Pol. J. Environ. Stud. Vol. 34, No. 2 (2025), 1945-1955
DOI: *10.15244/pjoes/185990* ONLINE PUBLICATION DATE: 2024-09-13

Original Research

Assessment of Soil Heavy Metals and Intake Risk in Typical Se-Rich Areas, Xianyushan Area, China

Bing Zhao, Duozhi Tu, Bowen Kong, Ting Yue, Xiaoguo Wu*

School of Ecology and Environment, Anhui Normal University, Wuhu, Anhui, 241002, PR China

Received: 29 December 2023
Accepted: 12 March 2024

Abstract

The concentrations of heavy metals (HMs) and the associated pollution risk and dietary exposure risk were assessed by collecting 154 soil samples and 6 food types from the Xianyushan area. The concentrations of Zn, Cd, and As in the soil were 1.2, 3.4, and 2.8 times the background values, and the percentages of samples exceeding the background value were 47%, 65%, and 96%, respectively. Concentrations of Cd, Cr, and As exhibited significant variations among the three land use types, which were potentially attributed to soil properties, agricultural amendments, and other contributing factors. Based on the integrated results from spatial distribution analysis, correlation analysis (CA), and principal component analysis (PCA), it can be inferred that these HMs mainly originated from soil parent materials. Pollution risk assessment indicated that Cd and As were the dominating soil pollutants. Although HM levels in different food types were lower than the limit defined in China's national food safety standards, the total estimated daily intakes (EDIs) of Cd and As exceeded permissible tolerable daily intake (PTDI) values as well as reference doses (RfD). The target hazard quotient (THQ) values indicated that Cd, As, and Hg present lower risks from dietary intake compared to other HMs, especially, since cereals present a high risk of HM intake and need further attention.

Keywords: heavy metals, soil, risk assessment, food type

Introduction

Soil heavy metal (HM) pollution is a significant global issue, and assessment of soil HM pollution has emerged as a crucial research topic for scholars in China and other countries [1, 2]. In China, soil HM pollution is a serious issue, with copper (Cu), zinc (Zn), lead (Pb), cadmium (Cd), chromium (Cr), mercury (Hg), and arsenic (As) being the primary pollutants according to the National Soil Pollution Survey Report released in 2014 [3]. The mining and smelting activities associated with nonferrous metal ores such as Pb, Zn, and Cu, along with atmospheric deposition, industrial emissions, fertilization, and other factors, have resulted in severe HM contamination across various regions in China [4, 5]. The accumulation of soil HMs through plant consumption significantly impacts human health. Therefore, studying the pollution status, spatial distribution, and ecological effects of HMs in soil holds immense value for guiding agricultural production and safeguarding human health.

Index evaluation methods are widely employed to evaluate soil HM pollution [6]. The single-factor index, Nemerow's index, and potential ecological risk index

*e-mail: wuxg1983@ahnu.edu.cn;
Tel.: +86-553-5910198;
Fax: +86-553-5910198

are used to calculate a comprehensive pollution index and assess the ecological risk associated with soil HM pollution [7, 8]. In recent years, geographic information system (GIS) technology has been effectively used to convert monitored point source data into point source data maps describing soil HM pollution evaluations alongside distribution maps complemented by tables and graphs. This technology has greatly enhanced researchers' ability to quantitatively assess and visualize spatial-temporal variations of soil HM [9, 10]. Dietary intake serves as the most critical pathway for HMs from contaminated soil entering the human body, therefore, analysis of residual levels of HMs present in food can elucidate human health risks faced by humans [11, 12]. The estimated dietary intake (EDI) and target hazard quotient (THQ) methods are commonly employed to evaluate HM risk from dietary exposure, which primarily focuses on noncarcinogenic health risks and cancer risks associated with HMs in food [13].

The Xianyushan area, located in Anhui Province, was the third naturally occurring Se-enriched area in China. According to a previous study, mean concentrations of Se in soil and rock samples of this region were 3.60 mg/kg and 15.1 mg/kg, respectively, and the Se intake of residents was estimated to be 261.2 µg/day [14]. Previous investigations on plants and soils have also found that Cd is more abundant in this region [15, 16]. However, little attention has been given to evaluating other HMs pollution and potential health risks. To improve the HMs pollution data, a study in Xianyushan was conducted by collecting 154 soil samples and 6 types of food. The objectives were: (1) to investigate the concentrations and spatial distribution of Cu, Zn, Pb, Cd, Cr, Hg, and As in soil; (2) to assess soil HM pollution and ecological risks; and (3) to evaluate the risk of HMs intake for residents. These findings provide the essential groundwork for sustainable utilization of selenium-rich soil resources and the safeguarding of residents' health in this region.

Material and Methods

Sampling and Sample Treatment

The location of the study is between 117°08'E-117°38'E and 29°50'N-30°15'N, within an approximate total area of 1918.4 km^2, with forest coverage exceeding 90%; making it a Se-enriched area in China, as depicted in Fig. 1. A total of 154 soil samples (0-20 cm) were collected and divided into three types: cultivated soil (paddy soil, n = 36), forest soil (yellow-brown soil, n = 54), and tea garden soil (yellow soil, n = 64). During soil surface sample collection, top debris was carefully removed first, and soil from a depth of 0-20 cm away from the surface was continuously obtained. At each sampling point, a radius of 100 meters was considered, and 5 points were selected to mix the same amount of soil, and the required number of samples was taken after mixing by the quartering method [3].

Food samples were collected and divided into vegetables (potato, n = 10; pepper, n = 12; cabbage, n = 11; radish, n = 11; eggplant, n = 11; cowpea, n = 13), cereal (rice and corn, 1 Kg for each sample), meat (pork, n = 3; beef, n = 3), fish (n = 21), poultry (chicken, n = 3; duck, n = 3) and eggs (chicken egg, n = 6; duck egg, n = 7). Vegetable samples were collected from the residents' vegetable gardens. Cereal samples were collected by plum blossom dot method and snake

