



Research Article

Heavy Metals and Organic Carbon in Sediments of Seagrass Sediments of Trang Province, Thailand

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Abstract

In this study, the accumulation of heavy metals (Cd, Cr, Cu, Fe, and Pb) in water, suspended sediments, sediments, and seagrass in Kalase Bay, Trang Province, Thailand, during the 2024 dry season was investigated. These findings indicate that the enrichment factor (EF) for all the metals was less than 1, suggesting that anthropogenic contamination is not a significant concern in the area. The translocation factor (TF) values were less than 1 for all the metals except Cu, whose TF was greater than 1; however, these values were not statistically significant, indicating limited phytoextraction capacity. The bioconcentration factor (BCF) values for Cu and Pb in seagrass were also less than 1, indicating no substantial accumulation of these metals in seagrass tissues. Additionally, the Cd and Cr concentrations were below the detection limits, further indicating their negligible presence. This study revealed a strong correlation between the concentrations of heavy metals and fine sediment fractions, with suspended sediments playing a critical role in the transport and distribution of these metals. The presence of sand bars and seagrass beds in the region likely influences sediment dynamics, contributing to the retention and bioavailability of metals. The sediment carbon-to-nitrogen (C/N) ratio ranged between 10 and 20, indicating that mixed organic matter sources are commonly associated with coastal sediments. This suggests that sediment organic matter originates from multiple inputs rather than from a single dominant source. These results underscore the importance of implementing a comprehensive monitoring program, including suspended sediment analysis, to track heavy metal accumulation and sediment changes, which is crucial for informed conservation and management of the unique ecosystem of Kalase Bay and its carbon storage potential.

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Introduction

Aquatic environments, especially marine environments, e.g., mangrove and seagrass environments, even those considered remote and pristine, are increasingly threatened by anthropogenic pollution, including heavy metals, excess nutrients, and various emerging contaminants (Haynes and Johnson, 2000; Na Talang et al., 2024). These pollutants pose significant risks to vital marine ecosystems, such as seagrass meadows, which provide essential ecological services and support biodiversity. Often referred to as ecosystem engineers (Bos et al., 2007), seagrasses play a vital role in carbon sequestration, shoreline stabilization, and providing nurseries for marine species. They help mitigate climate change by absorbing atmospheric carbon and storing it in sediments (Lima et al., 2023). In addition, seagrass meadows protect coastlines by reducing wave velocities, increasing sediment accumulation, and preventing erosion. Despite these crucial services, seagrass populations are rapidly declining worldwide, largely because of anthropogenic stressors, including land-based pollution and climate change (Ahmad-Kamil et al., 2013; Lincoln et al., 2021).

The sensitivity of seagrasses to environmental changes underscores the need for further research to understand and protect these ecosystems. Factors such as sediment composition—specifically carbon and nitrogen content—and physicochemical parameters of water, including salinity, pH, total alkalinity, and nutrient concentrations (phosphorus and nitrate), are known to directly influence seagrass growth. However, nutrient enrichment, while essential for growth, can also have detrimental effects. For example, excessive nitrate concentrations can be toxic to seagrasses (Burkholder et al., 1992). In Fatumaru Bay and Epau, high phosphate concentrations (9.33 mg P L⁻¹ and 7.33 mg P L⁻¹, respectively) have been linked to significant declines in seagrass cover (Lincoln et al., 2021). High salinity and low pH also negatively affect seagrass growth (Lincoln et al., 2021; Ramfos et al., 2024). Ilhami et al. (2020) reported that seagrasses thrive under slightly alkaline conditions (pH 8.10–8.50), with photosynthesis rates declining under more acidic conditions (pH 7.50–7.75).

Seagrass meadows in Southeast Asia, including Thailand, play a vital role in blue carbon storage but remain understudied compared with global estimates. Studies in southern Thailand (Stankovic et al., 2023) have revealed significant differences in carbon sequestration potential across seagrass species and sediment dynamics. Regional reviews (Ooi et al., 2018) highlight knowledge gaps and urgent conservation needs due to habitat degradation. Strengthening local monitoring and policy frameworks is critical to protect these ecosystems and their carbon sinks. In addition to environmental factors, anthropogenic pollutants, particularly heavy metals such as cadmium (Cd), lead (Pb),

and copper (Cu), pose a significant threat to seagrass meadows. Studies suggest that seagrass has potential as a natural phytoremediation agent, sequestering heavy metals from seawater (Lafratta et al., 2023). However, once seagrass meadows die, these metals may be released back into the environment. Seagrasses sequester heavy metals differently depending on species and environmental conditions, with certain metals accumulating in specific organs, such as roots or leaves (El Zrelli et al., 2023; Singh et al., 2021). Seagrasses can also serve as effective biomonitors for heavy metal contamination in marine environments with metals such as Cd and Pb, posing significant risks to human health because of their carcinogenic and neurotoxic properties. Regular monitoring of heavy metals in seawater, sediments, and seagrass is crucial for tracking contamination impacts, and preserving and restoring seagrass meadows is essential for maintaining marine ecosystem health (Ren et al., 2024). Rising global temperatures can threaten marine ecosystems, including seagrass meadows, which are crucial for carbon sequestration. This warming may also intensify the effects of pollution, making it essential to assess heavy metal contamination and its connection to climate change for effective seagrass management and restoration. The summer period from February to May 2024, which was hotter than the previous year, contributed to this decline. Consistent with these conditions, widespread coral bleaching was also observed in early 2024 as a consequence of ocean heating (Byrne et al., 2025).

