Front. Environ. Sci. Eng. 2016, 10(6): 7
DOI 10.1007/s11783-016-0873-7

RESEARCH ARTICLE

Assessment of pollution of potentially harmful elements in soils surrounding a municipal solid waste incinerator, China

Ying Han[1,2], **Huiting Xie**[1,2,3], **Wenbin Liu** (✉)[1,2], **Haifeng Li**[1,2], **Mengjing Wang**[1,2], **Xuebin Chen**[1,2], **Xiao Liao**[1,2], **Nan Yan**[1,2]

1 State Key Laboratory of Environmental Chemistry and Ecotoxicology, Research Center for Eco-Environmental Sciences, Beijing 100085, China
2 University of Chinese Academy of Sciences, Beijing 100049, China
3 Division of Environmental Management & Policy School of Environment, Tsinghua University, Beijing 100084, China

HIGHLIGHTS

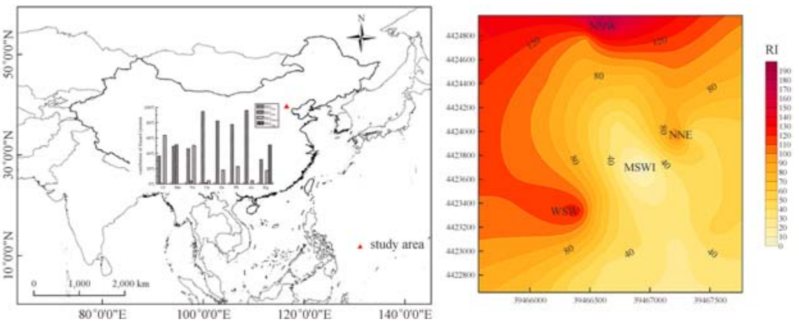
- Potentially harmful elements in ambient soil of a MSWI were assessed.
- Spatial distribution of potential ecological risk index was investigated.
- Health risk assessment of potentially harmful elements in soil was evaluated.
- Hg in the soil posed health hazards to the local population.

ARTICLE INFO

Article history:
Received 10 April 2016
Received in revised form 24 June 2016
Accepted 23 September 2016

Keywords:
Soil
Potentially harmful elements
Contamination
Kriging interpolation
Risk assessment
Wind direction

GRAPHIC ABSTRACT



ABSTRACT

We assessed the contamination levels of Mn, Zn, Cr, Cu, Ni, Pb, As and Hg and the risks posed by these potentially harmful elements in top-soils around a municipal solid waste incinerator (MSWI). We collected 20 soil samples, with an average pH of 8.1, and another fly ash sample emitted from the MSWI to investigate the concentrations of these elements in soils. We determined the concentrations of these elements by inductively coupled plasma–optical emission spectrometer (ICP-OES), except for Hg, which we measured by AF-610B atomic fluorescence spectrometer (*AFS*). We assessed the risks of these elements through the use of geoaccumulation index (I_{geo}), potential ecological risk index (*RI*), hazard quotient (HQ_i) and cancer risk ($Risk_i$). The results showed that concentrations of potentially harmful elements in soil were influenced by the wind direction, and the concentrations of most elements were higher in the area northwest of the MSWI, compared with the area southeast of the incinerator, with the exception of As; these results were in accordance with those results acquired from our contour maps. According to the I_{geo} values, some soil samples were clearly polluted by Hg emissions. However, the health risk assessment indicated that the concentrations of Hg and other elements in soil did not pose non-carcinogenic risks to the local populations. This was also the case for the carcinogenic risks posed by As, Cr, and Ni. The carcinogenic risk posed by As was higher, in the range 6.49×10^{-6}–9.58×10^{-6}, but this was still considered to be an acceptable level of risk.

1 Introduction

Along with rapid economic growth, there have been increases in the consumption of potentially harmful elements, which in turn leads to greater disposal of these elements to the environment [1]. A lack of landfill sites has caused incineration to become an effective and a common alternative for treating municipal solid waste (MSW) in China for significant waste reduction; for example, in 2012, 27% of the collected MSW were incinerated [2]. Potentially harmful elements are one of the major groups of pollutants that are emitted from incinerators and, may

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✉ Corresponding author
E-mail: liuwb@rcees.ac.cn

impact physicochemical and biological properties of ecosystems [1,3]. If current trends continue, large amounts of potentially harmful elements will continue be released from MSW incinerators (MSWIs).

The potentially harmful elements released from stacks can be deposited via wet and dry deposition to soil, and soils are an important indicator of environmental quality [4]. Potentially harmful elements may represent a serious threat to the environment and human health as a result of their physicochemical properties; for example, they are, non-biodegradable, have long biological half-lives, and bioaccumulation in the food chain [5]. If the elements appear to adversely impact human health or the environment, it is important to understand these impacts and to monitor these elements in the environment [6].

A number of recent studies have focused on levels of potentially harmful elements in ambient soil around MSWIs. Meneses et al. determined the temporal variations of element concentrations in soil near an old MSWI. With the exception of an increase in the levels of Be and Ni, the authors observed no significant differences between soils collected from two different years [7]. Loppi et al. found that Cu, and Hg concentrations in soil near a MSWI were similar to the concentrations in unpolluted areas, while Cr, Zn, and Cd concentrations were higher than in unpolluted areas, with Cd concentrations being particularly high [8]. Additionally, Zhang et al. discovered substantial temporal and spatial variations in the behavior of potentially harmful elements during MSW incineration. A large proportion of Hg was evaporated and enriched in the fine residues of the air pollution control system of the incinerator. Cr, Cu, and Ni were transferred into the residues and ash particles mainly by entrainment. Pb and Zn were transferred into the residues by both evaporation and entrainment. As was transported into the flue gas mainly via evaporation, with a lower coefficient [9].

