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Characterization of contamination levels of heavy metals in agricultural soils using geochemical baseline concentrations

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Abstract

Purpose It is currently very difficult to accurately evaluate the soil contamination by heavy metals (HMs) attributed to the unavailability of local geochemical background values (LGBVs). This study was performed to establish the geochemical baseline concentrations (GBCs), as an alternative for LGBVs to use for HM pollution assessment of agricultural soil.

Materials and methods GBCs of the HMs selected were determined using the cumulative frequency distribution curves (CFDCs). GBCs were then used to pursue the HM soil pollution and associated ecological risks, via calculation of geo-accumulation indices (I_{geo}), pollution load indices (*PLI*), as well as potential ecological risk indices (*RI*).

Results and discussion As to the soil investigated, the GBCs of Ni, Zn, Pb, and Cr were 29.34 mg/kg, 45.54 mg/kg, 21.81 mg/kg, and 33.65 mg/kg, respectively. I_{geo} values ranged from − 4.58 to 0.33 (Ni), from − 2.46 to 2.14 (Zn), from − 5.32 to 0.77 (Pb), and from − 3.83 to 0.96 (Cr), suggesting that the region was not polluted by these HMs. *PLI* values ranged from 0.08 to 2.45 with an average of 1.02. 49.6% of soil samples had the *PLI* values > 1.0, indicating that some of the soil may be moderately contaminated by HMs. The *RI* values of selected HMs were < 150, indicating a low potential ecological risk. Principal component analysis (PCA) implied Zn, Pb, and Cr were mainly sourced from parent (geological) materials, as well as agricultural activities, atmospheric deposition, etc., depending on the element.

Conclusions The present study illustrates the necessity of the characterization of GBCs at a regional scale, allowing for more accurate assessment of soil contamination by HMs. We hope that this will eventually lead to further development of better environmental management practices for agricultural soil polluted by HMs.

Keywords Agricultural soil · Ecological risk · Geochemical baseline concentrations · Heavy metals

1 Introduction

Due to our reliance on agricultural production, human health and ecological security are closely linked to soil contamination (Burges et al. 2015; He et al. 2017). The increasing demand for food coupled with growing population pressures has led to increasingly intensive development of farmland (de Vries et al. 2013; Garnett 2014; Tilman and Clark 2015). As a result, there have been elevated levels of heavy metals (HMs) in soils, mainly attributed to the significant utilization of fertilizers, pesticides, etc. (Tian et al. 2017; Zhou et al. 2017, 2019). HM accumulation in agricultural soils can have detrimental effects on soil ecosystems and can potentially threaten human health via food chain accumulation (Liu et al. 2013; Wongsasuluk et al. 2014; Singh and Prasad 2015; Yu et al. 2016; Li et al. 2018). Thus, agricultural soil

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contamination by HMs has become an increasingly prevalent environmental problem (Shao et al. 2016; Fan et al. 2017; Lawal et al. 2017). In fact, thousands of studies have evaluated the risks of agricultural soil contamination by HMs to human and environmental health (Marrugo-Negrete et al. 2017; Yang et al. 2017) by comparing the data with local geochemical background values (LGBVs).

LGBVs are references used to identify the differences between the natural and anthropogenically influenced concentrations of elements/compounds in potentially contaminated soil (Matschullat et al. 2000; Karim et al. 2015). Due to both the natural variability and anthropic influence, it has been almost impossible to determine true target LGBVs (Wei and Wen 2012; Karim et al. 2015). In this regard, the term "geochemical baseline concentrations" (GBCs) was coined and suggested to be used to represent the true background concentrations (Wei and Wen 2012). GBCs are often considered as the boundary dividing the anomalous values and the LGBVs. And, the data below the GBCs are thought to be the LGBVs while those above to be anomalous (Teng et al. 2009; Wei and Wen 2012). From a regulatory perspective, the GBCs provide an opportunity to identify whether the enrichment of elements has arisen. In general, the GBCs are the important benchmarks of soil pollution evaluation and soil quality management such as protection and legislation (Levitan et al. 2014; Zhang et al. 2014). The methods that have been used to determine the GBCs of the elements in the soils (Gałuszka 2007; Teng et al. 2009; Wei and Wen 2012) consist of statistical procedures generally employed as the concentration distributions are normal or log-normal, normalization methods for which reference materials such as Al and Li are used to normalize the data, substitution methods using the values of soil samples of deep layers as the GBCs, and integration methods combining two or more methods. The specific description for the estimation of GBCs can be available from Wei and Wen (2012), Gałuszka (2007), and Zhang et al. (2014). And, cumulative frequency distribution curves (CFDCs), an important method used by statistical procedures, have been used widely to build up the GBCs of the elements in soils. The projects and studies focused on GBCs in Europe, the USA, and other countries were conducted. This work had found out the GBCs of HMs in soils and sediments. However, current research in GBCs for agricultural soils, especially in terms of HM contamination, is seriously lacking. Therefore, the characterization of GBCs in local soils has been a priority to assess the agricultural soil HM contamination (Tian et al. 2017).