Moreover, the carbon concentration in seagrass sediment is particularly important, as higher levels indicate more productive seagrass meadows. Seagrass sediments are characterized by high organic carbon content because of the accumulation of decomposing plant material and fine-grained particles, which increase carbon sequestration. Moreover, seagrass increases sediment accumulation rates and reduces sediment grain size to silt (Bos et al., 2007). The grain size typically consists of silt and clay, promoting organic matter preservation. These sediments play a vital role in coastal carbon storage (Fourqurean et al., 2012a).

Despite the ecological importance of seagrass meadows, studies on trace metal accumulation, sediment characteristics, and blue carbon in seagrass habitats—particularly in areas connected to sandbars—are limited. This is the case in Kalase Bay, which is located in Sikao District, Trang Province, southern Thailand. This bay, home to seagrass meadows that serve as critical feeding grounds for dugongs, has experienced a decline in seagrass coverage. The findings of this study are expected to provide valuable insights to guide the conservation and restoration of seagrass ecosystems, ensuring the long-term health of this vital marine habitat.

Materials and methods

1) Study area and sample collection

The study area is in the seagrass region of Kalase Bay (Figure 1), within the Khao Mai Kaew subdistrict of Sikao District, Trang Province, southern Thailand, approximately 1 km north of the Laem Sai and Thung Thong fishing communities. Sample collection took place in early May 2024.

Kalase Bay features a prominent sandbar that is exposed during low tide, dividing the study area into inner and outer zones. The inner zone, which is located on the landward side of the sandbar near the seawall, includes sampling stations S11, S12, S13, and C1. The outer zone, seaward beyond the sandbar, includes stations S21, S22, S23, S31, S32, and S33. The C1 station is a designated area with seagrass growth enclosed by a 7 m × 7 m bamboo pole fence, with poles spaced 20–30 cm apart. The seagrass density is approximately 4 to 5 plants per 50 cm × 50 cm quadrant. The dominant species in this area is *Enhalus acoroides*.

To prevent cross-contamination, all the equipment used for surface sediment, water, and seagrass sampling was designated for specific stations and sample types. The equipment was thoroughly cleaned and acid-washed before use. Sampling was conducted in the afternoon during the ebbing tide. Surface sediment samples were

collected to a depth of 20 cm from all the stations and stored in plastic zip-lock bags, with three replicate samples taken at each station. These replicate samples were then combined to form one composite sample for analysis. Water samples were collected from 10 to 20 cm below the surface at stations S11, S12, and S13 using a 1 L polyethylene bottle and immediately preserved with 1% nitric acid (HNO₃). Nine whole seagrass plants were randomly collected from station C1 and placed into plastic zip-lock bags. A summary of the sample collections is provided in Table 1. During the sampling period, the water salinity ranged from 31.9 to 32.3 ppt, and the water temperature ranged from 33.44°C to 33.72°C. All samples, including surface sediment, water, and seagrass, were packed in coolers and stored at -20°C for subsequent laboratory analysis. In this report, the term “sediment” refers to surface sediment, and “suspended sediment” specifically refers to sediment that is suspended in the water column.

A key limitation of this study is that sampling was conducted only during the dry season, as site access during the monsoon season was limited by weather and field conditions. Because metal distribution and sediment dynamics in coastal environments vary seasonally, the results may not fully represent conditions during other periods of the year.