A number of epidemiologic studies have demonstrated relationships between exposure to potentially harmful elements and a wide range of adverse health outcomes. Pb contamination is widespread and is associated with a significant health risk, which may adversely affect human health, for instance, by resulting in impaired cognitive, physical and behavioral abilities [6]. As is an element often found in elevated concentrations in the environment. The persistence of As in soil and its potential toxicity to organisms has attracted a great deal of attention. As concentrations may be a human health hazard in some areas [10].

In this study, we collected 20 soil samples between 300 and 2000 m away from the stack of a typical modern MSWI, and we collected a fly ash (FA) sample to identify the distributions of potentially harmful elements and their environmental impacts. We assessed the risks posed by these potentially harmful elements to the local environment and to human health; we consider these parameters to be the primary indicators of soil pollution, and also evidence for successful remediation of polluted soils. The results of this study indicate the general accumulation and risk features of potentially harmful elements in the ambient soil released from the MSWI, and will be helpful in formulating effective contamination control measures by policy makers.

2 Materials and methods

2.1 Sample collection

The Sunrise MSWI is located in the city of Beijing (39° 59.923′ N, 116° 24.543′ E), Northeastern China (Fig. 1), and has a total area of 4.6×10^4 m^2. The MSWI began operating in 2003, and has a capacity of 0.58 million tons per year. As the terrain is high in the middle of the study site and low in the surrounding area, the contribution of natural sources to the potentially harmful elements in this site is negligible. The geology of the study area is particularly complex, and includes a combination of rock types; additionally there are a variety of industrial activities that take place in the vicinity. Restricted by ancient geography and influenced by long-term effects of fluvial activity, the subsoils of the study area are deposits of Quaternary Alluvial, the interaction layer of lacustrine coarse and fine particles. Mainland layers in the study site are distributed as follows (from top to bottom): artificial fill, silty clay, silt, sand, clay, and coarse sand. The prevalent winds are southeast in the summer and northwest in the winter. Cinnamon and fluvo-aquic were the dominant soils, with parent materials consisting of weathering rocks and loose quaternary sediment. The air pollution control system in the MSWI is composed of a semi-dry de-acidification unit, an active carbon injector, a bag filter, and an 80-m-high stack.

We collected 20 soil samples from around the Sunrise MSWI in August 2012. We chose the sampling sites using the atmospheric dispersion modeling system and the wind rose diagram, obtained from the local meteorological department (Fig. 1). For each soil sample, we collected about 1 kg of surface soil (0 – 10 cm) uniformly across an area of 25 m^2, and mixed five aliquots of soil evenly to produce a sample. After removing the plants and gravel, we sealed the soil samples in PVC bags, then stored them in the refrigerator at −20°C until analysis. We used a hand-held global positioning system device (Magellan GPS, San Dimas, USA) to determine the position of each sampling site. We also collected one FA sample from the outlet of the bag house filter in the MSWI in August 2012 to examine the relationship between the element concentrations in soil and their potential sources.

2.2 Sample treatment and analysis

We dried each soil sample at room temperature, then

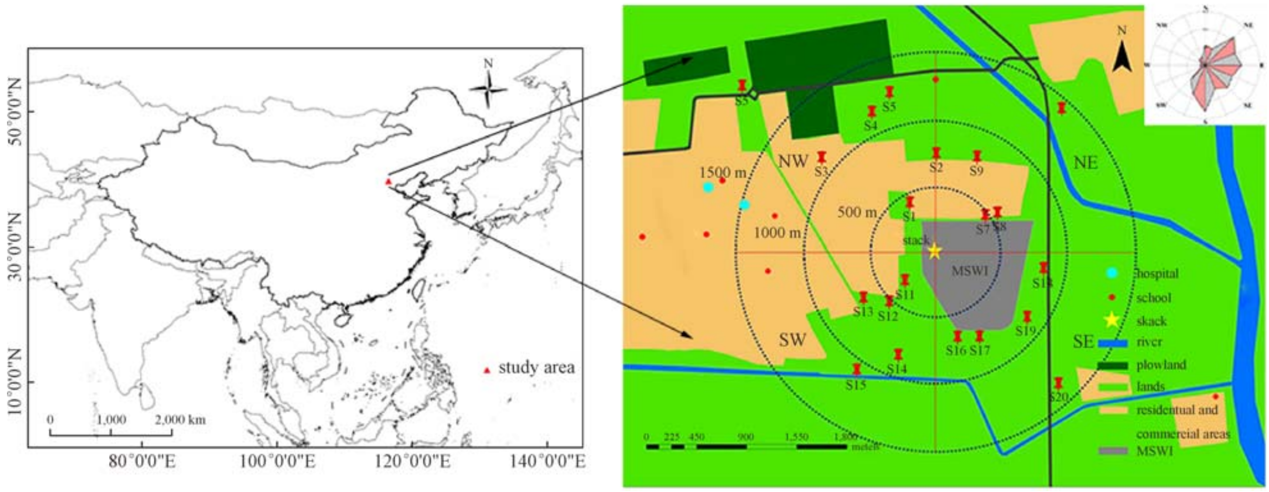

Fig. 1 The geo-localization of the study area and soil sampling sites around the MSWI

ground them and passed them through a 100-mesh sieve [11]. We digested about 0.5 g of each ground soil or FA sample in aqua regia (a 1:3 mixture of concentrated HNO_3 and concentrated HCl) and $HClO_4$, so that the As, Cr, Cu, Mn, Ni, Pb, and Zn concentration could be determined. We then diluted the mineral residues with deionized water to a final volume of 25 mL in a volumetric flask, and stored them in the refrigerator at 4°C before analysis by inductively coupled plasma–optical emission spectrometry (ICP-OES, Teledyne Leeman Labs, USA).