This study is focused to characterize the GBCs of HMs in agricultural soils, to determine the pollution levels and potential ecological risks of HMs selected, and to discuss the sources of the HMs targeted.

2 Materials and methods

The place involved (approximately 60 km^2) was located in the north of Huainan City (116° 21′ 21″ N~117° 11′ 59″ E, 32° 32′ 45″~33° 00′ 24″ N), Anhui Province, China. China is divided into North China and South China by the Qinling Mountains and Huai River. The study area, located at the south of Huai River, is part of the Jianghuai Hills zone. The GBCs of HMs in the soil from the Jianghuai Hills zone have not been studied yet. Geologically, it belongs to the Huainan terrace plain and the Huai River stratigraphic division zone. Shungeng Mountain appears in the northern part with the direction from the east to west. The area is completely covered by the Quaternary with the development from Archaeozoic to Mesozoic and Cenozoic. The thickness of loose layer is increased from the east and south to the west and north with the range of 0 m to 700 m, controlled by the paleotopography. Sedimentary facies change from the upper fluvial facies to the lacustrine facies. The vertical zoning of the aquifer is obvious with the transition from the upper HCO_3-type fresh water to the deep Cl-Na-type water. The groundwater flows from the north and west to the south and east, which is basically consistent with the surface water flow. The climate is characterized by monsoon humid weather in temperate zone. On average, the annual temperature, rainfall, and humidity were 15.7 °C, 970 mm, and 76.9%, respectively. The study area is predominantly covered by paddy soil but also contains zones with brown-red and yellow-brown soils. In addition to the stone mining activity for Shungeng Mountain, over several decades during twentieth, this area was mainly subjected to intense agricultural activities, predominantly producing rice, wheat, and rapeseed. During the sampling process, 112 samples (20 cm depth) were obtained (Fig. 1). The sampling, sample preparation, and element determination were operated based on the method from Niu et al. (2015).

In this study, CFDCs were used to determine the GBCs of HMs. Specifically, a curve with decimal coordinates was plotted by arranging all the element concentrations on the *X*-axis and their corresponding cumulative frequency on the *Y*-axis. Generally, the curve had two inflections: the lower inflection point (the upper limit of natural origin concentrations) and the higher inflection point (the lower limit of abnormal concentrations). This indicated HM concentration was above background level, which may or may not be related to the anthropogenic activities (Matschullat et al. 2000; Karim et al. 2015). However, occasionally, an approximately linear curve was obtained, which means that the concentration data is within the baseline range. To determine the baseline value, a linear regression method described by Wei and Wen (2012) was employed, with modifications (i.e., $p < 0.05$ and $R^2 > 0.95$ were used in this study instead of $p < 0.05$ and $R^2 > 0.90$).

To evaluate soil pollution by the HMs selected, geo-accumulation indices (I_{geo}), single-factor pollution indices

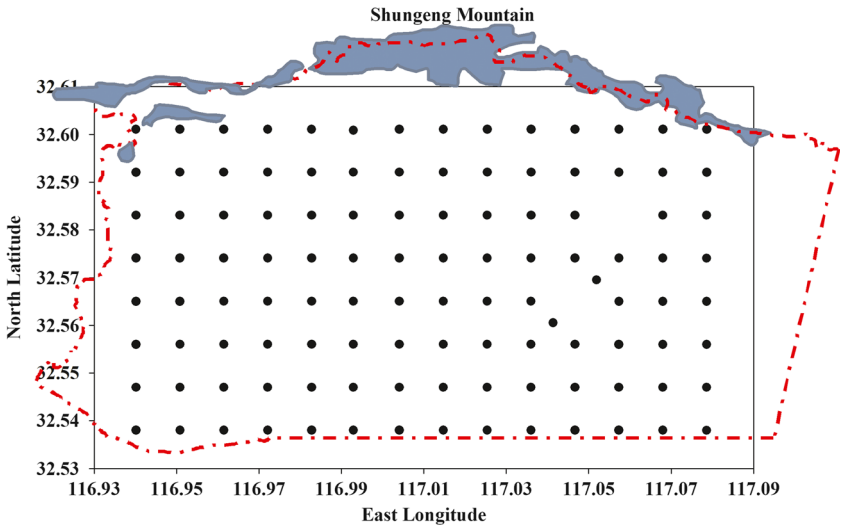

Fig. 1 Locations of the sampling sites

(*PI*), and the pollution load indices (*PLI*) are obtained using Eqs. (1) to (3), respectively:

$$I_{\text{geo}} = \log_2\left(\frac{C_i}{1.5B_i}\right) \quad (1)$$

$$PI_i = C_i/B_i \quad (2)$$

$$PLI = (PI_1 \times PI_{i2} \times PI_3 \times \cdots \times PI_n)^{\frac{1}{n}} \quad (3)$$

where C_i is the soil content of element i, B_i is the geochemical background value (GBV) of i in local soil, and n is the number of HMs investigated. The levels of HM pollution classified based on I_{geo} and *PLI* are summarized in Table 1.

Table 1 Contamination level or ecological risk of soil by HMs classified based on I_{geo}, *PLI*, and *RI*

Item	Critical range	Contamination level/ecological risk
I_{geo}	$I_{\text{geo}} \le 0$	Practically unpolluted
	$0 < I_{\text{geo}} \le 1$	Unpolluted to moderately polluted
	$1 < I_{\text{geo}} \le 2$	Moderately polluted
	$2 < I_{\text{geo}} \le 3$	Moderately to heavily polluted
	$3 < I_{\text{geo}} \le 4$	Heavily polluted
	$4 < I_{\text{geo}} \le 5$	Heavily to extremely polluted
	$I_{\text{geo}} > 5$	Extremely polluted
PLI	$PLI \le 1$	Unpolluted
	$1 < PLI \le 2$	Moderately polluted
	$2 < PLI \le 5$	Heavily polluted
	$PLI > 5$	Extremely heavily polluted
RI	$RI \le 150$	Low ecological risk
	$150 < RI \le 300$	Moderate ecological risk
	$300 < RI \le 600$	Considerable ecological risk
	$RI > 600$	Very high ecological risk

Potential ecological risk index (*RI*) was calculated by summing individual potential risk factor (E_i) using Eq. (4) (Ke et al. 2017; Liao et al. 2017; Tian et al. 2017)

$$RI = \sum_{i=1}^{n} E_i \quad (4)$$

$$E_i = T_i \times PI_i \quad (5)$$

where T_i is the biological toxicity factor of metal i (defined as Zn = 1, Ni = Pb = 5, Cr = 2, and Cd = 30) (Suresh et al. 2012). The ecological risk of HMs with respect to *RI* is listed in Table 1 (Ekere et al. 2017).

3 Results and discussion

3.1 Soil HMs

The following levels of HMs were detected in the soils: Ni = 0.45–61.10 mg/kg, Zn = 5.82–579.03 mg/kg, Pb = 0.16–70.95 mg/kg, and Cr = 1.10–132.17 mg/kg. Statistical analysis indicates that the datasets were normally distributed for Ni and Cr and log-normally distributed for Zn. Based on the average values, Zn (60.20 mg/kg) had the highest concentration followed by Cr (44.21 mg/kg), Ni (29.34 mg/kg), and Pb (24.02 mg/kg). The coefficient of variation value (CVV) is employed to show the variability degree of the soil concentrations of HMs: CVV < 20% with low variability, 20% ≤ CVV ≤ 50% with moderate variability, CVV > 50% with high variability, and CVV > 100% with exceptionally high variability (Xiao et al. 2015). In general, for HMs, the CVV governed by the anthropogenic sources is higher than that derived from the natural sources (Han et al. 2006; Guo et al. 2012; Karim et al. 2015).