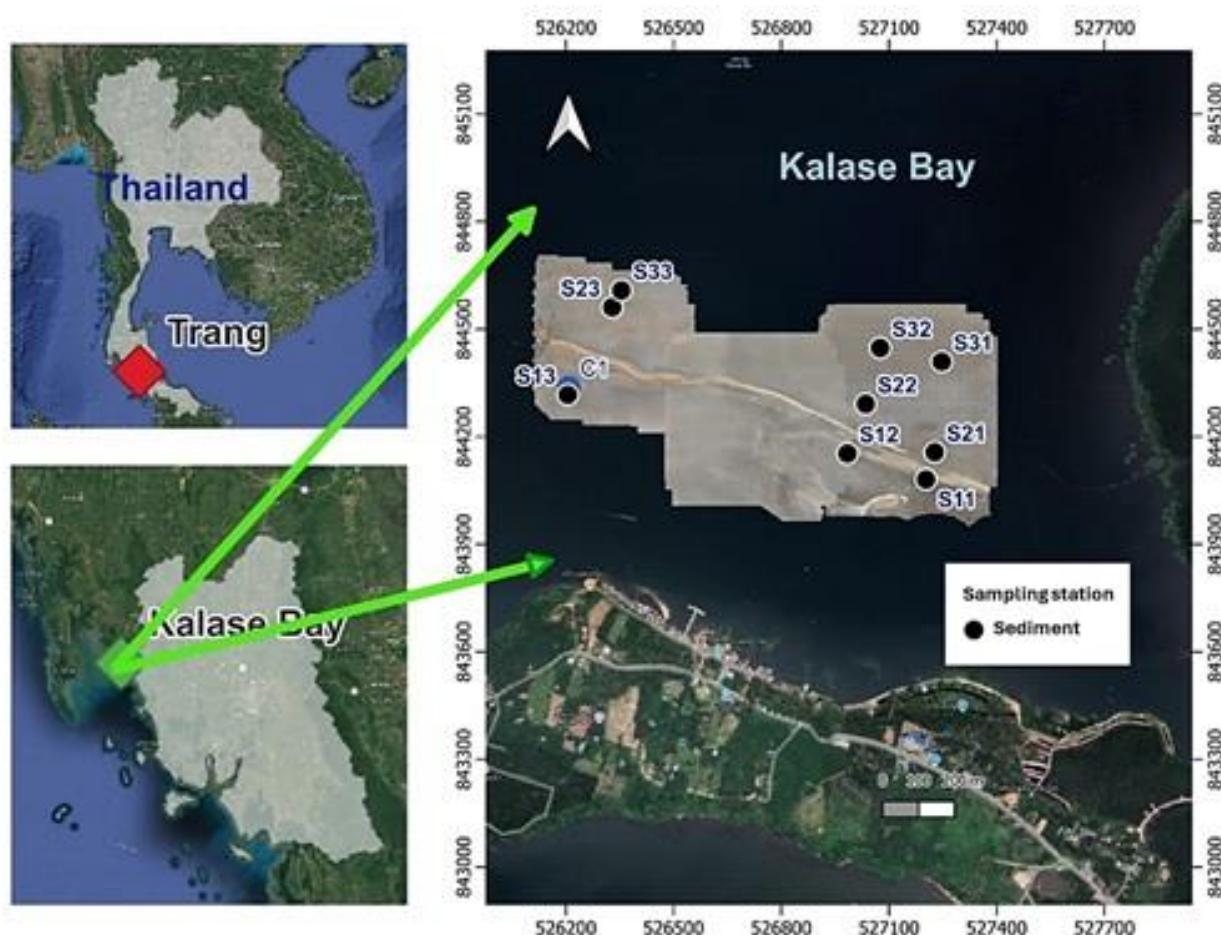


Figure 1 Sampling stations in the seagrass region in Kalase Bay, Trang Province, Thailand.

Table 1 Summary of sampling locations and collection at Kalase Bay, Trang Province, Thailand.

Sample collection	Inner zone sampling station				Outer zone sampling station					
	S11	S12	S12	C1	S21	S22	S23	S31	S32	S33
Sediment	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Water and suspended sediment	✓	✓	✓							
Seagrass				✓						

2) Heavy metal analysis

Sediment: Sediment samples were oven-dried at 105°C to remove all moisture content before further processing. The drying process was monitored until the samples reached a consistent weight, ensuring that they were fully dry. The dried sediment was then homogenized, ground in a mortar, and sieved through a 1-mm nylon sieve. More than 0.3 grams of sieved sediment was weighed and transferred into test tubes. To each tube, 2 mL of concentrated nitric acid (HNO₃) and 6 mL of concentrated hydrochloric acid (HCl) were added (Pradit et al., 2010). The mixture was then digested in a temperature-controlled water bath at approximately 95°C for 30 minutes. After digestion, the samples were allowed to cool to room temperature. Once cooled, the volume was adjusted to 10 mL with Milli-Q water, and the samples were stored for heavy metal analysis.

Water: Water samples were filtered through preoven-dried and weighed GF/C filter paper to separate the suspended sediments. The filtered water was then set aside for digestion. The suspended sediments collected on the filter paper were dried in an oven at 105°C for 1 h to remove moisture, cooled in a desiccator, and weighed to determine the total suspended solids (TSS). Owing to the low amount of suspended sediment, the filter containing the dried suspended sediment was carefully transferred into test tubes and subjected to the same digestion procedure as the sediment samples. After cooling, the samples were filtered with Whatman No. 1 paper, and the volume was adjusted with 10 mL of Milli-Q water. For the filtered water samples, 50 mL of the filtered water was mixed with 5 mL of concentrated HNO₃ and 2.5 mL of concentrated HCl. The mixture was digested in a temperature-controlled water bath at 95°C for 30 min. Once the samples were cooled to room temperature, the volume was adjusted to 50 mL with Milli-Q water, and the samples were stored for heavy metal analysis.

