Therefore, the decline in EC is greater during the 0–10-year period than during the 10–20-year period. This attribute is determined by the structure of the formula; our projections of dynamic EC for the Qinghai-Tibet region may therefore be biased. Further in-depth research is required to improve the dynamic EC equations to more precisely estimate the trend in EC for the Qinghai-Tibet region and to more accurately represent the real situation.

In analyzing the EC of this agro-pastoral area, we comprehensively considered factors such as geological background, land-use type, and soil type. However, the qualities and functions of the soil determine its EC, which is not fixed but varies depending on the soil properties. This variability depends on various factors, including soil type, the nature and type of pollutants present, the plants and soil microorganisms present, objective soil environmental conditions, and human activity. Therefore, to assess the EC of agro-pastoral soils more accurately, future research should comprehensively consider these factors to uncover the specific mechanisms influencing soil EC.

## Conclusions

The present study unveiled the dynamic and static annual ECs of HMs in the soils of the agro-pastoral region on the Tibetan Plateau, indicating that HMs such as Zn, Cr, Pb, Cu, and Ni exhibit larger ECs with relatively lower risks; whereas As, Co, Hg, and Cd display smaller ECs, suggesting relatively higher risks. This revelation underscores the necessity for vigilant monitoring and management of high-risk HMs, especially when their static environmental capacities are small. Given this context, we recommend prioritizing attention towards these high-risk HMs and devising corresponding management strategies to mitigate their potential hazards.

Furthermore, the study outcomes demonstrate that, apart from Zn and Hg, the RECs of other elements are significantly influenced by soil type, predominantly mountain meadow soils. This finding implies that grazing activities significantly impact soil HM content, suggesting that integrating agriculture and pastoralism can alleviate such impacts, thus achieving more efficient land resource utilization and maintaining ecological equilibrium.

Nevertheless, without external interventions, EC will inevitably decrease over time, highlighting the urgency to bolster soil pollution prevention and control efforts. This research assessed the impacts of agro-pastoralism on the soils of the Tibetan Plateau and offered preliminary recommendations for soil pollution prevention and control. However, due to the lack of comprehensive studies on the soil systems of the Tibetan Plateau, further in-depth investigations into other sources of HMs, as well as the effects of soil-forming matrices and parent rocks on these HMs, are necessitated.

To effectively safeguard the ecological environment of the Tibetan Plateau region, conservation measures must be formulated based on the specific characteristics of its regions. Our findings reveal that integrated agro-pastoralism enhances ecosystem stability, reduces the ecological and environmental pressures that might arise from agriculture or pastoralism independently, and stabilizes farmers' incomes. These insights provide scientific support for environmental protection and effective resource utilization in the Qinghai-Tibet region and offer concrete guidance to aid farmers and herders in improving their income. These contributions enrich the understanding of soil EC on the Tibetan Plateau and lay a theoretical groundwork for subsequent research.

**Author Contribution** H. S. Q.: Conceptualization, Methodology, Software, Investigation, Formal Analysis, Funding Acquisition, Resources, Writing—Original Draft; D. L.: Data Curation, Writing—Original Draft; L. Q. Y.: Visualization, Investigation; W. Y. N.: Resources, Supervision; N.Y.: Visualization, Writing—Review & Editing; X. K. L. (Corresponding Author): Conceptualization, Supervision, Writing—Review & Editing.

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**Data Availability** No datasets were generated or analysed during the current study.

**Declarations**

**Conflict of Interest** The authors declare no competing interests.

## References

Aarons, S. R., O'Connor, C. R., Hosseini, H. M., & Gourley, C. J. P. (2009). Dung pads increase pasture production, soil nutrients and microbial biomass carbon in grazed dairy systems. *Nutrient Cycling in Agroecosystems, 84*(1), 81–92. https://doi.org/10.1007/s10705-008-9228-5

Alli, A. (2020). *Environmental Capacity Assessment of Heavy Metals in Farmland Soils in Yanqi Basin, Xinjiang* [Master's Thesis]. Xinjiang Normal University. https://link.cnki.net/doi/https://doi.org/10.27432/d.cnki.gxsfu.2020.000182

Chai, Z., Ren, Y., & Cao, Y. (2022). Modernization measures for sustainable development of alpine grassland husbandry on the qinghai-tibet plateau: an example of countermeasures to pasture degradation. *Journal of Environmental and Agricultural Studies., 3*(3), 72–77.

Chen, H. M., Zheng, C. R., Tu, C., & Zhou, D. M. (2001). Studies on loading capacity of agricultural soils for heavy metals and its applications in China. *Applied Geochemistry., 16*(11), 1397–1403.

Du, J. H., Wang, J., & Wang, X. Z. (2007). Calculation on the soil environmental capacity for heavy metal elements in laoshan scenic spot. *Journal of Environmental Management College of China., 17*(01), 27–30.

Du, H. L., Wang, Y., Wang, J. S., Yao, Y. B., Zhou, Y., Liu, X. Y., & Lu, Y. L. (2021). Distribution characteristics and ecological risk assessment of soil heavy metals in typical watersheds of the qinghai-tibet plateau. *Environmental Science., 42*(09), 4422–4431.