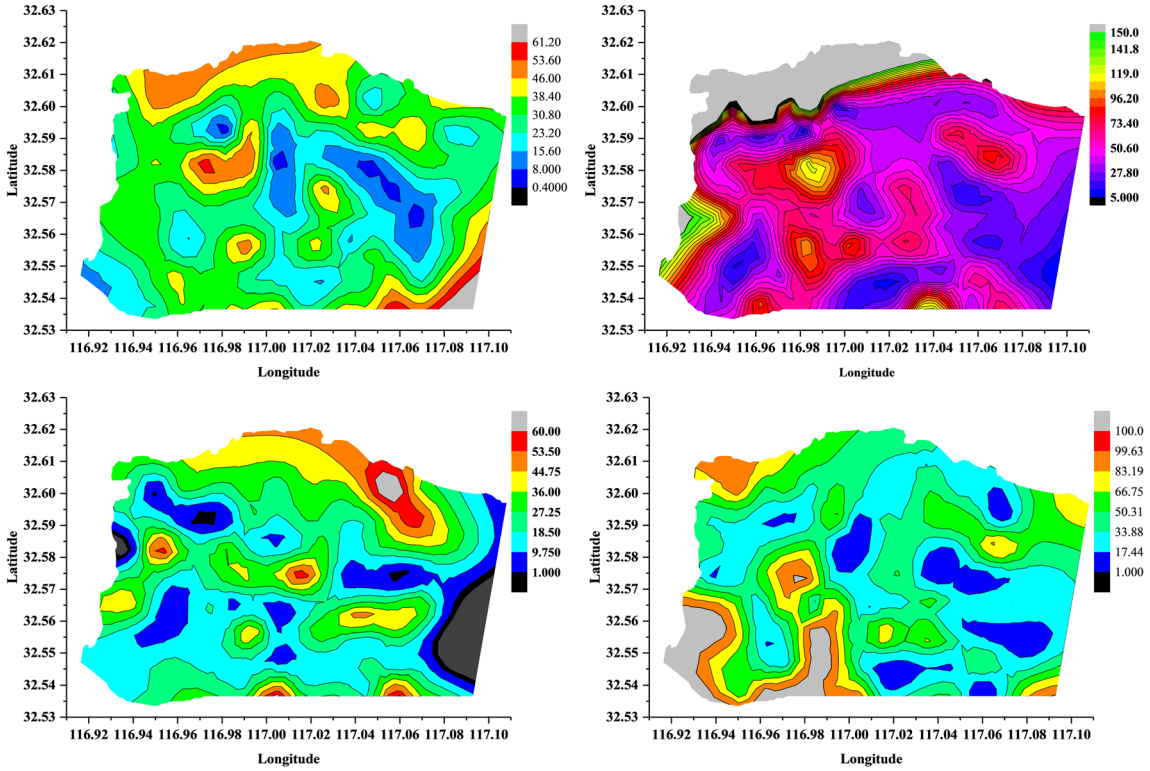

Fig. 2 Spatial distribution of HMs

The CVVs of HMs investigated were as follows: Zn (73%) > Cr (32%) > Pb (16%) > Ni (14%). As shown in Fig. 2, the highest contents of HMs selected, except for Cr, were found at a site near Shungeng Mountain. This is likely due to the previous stone mining activity. The highest Cr concentration appeared in the southeast, potentially from the discharge of vehicles from the He-Huai-Fu highway.

To comprehensively understand the HM pollution status of the agricultural soils investigated, we compare our result to the data from other regions of China (Table 2). In China, the HM concentrations in agricultural soils changed remarkably depending on sample site, ranging from 15.5 to 40.9 for Ni, from 42.2 to 414.0 for Zn, from 6.74 to 159.25 for Pb, and from 7.17 to 285.8 for Cr, respectively. The highest levels of Zn and Pb occurred in Zhuzhou, likely due to Pb-Zn mining activity that occurred in that area. Interregional comparisons show that Zn, Pb, and Cr in the soils had lower levels than those reported by the literature ($p < 0.05$, independent-samples t test), while across all the soils, Ni was similar ($p > 0.05$). However, HM concentrations varied remarkably within this region, indicating that due to the spatial variation of soil heterogeneity, HM concentrations in selected soils may not represent the actual pollution level. Therefore, it is almost impossible to evaluate HM soil pollution solely based on their concentrations, so geological background concentrations should be considered in risk assessment practices (Tian et al. 2017).

Pearson's correlation analysis was conducted among the HMs selected (Table 3). In summary, there was no significant relationship between Zn, Pb, and Cr ($p < 0.001$). Therefore, the main sources of Zn, Pb, and Cr were different. However, Ni had a significantly positive correlation with the other three HMs ($p < 0.001$), indicating that Ni in the sampled soil came from at least three sources.

Principal component analysis (PCA) has also been employed by many studies to pursue the sources of trace element in the soils (Chabukdhara and Nema 2012; Cai et al. 2015). To supplement our analysis, the result of PCA is used to determine the sources of the HMs. The results of the PCA for the selected HMs are shown in Tables 4 and 5. The agricultural production was the only predominant activity, and there were almost no factories discharging the solid waste or wastewater containing HMs in this region. Moreover, the previous small-scale stone mining activity for Shungeng Mountain could only affect the accumulation of HMs in the soils near the mountain. However, there were several factories and coal-fired power plants with gas emissions not far away. In this regard, based on the result of PCA and the literatures, the sources of HMs in the region can be easily identified.

