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Ecological risk, seasonal and depth variations of toxic elements in a closed landfill in Vietnamese Mekong Delta province

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Abstract: The study was carried out to assess ecological risks at the closed landfill in Cai Dau town, Chau Phu district, An Giang province. Soil samples were collected at six locations (S1–S6) at depths of 0–20 cm and 60–80 cm in the rainy season (November 2020) and dry season (March 2021). The soil quality was evaluated using eight heavy metals (Mn, Fe, Cu, Zn, Ni, Pb, Cr and As). The results showed that most monitoring sites had heavy metal contents within the allowable limits for agricultural land of QCVN 03-MT:2015/BTNMT. Some heavy metals have been found to migrate to the soil layer of 60–80 cm. The heavy metals accumulation (I_{geo}) in the two soil layers did not differ; Ni, Cr, Pb and As accumulated at non-polluting levels – moderate to high in the rainy season, while Cr and As only accumulated in the dry season. The composite pollution index (*PLI*) indicated that the locations around the landfill were polluted; however, the ecological risk ranged from low to moderate (potential ecological risk index (*PERI*) = 102–195) in the rainy season. Only about 50% of study sites during the dry season were contaminated and the risk was low (*PERI* = 44–68). However, the area around the landfill always poses potential risks due to the presence of heavy metals, including Ni, Cr, As and Pb. Monitoring the heavy metals in the surrounding landfill for the potential risks to human health and environment is needed.

Keywords: environmental risks, heavy metals, landfill, soil properties

INTRODUCTION

Nowadays, along with the rapid economic growth, industrialisation and modernisation process, a large amount of waste is generated. The global municipal solid waste generation was approximately 1.3 bln Mg, and it was forecasted to increase to approximately 2.2 bln Mg by the year 2025 (Hoang *et al.*, 2019). The amount of solid waste received at landfills is increasing day by day and is of interest to all sectors and levels. The total national solid waste generated in 2007–2014 increased by 12% per y (17,682–32,000 Mg·day^{-1}) (Bộ Tài Nguyên và Môi Trường, 2017). Solid waste is generated at the rate of 0.3–1.2 kg·person^{-1}·day^{-1} depending on the rural areas and different types of urban areas, on average, about 0.75 kg·person^{-1}·day^{-1} (Bộ Tài Nguyên và Môi Trường, 2011); this rate in the Mekong Delta region is about 0.61 kg·person^{-1}·day^{-1} (Bộ Tài Nguyên và Môi Trường, 2017) corresponding to the amount of solid waste generated – 10,675 Mg·day^{-1}. According to the National State of the Environment Report for the period of 2011–2015 (Bộ Tài Nguyên và Môi Trường, 2015b), about 46% of solid waste generated is municipal solid waste, 17% of solid waste is from industrial production activities, and the rest of solid waste – in rural areas, craft villages and healthcare. Most of the solid waste is collected and treated in the improperly designed landfill, such as solid waste that has not been classified or immediately buried (Bộ Khoa học, Công nghệ và Môi trường – Bộ Xây dựng, 2001). This leads to many centralised landfills being overcrowded. In addition, landfills are not properly covered, and leachate is not treated. Leachate is water that oozes from a landfill and contains various pollutants. Rainwater and solid waste decomposition cause leachate to move into environmental components. Previous studies have shown that leachate from landfills contains many heavy metals and organic pollutants (Hussein *et al.*, 2021; Essien *et al.*, 2022; Wijekoon *et al.*, 2022). The concentration of heavy metals in leachate depends on the composition of the waste and is

considered hazardous waste because of its durable properties, not being degraded by microorganisms, and the ability to penetrate and accumulate in the wastewater, ecosystems, plants and animals, and ultimately to humans through the food chain (Klinsawathom, Songsakunrungrueng and Pattanamahakul, 2017; Rashed, 2018). Currently, there are only a few studies on the assessment of soil quality around landfills in the Mekong Delta, such as Nhien and Giao (2019), Nhien, Mi and Giao (2022); nevertheless, very little research has been done on the closed landfills in this area. This study was conducted to assess the current status of soil environmental quality and risks in the area around the landfill that has been treated and closed. The research results are basic to help managers find suitable treatment options for contaminated sites and monitor the remaining landfills in the locality.

MATERIALS AND METHODS

SOIL SAMPLING AND ANALYSIS

The sampling soil was collected in both seasons, including the rainy season (November 2020) and the dry season (March 2021). The sampling location is typical for the area affected by waste sources, the operating area of the landfill. The control field sample located about 800 m from the landfill is a pooled sample collected at 5 different locations at depths of 0–20 cm and 60–80 cm. Due to the large area, aggregate samples were collected from 5 locations at depths of 0–20 cm and 60–80 cm on the closed landfill site. Soil samples at two floors (0–20 cm and 60–80 cm) at locations where: the landfill is adjacent to the infield canal, the landfill bank is adjacent to the lake, the infield canal is 100 m away from the landfill, the landfill bank is adjacent to the fruit orchard, and on the surface of the landfill, were collected in the form of a pooled sample. Thus, a total of 12 pooled samples were collected, with 6 sampling sites in each depth. Details are presented in the sampling scheme (Fig. 1). The samples from the sites were put in plastic bags, labelled, and brought back to the laboratory. Then, all collected samples were dried under room temperature conditions, ground and passed through a 2 mm sieve to remove debris before analysing.