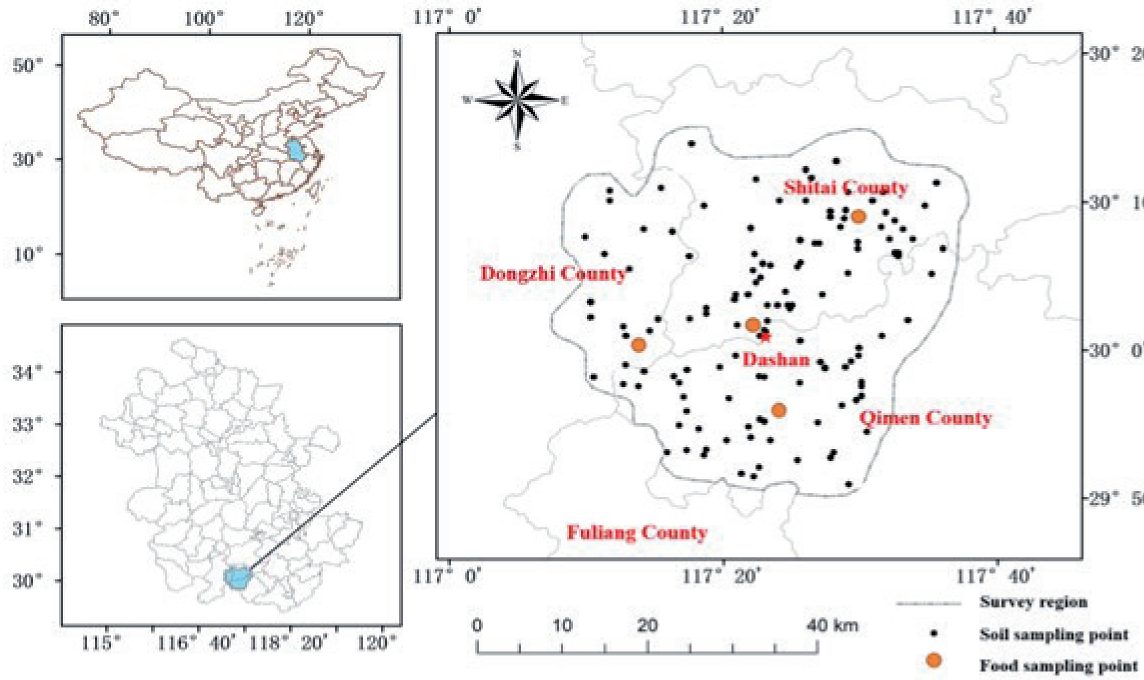

Fig. 1. Sampling locations in Xianyushan, China.

method according to farmland area and terrain. Fish samples were collected by fishermen from the river. Meat, poultry, and egg samples were purchased from local farmers. The fish, poultry, and meat were weighed in the laboratory and the fresh weight of the muscle sample was crushed and ground before being frozen and stored in a centrifugal tube for testing. Eggs were weighed for fresh weight, shelled, homogenized, and frozen [13].

Sample Analysis and Quality Control

The analysis of HMs in samples was determined using established methods [13, 17, 18]. The samples were precisely weighed and placed in a polytetrafluoroethylene (PTFE) dissolving bottle, followed by digestion with a mixed acid solution. Inductively Coupled Plasma Optical Emission Spectrometry (ICP-OES, Agilent ICP 700-ES, Agilent Technologies Inc., US) was used for the analysis of HMs. Standard samples of twigs and leaves (GBW07603, GSV-2), meat (GBW08552, Pork), and soil (GBW07405, GSS-5) were used for quality control purposes and to evaluate the validity of the analytical process. The recoveries ranged from 91% to 104%. Data analysis was performed with SPSS 20.0. All spatial mapping activities were conducted within the Arc Geographic Information System (ArcGIS 10.3).

Calculation of the Pollution Index, EDI, and THQ

The values of the single-factor pollution index, Nemerow's synthetic pollution index, and potential ecological risk index were calculated based on the elemental concentrations in the soil samples.

The formula for calculating the single-factor pollution index is as follows [8]:

$$P_i = Ci/Si$$

where *Pi* represents the pollution index of HM elements at each sampling point *i*; *Ci* is the measured value of HM elements at each sampling point *i*; *Si* is the evaluation standard of *i* HM element. In this study, the screening values of HM elements in soil Environmental Quality and Soil Pollution Risk Control Standard for Agricultural Land (Trial) (GB15618-2018) were used as evaluation criteria.

The formula of Nemerow's synthetical pollution index is as follows [8]:

$$P_n = \sqrt{(P_{imax}^2 + P_{iave}^2) \times 0.5}$$

where *Pn* represents the comprehensive pollution index of sampling point *i*; *Pimax* is the maximum value among all single-factor pollution index of HM elements at sampling point *i*. *Piave* is the average value among all single-factor pollution indices of HM elements at sampling site *i*. All parameters are dimensionless in these calculations.

The formula of the potential toxicity response index for a single HM is as follows [7]:

$$E_r^i = T_r^i \times Pi$$

where E_r^i is the response coefficient for the toxicity of a specific HM. This formula reveals the hazards to human and aquatic ecosystems caused by HMs, reflecting their toxic levels and ecological sensitivity to HM pollution.

The potential health risks of HMs were assessed by calculating the values of EDI and THQ for HM, which involved multiplying the mean elemental concentration in the food samples by the average aquatic products consumed per individual in China. The formula for EDI, THQ, and TTHQ are as follows:

$$\text{EDI} = (\text{MC} \times \text{DC})/\text{BW}$$

where EDI (μg/kg/day BW) represents the estimated daily intake; MC is the mean contaminant content (μg/g ww); DC is the daily aquatic product consumption for the Chinese population (g/day), as reported by the National Bureau of Statistics of China (NBSC); BW is the average human body weight (70 kg for an adult person) [13].

$$\text{THQ} = \left[\frac{\text{EFr} \times \text{ED} \times \text{FIR} \times \text{MC}}{\text{RfD} \times \text{BW} \times \text{AT}}\right] \times 10^{-3}$$

$$\text{TTHQ} = \text{THQ1} + \text{THQ2} + \cdots \text{THQn}$$

where THQ is the target hazard quotient; EFr is the exposure frequency (365 days/year); ED is the exposure duration (70 years) equivalent to an average human lifetime; FIR is the food ingestion rate (g/day/person); MC is the element content in samples on fresh wet weight basis (mg/kg); RfD is the oral reference dose (mg/kg/day), available only for Cu, Zn, Pb, Cd, and Se, which represent an estimate of the daily exposure over a lifetime without an appreciable risk of deleterious effects; BW is average body weight for an adult (70 kg); AT is the averaging time for non-carcinogens (assuming 70 years). THQ is classified into five categories: no significant risk: THQ≤1; low risk: 1<THQ<9.9; moderate risk:10<THQ<19.9; high risk: 20<THQ<99; serious risk: ≥100. Summing up individual food type's THQ values and expressed as total THQ (TTHQ) [13].