For Hg analyses, we added 0.1 g of a ground soil or FA sample to a 20 mL ampule, then added 2 mL of HNO_3 and 1 mL of deionized water. After allowing pre-digestion to occur for 30 minutes, we sealed the ampule using a Relatives for Justice (RFJ) model manual ampule sealer (Jishou Zhongcheng Pharmacy Machine Co., Ltd., China). We heated the ampule for 2 h in boiling water, then allowed to cool to room temperature before opening. We diluted the digested solution with deionized water to 10 mL in a 15-mL polyethylene terephthalate vial, mixed it thoroughly, then determined using an AF-610B atomic fluorescence spectrometer (AFS; Beijing Rayleigh Analytical Instrument Co., China).

For each sample, we averaged the concentration of three replicate samples to determine the final concentration. We also used reagent blanks and standard reference materials (GSS-6 soil; China National Center for Standard Material, Beijing, China) for data quality control and assurance. The recoveries of the eight metals that we analyzed ranged from 89% to 109%.

We determined the total organic carbon (TOC) content of the soil samples using a TOC analyzer (OI Analytical, College Station, USA). The TOC contents ranged from 1.26% to 8.81%, and the average was 2.75%±1.70%. We mixed 10 g of the soil samples with 25 ml distilled water for 1 minute, then let it stand for 30 minutes prior to analysis. We determined the pH of each soil sample in the soil solution (1:2.5) using a combination pH electrode.

2.3 Assessment of potentially harmful elements contamination

We selected the geoaccumulation index (I_{geo}) to assess the environmental contamination levels of potentially harmful elements in soil around the MSWI by comparing the current and pre-industrial concentrations [12]. We calculated the I_{geo} was calculated using Eq. (1),

$$I_{geo} = \log_2 \frac{C_n}{1.5B_n}, \tag{1}$$

where C_n is the measured concentration and, B_n is the expected background concentration of element n in the soil. The constant (1.5) accounts for natural fluctuations in the concentration of a given substance in the environment, and allows for the identification of very small anthropogenic influences [13]. The I_{geo} includes seven grades, and the highest grade reflects a value 100 times greater than the background values (Table 1).

Table 1 The seven grades of the I_{geo} [14]

grade	value	soil quality
0	$I_{geo} \leqslant 0$	practically uncontaminated
1	$0 < I_{geo} \leqslant 1$	uncontaminated to moderately contaminated
2	$1 < I_{geo} \leqslant 2$	moderately contaminated
3	$2 < I_{geo} \leqslant 3$	moderately to heavily contaminated
4	$3 < I_{geo} \leqslant 4$	heavily contaminated
5	$4 < I_{geo} \leqslant 5$	heavily to extremely contaminated
6	$5 < I_{geo}$	extremely contaminated

2.4 Potential ecological risk

The potential ecological risk index (*RI*) method for assessing the risks posed by potentially harmful elements is the only method that takes both the concentrations and toxic response factors into consideration, and is the most commonly used method for assessing risks posed by potentially harmful elements [15]. We calculated the *RI* using Eq. (2),

$$RI = \sum_{i=1}^{m} E_r^i = \sum_{i=1}^{m} \left(T_r^i \times \frac{C_D^i}{C_B^i} \right), \tag{2}$$

where C_D^i is the measured concentration in the sample, C_B^i is the background concentration in the soil, T_r^i is the biological toxicity factor for each element, E_r^i is the potential ecological risk factor for each element, and *RI* is the potential ecological risk index for each element. The T_r^i constants commonly used for As, Cr, Cu, Hg, Mn, Ni, Pb, and Zn are 10, 2, 5, 40, 1, 5, 5, and 1, respectively; these are the values calculated by the principle of Hakanson [16]. We assessed E_r^i, for each substance according to five grades, and assessed the *RI* for a given area according to four grades [17].

The geostatistical interpolation technique has been widely used to create continuous contour maps, in which the levels of risk associated with pollutants are described by a series of equivalent lines on a map [18]. We applied the widely–used ordinary Kriging interpolation technique [19,20] to create a contour map of the *RI* values in the soil samples.