Eight heavy metals were analysed in the study. After acid digestion of samples with HNO_3–HCl, the flame and electro-thermal atomic absorption spectroscopy methods were used to analyse Zn, Fe, Ni, Pb, Cr and Cu. Mn and As were analysed by persulfate and graphite furnace atomic absorption

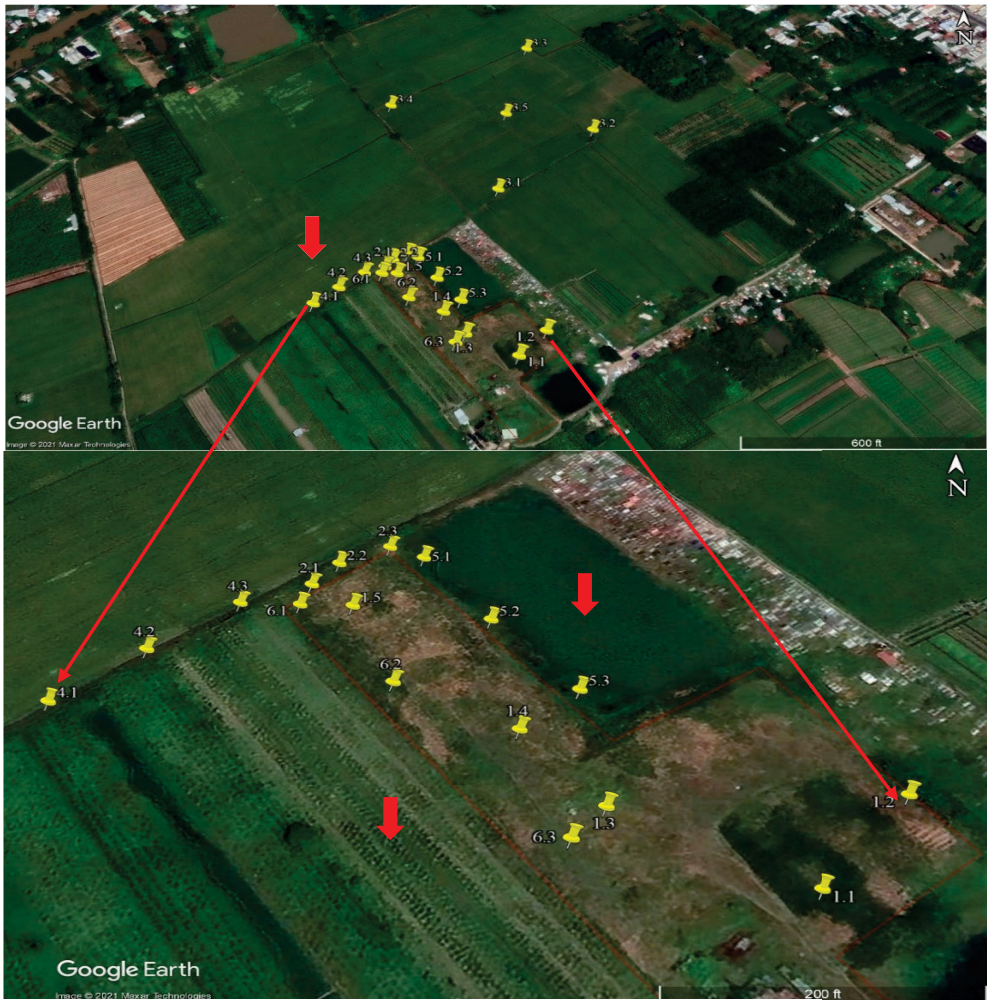

Fig. 1. Location map of soil sample collection in the study area; 1.1–1.5 = positions of pooled sample S1, 2.1–2.3 = positions of pooled sample S2, 3.1–3.5 = positions of pooled sample S3, 4.1–4.3 = positions of pooled sample S4, 5.1–5.3 = positions of pooled sample S5, 6.1–6.3 = positions of pooled sample S6; source: own elaboration based on Google Earth

spectrophotometry methods (SW-846 Test Method 7010A), respectively. The heavy metal concentrations were observed by using atomic absorption spectroscopy (Model: AAnalyst 400, Perkin Elmer). All the above analytical methods have been certified by VIMCERTS 041 of the Bộ Tài Nguyên và Môi Trường (2019).

DATA PROCESSING

Eight parameters for assessing the quality of the soil environment in the study area are statistically based on six sampling sites, depth (0–20 cm and 60–80 cm) and sampling time (rainy season and dry season). Evaluation of the mean difference of sampling criteria by the depth and by season was analysed by the independent sample T-Test method using IBM SPSS 20.0 Windows statistical software. The concentration of heavy metals was compared with the national technical regulation QCVN 03-MT:2015/BTNMT on the allowable limit of some heavy metals in agricultural land (Bộ Tài nguyên và Môi trường, 2015a). The geoaccumulation index (I_{geo}) is used to estimate pollution levels of individual heavy metals; meanwhile, the pollution load index (*PLI*) – for the quantitative application of the overall pollution level of heavy metals and potential ecological risk index (Tomlinson *et al.*, 1980). The potential ecological risk index (*PERI*) was used to identify potential ecological risks posed by heavy metals and was proposed by Hakanson (1980). These indices can be calculated according to the following formulas:

– geoaccumulation index

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

– pollution index

$$PI = \frac{C_i}{B_i} \quad (2)$$

– pollution load index

$$PLI = \sqrt[n]{PI_1 \cdot PI_2 \cdot PI_3 \cdot \ldots PI_i} \quad (3)$$

– potential ecological risk index

$$PERI = \sum_{i=1}^{n} T_x^i \cdot PI_i \quad (4)$$

where: C_i = actual monitoring value of parameter *i*, B_i = average concentration of heavy metal *i* in the background level (with Fe = 24,733; Mn = 1,226; Pb = 15; Zn = 47.16; Cu = 19.78; Cr = 3.4; Ni = 1.7; As = 1.9) (Kabata-Pendias and Pendias, 2001; Nguyen *et al.*, 2021), 1.5 = correction factor for the background matrix due to the petrographic effect, *n* = amount of heavy metal, *PI* = pollution index, PI_1, PI_2, PI_3 = heavy metal pollution index 1, 2, 3; PI_i = heavy metal pollution index *i*; T_x^i = ecotoxicity coefficient (Mn = Zn = 1, Cr = 2, Cu = Ni = Pb = 5, As = 10) (Hakanson, 1980). The values of each indicator are then evaluated against the cut-off intervals shown in Table 1.

Table 1. Assessment of heavy metal pollution in soil

Index	Ranges	Pollution indication
I_{geo}	$I_{geo} \leq 0$	not polluted
	$0 < I_{geo} \leq 1$	not polluted to moderate
	$1 < I_{geo} \leq 2$	moderate
	$2 < I_{geo} \leq 3$	moderate to high
	$3 < I_{geo} \leq 4$	high
	$4 < I_{geo} \leq 5$	high to very high
	$5 < I_{geo}$	very high
PLI	$PLI \leq 1$	not polluted
	$1 < PLI \leq 2$	moderate
	$PLI > 2$	high
PERI	$PERI < 150$	low
	$150 \leq PERI < 300$	moderate
	$300 \leq PERI < 600$	high
	$PERI \geq 600$	very high

Explanations: I_{geo} = geoaccumulation index, *PLI* = pollution load index, *PERI* = potential ecological risk index.
Source: own elaboration based on Thongyuan *et al.* (2020).

RESULTS AND DISCUSSION

EVALUATION OF HEAVY METAL FLUCTUATIONS BY DEPTH AND BY SEASON

Mn occurs naturally in water, soil and plants, which is essential for the normal growth of animals and plants. The Mn content in the study at the 0–20 cm layer had an average value in the range of 81.45 ±12.59–237.64 ±103 mg·kg^{-1}, the lowest at position S2 and the highest at position S1. Meanwhile, Mn at the 60–80 cm layer fluctuates in the range of 47.38 ±5.30–313.72 ±204 mg·kg^{-1}, the lowest Mn level is at position S3 and the highest – at position S1. This shows that Mn level was affected by landfills. Mn at most positions between two layers was not statistically significant, except for the positions S4 and S5 scores (Fig. 2a). Mn tends to accumulate high in the 0–20 cm horizon except for the positions S1 and S2. Research by Nguyen (2020) shows that among the studied heavy metals, the Mn content at Dong Thang landfill in the city of Can Tho has a high value with an average of 0–25 cm, ranging from 234.0 ±8 to 321.0 ±2 mg·kg^{-1}, higher than in the present study. Mn content between two seasons at locations ranges from 44.57 ±27.84 to 135.3 ±8.99 mg·kg^{-1} (rainy season) and 85.09 ±29 to 416.06 ±92.34 mg·kg^{-1} (dry season) (Fig. 2b). Mn fluctuates seasonally, with Mn level being higher in the dry season than in the rainy season.

Fe content at each position in the 0–20 cm layer varied on average from 21,926.55 ±1,584.94 to 29,314.68 ±1,864.15 mg·kg^{-1}, the lowest content was at S3 and the highest – at S6. In the 60–80 cm layer, Fe has an average value in the range of 23,617.60 ±1,042.36–26,271.57 ±758.50 mg·kg^{-1}, the lowest and highest levels were at S2 and S1, respectively. At locations S1, S2, S3 and S6, the Fe content between the two floors was statistically significant. In contrast, positions S4 and S5 had no significant difference (Fig. 3a). Fe content in the control field (S3) tended to

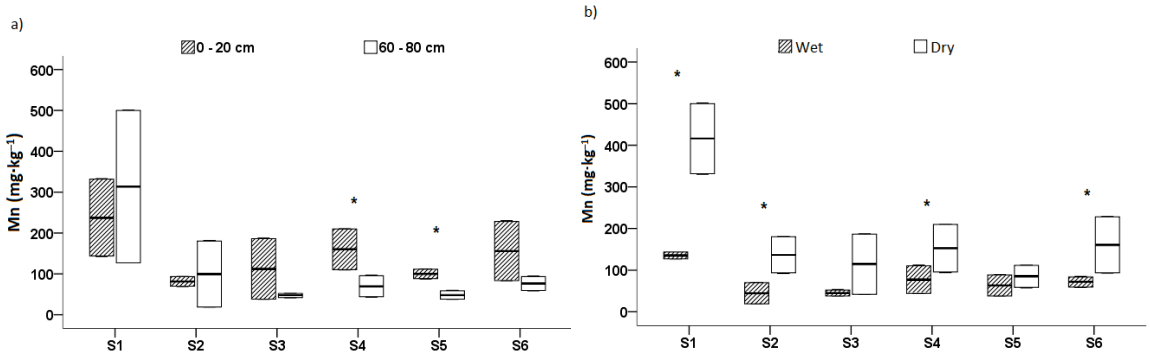

Fig. 2. Variation of Mn content (mg·kg^{-1}) with: a) depth, b) season; S1–S6 as in Fig. 1; source: own study