It is found that the first three principal components (PCs) could explain the 85.92% variance, according to

Table 2 HMs in agricultural soils from this study and other areas in China (mg/kg)

Area	Ni	Zn	Pb	Cr	Source
This study	29.34	60.2	24.02	44.21	
Bazhong	–	–	24.36	–	Li and Chen (2014)
Beijing	–	81.1	18.48	75.74	Liu et al. (2005)
Chengdu	–	227	77.27	59.5	Liu et al. (2006a)
Cixi	–	88.91	28.06	–	Liu et al. (2006b)
Daye	25.8	159	43.7	60.7	Du et al. (2015)
Dongguan	21.8	76.63	65.83	38.86	Dou et al. (2008)
Fuyang	21.85	42.20	16.80	–	Chen et al. (2013)
Fuxin	22.05	45.76	–	–	Xu et al. (2007)
Gansu	–	–	21.44	38.82	Li et al. (2008)
Ganzhou			33.31	15.01	Song (2008)
Guangzhou	–	162.6	58	64.65	Li et al. (2009)
Hailun	25.7	61.7		58.50	Chen et al. (2015)
Hainan	15.51	52.17	48.01	22.67	Zhao et al. (2007a)
Huangshi	–	74.4	82.6	–	Wei (2009)
Jiamusi	25.2		22.7		Li and Li (2012)
Jiangmeng	–	–	6.74	7.17	Qiu et al. (2017)
Jinghe	–	–	22.44	44.21	Zheng (2008)
Kashgar	–	100.1	34.3	109.6	Zha et al. (2016)
Kunshan	31.08	105.93	30.48	87.73	Chen and Pu (2007)
Laiwu	36.10	74.54	28.81	74.21	Yu et al. (2016)
Nantong	–	–	19.47	51.41	Gong et al. (2014)
Ningbo	–	–	48.05	42.11	Zhou et al. (2016)
Pingdingshan	25.9	71.6	32.8	56.4	Chen et al. (2014)
Qianan	21.98	50.33	20.54	52.05	Wang et al. (2006)
Shanghai		106.2	90.7	285.8	Meng et al. (2008)
Shenyang	–	–	22.00	37.81	Li (2010)
Suzhou	–	–	28.16	53.38	Shen et al. (2010)
Taian	37.89	65.56	15.37	60.04	Li et al. (2011)
Taihang	25.04	69.96	18.8	57.77	Yang et al. (2009)
Taiyaun	29.74	90.76	27.87	74.10	Liu et al. (2015)
Wuxi	–	112.9	46.7	58.6	Zhao et al. (2007b)
Xiangyang	30.2	81.6	23.8	60.1	Zhao (2014)
Xuzhou	–	149.68	56.2	–	Liu et al. (2006c)
Yangzhou	38.5	98.1	35.7	77.2	Huang et al. (2007)
Zhangye	40.9	86.79	27.98	77.42	Zhu and Yang (2014)
Zhengzhou	–	–	17.11	60.67	Liu et al. (2007)
Zhuzhou	–	414	159.25	87.55	Li and Chen (2016)
China soil	26.9	74.2	26.0	61	Tian et al. (2017)
World soil	70.0	29	27.0	59.5	Tian et al. (2017)
Crustal average	70	20	15	100	Tian et al. (2017)

their eigenvalues, which are over 1.0 after varimax. Moreover, the first two extracted factors had the eigenvalues of > 1, while the third gradually became > 1. The component matrix rotated indicates that Ni and Cr fell in F1, Pb in F2, and Zn in F3.

F1 (30.3% of the total variance) could be regarded as a lithogenic component, because the variabilities of Ni and Cr were likely to be governed by the parent materials (Sun et al. 2013). This result arose in consistency with the geostatistical analyses presented in Fig. 2, showing that in the study area, there was no significant anthropic contribution to Ni and Cr, based on their spatial distribution. The result is in accordance with the previous studies that parent materials, together with pedogenic processes, predominantly affect the distribution of Ni and Cr (Rodríguez et al. 2008), while anthropic contribution including fertilizers, manure, and limestone is typically limited (Facchinelli et al. 2001).

Table 3 Pearson's correlation analysis between select HMs

Element	Ni	Zn	Pb	Cr
Ni	1.000			
Zn	0.254**	1.000		
Pb	0.308**	0.045	1.000	
Cr	0.395**	0.188	0.077	1.000

**$p < 0.01$, significant (two-tailed)

Table 4 Total variance explanation

Component	Total variance explained								
	Component initial eigenvalues			Extraction sums of squared loadings[a]			Rotation sums of squared loadings[b]		
1	1.601	40.026	40.026	1.601	40.026	40.026	1.212	30.312	30.312
2	1.009	25.217	65.243	1.009	25.217	65.243	1.173	29.328	59.639
3	0.827	20.680	85.923	0.827	20.680	85.923	1.051	26.284	85.923
4	0.563	14.077	100.000						