Results and Discussion

HM Concentration, Source, and Risk Assessment in Soil

Concentration of HMs in Soil

The concentrations of HMs in the soil in Xianyushan are shown in Table 1. The mean concentrations of HMs in descending order were Zn>Cr>Pb>Cu>

Table 1. Concentrations of HMs in the soil of Xianyushan area.

Index	Concentration (mg/kg)						
	Cu	Zn	Pb	Cd	Cr	Hg	As
Min	2.7	13	9.2	0.001	12	0.001	3.2
Max	76	339	136	1.9	182	0.79	69
mean	23	94	26	0.33	57	0.06	31
SD (±)	11	51	22	0.41	36	0.10	14
Coefficient of variation	0.51	0.54	0.85	1.2	0.63	1.7	0.43
Background value of China [19]	23	74	26	0.1	61	0.1	11
The number of points exceeded the background value	24	73	32	100	30	17	148
Percentage of points out of background value (%)	16	47	21	65	20	11	96
Screening value (GB15618-2018) [20]	100	250	120	0.3	200	2.4	30
Control value (GB15618-2018) [20]	-	-	700	3.0	1000	4.0	120

As>Cd>Hg. The mean concentrations of Zn, Cd, and As were 1.2, 3.4, and 2.8 times the background value for soil HMs in China respectively, with percentages exceeding the background value being 47%, 65%, and 96%, respectively [19]. Although Cd and As slightly exceeded the screening value but remained lower than the control value compared to the soil environmental quality control standard (GB15618-2018) [20], indicating varying degrees of HM pollution (values exceeding the background values) in the Xianyushan area where Cd and As posed high risks compared to other HMs. Moreover, the variation coefficients of Cd and Hg exceeded 1, indicating that the concentration and dispersion of Cd and Hg in the soil of the study area were relatively high and likely influenced significantly by anthropogenic activities [6].

A comparison of pH, organic matter (OM), and HM concentrations among three land use types of forest soil, tea garden soil, and cultivated soil were presented in Table 2. The concentration of Cd was significantly higher in cultivated soil and forest soil compared to tea garden soil ($p<0.05$), while Cr and As exhibited significantly higher levels in cultivated soil than those in tea garden soil ($p<0.05$). Previous studies have demonstrated significant variations among different land use types due to water contamination, physical and chemical properties of soil, microbial activities, crop absorption, agricultural fertilization, as well as human activities [21].

Source and Influence Factor Analysis of HMs in Soil

The spatial distribution of HMs in soil was analyzed using the inverse distance weighting method by ArcGIS, as shown in Fig. 2. The distribution patterns of high and low values of HMs were found to be inconsistent, indicating multiple sources of pollution rather than a single source. Correlation analysis results for HMs in different land use types are presented in Table 3. Significant positive correlations ($p<0.05$) were observed between Zn-Cr, Cr-Hg, and Cd-Pb in tea garden soil, and significant positive correlations ($p<0.05$) were also found between Cd-As and Cr-As in cultivated soil, moreover, an extremely significant positive correlation ($p<0.01$) was identified between Hg and Pb levels. Additionally, forest soil exhibited significant positive correlations ($p<0.05$) among Cr-Pb, Cr-Hg, Pb-As, Pb-Cd, and As-Hg. These correlation analysis findings indicated that the mentioned HMs may have similar sources across all three land use types, based on previous studies [17, 18]. Principal component analysis was employed to further identify the sources of HMs in soil [22]. Three principal components with a cumulative variance contribution rate exceeding 70% for each land use type, and these PCA results were consistent with

Table 2. Values of pH, organic matter, and HMs in soil of different land-use types.

	Tea garden soil (n = 64)	Cultivated soil (n = 36)	Forest soil (n = 54)
pH	6.7±0.26a	6.7±0.23a	6.7±0.25a
OM (%)	2.1±0.48b	2.5±0.57a	2.0±0.57b
Cu (mg/kg)	20±9.4a	24±11a	22±13a
Zn (mg/kg)	77±46a	111±50a	85±51a
Pb (mg/kg)	25±22a	22±11a	30±28a
Cd (mg/kg)	0.25±0.34b	0.35±0.45a	0.35±0.41a
Cr (mg/kg)	49±16b	68±44a	53±37ab
As (mg/kg)	21±13b	32±13a	32±15a
Hg (mg/kg)	0.06±0.12a	0.05±0.03a	0.08±0.12a

Different letters (a and b) of the same indicator indicate significant differences ($p<0.05$).

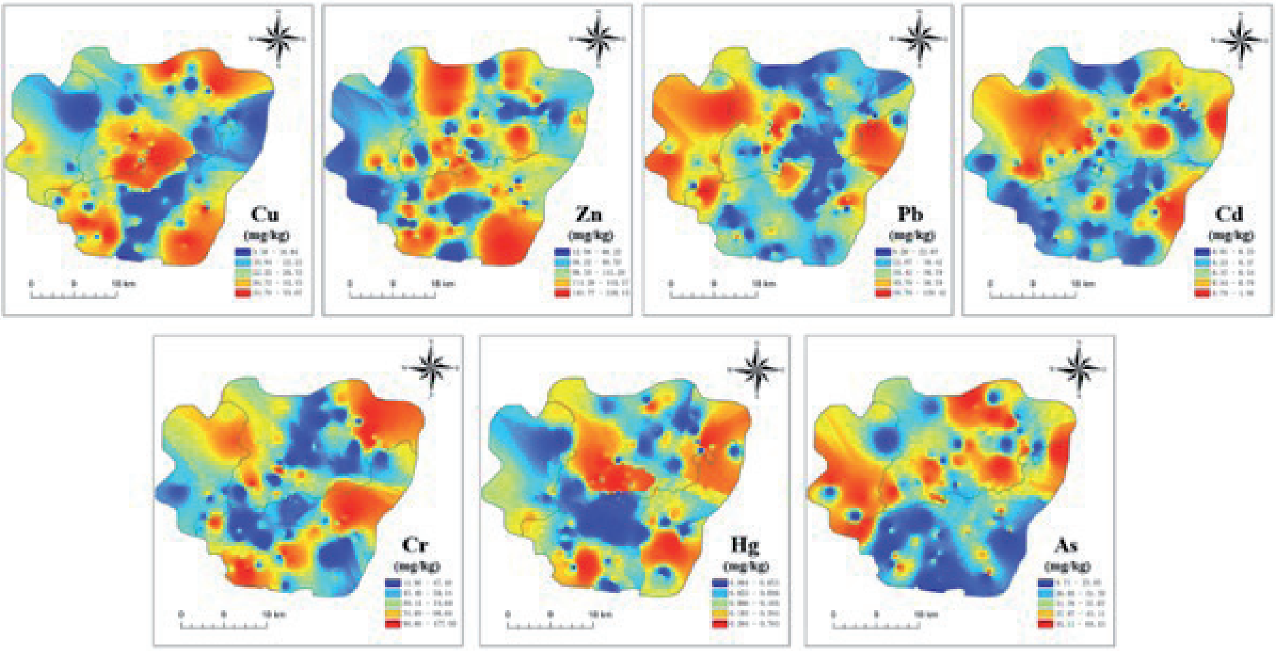

Fig. 2. Spatial distribution of HMs in Xianyushan.