fluctuate in the opposite direction compared to the other positions. Fe content in the surface layer 0–20 cm is lower than that in the layer 60–80 cm due to the process of moving and accumulating Fe over time (Cung, 2008; Nguyen, 2020). Fe content in the rainy season and dry season at the locations was in the range of 24,529.47 ±493.54–29,669.19 ±1,344.72 mg·kg^{-1} and 21,028.2 ±1,937.24–27,134.21 ±1,022 mg·kg^{-1}, respectively (Fig. 3b). A significant difference in Fe content was only recorded between the two seasons at positions S3, S5 and S6 ($p < 0.05$), in which Fe in the rainy season was higher than in the dry season except for positions S1 and S2.

Cu content at depths of 0–20 cm and 60–80 cm was 17.97 ±4.48–27.27 ±4.10 mg·kg^{-1} and 19.03 ±3.19–31.39 ±8.75 mg·kg^{-1}, respectively. The study results of Cu content between the two soil layers at all locations did not have a statistically significant difference (Fig. 4a). The fluctuation of Cu content at positions S1, S4 and S6 at the depth of 60–80 cm tended to accumulate higher than in the 0–20 cm layer because the mobility of heavy metals in the soil depends on many factors. Factors include time, the concentration of pollutants and soil properties (Hoàng, 2018). Cu content in the rainy season ranges from 21.97 ±0.38 to 35.2 ±4.59 mg·kg^{-1}; meanwhile, the dry season was assessed to have a significantly lower value (15.03 ±1.38–23.46 ±0.15 mg·kg^{-1}) (Fig. 4b). Statistical analysis also showed a significant difference between the two seasons at all locations ($p < 0.05$).

The average concentration of Zn at each location by depth in the 0–20 cm layer ranged from 47.25 ±6.56 to 61.05 ±6.54 mg·kg^{-1}, the lowest was at S2 and the highest – at S4. Zn in the layer 60–80 cm averaged from 40.34 ±3.74 to 61.50 ±16.07 mg·kg^{-1}, the lowest was at S2 and the highest – at S1. Zn in all positions between the two locations was not statistically significant (Fig. 5a). The average value of Zn in the rainy and dry seasons at each location is in the range of 48.25 ±5.5–70.32 ±6.64 mg·kg^{-1} and 38.87 ±3.48–48.88 ±7.68 mg·kg^{-1}, respectively (Fig. 5b). The Zn content at the studied sites between the two seasons was statistically significant ($p < 0.05$). In contrast to Mn, the mean value of Zn in the wet season is higher than that of Zn in the soil in the dry season; this is similar to the fluctuations of Fe and Cu.

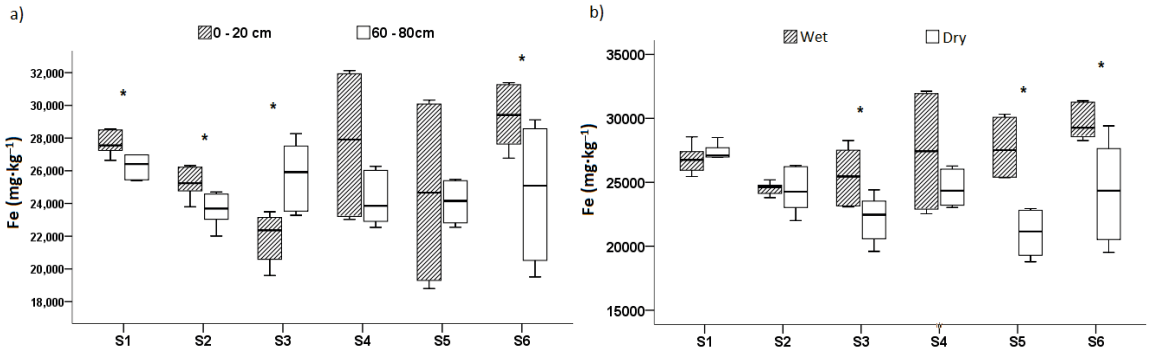

Fig. 3. Variation of Fe content (mg·kg^{-1}) with: a) depth, b) season; S1–S6 as in Fig. 1; source: own study

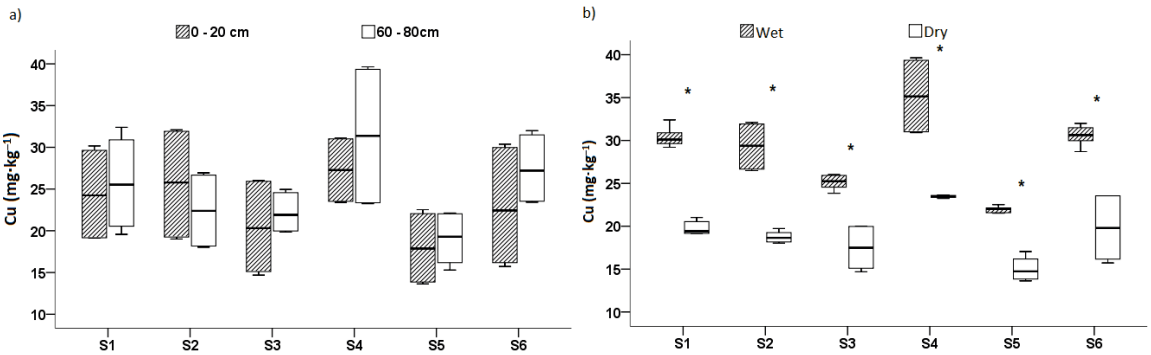

Fig. 4. Variation of Cu content (mg·kg^{-1}) with: a) depth, b) season; S1–S6 as in Fig. 1; source: own study