[a] Extraction method: principal component analysis

[b] Rotation method: varimax with the Kaiser normalization

And, F2 explained the 29.3% variance. Pb is generally sourced from industrial fumes, vehicle exhaust, pesticides, and sewage sludge (Francouría et al. 2009; Lu et al. 2012). Usually, Pb, which is from car exhaust, cannot extend appreciably to the place 30 m away from the roadside (Smith 1976); therefore, vehicle discharge could not be considered a significant input in this area. However, the exhaust fumes can move over longer distance via wind action (Facchinelli et al. 2001), so atmospheric deposition could be one of the significant contributors to soil Pb enrichment (Davis et al. 2001). Considering the location of the study area, industrial fumes, coal-burning exhaust, and domestic waste are likely to be the most important contributors of soil Pb.

F3 (26.3% of the total variance) included Zn. Zn in the soils can be from lithogenic sources, with the forms of various soluble (e.g., nitrates and chlorides) and insoluble (e.g., sulfides, phosphates, carbonates, silicates) salts. In addition to the pedogenic processes, Zn enrichment is also close to the utilization of fertilizers containing Zn (Sun et al. 2013; Moreno-Jiménez et al. 2016). Hence, the concentrations of Zn found in the soils could be changed depending on the specific human activities. Forty-six percent of the samples had the Zn concentrations higher than the corresponding GBV (Niu et al. 2015), indicating that Zn was sourced from both the parent materials and applied fertilizers.

3.2 GBCs of selected HMs in soil

Due to anthropogenic influence, in recent practice, it has been impossible to obtain a natural background for elemental composition based on the pristine geochemical characteristics of soil. As a result, the geochemical baseline has been suggested as an alternative to use as a natural comparison (Karim et al. 2015). The GBCs of HMs selected were characterized and demonstrated in Fig. 3.

CDFCs of Zn, Pb, and Cr had only one inflection, respectively. Ni had no inflection, implying that concentrations of Ni in soil were not human induced. GBCs of HMs are presented in Tables 4 and 5. For the soil sampled, Ni, Zn, Pb, and Cr had baseline concentrations of 29.34 mg/kg, 45.54 mg/kg, 21.81 mg/kg, and 33.65 mg/kg, respectively. A comparison of GBCs of selected HMs with respect to regional land use type is carried out and shown in Table 6. According to the literatures, the GBCs of Ni, Zn, Pb, and Cr arose with the range of 18.7395.5 mg/kg, from 32.8 to 218.2 mg/kg, from 14.96 to 58.2 mg/kg, and from 12.9 to 340.7 mg/kg, respectively, depending on the regional land use type. Even in the same land type like agricultural soil, the GBCs in soils also varied greatly. Overall, the baseline value of Ni established for the present study is within the general range of 20–30 mg/kg reported by the literatures, suggesting that this value is comparable to the other values, whereas the GBCs of Zn, Pb, and Cr are generally smaller

Table 5 Component matrices of the first three principal components for selected HMs in soil

Element	Component matrix			Rotated component matrix		
	First component (F1)	Second component (F2)	Third component (F3)	F1	F2	F3
Ni	0.812	0.130	− 0.081	0.563	0.531	0.290
Zn	0.562	− 0.425	0.690	0.091	0.018	0.982
Pb	0.465	0.815	0.102	− 0.030	0.943	− 0.030
Cr	0.640	− 0.383	− 0.578	0.941	−0.047	0.040