2.5 Human health risk assessment

We also evaluated the health risks posed to the local population by exposure to potentially harmful elements in soil. Human exposure to these elements in soil can occur through three main paths: soil ingestion (CDI_{ing}), air inhalation (CDI_{inh}), and dermal contact (CDI_{derm}). Humans can also be exposed to Hg through inhaling Hg in the vapor form (CDI_{vap}) [21]. We assessed the risks posed by exposure through these four paths using equations 3–6 [19].

$$CDI_{\mathrm{ing}} = \frac{C \times IFP \times EF \times ED}{BW \times AT} \times 10^{-6}, \tag{3}$$

$$CDI_{\mathrm{inh}} = \frac{C \times IR \times EF \times ED}{BW \times AT \times PEF}, \tag{4}$$

$$CDI_{\mathrm{derm}} = \frac{C \times SA \times AF \times ABS \times EF \times ED}{BW \times AT} \times 10^{-6}, \tag{5}$$

$$CDI_{\mathrm{vap}} = \frac{C \times IR \times EF \times ED}{BW \times AT \times VF}, \tag{6}$$

In Eqs. (3)–(6), *CDI* is chronic daily intake ($\mathrm{mg \cdot kg^{-1} \cdot d^{-1}}$), *C* is the concentration of the element in soil ($\mathrm{mg \cdot kg^{-1}}$), *IFP* is the soil ingestion rate ($\mathrm{m^3 \cdot kg^{-1}}$), *EF* is the exposure frequency ($\mathrm{d \cdot yr^{-1}}$), *ED* is the exposure duration (yr), *BW* is the average body weight (kg), *AT* is the average time (d), *IR* is the inhalation rate ($\mathrm{m^3 \cdot d^{-1}}$), *PEF* is the particulate emission factor, *SA* is the surface area of the skin that comes into contact with the dust ($\mathrm{cm^2 \cdot d^{-1}}$), and *AF* is the adherence factor soil to skin ($\mathrm{mg \cdot cm^{-2}}$), *ABS* is the dermal absorption factor, *VF* is the volatilization factor ($\mathrm{m^3 \cdot kg^{-1}}$). All of these factors are shown in Table S1 (Supplementary material).

The eight elements that were analyzed may all potentially cause toxic effects in humans, and As, Cr, and Ni are considered to be probable carcinogens. We considered both non-carcinogenic and carcinogenic risks from exposure to As, Cr, and Ni through the four exposure routes described above.

The hazard quotient (HQ_i) and hazard index (HI) can be calculated using equation 7 to determine the non-cancer health risks posed by potentially harmful elements. The CDI_i is the chronic daily intake of an element through each exposure route, and the RfD_{i} is the reference dose for the metal. In the cancer risk ($Risk_i$) calculations (equation 8), the CDI_i is multiplied by the appropriate slope factor (*SF*), and $Risk_T$ is the total cancer risk [22].

$$HQ_i = CDI_i / RfD_i,\ HI = \sum HQ_i, \tag{7}$$

$$Risk_i = CDI_i \times SF_i,\ Risk_T = \sum Risk_i. \tag{8}$$

As a rule, a *HI* or *HQ* less than or equal to 1 is assumed to indicate that the exposure pathway or chemical is negligible, and an *HQ* value greater than 1 is assumed to indicate that there are likely to be adverse health effects [23]. In general, $Risk_i$ values below 10^{-6} are not considered to indicate a significant cancer risk. $Risk_i$ values greater than 10^{-4} are considered to be unacceptable [24], which means that the potential for cancer is too high and the surroundings need to redesign. $Risk_i$ values between 10^{-4} and 10^{-6} are generally considered to be acceptable, depending on the circumstances of exposure to the residents and the local environment [22].

The descriptions of the different parameters that were used in the risk assessment, and the values that were used to calculate these parameters, are summarized in Tables S1 and S2 (Supplementary material).

3 Results and discussion

3.1 Concentrations of potentially harmful elements

The As, Cr, Cu, Hg, Mn, Ni, Pb, and Zn concentrations in the soil and FA samples are shown in Table 2 and Table S3 (Supplementary material). The pH values of the soil

Table 2 Heavy metal concentrations observed in the study area and in other areas around the world (mg·kg^{-1})

metal	soil analyzed in this study	soil from Italy [26]	soil from Newcastle, UK [27]	soil from Shenzhen [28]	background soil [25]	ash analyzed in this study	ash from Shanghai [29]	ash from Singapore [30]
	mean±sd	mean±sd	mean	mean	mean±sd	mean	mean	mean
Cr	52±8.2	85±24	55	52	68±16	156	157	72
Mn	546±40	–	–	–	705±160	994	704	309
Ni	24±3.8	73±24	30	6.0	29±7.5	88	71	22
Cu	28±8.6	69±43	233	12	24±4.7	402	563	570
Zn	100±40	127±135	419	47	103±35	1614	3269	6288
Pb	19±9.0	39±35	350	28	25±6.3	328	1515	2000
As	7.7±2.6	5.9±1.9	20	9.4	9.7±2.5	56	26	–
Hg	0.088±0.064	0.17±0.11	0.50	0.058	0.069±0.051	5.9	3.6	–

samples were between 7.9 and 8.6, and the average was 8.1.

Mn is commonly one of the most abundant trace elements in soils [25]. The average Mn concentration in the soil samples (546±40 mg·kg^{-1}) was higher than that of the other elements; the concentrations of the other elements occurred in the following, decreasing order: Zn (100±40 mg·kg^{-1}), Cr (52±8.2 mg·kg^{-1}), Cu (28±8.6 mg·kg^{-1}), Ni (24±3.8 mg·kg^{-1}), Pb (19±9.0 mg·kg^{-1}), As (7.7±2.6 mg·kg^{-1}), and Hg (0.088±0.064 mg·kg^{-1}). The concentrations of these elements in the FA samples were relatively high: the mean Hg concentration was almost 67 times higher than the mean soil concentration, and the Pb, Zn, and Cu concentrations were 17.6, 16.1, and 14.4 times the mean soil concentrations, respectively.