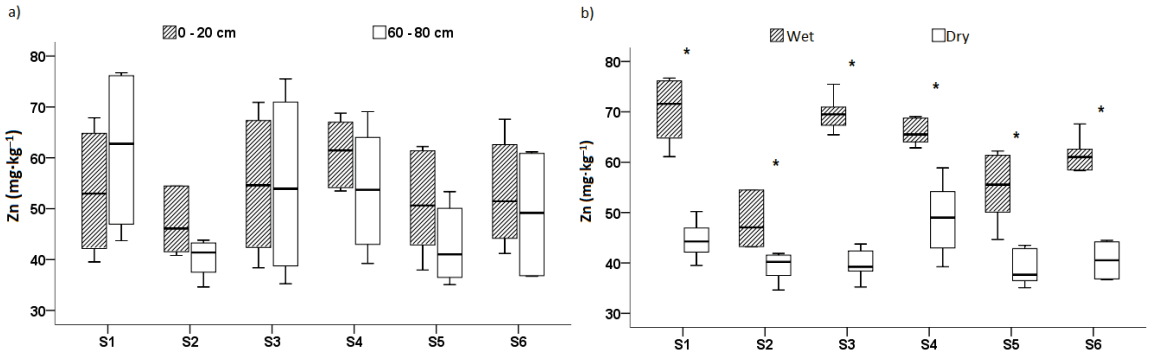

Fig. 5. Variation of Zn content (mg·kg^{-1}) with: a) depth, b) season; S1–S6 as in Fig. 1; source: own study

The average concentration of Ni at each position in the 0–20 cm layer ranged from 8.99 ±9.85 to 12.67 ±13.98 mg·kg^{-1}, the lowest was at S3 and the highest – at S6. Ni at the 60–80 cm layer is in the range of 8.45 ±9.33–16.85 ±18.62 mg·kg^{-1}, the lowest concentration was at S5 and the highest – at S1. The difference in Ni content between the two strata was not statistically significant (Fig. 6a). Similar to the study by Nguyen (2020) at the Dong Thang landfill in Can Tho, the Ni content also tends to be similar, increasing gradually with depth. However, the Ni concentration at Dong Thang landfill is relatively higher than in the current study area. Ni content was not detected during dry season monitoring; this resulted in statistically significant differences at all locations. Figure 6b shows that in the rainy season, the Ni value at each location ranges from 19.44 ±3.24 to 27.76 ±7.05 mg·kg^{-1}, and the highest occurs at S1 – landfill.

The average concentration of Pb at each position in the 0–20 cm layer ranged from 19.66 ±6.04 to 24.88 ±13.07 mg·kg^{-1}, the lowest was at S1 and the highest – at S3. Meanwhile, the concentration of Pb in the layer 60–80 cm varied from 24.73 ±2.64 to 29.74 ±1.59 mg·kg^{-1}, the lowest and highest were found at S5 and S1, respectively. Pb at most positions between the two sampling stages was different without statistical significance except for positions S1, S5 and S6 (Fig. 7a). In the rainy season, the average value of Pb at each location ranged from 23.95 ±3.67 to 34.41 ±2.94 mg·kg^{-1} (Fig. 7b). In the dry season, the Pb content fluctuates between 18.47 ±5.99 and 23.12 ±3.63 mg·kg^{-1}. There was a statistically significant difference between the two seasons at positions S3, S4 and S6, in which lead concentration tended to be higher in the rainy season than in the dry season.

The average content of Cr at each location ranged from 28.51 ±16.21 to 40.01 ±30.57 mg·kg^{-1} in the 0–20 cm layer, the lowest content was at S3 and the highest – at S6. At the layer 60–80 cm, the average content of Cr ranged from 28.68 ±16.44 to 38.34 ±24.00 mg·kg^{-1}, the lowest and highest were at S5 and S1, respectively. The results showed that the Cr content between the two strata at all locations was different without statistical significance ($p > 0.05$, Fig. 8a). Cr concentration at each location in the study, in the rainy and dry seasons, ranged from 48.84