Seagrass: Seagrass samples were divided into three replicates for analysis. Before analysis, the seagrass was washed with distilled water to remove mud, sand, and other adhered particles. After cleaning, the seagrass was divided into aboveground (leaf) and below-ground (root) components. The separated samples were then oven-dried at 60–70°C until fully dry, as indicated by a consistent weight across multiple measurements. The dried samples were ground into a fine powder and stored in clean polyethylene bags. More than 0.5 g of each powdered sample was placed

into test tubes, and 2 mL of HNO₃ was added. The mixture was digested in a water bath at 95°C for 1 hour. Afterward, 0.5 mL of 30% H₂O₂ was added. The samples were then left to cool at room temperature. After cooling, the samples were then filtered with Whatman No. 1 paper, and the volume was adjusted with 10 mL of Milli-Q water and stored for heavy metal analyses. The analysis of total heavy metals in the plant samples was conducted following the AOAC method (Horwitz and Latimer, 2005). All the elements were measured with an inductively coupled plasma optical emission spectrophotometer (ICP-OES), Perkin-Elmer, model Optima 4300 DV.

Quality control: The analytical accuracy was verified using the marine sediment-certified reference material MESS-4, with recovery percentages for Cu, Pb, and Fe ranging from 80–120%, which were within the acceptable range. The reagent blanks were below the detection limits, indicating that there was no significant contamination. The detection limits were approximately 0.03 mg kg⁻¹ for Cu and 0.15 mg kg⁻¹ for Pb in sediment and seagrass samples on the basis of instrument detection limits and dilution factors. The calibration curves achieved R² > 0.995 for all the elements. Although duplicate analyses were not performed, the use of CRM and blanks provides confidence in the data quality.

3) Determination of organic carbon and nitrogen in sediment

The total organic carbon (TOC) and total nitrogen (TN) in the sediment samples were analyzed using a PerkinElmer® CHNS/O Elemental Analyzer (model 2400 Series II). Prior to analysis, inorganic carbon, mainly present as CaCO₃, was removed by adding 10% (v/v) HCl to 2 g of dried sediment in a glass tube. The solution was left until it became clear, after which the solution was discarded. The acid addition was repeated until no more gas bubbles appeared, and the solution remained clear. Afterward, the sediment was rinsed with DI water until the pH was equal to that of distilled water. After rinsing, the sediment was dried and finely ground before being placed in a precleaned tin capsule. The sample was weighed using a Perkin Elmer™ AD 6 Auto Balance Controller with a precision of 5 decimal places. The weighed sample was then packed in a tin capsule for analysis. To determine the percent reproducibility (%RPD), 10% duplicate samples were analyzed. The accuracy of the method was maintained and monitored by analyzing the acetanilide standard (0240

–1121) for every 20 samples. The recommended percentage of C:H:N for the standard is 16.5:50:6, respectively. The results are reported as %C for organic carbon (OC).

4) Sediment grain size analysis

Particle size analysis was performed using the hydrometer method (Gee and Bauder, 1986). Additionally, laser particle size analysis was performed with the ANALYSETTE 22 nanoTec plus (Fritsch) for samples from stations C1, S12, S22, and S32.

5) Data processing and analysis

All calculations and statistical analyses were carried out using MS Excel 2007 (Office Professional Plus 2019) software. The normality of the data distribution was tested using the Shapiro–Wilk method. Variables that were not normally distributed ($p < 0.05$) were identified. Consequently, to examine the relationships between the physicochemical parameters and heavy metal concentrations, Spearman's rank correlation coefficient was employed as a nonparametric test.

Quantitative contamination indices and their characterization standards were used to evaluate the level of trace metal pollution at the study sites. These included the enrichment factor (EF), the geoaccumulation index (I_{geo}), and the ecological risk index (R_i).

The enrichment factor (EF) is an important indicator that reflects the influence of environmental trace metal pollution and anthropogenic activities on heavy metal enrichment. The elemental concentrations of the samples were compared to those of stable elements found in specific soils, aiming to identify and characterize anthropogenic contamination of trace metal elements in environmental media (Akoto and Anning, 2021). The enrichment factors for the trace metals were calculated using Eq. 1.

$$EF = (C_i/C_{Fe})_{Sample} / (C_i/C_{Fe})_{Background} \quad (\text{Eq. 1})$$

where (C_i/C_{Fe}) is the ratio of the metal concentration in the sample to the concentration of Fe and $(C_i/C_{Fe})_{Background}$ is the ratio of the metal concentration in the background to the concentration of Fe (Esfandiari and Hakimzadeh, 2022).