those obtained from elemental correlation analysis as shown in Table 4. Based on both PCA and correlation analysis, it can be inferred that HMs exhibiting high loads and strong correlations within the same principal component likely share common origins, furthermore, it is evident that environmental background factors along with human activities (domestic waste, irrigation, fertilization, and pesticides) contribute significantly to HMs presence within soils [23]. Notably, there exist variations regarding the sources of soils from different land use types due to diverse influencing factors such as irrigation, fertilization methods, and plant species composition [24]. Forest soil is a soil type that remains minimally influenced by human activity, therefore, the presence of HMs mainly originates from a local environmental background. However, Tang et al. (2015) [25] have suggested that atmospheric deposition plays a significant role in the accumulation of Cd and Pb. Previous studies have indicated that the long-term growth of trees leads to the absorption and enrichment of HMs from soil, resulting in higher HM concentrations in surface forest soil [26]. Tea garden soil and cultivated soil are more susceptible to fertilization due to long-term tillage practices. Liao et al. (2021) [27] discovered that organic fertilizer and phosphate fertilizer are major sources of Pb and Zn in agricultural soils, while Cd primarily originates from phosphate fertilizer in Jinhua City, China. Hu et al. (2020) [28] suggested that substances added during agricultural production such as fertilizers, pesticides, and sewage irrigation serve as important sources of Cr, Cd, Hg, and As contamination in farmland across Zhejiang province, China.

In addition, no significant correlation was observed between pH and the selected HMs in all land use types, indicating that pH does not play a significant role in influencing the distribution of these eight elements. However, a positive correlation ($p<0.05$) was found between Cu, Hg, Pb, and OM in tea garden soil, cultivated soil, and forest soil, respectively, indicating that the distribution of Cu, Hg, and Pb was controlled by OM. Generally, OM has dual effects on the fate of HMs: it can enhance HMs mobility by promoting their movement and absorption by plant roots, while also potentially immobilizing them through metal-organic matter interactions [29, 30]. Ning et al. (2017) [31] reported a higher accumulation of Zn, Cd, and Cr in soil due to the long-term application of organic fertilizer. Organic manure used as an important source of OM in agricultural land contains substantial concentrations of trace metals, such as Pb, Cu, Cr, and Zn, which are fed to animals for health improvement as trace metal supplements [18].

Evaluation of Soil Environmental Quality

The pollution index method has been widely used to evaluate soil environmental quality, employing distinct boundaries to differentiate and quantify the extent of soil contamination [32, 33]. In the study area, there were a total of 3, 3, 21, and 71 contaminated sites with Zn, Pb, Cd, and As, respectively. Among these sites, three Zn and Pb sites exhibited slight pollution levels, and the site number of seriously polluted, moderately polluted, and slightly polluted sites for Cd were 1, 8, and 12, respectively. Regarding As contamination, three sites were moderately polluted, and sixty-eight sites showed slight pollution levels. These results showed that Cd and As are the primary HM pollutants followed by Zn and

Table 3. Pearson correlation coefficients for HMs in soil.

	pH	OM	Cr	Pb	Cu	Zn	Hg	As	Cd
Tea garden soil									
pH	1								
OM	-0.07	1							
Cr	0.09	-0.04	1						
Pb	-0.14	-0.10	-0.22	1					
Cu	0.30	0.37*	0.08	0.02	1				
Zn	-0.18	0.18	0.38*	0.05	-0.13	1			
Hg	-0.15	0.02	0.41*	-0.04	0.13	0.32	1		
As	0.01	0.01	0.15	0.06	0.26	-0.20	0.10	1	
Cd	-0.22	-0.18	0.03	0. 41*	-0.18	0.07	0.13	0.01	1
Cultivated soil									
pH	1								
OM	-0.06	1							
Cr	0.05	-0.15	1						
Pb	0.26	0.09	-0.24	1					
Cu	-0.05	0.16	-0.05	-0.10	1				
Zn	-0.29	-0.22	0.07	-0.04	0.16	1			
Hg	0.08	0.34*	-0.28	0.42**	0.08	-0.03	1		
As	0.03	0.27	0.36*	0.12	-0.09	-0.11	-0.11	1	
Cd	0.22	0.12	0.21	-0.04	-0.02	-0.09	0.09	0.38*	1
Forest soil									
pH	1								
OM	0.02	1							
Cr	0.05	-0.03	1						
Pb	-0.25	0.05	0.31*	1					
Cu	-0.05	0.01	-0.01	-0.13	1				
Zn	0.06	0.05	-0.21	-0.13	0.21	1			
Hg	-0.11	0.04	0.30*	-0.15	0.03	0.07	1		
As	0.05	0.11	-0.01	0.27*	0.04	0.15	0.33*	1	
Cd	-0.09	0.32*	0.04	0.28*	0.02	0.14	0.05	0.06	1

*Correlation is significant at the 0.05 level (2-tailed); **Correlation is significant at the 0.01 level (2-tailed)

Pb, however, Cu, Cr, and Hg did not contribute to any pollution. The spatial distribution of soil HM risk based on the single-factor pollution index method is shown in Fig. 3. In terms of pollution levels of each HM, the proportion of As pollution above the "precaution" level was higher than that of Pb and Cd, so it shows more serious pollution. However, single-factor analysis could not fully reflect the comprehensive HM pollution in the study area. Nemerow's synthetic pollution index was used to evaluate the risk of soil contamination, and the corresponding results are presented in Fig. 3 [8]. Out of the total data points, sixty-eight points (44%) were classified within the safety domain, while 41 points (27%) fell into the precautionary domain. Additionally, 45 polluted points accounted for 29 % of all data points, among which 42 were slightly polluted (27%) and 3 were moderately polluted (1.9%). Overall, slightly polluted sites accounted for a significant majority with a proportion of 93%, whereas moderately polluted sites only accounted for a mere fraction of 6.7% among all

Table 4. Principal component analysis.