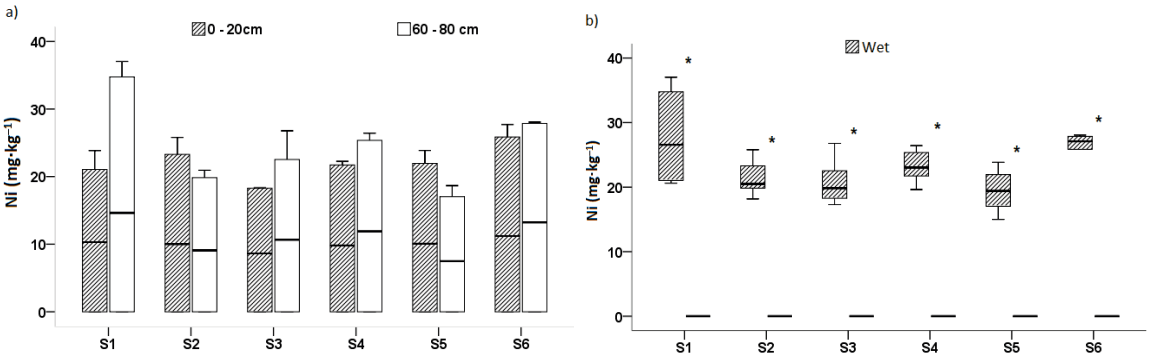

Fig. 6. Variation of Ni content (mg·kg^{-1}) with: a) depth, b) season; S1–S6 as in Fig. 1; source: own study

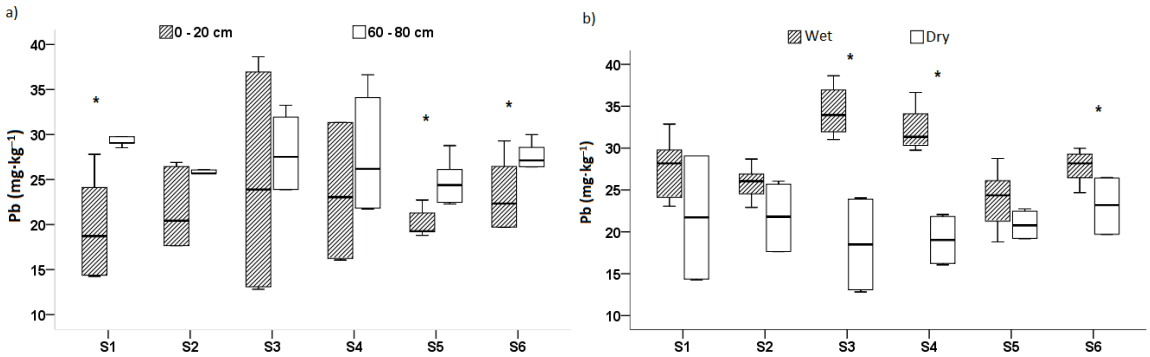

Fig. 7. Variation of Pb content (mg·kg^{-1}) with: a) depth, b) season; S1–S6 as in Fig. 1; source: own study

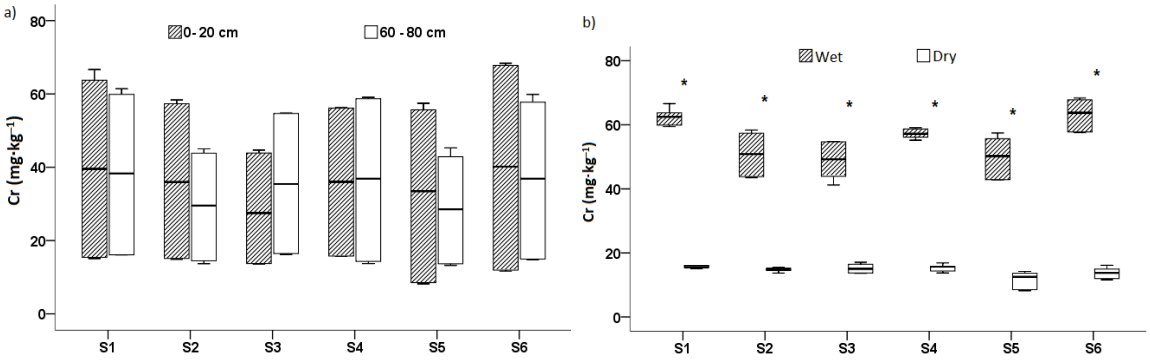

Fig. 8. Variation of Cr content (mg·kg^{-1}) with: a) depth, b) season; S1–S6 as in Fig. 1; source: own study

±6.21 to 63.17 ±5.26 mg·kg^{-1} and from 11.59 ±2.64 to 15.9 ±0.74 mg·kg^{-1}, respectively. The difference in Cr content was statistically significant ($p < 0.05$) at all locations between the two seasons (Fig. 8b). This significant disparity can be explained by the addition of a large amount of water in the rainy season, leading to an increase in leachate from the closed landfill into the soil environment. This variation trend was found for similar metals such as Fe, Cu, Zn, Ni and Pb.

The average concentrations of As in the surface and deep layers were 3.62 ±0.59–7.00 ±1.23 mg·kg^{-1} and 3.44 ±1.68–7.69 ±0.39 mg·kg^{-1}, respectively (Fig. 9a). As is highest at position S4 and lowest at position S1 of the 0–20 cm layer. Meanwhile, at the level of 60–80 cm, the average As value is the lowest at S5 and the highest at S1. As content had no statistically significant difference ($p > 0.05$) at all positions between the two sampling strata. The average concentration of As at each study site in the rainy and dry seasons ranged from 3.32 ±1.55 to 6.73 ±0.97 mg·kg^{-1} and from 4.16 ±0.32 to 7.96 ±0.32 mg·kg^{-1}, respectively. As content fluctuated significantly according to the season; As concentration in most locations in the dry season was higher than in the rainy season (Fig. 9b). Particularly for positions S3 and S6, the difference in As content was not statistically significant ($p > 0.05$).