Metal concentrations were normalized using iron (Fe) because of its natural abundance and status as a conservative element in the Earth's composition (Ibrahim and Selim, 2022). In the absence of site-specific background values (e.g., from deep sediment cores) for the intertidal seagrass environment, background reference values were adopted from offshore Gulf of Thailand sediments reported by Shazili et al. (1999). We acknowledge that offshore sediments represent a different depositional environment with potentially different natural metal concentrations because of variations in grain size, organic

matter content, hydrodynamic conditions, and sediment sources. Intertidal seagrass sediments typically have relatively coarse grain sizes and high organic matter inputs from seagrass production and experience fluctuating redox conditions because of tidal cycles. Therefore, enrichment factors and geoaccumulation indices calculated using these offshore references should be interpreted as relative indicators rather than absolute measures of anthropogenic enrichment.

The enrichment factor (EF) was calculated to assess metal enrichment relative to the background. EF values were interpreted following Trajković et al. (2024), where $EF < 1$ indicates depletion relative to background; $EF = 0.5–1.5$ suggests metals derived from crustal materials or natural processes; and $EF > 1.5$ indicates anthropogenic influence.

The geoaccumulation index (I_{geo}) was proposed by Müller (1969) to evaluate the pollution levels of each trace metal in marine environments. I_{geo} was determined using Eq. 2.

$$I_{geo} = \log_2 C_n / 1.5B_n \quad (\text{Eq. 2})$$

where C_n is the total concentration of the element in the sample and B_n is the geochemical background value of the trace metals. A factor of 1.5 is the background matrix correction factor to account for the influences of weathering and lithogenic factors (Chen et al., 2023). The index of geoaccumulation consists of seven classes: extremely contaminated ($I_{geo} > 5$), strongly to extremely contaminated ($4 < I_{geo} \leq 5$), strongly contaminated ($3 < I_{geo} \leq 4$), moderately to strongly contaminated ($2 < I_{geo} \leq 3$), moderately contaminated ($1 < I_{geo} \leq 2$), uncontaminated to moderately contaminated ($0 < I_{geo} < 1$), and uncontaminated ($I_{geo} < 0$).

The potential ecological risk index (R_i) is widely used to assess the degree of potential ecological risk of trace metals in the environment. The R_i for trace metals is calculated as the sum of the risk factors using the equations proposed by Hakanson (1980). The formula for the potential ecological risk index is shown in Eqs. 3 and 4.

$$R_i = \sum_{i=1}^n E_i \quad (\text{Eq. 3})$$

where:

$$E_i = T_i(C_i/C_0) \quad (\text{Eq. 4})$$

In these equations, E_i represents the individual ecological risk factor for each trace metal, T_i is the toxicity-response factor for the specific metal, C_i is the concentration of the metal concentration in the samples, and C_0 is the background concentration of the metal. The potential ecological risk index is obtained by summing the individual risk factors, E_i , for all n metals

In accordance with Hakanson (1980), the following toxicity response factors were used: Pb = 5, and Cu = 5 (Fadlillah et al., 2023; Hakanson, 1980). To describe the ecological risk factor, the following terminology was used: low ($R_i < 150$), moderate ($150 \leq R_i \leq 300$), high ($300 \leq R_i \leq 600$), and high ($R_i > 600$).

Bioconcentration factors (BCFs) indicate the efficiency with which a plant species absorbs a specific element from sediment and accumulates it in its tissues. The BCF of individual metals in seagrass (both leaf and root) was calculated using Eq.5 (Zhang et al., 2024).

$$BCF = C_{\text{seagrass}}/C_{\text{sediment}} \quad (\text{Eq. 5})$$

where C_{seagrass} and C_{sediment} are the metal levels (mg kg^{-1} dry weight) in the seagrass tissues and the sediment (mg/kg dry weight), respectively.

A $BCF > 1$ indicates active uptake by seagrass, whereas a $BCF < 1$ indicates higher heavy metal concentrations in soil than those taken up by plants (Jeong et al., 2021).

The translocation factor (TF) reflects the mobility and transport of heavy metals within a plant, specifically from the roots to the stems and leaves. The TF is calculated using Eq. 6.

$$TF = C_{\text{aboveground tissues}}/C_{\text{belowground tissues}} \quad (\text{Eq. 6})$$

where $C_{\text{aboveground tissues}}$ and $C_{\text{belowground tissues}}$ represent the heavy metal concentrations in the aboveground and belowground tissues of seagrass, respectively (Pradit et al., 2018).

When the TF is ≤ 1 , the ability of the plant to translocate heavy metals from the roots to the above-ground parts is low. In contrast, a $TF > 1$ indicates efficient translocation of heavy metals from the roots to the above-ground parts of the plant.