Dung, T. T. T., Cappuyns, V., Swennen, R., & Phung, N. K. (2013). From geochemical background determination to pollution assessment of heavy metals in sediments and soils. *Reviews in Environmental Science and Bio/technology, 12*(4), 335–353. https://doi.org/10.1007/s11157-013-9315-1

Guan, Z. H., Li, X. G., & Wang, L. (2018). Heavy metal enrichment in roadside soils in the eastern Tibetan Plateau. *Environmental Science and Pollution Research International., 25*(8), 7625–7637.

Holt, J. A. (1997). Grazing pressure and soil carbon, microbial biomass and enzyme activities in semi-arid northeastern Australia. *Applied Soil Ecology., 5*(2), 143–149.

Jia, Y., Cui, N. B., Wei, X. P., Gong, D. Z., & Hu, X. (2016). Applicability evaluation of different algorithms for reference crop evapotranspiration in Yangtze River Basin based on inverse distance weighted method. *Transactions of the Chinese Society of Agricultural Engineering., 32*(06), 130–138.

Li, Y. Q., & Zhao, J. B. (2005). Effects of overgrazing on ecological and environmental construction and measurement. *Ournal of Desert Research., 25*(03), 404–408.

Li, W. M., Sun, Z., & Chen, X. Y. (2023a). Evaluation and source of heavy metal pollution in surface soils in typical alpine agricultural areas of qinghai province. *Rock and Mineral Analysis., 42*(03), 598–615.

Li, W., Xie, P., Ding, Z., Yang, B., Zhang, N., Xiong, J., & Lyu, X. (2023). Evaluation and analysis of soil heavy metal characteristics in typical plateau urban green spaces. *Environmental Science & Technology*, 1–14. https://doi.org/10.19672/j.cnki.1003-6504.1341.23.338

Li, Y. (2022). *Risk evaluation, driving factors and management of soil contaminated by heavy metal in regional area* [Master's Thesis]. Zhejiang University. https://link.cnki.net/doi/https://doi.org/10.27461/d.cnki.gzjdx.2022.000826

Lin, M. L., Hu, Z. Q., Peng, W. H., Ye, W. L., Zhang, C. L., Huang, X. R., Chen, S., & Gui, H. R. (2024). Pollution Assessment and Source Apportionment of Heavy Metals in the Surrounding Soil of Typical Mining Areas in Tongling, Anhui Province. *Environmental Science*, 1–14. https://doi.org/10.13227/j.hjkx.202307274

Liu, X. L., Gao, W. H., Wei, T., Dong, Z. W., & Shao, Y. P. (2024b). Distribution characteristics of heavy metals in tibetan plateau surface soils and its significance for source tracing of heavy metal deposition in surrounding glacial areas. *China Environmental Science., 44*(04), 2198–2207.

Liu, K., Li, Y., Yu, H., Zhou, L., Li, H., & Wang, J. (2024). Pollution Characteristics, Risk Assessment and Sources Analysis of Soil Heavy Metals in an Industrial Park of Chongqing. *Environmental Monitoring in China*, 1–10. https://doi.org/10.19316/j.issn.1002-6002.2024.02.09

Ma, H. Y., Yang, X. D., & Lyu, G. H. (2016). Tendency prediction about soil environmental capacity of heavy metals in

Jinghe County in recent 100 years. *Xinjiang Agricultural Sciences., 53*(08), 1521–1532.

Mai, X., Tang, J., Tang, J., Zhu, X., Yang, Z., Liu, X., Zhuang, X., Feng, G., & Tang, L. (2024). Research progress on the environmental risk assessment and remediation technologies of heavy metal pollution in agricultural soil. *Journal of Environmental Sciences., 149*, 1–20.

Marcott, S. A., Shakun, J. D., Clark, P. U., & Mix, A. C. (2013). A reconstruction of regional and global temperature for the past 11,300 years. *Science, 339*(6124), 1198–1201. https://doi.org/10.1126/science.1228026

Qian, S., Mao, L. X., Hou, Y. Y., Fu, Y., Zhang, H. Z., & Du, J. (2007). Livestock car rying capacity and balance between car rying capacity of gr assland with added for age and actual livestock in the qinghai-tibet plateau. *Journal of Natural Resources., 22*(3), 389–397.

Sai, N. G., Qi, J., & Jia, Y. W. (2022). Evaluation of soil heavy metal pollution under different land use patterns in the eastern Qilian Mountains. *Acta Prataculturae Sinica., 31*(10), 99–109.

Su, Y., & j. (2011). A quantitative analysis on grassland animal husbandry of sichuan tibetan region after the liberation. *Animal Husbandry and Feed Science, 3*, 4–41.

Tang, Z., Guan, S. Q., Zhao, M. L., Wang, L., Liu, Y. J., & Fan, Y. B. (2022). Grassland transfer and its income effect: evidence from pastoral areas of the qinghai-tibet plateau. *Land, 11*(11), 2020.