To facilitate the evaluation, we compared the concentrations of these eight elements in the soil samples with those in soil samples from around the world (Table 2). Average concentrations of these elements in most soil samples were lower than those observed in soil near MSWIs in Italy [26], Newcastle, UK [27], and the background soil from a remote area [25], but slightly higher than those from soil in Shenzhen, China [28]. The MSWIs from the study areas in Italy, UK, and Shenzhen, have pollution control measures similar to those of the Sunrise MSWI. In all cases, the flue gases are processed with a bag house filter and, electrostatic precipitator, and are then neutralized in a wet scrubber by water and lime [26-28]. Simultaneously, the levels of these elements except for Mn were compared with the average background values (ABVs) in Chinese soils (without standards for Mn) [31]. The ABVs in China for Cr, As, Hg, Pb, Cu, Zn and Ni were 61.0, 11.2, 0.065, 26.0, 22.6, 74.2 and 26.9, respectively [31]. The mean concentrations of Cu, Zn, and Hg were slightly higher (1.239, 1.348, and 1.354 times higher, respectively) than their corresponding ABVs. However, the mean concentrations of all seven elements met the soil quality standards of China (Grade Ⅰ) [32]. The Cu, Pb, and Zn concentrations in the FA samples were lower than those found in FA from Shanghai, China [29] and Singapore [30], but the Mn and Ni concentrations were slightly higher.

Concentrations of the potentially harmful elements usually decrease with distance from their source, as a result of dilution and dispersion. The distances of sampling sites from the MSWI were in the range of 500 m to over 1500 m, and distance was a very important factor in element concentrations in soil. We would expect that the concentrations of the elements would be highest within 500 m of the MSWI, and the lowest at sites more than 1500 m away. However, the data demonstrate that the elements in this study area did not follow this trend (Table 3). For the elements other than Cr and As, concentrations tended to decrease up to a distance of 1000 m. For most elements, the highest concentrations occurred in soil samples at distances of 500–1000 m, and the maximum concentrations of some elements occurred at distances beyond 1500 m. This phenomenon may be the result of disturbances to the surrounding area. The intrusion of other contamination sources introduced through anthropogenic activities or atmospheric deposition of pollutants from other emission regions (Fig. 1) would increase the element concentrations within this range, which may explain the sudden increase

Table 3 Average concentrations of eight elements in soils from a range of distances from the MSWI (mg·kg^{-1})

distance	Cr	Mn	Ni	Cu	Zn	Pb	As	Hg
< 500 m	48	555	23	26	99	18	8.0	0.073
500 – 1000 m	49	523	23	23	82	14	8.5	0.067
1000 – 1500 m	59	577	25	35	129	25	6.5	0.11
>1500 m	51	551	25	36	110	29	6.6	0.19

in element concentrations at distances further from the stack.

We performed cluster analysis of eight potentially harmful elements in soil by SPSS 18.0 to determine all the possible sources of the metals by connecting the spatial correlations between potentially harmful elements with their distribution characteristics during MSW incineration. Based on the results of these elements, the elements were divided into three groups: (I) Cu, Pb, Zn, Cr, Mn; (II) As, Ni; (III) Hg (Fig. S1).

3.2 Relationship between wind direction and concentrations of potentially harmful elements

We divided the study area into four zones: northwest (NW), northeast (NE), southwest (SW), and southeast (SE), to examine the influence of wind on the distribution of potentially harmful elements (Fig. 2).

Figure 2 shows clear differences in the concentrations of these eight elements between the four zones. The statistic is analyzed using Pearson's correlation analysis. The relationship between wind direction and element concentration was similar for all elements (except for As): the highest concentrations occurred in the zone that corresponded to the dominant summer wind direction. The inorganic and organic forms of As both occur in soil. Some forms of As are equally accessible to the microbes, while other forms, such as As sulfides are relatively stable and resistant to microbial uptake; which also related to the organic carbon, indicating that there are obstructions to As transport As in soil [33]. Our results demonstrated no significant differences in As concentrations between the NW and SE zones. The spatial distributions over the other seven elements were similar to one another. For instance, the concentration of Hg in the NW zone, downwind of the MSWI, was significantly higher than the concentration in the SE zone, upwind of the MSWI ($P = 0.012 < 0.05$); Hg concentrations did not differ significantly between the NE and SW zones ($P = 0.47 > 0.05$). The flue gas emitted from the stack was influenced by the wind direction, and diffused in the downwind direction. We performed sampling in late summer, when there was a prevailing southeast wind, which resulted in accumulation of Hg in the NW zone. In winter, the prevailing wind at the sampling site was a northwest wind. Because the weather was cold and dry, with little snowfall, dry deposition was the primary means of deposition of Hg in the study area. Consequently, the amount of Hg deposited in winter was less than that in summer. Additionally, the long–term effects of leaching in rain also resulted in the washing away of soluble Hg, even downwind of the MSWI in the NE zone in winter [34], which could explain why the differences in Hg levels were smaller between the NE and SW zones.