Statistical analysis results also show that all heavy metals have a statistically significant difference in content between the two seasons; they tended to be lower in the dry season, with up to 6 out of 8 heavy metals contents recorded higher in the rainy season. Therefore, the depth and seasonal variation of soil sampling also significantly influenced the concentrations of the metals studied. The accumulation of excessive heavy metals in the ecosystem will greatly affect the ecology and public health (Hussein *et al.*, 2021; Karimian, Shekoohiyan and Moussavi, 2021).

ECOLOGICAL RISK ASSESSMENT

The $I_{geo} > 0$ indicates that most soil samples can be considered to have heavy metal accumulation in the soil (Tab. 2). In particular, according to the rating scale, the concentration of heavy metals appears at medium to very high levels compared to the natural background value and the typical soil value in Chau Phu district, except for Ni in the dry season (Tab. 2).

Accumulation levels of Cr and Ni were determined to be high, and Pb – from uncontaminated to moderate pollution at all study sites in the rainy season. Besides, locations S1, S2 and S6 in the rainy season are also moderately polluted with As. Accumulation was found to be lower in the dry season in both soil layers, typically of Cr and Ni. This can be explained by water as one of the main factors controlling the movement of these trace elements in the soil by changing the pH (Kim and Lee, 2009). Therefore, enhanced soil metal transport can occur during rainy periods. In addition, Cr is classified as an element with limited mobility; therefore, its accumulation occurs locally (Kabata-Pendias and Pendias, 2001). The locations showed different contamination levels for each metal because no clear trend was observed for all sites. Even so, a trend can be easily seen for the accumulation of Ni and Cr at positions S1 and S6 in the rainy season (Tab. 3); specifically, there was the highest accumulation at S6 in the 0–20 cm layer, while in the 60–80 cm strata was recorded the highest accumulation at S1. This may indicate an effect on perennial crop cultivation (S6) and the decomposition of the landfilled litter (S1). In addition, the Ni and Cr accumulation values at the 0–20 cm layer in the rainy season are at a longer distance than the landfill (S3) with a lower pollution level. This trend may explain the potential impacts of long-term landfilling.

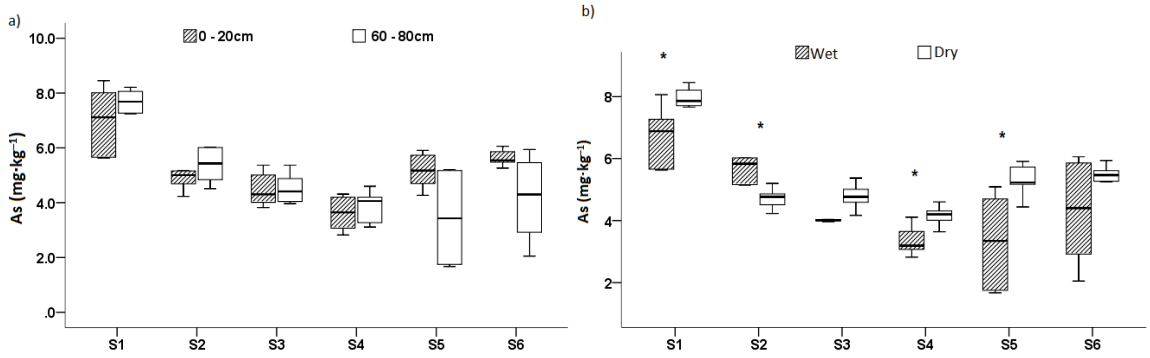

Fig. 9. Variation of As content (mg·kg^{-1}) with: a) depth, b) season; S1–S6 as in Fig. 1; source: own study

Table 2. Indicators of heavy metal accumulation (I_{geo}) in the rainy season

Soil layer (cm)	Sites	Mn	Fe	Cu	Zn	Ni	Pb	Cr	As
0–20	S1	–3.68	–0.43	0.00	–0.13	3.10	0.15	3.66	1.06
	S2	–4.72	–0.59	0.11	–0.41	3.18	0.18	3.49	0.93
	S3	–5.59	–0.67	–0.20	–0.06	2.82	0.71	3.08	0.47
	S4	–4.06	–0.22	0.06	–0.09	3.06	0.45	3.45	0.16
	S5	–4.38	–0.31	–0.43	–0.23	3.11	–0.10	3.46	0.72
	S6	–4.45	–0.27	0.00	–0.17	3.31	0.25	3.74	1.03
60–80	S1	–3.85	–0.51	0.07	0.10	3.72	0.43	3.56	1.40
	S2	–6.58	–0.60	–0.15	–0.70	2.95	0.22	3.11	1.05
	S3	–5.14	–0.42	–0.28	0.02	3.21	0.51	3.42	0.65
	S4	–5.38	–0.70	0.41	–0.12	3.30	0.58	3.52	0.30
	S5	–5.59	–0.55	–0.44	–0.52	2.73	0.26	3.10	–0.55
	S6	–4.95	–0.37	0.08	–0.23	3.43	0.35	3.52	–0.04

Explanations: S1–S6 as in Fig. 1.
Source: own study.