Element	Tea garden soil			Cultivated soil			Forest soil		
	Principal component			Principal component			Principal component		
	1	2	3	1	2	3	1	2	3
Cr	0.64	0.24	-0.10	0.69	-0.11	-0.12	-0.57	0.65	-0.15
Pb	-0.35	-0.01	0.77	-0.17	0.00	0.77	-0.35	0.71	-0.07
Cu	-0.41	0.56	-0.03	0.31	0.58	-0.19	0.58	-0.03	-0.57
Zn	0.65	-0.33	0.23	0.42	0.57	-0.22	0.67	0.16	0.10
Hg	0.77	-0.20	0.29	-0.13	0.23	0.76	0.19	-0.43	0.76
As	0.01	0.79	-0.02	0.62	-0.46	0.14	-0.45	0.18	0.75
Cd	0.13	-0.39	0.73	0.78	-0.15	0.08	0.76	0.22	-0.37
Eigenvalues	2.4	1.7	1.5	2.4	2.1	1.6	2.5	2.2	1.8
Contribution rate (%)	33	23	21	28	24	19	31	27	22
Cumulative contribution rate (%)	33	56	77	28	52	71	31	58	80

Table 5. Comparison with the standard of heavy metals in food.

Element	Cereal (mg/kg)		Vegetable (mg/kg)		Fish (mg/kg)		Meat and Poultry (mg/kg)		Egg (mg/kg)	
	Limit	Xianyushan	Limit	Xianyushan	Limit	Xianyushan	Limit	Xianyushan	Limit	Xianyushan
Cu	-	1.1	-	0.15	-	1.1	-	5.4	-	2.5
Zn	-	20	-	1.3	-	37	-	81	-	70
Pb	0.20	0.17	0.30	0.02	1.0	0.20	0.20	0.13	0.20	0.10
Cd	0.20	0.14	0.20	0.02	0.10	0.04	0.10	0.08	0.05	0.04
Cr	1.0	0.35	0.50	0.25	2.0	0.76	1.0	0.32	-	0.30
As	0.50	0.26	0.50	0.20	-	0.41	0.50	0.28	-	0.29
Hg	0.02	0.02	0.01	0.001	-	0.03	0.05	0.04	0.05	0.03

*The limit data refers to the limit of contaminants in food according to national Food Safety Standards (GB 2762-2017) [20].

contaminated locations, and no severely polluted sites were identified. Consequently, it can be inferred that Xianyushan soil exhibited some level of HM risk, however, the extent of contamination remained relatively low.

The study area did not exhibit any HMs with a high or serious ecological hazard level according to the result of the single potential ecological risk index (Fig. 4). Amongst all HMs, Cd was found to cause greater ecological damage. Specifically, there were 3 points classified as having a high ecological hazard level, 13 points classified as having a moderate ecological hazard level, and 138 points classified as low ecological hazard level. These accounted for 2.0%, 8.4%, and 90% of the total points, respectively. Furthermore, the calculation results of the comprehensive potential ecological risk index demonstrated that the comprehensive potential ecological risk posed by HMs in the study area was at a low level (Fig. 4).

Risk Assessment of HMs and Their Dietary Intake via Food

Benchmarking Analysis of HMs in Food

Food samples were categorized into six types according to the classification of the food safety national standard-contaminant limits in food (GB 2762-2017) [20]. Additionally, the safety level of all types of food was analyzed by comparing the limit index, and the results are presented in Table 5. Based on these comparisons, it was observed that the concentration of each element in vegetables, fish, meat, and eggs remained below the national safety limit, indicating their suitability for consumption. However, cereals exhibited that mercury (Hg) concentration matched the limit while the concentrations of other elements were lower than the safety limit. Therefore, the Hg concentration in cereals posed some dietary exposure risk within this region.

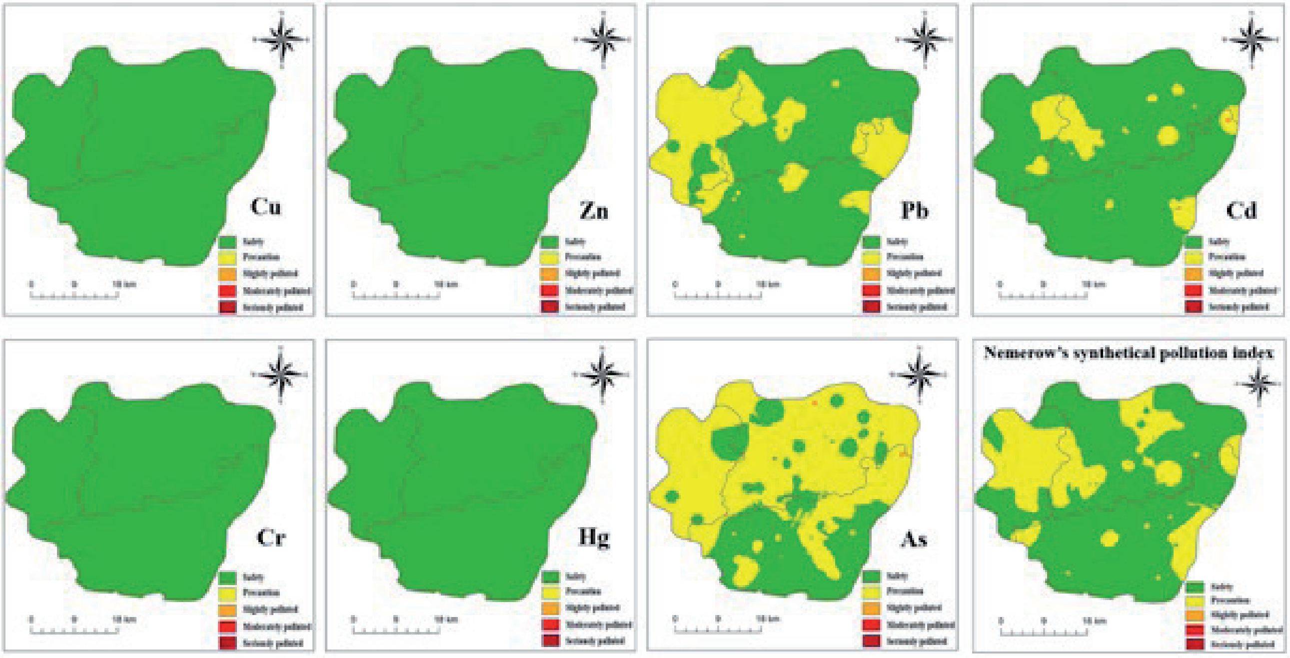

Fig. 3. Assessing map of soil environmental quality based on single-factor and Nemerow's synthetical pollution index.