The concentrations of potentially harmful elements in the soil parent material and in the MSWI emissions both probably contributed to the large fluctuations in concentrations observed for most of the elements in soils downwind of the MSWI. However, the fluctuations in the element concentrations in soils upwind of the MSWI were not significant. This is possibly because the MSWI emissions had little impact on the soil concentrations in those areas because dominant wind direction prevented most of the emissions from traveling in that direction. Ghassen et al. investigated the mobility of potentially harmful elements in carbonated soils contaminated by old mine tailings, and found that the eolian transport had a substantial impact on the redistribution of potentially harmful elements, especially in the dominant wind direction [35]. In this study, the study area was polluted, though the pollution was located topographically above the mine site. However, it faced the prevailing northwest winds and went through the spread of potentially harmful elements. The existence of massive limestone hills prevented the dispersion of potentially harmful elements toward the inhabited areas, located southwest of the hills. Additionally, the long–range atmospheric transportation and atmospheric deposition of potentially harmful elements released from other pollution sources may also contribute to the levels of the elements observed here.

3.3 Assessment of the risks posed by potentially harmful element contamination around the MSWI

We calculated the I_{geo} for each sampling site to identify the potentially harmful elements that were present in concentrations that posed relatively high risks in the soil samples (Fig. 3).

All of the samples had I_{geo} values less than or equal to 0 for As, Cr, Mn, and Ni, indicating that the soil was unpolluted by these elements. For Hg, seven samples (S3, S4, S6, S7, S9, S10, and S13) had I_{geo} values of $0 < I_{\text{geo}} \leqslant 1$ (unpolluted to moderately polluted) and one sample (S5) had an I_{geo} value of $1 < I_{\text{geo}} \leqslant 2$ (moderately polluted). The I_{geo} values for Cu in five of the samples (S3, S4, S5, S6, and S7) indicated that the samples were unpolluted or moderately polluted. The I_{geo} values for Zn and Pb in sample S3 indicated that that soil was unpolluted or moderately polluted.

Soil in the NW zone was more affected by emissions of potentially harmful elements than those in the other zones. The influence of potentially harmful elements in soil samples occurred in the following order: NW>NE>SW>SE.

3.4 Potential ecological risk assessment of potentially harmful elements and risk mapping

We performed an ecological risk assessment to determine the level of potential ecological risk posed by the potentially harmful elements in soil around the MSWI. We calculated the potential ecological risk from the individual elements (E_r^i, Table S4, Supplementary mate-

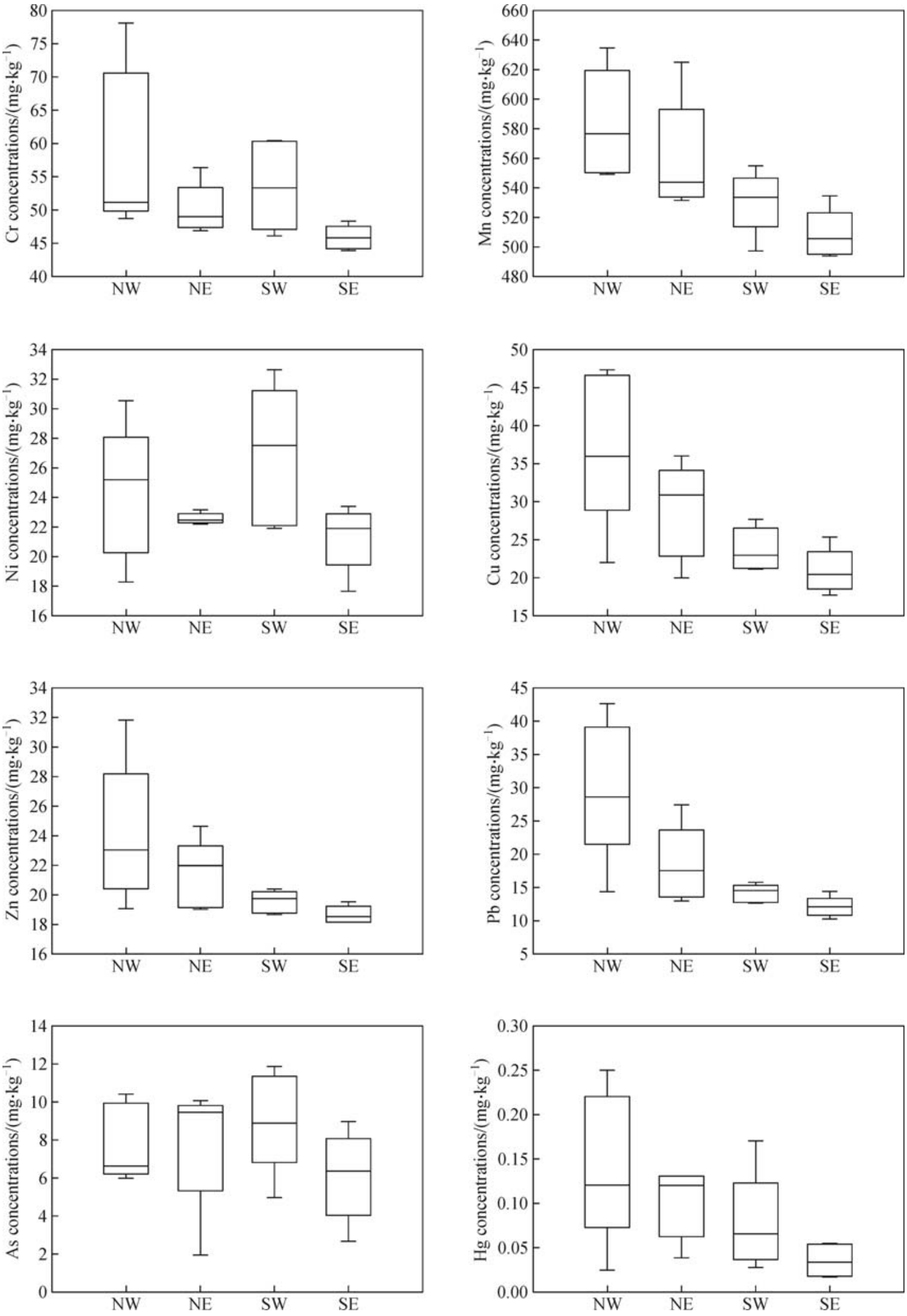

Fig. 2 Relationship between wind direction and the concentrations of potentially harmful elements in soil