Results and discussion

1) Heavy metals in surface sediment

The total suspended sediment concentration in the seawater samples ranged from 20.0 to 24.8 mg L^{-1} , with an average of $24.3 \pm 4.2 \text{ mg L}^{-1}$, indicating typical low suspended sediment levels for coastal waters in the area. The concentrations of heavy metals in the sediment, seawater, suspended sediment, and seagrass in Kalase Bay, Trang Province, are summarized in Table 2. Cu and Pb were the only metals detected, as Cd and Cr were below detectable levels (0.01 mg L^{-1}). In the sediment, the concentrations of the observed heavy metals varied spatially. The average concentration of Cu in the sediment ($1.10 \pm 0.23 \text{ mg kg}^{-1}$) was lower than that of Pb ($4.44 \pm 0.95 \text{ mg kg}^{-1}$). Similarly, the average concentration of Cu in the suspended sediment ($0.52 \pm 0.00 \text{ mg kg}^{-1}$) was lower than that of Pb ($2.36 \pm 1.91 \text{ mg kg}^{-1}$). In seawater, however, the average concentration

of Cu ($0.15 \pm 0.05 \text{ mg L}^{-1}$) was higher than that of Pb, which was below detectable levels ($< 0.01 \text{ mg L}^{-1}$). This suggests that Pb is more strongly adsorbed (Wu and Ren, 2011) and is more likely to adsorb to and be transported with suspended particles.

Spatial distribution mapping (Figure 2) indicated that Cu and Pb concentrations were generally higher in the inner zone (landward side of the sandbar) than in the outer zone (seaward side). Although hydrodynamic measurements were not available to confirm the controlling mechanisms, this pattern is likely related to reduced hydrodynamic energy, as sandbars can attenuate wave energy and lower current velocities, promoting the settling of suspended particles and metal-bound fine sediments (Wang and Zou, 2016). Increased water residence time in the semienclosed inner zone may further increase particle settling and metal adsorption onto sediments (Valle-Levinson, 2022). This interpretation is supported by the strong correlation of Cu and Pb with silt and clay suggesting that fine-grained sediments with high adsorption capacity preferentially accumulate under lower-energy conditions. In addition, the closer proximity of the inner zone to terrestrial sources may also contribute to the observed enrichment.

It is recommended that areas with sand bars implement comprehensive monitoring and control measures for heavy metals, including suspended solids and their associated metal concentrations. If the sandbar indeed traps metal-contaminated sediments, the inner zone may become increasingly vulnerable to heavy metal pollution over time, particularly if terrestrial inputs increase. This highlights the need for integrated coastal management that considers both water quality and geomorphological features. Early monitoring and proactive conservation efforts are essential to prevent further pollution and protect this fragile coastal ecosystem.

The results of this study (Table 3) were compared with sediment quality guidelines from the U.S. Environmental Protection Agency (1989), the Canadian Council of Ministers of the Environment (CCME) (2025), and the National Oceanic and Atmospheric Administration (NOAA) (1999). These guidelines were selected because they are widely used in international coastal sediment research, facilitate comparisons with studies from other regions, and provide multiple assessment frameworks (pollution classification, ISQG/PEL, ERL/ERM) that offer complementary perspectives on sediment quality. While official numerical values for Thailand's sediment quality guidelines (SQGTs) are not publicly available (Lapat-atiwat, 2021), a site-specific SQG has recently been developed for a Thai coastal environment; Jiwrungrueangkul et al. (2024) established locality-adapted SQGs for Bandon Bay, Thailand, using an equilibrium partitioning approach, providing values of 84.6 mg kg^{-1} for Cu and 34.6 mg kg^{-1} for Pb on a

normalized basis. These values offer a valuable regional benchmark for comparison. The mean concentrations of Cu (1.10 mg kg^{-1}) and Pb (4.44 mg kg^{-1}) in this study are well below both the international guidelines and this Thai site-specific reference value.

Table 2 Heavy metal concentrations in sediment, seawater, suspended sediment, and seagrass samples from Kalase Bay, Trang Province

Samples	Values	Heavy metal concentration		
		Cu	Pb	Fe
Sediment (mg/kg) (n=10)				
All	Average $\pm$ S.D.	1.10 $\pm$ 0.23	4.44 $\pm$ 0.95	7,512.35 $\pm$ 842.63
	Min	0.90	3.45	6,247.51
	Max	1.80	5.95	8,564.65
Inner zone	Average $\pm$ S.D.	1.50 $\pm$ 0.40	4.82 $\pm$ 1.05	8,417.93 $\pm$ 664.58
	Min	0.94	3.45	6,247.51
	Max	1.80	5.95	8,564.65
Outer zone	Average $\pm$ S.D.	1.07 $\pm$ 0.18	4.32 $\pm$ 0.75	7,050.32 $\pm$ 690.17
	Min	0.90	3.64	6,247.51
	Max	1.41	5.51	7,835.14
Seawater (filtered) (mg L ⁻¹) (n=3)	Average $\pm$ S.D.	0.15 $\pm$ 0.05	ND	ND
	Min	0.09	ND	ND
	Max	0.18	ND	ND
Suspended sediment (mg kg ⁻¹) (n=3)	Average $\pm$ S.D.	0.52 $\pm$ 0.00	2.36 $\pm$ 1.91	603.14 $\pm$ 237.49
	Min	0.52	0.72	444.20
	Max	0.52	4.52	876.14
Seagrass				
Leave (mg kg ⁻¹) (n=3)	Average $\pm$ S.D.	1.07 $\pm$ 0.48	1.04 $\pm$ 0.15	957.66 $\pm$ 269.85
	Min	0.63	0.90	691.45
	Max	1.59	1.20	1,231.00
Root (mg kg ⁻¹) (n=3)	Average $\pm$ S.D.	0.55 $\pm$ 0.71	1.49 $\pm$ 2.00	1335.49 $\pm$ 187.68
	Min	0.38	0.97	1,202.78
	Max	0.71	2.00	1,468.20