Tian, K., Li, M., Hu, W., Fan, Y. N., Huang, B., & Zhao, Y. (2022). Environmental capacity of heavy metals in intensive agricultural soils: Insights from geochemical baselines and source apportionment. *Science of the Total Environment, 819*, 153078.

Wang, N., & EZIZ, M., & SIDIKJAN, N. (2023). Heavy metals in soils of an urbanization gradient zone in Urumqi City: evaluation of environmental capacity. *Environmental Science & Technology., 46*(12), 1–9.

Wang, G., Yan, X., Zhang, F., Zeng, C., & Gao, D. (2014). Influencing factors of heavy metal concentration in roadside-soil of Qinghai Tibet Plateau. *Acta Scientiae Circumstantiae., 34*(02), 431–438.

Wang, W., Ji, X., Abakumov, E., Polyakov, V., Li, G., & Wang, D. (2022). Assessing sources and distribution of heavy metals in environmental media of the tibetan plateau: A critical review. *Frontiers in Environmental Science, 10*, 874635.

Wang, D., Li, J., & Yang, H. (2024a). Evaluation of heavy metal pollution characteristics, sources and ecological risks of soil around mining areas in Luanchuan County. *Luoyang City. Environmental Chemistry., 43*(10), 1–14.

Wang, H. G., Wang, Q. T., Yang, P., Ge, X. W., Gu, X. Y., Zhang, J. H., Yu, X. C., Mao, F. S., & Chen, L. (2024). Spatial Distribution, Pollution Assessment, and Source Analysis of Soil Heavy Metals Under Different Land Use Types in the South Area of Laizhou Bay. *Environmental Science*, 1–18. https://doi.org/10.13227/j.hjkx.202312225

Wu, J. F., Wang, H. M., & Li, Y. T. (2023). Progress of the Theory, Calculation Methods and Application of Soil Environmental Capacity. *Ecological Economy*, *39*(06), 182–188+227. https://kns.cnki.net/kcms2/article/abstract?v=Xhw-7KfLOFktaBAqOE6U7lR69LRBu6oU_tXPz6-BcEWf2cE3JdX72KMf4aa1_UUDwWLxTblxT_JehrQgj5mlsOSxgqQzv9cqiKmJ3rHIJvESRsO7GuBF4NXT-v7aZcCDFGPl2gdamlRGdQlmEYCkVA==&uniplatform=NZKPT&language=CHS

Wu, F. (2017). *The research of heavy metals environmental capacity in main soil types of Yellow River Delta* [Master's Thesis]. Shandong University. https://kns.cnki.net/kcms2/article/abstract?v=Xhw-7KfLOFkIzkSN2Ac10T8uhPs66YqVfPizV2SO29hHNnX2j5HjQGqCh4CYVl8dkY7UFb958iQvDh9zMyfpqFt5UP0xg7KmO7AuwJtjACFAdZYdglAQvsia7oyA_5mLrsKTLEPKtLw8OdJk0VSudQ==&uniplatform=NZKPT&language=CHS

Wu, J. F. (2023). *Research on Regional Soil Environmental Capacity based on Risk Constraint - Taking a Typical Copper Smelting Area in Watershed as an Example* [Master's Thesis]. Chinese Research Academy of Environmental Sciences. https://link.cnki.net/doi/https://doi.org/10.27510/d.cnki.gzhky.2023.000082

Yang, A., Wang, Y. H., Hu, J., Liu, X. L., & Li, J. (2020). Evaluation and Source of Heavy Metal Pollution in Surface Soil of Qinghai-Tibet Plateau. *Environmental Science., 41*(02), 886–894.

Yang, J., Xiong, J., Li, W., Huang, R., Yang, B., & Xie, P. (2024). Soil heavy metal risk evaluation and source analysis of Zhaxigang wetland in Tibet based on PCA-APCS-MLR Model. *Chinese Journal of Ecology, 43*(6), 1–13.

Yao, T. D., Thompson, L. G., Mosbrugger, V., Zhang, F., Ma, Y., Luo, T., Xu, B., Yang, X., Joswiak, D. R., Wang, W., Joswiak, M., Devkota, L. P., Tayal, S., Jilani, R., & Fayziev, R. (2012). Third Pole Environment (TPE). *Environmental Development., 3*, 52–64.

Zhang, J., & Duan, P. (2023). Influence of terrain factors on optimal order distance of Inverse Distance Weighted (IDW). *Scientia Geographica Sinica., 43*(07), 1281–1290.

Zhang, Y., Liu, L., Wang, Z., Bai, W., Ding, M., Wang, X., Yan, J., Xu, E., Wu, X., Zhang, B., Liu, Q., Zhao, Z., Liu, F., & Zheng, D. (2019). Spatial and temporal characteristics of land use and cover changes in the Tibetan Plateau. *Chinese Science Bulletin., 64*(27), 2865–2875.

Zhao, B., Wang, L., Wei, Y., Zhu, W., Cao, H., Zhang, H. (2024). Status and Prospect of Standard Systems for Agricultural Soil Heavy Metal Contamination Prevention and Control in China. *Research of Environmental Sciences*. https://doi.org/10.13198/j.issn.1001-6929.2024.01.06


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