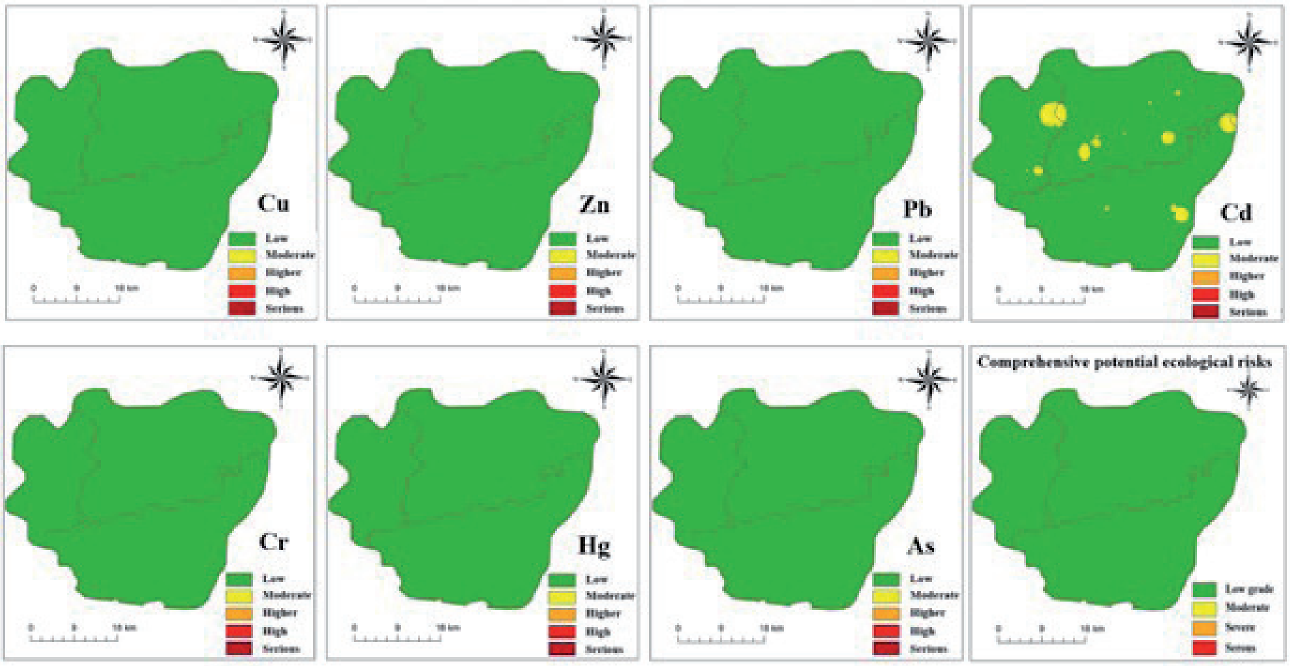

Fig. 4. Assessing map of soil environmental quality based on a single-factor and comprehensive potential ecological risk index.

Risk Assessment of HM Intake via Food

As shown in Table 6, the estimated daily intake (EDI) of Cu, Zn, Pb, Cr, Hg, and As from a single food type was found to be lower than the reference dose (RfD) and provisional tolerable daily intake (PTDI). However, the EDI value of Cd in cereals exceeded the PTDI [34, 35]. To comprehensively evaluate the risk of intake, the total EDI was calculated and observed that the total EDI of Cd and As were higher than the PTDI and RfD values [36]. The levels of other elements have remained within permissible limits. These results indicate that residents face potential health risks due to excessive consumption of Cd and As via their daily diet, despite individual food types being within acceptable limits. Previous studies have shown that grain consumption is a primary pathway for HM exposure, particularly for rice, which can accumulate Cd and As from soil into its grains [37,

Table 6. The value of EDI in different food types.

Element	EDI (μg/kg/day)						Total EDI (μg/kg/day)	PTDI (μg/kg/day)	RfD (μg/kg/day) [36]
	Cereal	Vegetable	Fish	Meat	Poultry	Egg			
Cu	7.7	0.72	0.47	4.6	0.81	1.1	15	500 [34]	40
Zn	138	6.1	16	69	19	30	279	857 [35]	300
Pb	1.2	0.11	0.09	0.11	0.03	0.04	1.5	3.6 [34]	4
Cd	0.95	0.08	0.02	0.06	0.02	0.02	1.2	0.83 [34]	1
Cr	2.4	1.2	0.33	0.27	0.07	0.13	4.4	300 [34]	1500
As	1.8	0.94	0.18	0.24	0.09	0.12	3.4	2.1 [34]	3
Hg	0.14	0.001	0.01	0.03	0.01	0.01	0.20	0.57 [34]	0.5

Table 7. The value of THQ and TTHQ in different food types.

Food types	THQ							TTHQ
	Cu	Zn	Pb	Cd	Cr	As	Hg	
Cereal	0.45	1.1	0.67	2.2	0.004	14	3.3	22
Vegetable	0.04	0.05	0.06	0.19	0.002	7.3	0.11	7.8
Fish	0.03	0.12	0.05	0.04	0.001	1.4	0.30	1.9
Meat	0.27	0.54	0.06	0.15	0.001	2.0	0.80	3.7
Poultry	0.05	0.15	0.02	0.05	0.001	0.7	0.30	1.2
Egg	0.06	0.23	0.03	0.04	0.001	1.0	0.30	1.6

38]. Therefore, residents should reduce rice consumption or substitute it with alternative grains to mitigate health hazards.