Note: ND = not detected (concentration below the detection limit). Detection limits: 0.01 mg/L for seawater; 0.03 mg/kg for Cu; and 0.15 mg/kg for Pb in sediment and seagrass (based on method detection limits). Cadmium (Cd) and chromium (Cr) were below the detection limits in all the samples and were therefore excluded from this table.

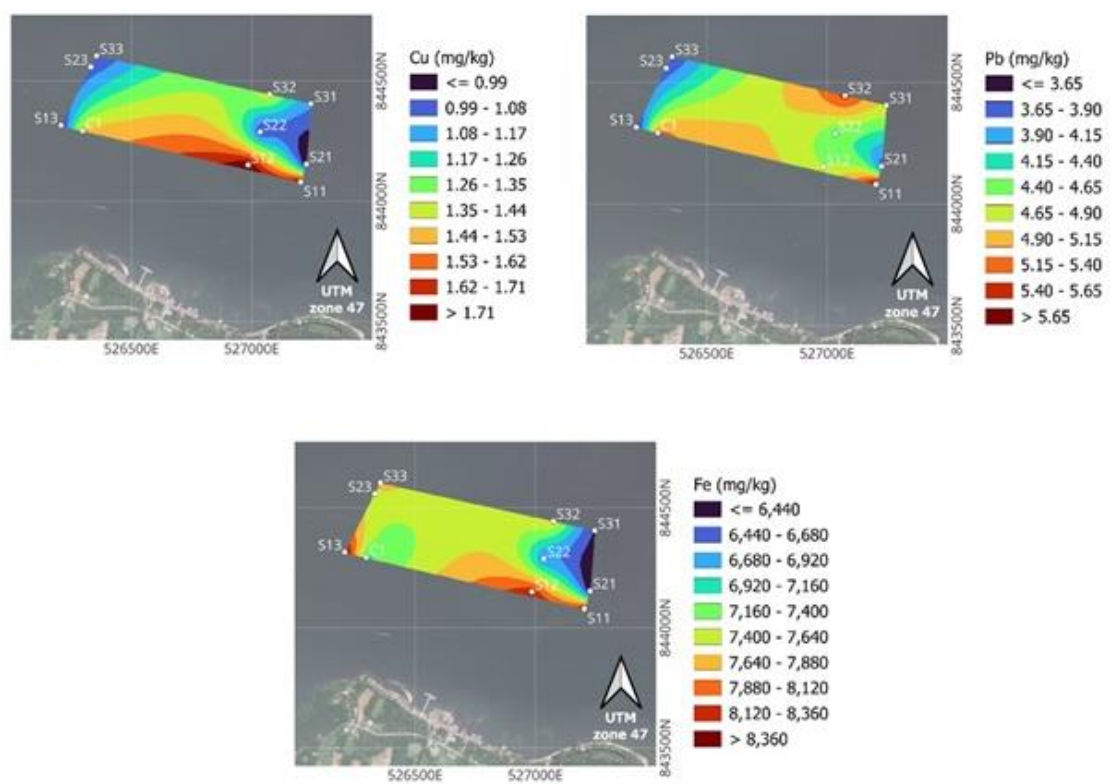


Figure 2 Distribution of heavy metals in the surface sediment.

Table 3 Sediment quality guidelines (mg kg⁻¹) were compared with Thai site-specific reference values

Metals	Not polluted	U.S EPA. [§]	Heavily polluted	CCME [£]		NOAA [§]		Thailand* (Site-Specific SQG)*
		Moderately polluted		ISQG [£]	PEL [£]	ERL [§]	ERM [§]	
Cd	<25	25-75	>75	0.7	4.2	1.2	9.6	-
Cr	<25	25-75	>75	52.3	160.0	81	370	-
Cu	<25	25-50	>50	18.7	108.0	34	270	84.6
Pb	<40	40-60	>60	30.2	112.0	46.7	218.0	34.6
Fe	<17,000	17,000-25,000	>25,000	ND	ND	ND	ND	-

Note: ND = no available data.

[§] US EPA guidelines (1989).

[£] CCME guidelines: ISQG = Interim sediment quality guidelines, PEL = Probable effect levels (2025).

[§] NOAA guidelines: ERL = Effects range-low, ERM = Effects range-median (1999).

* Thai site-specific SQG values from Jiwarungrueangkul et al. (2024) for Bandon Bay, Thailand.