Table 3. Indicators of heavy metal accumulation (I_{geo}) in the dry season

Soil layer (cm)	Sites	Mn	Fe	Cu	Zn	Ni	Pb	Cr	As
0–20	S1	–2.47	–0.42	–0.63	–0.75	–	–0.65	1.59	1.50
	S2	–4.31	–0.52	–0.62	–0.77	–	–0.34	1.56	0.69
	S3	–3.30	–0.85	–0.99	–0.77	–	–0.79	1.43	0.81
	S4	–3.13	–0.66	–0.33	–0.35	–	–0.47	1.66	0.51
	S5	–4.04	–0.94	–1.10	–0.77	–	–0.23	0.90	0.98
	S6	–3.01	–0.41	–0.89	–0.71	–	–0.18	1.25	0.94
60–80	S1	–1.88	–0.49	–0.54	–0.59	–	0.37	1.69	1.46
	S2	–3.35	–0.70	–0.71	–0.93	–	0.20	1.51	0.77
	S3	–5.43	–0.64	–0.57	–0.90	–	0.09	1.70	0.68
	S4	–4.27	–0.53	–0.34	–0.74	–	–0.04	1.51	0.58
	S5	–4.97	–0.70	–0.88	–0.96	–	0.00	1.42	0.79
	S6	–4.30	–0.85	–0.34	–0.90	–	0.23	1.58	0.96

Explanations: S1–S6 as in Fig. 1.
Source: own study.

In addition, the *PLI* and *PERI* indices at S1 (on the landfill surface) and S6 (fruit growing area) in the 0–20 cm layer in the rainy season can also indicate that the soil has been at high levels of heavy metal pollution and moderate ecological risk; meanwhile, in the soil at the 60–80 cm layer, high pollution levels and medium ecological risks were recorded only at site S1. The *PLI* values at the remaining sites were found to be in medium contamination level with low ecological risk considering the total number of metals studied and ranged between 1.57 (rice fields) and 1.92 (canal near the landfill) for 0–20 cm level, and between 1.31 (banks adjacent to the lake) and 1.78 (rice fields) for 60–80 cm soil layer (Fig. 10a). From the above observations, it is

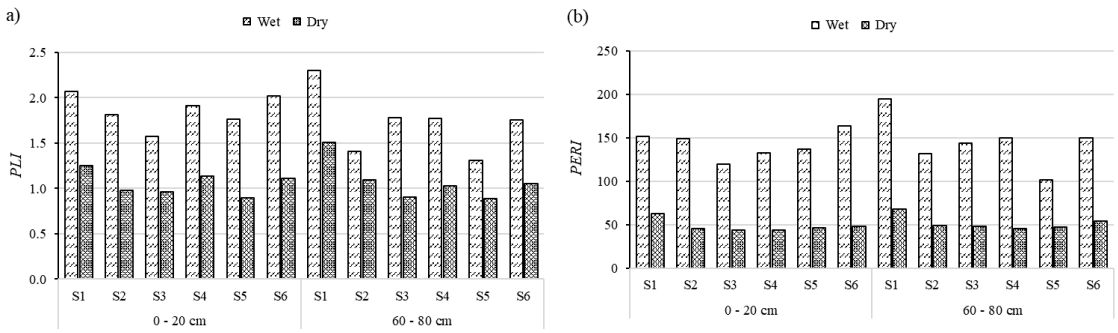

Fig. 10. Pollution levels and ecological risks by season and depth: a) pollution load index (*PLI*), b) potential ecological risk index (*PERI*); S1–S6 as in Fig. 1; source: own study

shown that the ecological risk for the area in the rainy season is higher than in the dry season and in the 60–80 cm layer is higher than in the 0–20 cm layer (Fig. 10b). Several other studies reported contamination by trace metals in the soil around municipal solid waste landfills, posing a risk of ecological toxicity (Ferronato and Torretta, 2019; Vongdala *et al.*, 2019). The study's results demonstrated that the soil was contaminated with heavy metals, especially Cr, Ni, As and Pb at the landfill and in the surrounding areas. This could cause concern for people cultivating in the area, especially in the fruit-growing area.

CONCLUSIONS

The study results of the soil environment around the Cai Dau landfill show that the metal content has not significantly impacted the soil environment. All soil quality parameters' concentrations have seasonal fluctuations, while only Mn, Fe, and Pb concentrations have been recorded to fluctuate with depth. The soil layer of 0–20 cm tended to be higher than that of the soil layer of 60–80 cm. Mn, Zn and Fe are mostly retained in the 0–20 cm layer, but Cu, Pb and As penetrate the 60–80 cm layer deeper. At the same time, no certain trend was recorded in Ni and Cr contents. There is lower heavy metal content in the dry season than in the rainy season. The I_{geo} values of Ni and Cr showed that all soil samples had high levels of heavy metal accumulation, Pb and As – from no pollution to medium pollution. The *PLI* value showed that all soil samples were moderate to highly contaminated concerning heavy metal concentrations, except for the area of rice fields (S3) and adjacent to lakes (S5). The value of *PERI* indicates a low to moderate potential ecological risk. Ni, Cr, Pb and As contributed the most to the ecological risks associated with heavy metals in the study area. Therefore, the possibility of causing heavy metal pollution to crops is very large, specifically the location S6 – the adjacent bank of the fruit tree landfill area, by the land in the area for agricultural purposes. Studies on the possibility of heavy metal accumulation in crops and landfill-related pollution remediation measures are needed in the future. The risk assessment methods in this study can be applied to landfills to assess the impact of landfill activities on the environment and health and have solutions for prevention and response measures.

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