The target hazard quotient (THQ) assumes that the absorbed dose is equivalent to its ingested dose, using the ratio of the measured human ingested dose to the reference dose as the standard for evaluation [13]. It indicates that there is no significant health risk in the exposed population when the value is below 1, otherwise, there is a potential health risk [13]. As presented in Table 7, THQ values for Cu, Pb, and Cr in all food types were below 1. However, THQ values for Zn, Cd, As, and Hg in cereals exceeded 1. Moreover, THQ values for As in vegetables, fish, and meat were also above 1 but remained at a low risk level. However, the THQ of As in cereal was 14, which indicated a moderate level of risk. These results indicate that Zn, Cd, As, and Hg pose higher risks through dietary intake compared to other HMs. Furthermore, the combined risk assessment considering multiple HMs for six food types revealed cereal as having the highest total target value of TTHQ, while poultry had the lowest TTHQ value. The order of HM risk based on the TTHQ was cereal > vegetable > meat > fish > egg > poultry, which can be concluded that cereals were identified as presenting greater HM intake risks than other food types. Therefore, further attention should focus on cereal consumption as an important pathway for HM exposure.

The valuation of THQ is predicated on the absorbed dose equivalent to the intake of HMs, however, some HMs are discharged from the human body via the action of the digestive system, and merely some HMs accumulate in various organs of the human body. Consequently, variations exist due to individual differences, internal and external environmental fluctuations, and food intake levels [39]. Generally, the THQ is employed as a reference for assessing the risk associated with dietary exposure to HMs. However, when the THQ of HMs exceeds 1, the exposed population may not show negative health symptoms, but the potential risk from dietary intake still exists.

Conclusions

The levels of Cd and As slightly exceeded the screening value, making them the predominant soil pollutants in the study area, followed by Zn and Pb. However, Cu, Cr, and Hg levels did not indicate pollution levels. Among these HMs, Cd presented a higher ecological risk compared to other HMs, nevertheless, the comprehensive potential ecological risk remained at a low level. The combined results of field investigations, correlation analyses, and principal component analysis indicated that the detected HMs in Xianyushan originated mainly from soil parent material.

Nonetheless, contributions from anthropogenic sources such as road traffic, pesticides, and fertilizers usage, as well as household waste cannot be neglected. The concentrations of HMs in different food types were found to be lower than the limit defined by China's national food safety standards, however, dietary intake risks were observed for Cd and As in cereals due to their total EDI exceeding both PTDI and RfD. Cereals exhibited the highest value of TTHQ, indicating their significance as a major source of HMs and should be given further attention.

Acknowledgments

This work was supported by the Anhui Provincial Department of Education Scientific Research Project (2023AH050133) and the Natural Science Foundation of Anhui Province (2108085Y15).

Conflict of Interest

The authors declare no conflict of interest.

References

1. CHANDA A., SWAIN S., DAS S., SEAL A., GHOSH T., HAZRA S. Spatial variation of heavy metal accumulation in the sediments adjacent to different mangrove plant species within a mixed mangrove stand. Soil & Sediment Contamination, **30** (5), 548, **2021**.
2. HUANG B., YUAN Z.J., LI D.Q., ZHENG M.G., NIE X.D., LIAO Y.S. Effects of soil particle size on the adsorption, distribution, and migration behaviors of heavy metal(loid)s in soil: a review. Environmental Science-Processes & Impacts, **22** (8), 1596, **2020**.
3. WAN X.M., YANG J.X., SONG W. Pollution status of agricultural land in China: impact of land use and geographical position. Soil and Water Research, **13** (4), 234, **2018**.
4. ZHAO F.J., MA Y.B., ZHU Y.G., TANG Z., MCGRATH S.P. Soil contamination in China: current status and mitigation strategies. Environmental Science & Technology, **49** (2), 750, **2015**.
5. HU Y.N., CHENG H.F., TAO S. The challenges and solutions for cadmium-contaminated rice in China: A critical review. Environment International, **92-93**, 515, **2016**.
6. WANG H., LI W.Y., ZHU C.M., TANG X.B. Analysis of heavy metal pollution in cultivated land of different quality grades in Yangtze River Delta of China. International Journal of Environmental Research and Public Health, **18** (18), 9876, **2021**.
7. MIN X.B., XIE X.D., CHAI L.Y., LIANG Y.J., LI M., KE Y. Environmental availability and ecological risk assessment of heavy metals in zinc leaching residue. Transactions of Nonferrous Metals Society of China, **23** (1), 208, **2013**.
8. CHENG J.L., SHI Z., ZHU Y.W. Assessment and mapping of environmental quality in agricultural soils of Zhejiang Province, China. Journal of Environmental Sciences, **19** (1), 50, **2007**.
9. 9. GU Y.G., GAO Y.P. An unconstrained ordination- and GIS-based approach for identifying anthropogenic sources of heavy metal pollution in marine sediments. Marine Pollution Bulletin, **146**, 100, **2019**.
10. HASSAN H.B., MOHAMED W.M. The distribution of heavy metal pollutants in Suez Bay using geographic information system (GIS). Arab Journal of Nuclear Sciences and Applications, **50** (2), 258, **2017**.
11. ALI H., KHAN E. Trophic transfer, bioaccumulation, and biomagnification of non-essential hazardous heavy metals and metalloids in food chains/webs-Concepts and implications for wildlife and human health. Human and Ecological Risk Assessment, **25** (6), 1353, **2019**.
12. ISMAIL A., RIAZ M., AKHTAR S., ISMAIL T., AMIR M., ZAFAR-UL-HYE M. Heavy metals in vegetables and respective soils irrigated by canal, municipal waste and tube well waters. Food Additives & Contaminants Part B-Surveillance, **7** (3), 213, **2014**.
13. LIU H.Q., LIU G.J., WANG S.S., ZHOU C.C., YUAN Z.J., DA C.N. Distribution of heavy metals, stable isotope ratios (delta C-13 and delta N-15) and risk assessment of fish from the Yellow River Estuary, China. Chemosphere, **208**, 731, **2018**.
14. ZHAO B., XING C., ZHOU S.B., WU X.G., YANG R.Y., YAN S.K. Sources, fraction distribution and health risk assessment of selenium (Se) in Dashan Village, a Se-rich area in Anhui Province, China. Bulletin of Environmental Contamination and Toxicology, **104**, 545, **2020**.
15. YANG B.B., YANG C., SHAO Z.Y., WANG H., ZAN S.T., ZHU M., ZHOU S.B., YANG R.Y. Selenium (Se) does not reduce cadmium (Cd) uptake and translocation in rice (*Oryza sativa L.*) in naturally occurred se-rich paddy fields with a high geological background of Cd. Bulletin of Environmental Contamination And Toxicology, **103** (1), 127, **2019**.
16. ZHAO B., ZHAO J., ZHOU S.B., WU X.G., XU X., YANG R.Y., XU X. Selenium and toxic metals in human hair of the Dashan Region, China: Concentrations, sources, and antagonism effect. Ecotoxicology and Environmental Safety, **250**, 114479, **2023**.
17. LIU H.Q., LIU G.J., ZHOU C.C., YUAN Z.J., DA C.N. Geochemical speciation and ecological risk assessment of heavy metals in surface soils collected from the Yellow River Delta National Nature Reserve, China. Human and Ecological Risk Assessment, **23** (7), 1585, **2017**.
18. XU Q.T., ZHANG M.K. Source identification and exchangeability of heavy metals accumulated in vegetable soils in the coastal plain of eastern Zhejiang province, China. Ecotoxicology and Environmental Safety, **142**, 410, **2017**.
19. SEPA (State Environmental Protection Agency). Background values of soil elements in China. China Environmental Press. **1990** [In Chinese].
20. YANG S.M., LI Q.H., LI C.F., WANG H., CHEN C., LONG J., WEI X., SONG L.H., WEN X.F. Geochemical characteristics and risk assessment of heavy metals in soil and fruit of major blueberry growing areas in Guizhou Province. Polish Journal of Environmental Studies, **32** (1), 913, **2023**.
21. MALAKOOTIAN M., MOHAMMADI A., NASIRI A., CONTI G.O., FARAJI M. Correlation between heavy metal concentration and oxidative potential of street dust. Air Quality Atmosphere and Health, **15** (4), 731, **2022**.