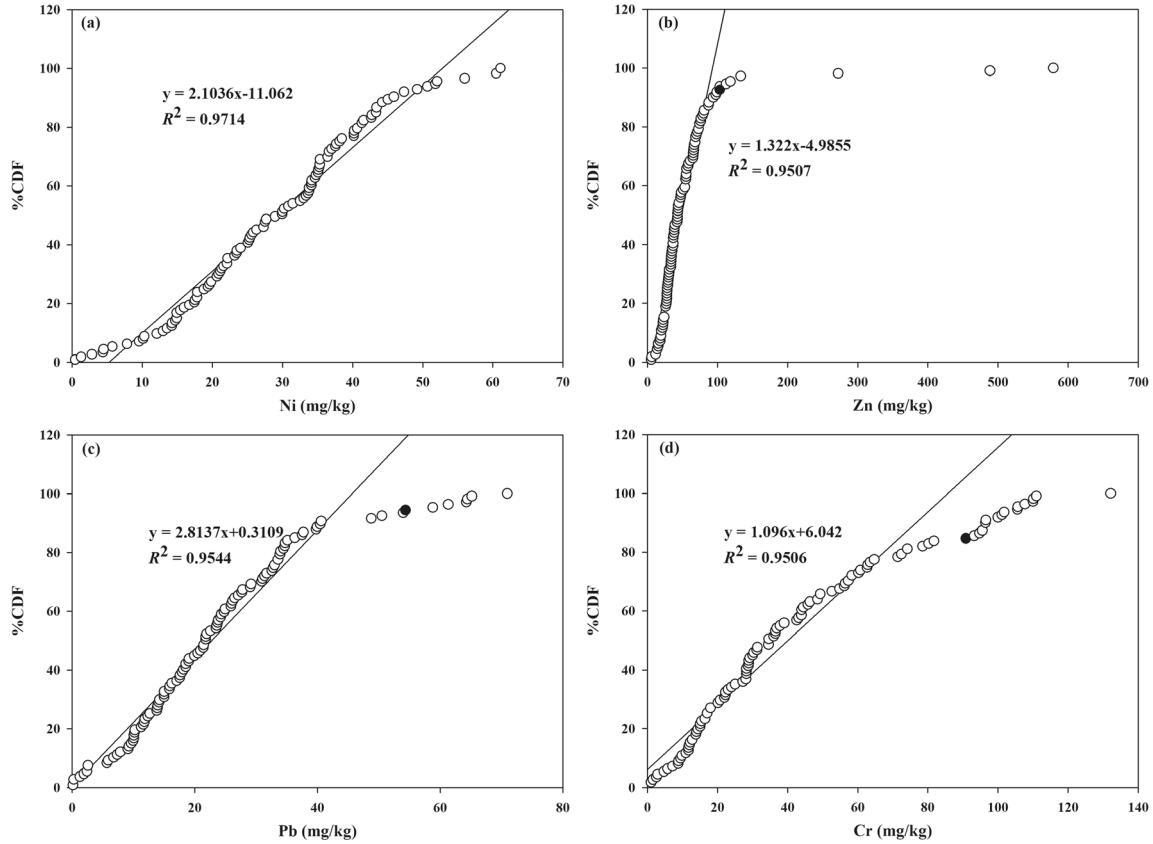

Fig. 3 CDFCs of HMs. **a** Ni. **b** Zn. **c** Pb. **d** Cr

than those from the other regions. And, the variation of GBCs of Ni, Zn, Pb, and Cr is in accordance with the change of their contents discussed above. This work suggests that regional GBCs for HMs towards specific land use type should be proposed in order to easily facilitate the identification of soil contamination and risk assessment. However, the LGBVs for these HMs were reported as 25.74 mg/kg for Ni, 80.81 mg/kg for Zn, 30.47 mg/kg for Pb, and 64.93 mg/kg for Cr (Niu et al. 2015). Thus, the GBCs derived are significantly lower than the LGBVs from the literature, indicating that ecological risk assessments based on the previously reported background values may underestimate risk. However, due to regional differences, these results of HM GBCs may not apply the different environments (Tables 4 and 5). Therefore, a more concerted effort needs to be made to determine if the GBCs for HMs can be applied across the different geographic regions so that we can accurately evaluate the risks from HMs to human and environmental health.

3.3 I_{geo} of soil HMs

I_{geo} values of HMs were calculated and are given in Fig. 4. The I_{geo} of Ni, Zn, Pb, and Cr ranged from −4.58 to 0.33, from −2.46 to 2.14, from −5.32 to 0.77, and from −3.83 to 0.96, respectively. All the mean I_{geo} values were lower than 0. Hence, most of the studied area had not been contaminated by Ni, Zn, Pb, and Cr. However, some sampled areas were subjected to moderate HM contamination. It

Table 6 Comparison of estimated GBCs with the data from the literature (mg/kg)

Soil type	Ni	Zn	Pb	Cr	Data source
Agricultural soils	29.34	45.54	21.81	33.65	Present study
	28	83	28	36	Micó et al. (2007)
	35.18	94.54	31.27	82.06	Lu et al. (2018)
	62.7	218.2	70.6	340.7	Wu (2016)
	95.5	90.2	37.3	–	Ding (2018)
	18.70	52.62	48.39	146.21	Fan et al. (2014)
	24.5	62	28.19	66.41	Li et al. (2012)
	29.9	78.44	26.18	66.78	Wu et al. (2018)
Urban soils	31.2	161	58.2	63.1	Jarva et al. (2014)
	–	123.03	56.23	12.9	Karim et al. (2015)
Greenhouse soils	21.95	47.83	14.96	47.76	Tian et al. (2017)
Coal mining area	–	32.80	20.47	42.67	Zakir et al. (2017)
Metal mining area	45.67	76.92	28.53	281.58	Teng et al. (2002)