rial) and the comprehensive potential ecological index (*RI*, Fig. 4). The mean E_r^i values for the potentially harmful elements occurred in the following order: Hg>As>Cu>Ni>Pb>Cr>Zn>Mn, indicating that Hg is associated with higher levels of potential risks than the other elements.

Three areas (the NNW, NNE, and WSW zones) with

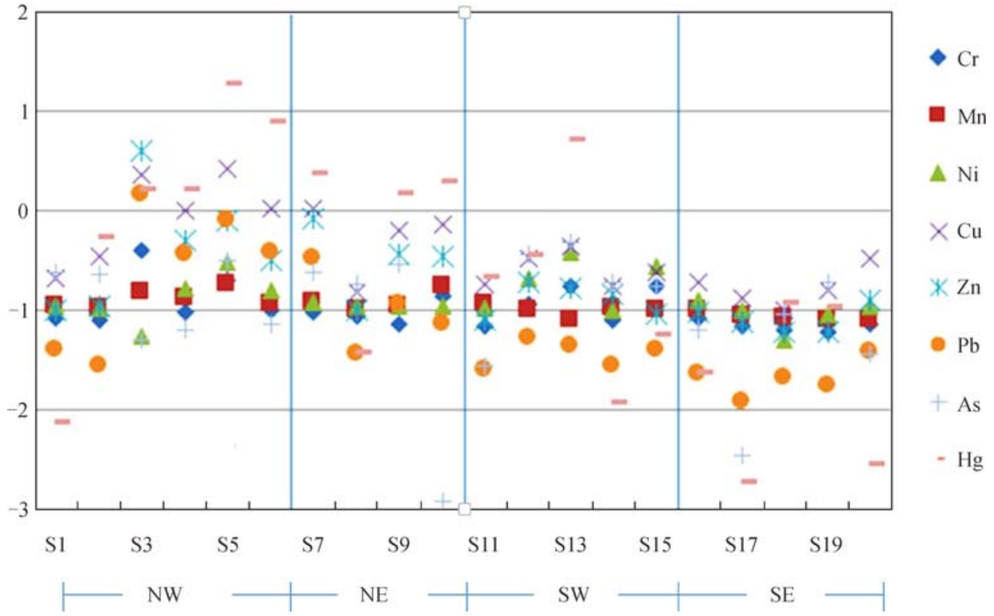

Fig. 3 Geoaccumulation indices (I_{geo}) for the potentially harmful elements in the soil samples

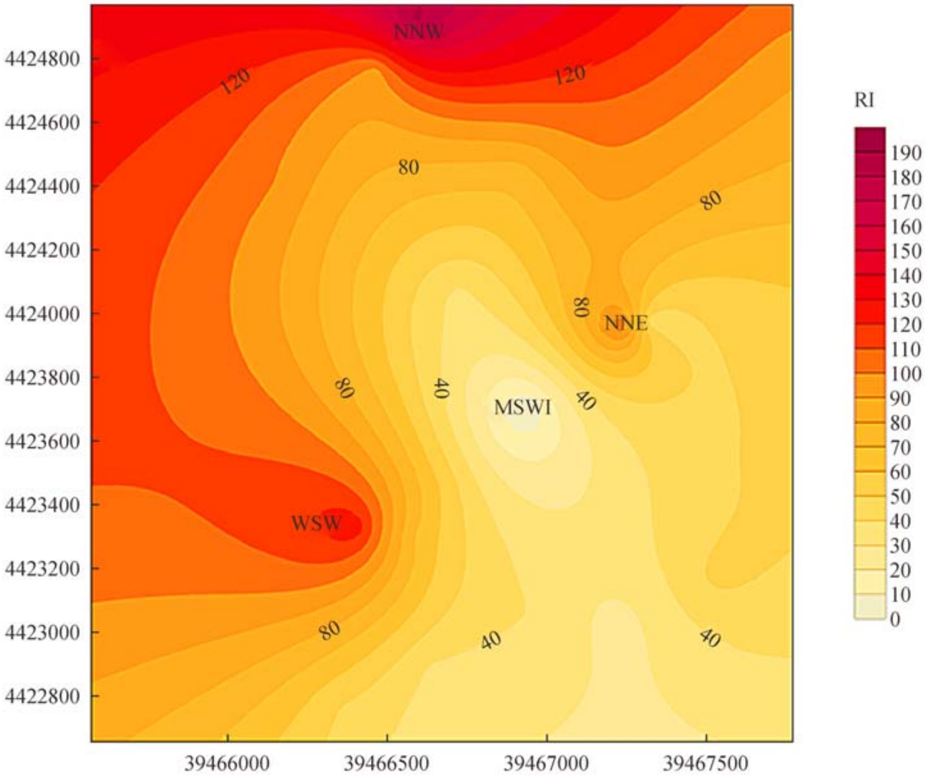

Fig. 4 Spatial risk map associated with potentially harmful elements based on the *RI* values (unitless)

high *RI* values can be identified from Fig. 4. The *RI* values in the NNW zone were higher than 150, indicating moderate risks in that district, and the *RI* values in the NNE and WSW zones have low levels of risks. From the viewpoint of the entire study area, the mean *RI* (75.7±41.4) value belonged to the low risk level, and Hg was the primary contributor to the *RI*. The results obtained from this method are similar to those obtained from the examination of the relationship between wind direction and concentrations of potentially harmful elements with respect to the distribution of the risks associated with Hg emissions from the MSWI.