22. HAN Y.M., DU P.X., CAO J.J., POSMENTIER E.S. Multivariate analysis of heavy metal contamination in urban dusts of Xi'an, Central China. Science of the Total Environment, **355** (1-3), 176, **2006**.
23. AL-KHASHMAN O.A. Heavy metal distribution in dust, street dust and soils from the workplace in Karak Industrial Estate, Jordan. Atmospheric Environment, **38** (39), 6803, **2004**.
24. ZHANG X.W., WEI S., SUN Q.Q., WADOOD S.A., GUO B.L. Source identification and spatial distribution of arsenic and heavy metals in agricultural soil around Hunan industrial estate by positive matrix factorization model, principle components analysis and geostatistical analysis. Ecotoxicology and Environmental Safety, **159**, 354, **2018**.
25. TANG R.G., LUO J., SHE J., CHEN Y.C., YANG D.D., ZHOU J. The cadmium and lead of soil in timberline coniferous forests, Eastern Tibetan Plateau. Environmental Earth Sciences, **73** (1), 303, **2015**.
26. NIE M., WAN J.R., CHEN X.F., WANG L., LI B., CHEN J.K. Comparison of heavy metal elements between natural and plantation forests in a subtropical montane forest. Spectroscopy and Spectral Analysis, **31** (11), 3098, **2011**.
27. LIAO S.Y., JIN G.Q., KHAN M.A., ZHU Y.W., DUAN L.L., LUO W.X., JIA J.W., ZHONG B., MA J.W., YE Z.Q., LIU D. The quantitative source apportionment of heavy metals in peri-urban agricultural soils with UNMIX and input fluxes analysis. Environmental Technology & Innovation, **21**, 101232, **2021**.
28. HU B.F., ZHOU Y., JIANG Y.F., JI W.J., FU Z.Y., SHAO S., LI S., HUANG M.X., ZHOU L.Q., SHI Z. Spatio-temporal variation and source changes of potentially toxic elements in soil on a typical plain of the Yangtze River Delta, China (2002-2012). Journal of Environmental Management, **271**, 110943, **2020**.
29. ROSEN V., CHEN Y.N. Effects of compost application on soil vulnerability to heavy metal pollution. Environmental Science and Pollution Research, **25** (35), 35221, **2018**.
30. DAI J., BECQUER T., ROUILLER J.H., REVERSAT G., BERNHARD-REVERSAT F., LAVELLE P. Influence of heavy metals on C and N mineralisation and microbial biomass in Zn-, Pb-, Cu-, and Cd-contaminated soils. Applied Soil Ecology, **25** (2), 99, **2004**.
31. NING C.C., GAO P.D., WANG B.Q., LIN W.P., JIANG N.H., CAI K.Z. Impacts of chemical fertilizer reduction and organic amendments supplementation on soil nutrient, enzyme activity and heavy metal content. Journal of Integrative Agriculture, **16** (8), 1819, **2017**.
32. LI W.X., ZHANG X.X., WU B., SUN S.L., CHEN Y.S., PAN W.Y., ZHAO D.Y., CHENG S.P. A comparative analysis of environmental quality assessment methods for heavy metal-contaminated soils. Pedosphere, **18** (3), 344, **2008**.
33. LUO W., LU Y.L., GIESY J.P., WANG T.Y., SHI Y.J., WANG G., XING Y. Effects of land use on concentrations of metals in surface soils and ecological risk around Guanting Reservoir, China. Environmental Geochemistry and Health, **29** (6), 459, **2007**.
34. JECFA (JOINT FAO/WHO EXPERT COMMITTEE ON FOOD ADDITIVES). Evaluations of the Joint FAO/WHO Expert Committee on Food Additives, **2015**.
35. WHO. Guidelines for Drinking-Water Quality. World Health Organization, Geneva, Switzerland, **1996**.
36. USEPA. Risk-based concentration table. Washington, United States Environmental Protection Agency, **2000**.
37. 37. KIBRIA K.Q., ISLAM M.A., HOQUE S., SIDDIQUE M.A., HOSSAIN M.Z., ISLAM M.A. Variations in cadmium accumulation among amon rice cultivars in Bangladesh and associated human health risks. Environmental Science and Pollution Research, **29** (26), 39888, **2022**.
38. SUN S.K., CHEN Y., CHE J., KONISHI N., TANG Z., MILLER A.J., MA J.F., ZHAO F.J. Decreasing arsenic accumulation in rice by overexpressing OsNIP1;1 and OsNIP3; 3 through disrupting arsenite radial transport in roots. New Phytologist, **219** (2), 641, **2018**.
39. HORIGUCHI H., OGUMA E., SASAKI S., MIYAMOTO K., IKEDA Y., MACHIDA M., KAYAMA F. Dietary exposure to cadmium at close to the current provisional tolerable weekly intake does not affect renal function among female Japanese farmers. Environmental Research, **95** (1), 20, **2004**.