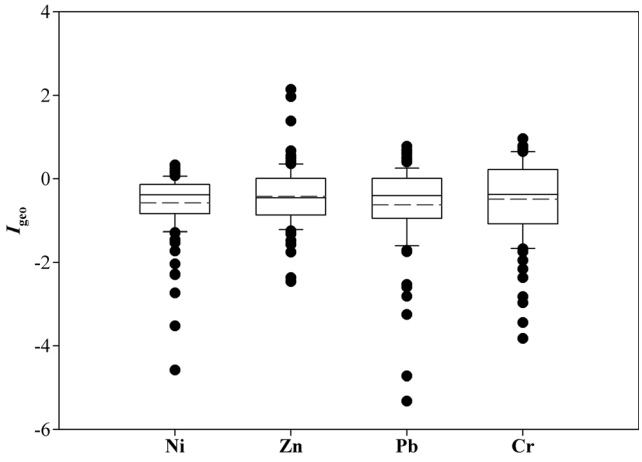

Fig. 4 I_{geo} of HMs

should be highlighted that I_{geo} just can show the accumulation degree of the HMs selected by comparing the contents obtained with GBCs whereas it cannot give the implication of ecological risk. As a result, the condition with small I_{geo} but high ecological risk is likely to arise as GBCs are high.

3.4 *PLI* and ecological risk of soil HMs

The *PI* values of Ni, Zn, Pb, and Cr ranged from 0.02 to 2.08, from 0.13 to 12.71, from 0.01 to 3.25, and from 0.03 to 3.93, respectively (Table 7). Overall, Zn had the highest *PI* value followed by Cr and Pb and then Ni. The average *PI* values were mainly in the range of 1 and 2, suggesting that the study soil had been moderately polluted by individual elements. However, some areas in the region were subjected to high HM contamination ($PI > 2$) by Zn (11.7% of samples), Pb (9.3% of samples), and Cr (22.5% of samples). Ni *PI* values mainly remained < 1.0, meaning that Ni pollution had not taken place in the study area. The *PLI* of the site ranged from 0.08 to 2.45, with an average of 1.02. However, 49.6% of samples had the *PLI* values > 1.0, suggesting that the soil may have been moderately contaminated by HMs. Soil with higher *PLI* values was mostly sampled around the mountain, and most contamination was attributed to significant accumulation of Zn in soil. However, the *PLI* of this region was 0.87 with the elemental *PLI* of 0.84 for Ni, 0.98 for Zn, 0.80 for Pb, and 0.92 for Cr, which means that overall there was no pollution by HMs.

RI is employed commonly to find out the ecological risks of trace element. As shown in Fig. 4, all the samples had the *RI* values less than 150, indicating a low potential ecological risk from the HMs, and the HMs, despite the huge amount of fertilizers and pesticides, had been used in agricultural practices.

4 Conclusions

In the region investigated, the concentrations of Zn, Pb, and Cr were generally lower than those reported by the literature. The GBCs of Ni, Zn, Pb, and Cr were 29.34 mg/kg, 45.54 mg/kg, 21.81 mg/kg, and 33.65 mg/kg, respectively. The I_{geo} values indicated that most of the study area was not contaminated by the elements selected, the *PLI* values suggested a moderate HM contamination, but the *RI* values indicated a low potential ecological risk. Therefore, the variation between these different risk evaluation methods should be further investigated. Moreover, further study towards the depth profile of GBCs of HMs should also be carried out in pursuing the contamination levels, sources, and ecological risk of this region well in the future. Ni and Cr appeared in the soils mainly from the parent materials. Industrial fume and coal burning exhaust were likely the significant sources of Pb. Zn was sourced not only from the parent material (pedagogic processes) but also from the fertilizer application. For environmental risk assessment, we commonly assume that the reference soil used in HM characterization has not been influenced by human activity. Indeed, as it has been impossible to obtain a *natural soil* (with quantified GBCs), the commonly used HM background values are obtained via characterization of sample soil from a *reference site*. However, in this study, we have established that this practice may be resulting in underestimation of HM soil contamination and thus underestimating the risk HMs are currently posing to human and environmental health. Therefore, it is urgent to establish GBCs for agricultural soils at the regional scale, to enhance ecological risk management practices, and to improve regulatory controls.

Table 7 *PLI* values and potential ecological risk of selected HMs in soil samples

Metal	*PI*				*PLI*	E_i				*RI*
	Ni	Zn	Pb	Cr		Ni	Zn	Pb	Cr	
Min.	0.02	0.13	0.01	0.03	0.08	0.08	0.13	0.04	0.07	1.36
Max.	2.08	12.71	3.25	3.93	2.45	10.41	12.71	16.27	7.86	30.20
Average	1.00	1.32	1.10	1.31	1.02	5.00	1.32	5.51	2.63	14.09

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