3.5 Health risk assessment for the potentially harmful elements

We estimated the impact of potentially harmful elements on the health of individuals living in the area surrounding the MSWI (Table S5, Supplementary material). For the non-carcinogenic risks, we found that soil ingestion is the main exposure pathway for As, Cu, Pb, and Zn, contributing to more than 70% of the total risk. Dermal contact was found to be the main exposure pathway for Cr and Ni, and inhalation (followed by soil ingestion) was found to be the main exposure pathway for Mn. Aside from this, the inhalation of Hg in the vapor form was found to be the main exposure route for Hg, which posed significant risks to humans living near the MSWI (Fig. 5).

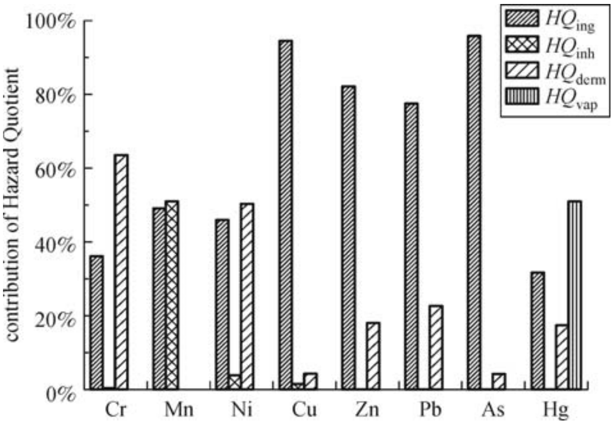

Fig. 5 Contributions of the HQs associated with each exposure pathway for the eight elements ($HQ_{\text{ing}} = HQ$ for ingestion, $HQ_{\text{inh}} = HQ$ for inhalation, $HQ_{\text{derm}} = HQ$ for dermal exposure, and $HQ_{\text{vap}} = HQ$ for Hg vapor)

The calculated HI values for the soil samples from the NW zone occurred in the following order: Cr>As>Pb >Mn>Ni>Hg>Cu>Zn. The HI values from the NE, SW, and SE zones occurred in the following order: Cr>As>Mn>Pb>Ni>Cu>Hg>Zn. The elements that posed non-carcinogenic risks to the local population were As, Cr, Mn, and Pb. However, the HQ and HI values were all below 1, indicating that the risks posed by all of the potentially harmful elements were negligible.

We only assessed the carcinogenic risks to human health for three elements: As, Cr, and Ni, which have defined slope factors. The $Risk_T$ values for Cr and Ni were below 10^{-6}, meaning that they posed nonsignificant risks to the local population. However, the $Risk_T$ value for As was 6.49×10^{-6}–9.58×10^{-6}; these values are considered to be an acceptable level of carcinogenic risk to the health of the local residents.

4 Conclusions

We determined the concentrations, spatial distribution, level of contamination, and risks of the potentially harmful elements in surrounding soil from the Sunrise MSWI in Beijing and in FA from the MSWI. The soil concentrations of potentially harmful elements around the MSWI were influenced by emissions from the MSWI. Concentrations of Hg in soil samples were found to be influenced more heavily by the MSWI more than were other elements. The potentially harmful element concentrations in soils were clearly influenced by the wind direction, and they were relatively high in the NW zone, downwind of the MSWI, compared with the concentrations in the SE zone, upwind of the MSWI; these findings were consistent with the results confirmed by results of the geoaccumulation index method. We also used the Kriging interpolation method to illustrate potential ecological risks posed by the potentially harmful elements in soil around the MSWI. Soils in the study area in general had a low potential ecological risk. The three areas that contained soil that had the highest risk were the NNW, NNE, and WSW zones. Special attention should be paid to target the lowest threat to environment. We used a health risk assessment to assess human exposure to potentially harmful elements from soils around the MSWI. There were no serious non-carcinogenic or carcinogenic risks to residents, as the risk values were lower than the threshold safe level for all elements expect for As. The carcinogenic risk value for As (6.49×10^{-6}–9.58×10^{-6}) was slightly higher than the threshold value of 10^{-6}, indicating that residents living nearby the MSWI do face a slight risk with respect to As. Based on these findings, we highly recommend that the release of pollutants from the Sunrise MSWI should be reduced, and relevant soil remediation activities should be carried out to avert potential ecological disasters.

Acknowledgements This study was supported by The National Basic Research Program of China (Grant No. 2015CB453103), the Strategic Priority Research Program of the Chinese Academy of Sciences (Grant No. XDB14020100) and the National Natural Science Foundation of China (Grant Nos. 21477150 and 21321004).

Electronic Supplementary Material Supplementary material is available in the online version of this article at http://dx.doi.org/10.1007/s11783-016-0873-7 and is accessible for authorized users.

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