The mean values of Cu and Pb were below all guidelines. Excessive Cu can impair fish osmoregulation by disrupting their salt balance. The Cu concentrations observed in this study are consistent with the findings of Kobkeathhawin et al. (2021), who reported similar levels in the bare mudflat near the seagrass area at Libong Island, Trang Province. However, the mean Pb concentration in this study was more than three times greater than that in the mudflat at Libong Island. This discrepancy may be due to the higher retention of Pb within the seagrass meadow than within the bare mudflat. The individual characteristics and distribution patterns of Cu and Pb in the sediments of Kalase Bay are discussed separately below.

Copper (Cu)

The Cu concentrations ranged from 0.90 to 1.80 mg kg⁻¹ (mean 1.10 ± 0.23 mg kg⁻¹), with relatively low spatial variation. Cu was strongly correlated with fine-grained sediments (silt: $r = 0.875$, $p < 0.01$; clay: $r = 0.904$, $p < 0.01$) and strongly negatively correlated with sand ($r = -0.919$, $p < 0.01$) (Table 5). These findings indicate that the Cu distribution is primarily controlled by sediment texture, with greater accumulation in finer sediments where the surface area and organic matter content are greater. The enrichment factor for Cu (EF = 0.11–0.21) indicates depletion relative to the background, suggesting natural sources rather than anthropogenic inputs. As an essential micronutrient, Cu at these low concentrations (below all international guidelines) poses no ecological risk and may even benefit seagrass physiology.

Lead (Pb)

The Pb concentrations ranged from 3.45 to 5.95 mg kg⁻¹ (mean 4.44 ± 0.95 mg kg⁻¹), approximately four times higher than the Cu concentration. Pb was also positively correlated with fine sediments (silt: $r = 0.623$, $p < 0.05$; clay: $r = 0.604$, $p < 0.05$), although the correlations were weaker than those for Cu, suggesting the presence of additional controlling factors beyond simple grain size control. The enrichment factor for Pb (EF =

0.33–0.62) was higher than that for Cu but still < 1 , indicating relative depletion. EF values closer to the 0.5–1.5 range are considered indicative of crustal sources (Trajković et al., 2024).

Pb is a nonessential element with no biological function and is toxic even at low concentrations. Lead has a high adsorption affinity for sediment particles, particularly organic matter and iron/manganese oxides, which explains its tendency to accumulate in depositional environments. The higher Pb concentrations than Cu concentrations, despite both having EFs < 1 , may reflect (i) a greater natural abundance of Pb in source rocks; (ii) stronger binding to sediments, leading to greater retention; and (iii) possible historical atmospheric deposition (although the current inputs appear minimal).

2) Heavy metals in seagrass

Seagrasses are widely used as bioindicators because of their ability to accumulate metals. When in contact with sediments, seagrasses absorb metals both from the water column and from the sediment. The average concentration of heavy metals in seagrass leaves was 1.07 mg kg⁻¹ for Cu (range: 0.63–1.59 mg kg⁻¹) and 1.04 mg kg⁻¹ for Pb (range: 0.90–1.20 mg kg⁻¹) (Table 2). In seagrass roots, the average concentration was 0.55 mg kg⁻¹ for Cu (range: 0.38–0.71 mg kg⁻¹) and 1.49 mg kg⁻¹ for Pb (range: 0.97–2.00 mg kg⁻¹) (Table 2). Seagrasses take up nutrients and heavy metals from the sediment via their roots or from the water column via their shoots (Martin et al., 2022). Additionally, several factors influence the accumulation of Pb and Cu in seagrasses, including seagrass species, tissue type, redox reactions, seasonal variations, the specific metals involved, metal concentrations in the water and sediments, and the characteristics of the sediments themselves (Yan et al., 2020).

Despite these findings, given the unique and pristine nature of this environment, owing to valuable seagrass ecosystems, a continuous monitoring program of heavy metals in sediments, seawater, suspended sediments, and seagrasses remains essential to identify the early detection of any emerging threats and enables timely

management interventions to protect the health and sustainability of this ecosystem.

3) Grain size distribution

The composition of each sample, which included clay, sand, and silt, is shown in Figure 3, with sand constituting the greatest proportion, followed by clay and silt. The highest percentage of silt (9.95%) was found at station S11, whereas the lowest (1.09%) was observed at station S23. Compared with the other samples, station S23 also had the lowest clay content (13.9%). On the other hand, the sand content at station S23 was the highest (85.02%), while that at station S11 was the lowest (70.52%). The water content ranged from 20% to 24%.

The sediment grain sizes varied significantly across the locations, with a range of grain types, including clay, sand, and silt, but no gravel. The grain sizes typically ranged from fine silt to coarser sand particles. This variation is likely due to the low-energy wave environment, which promotes the settling of fine particles. Understanding grain size is crucial for assessing environmental conditions, as it influences the types of organisms that can inhabit the area and the ability of the sediment to retain pollutants, such as heavy metals. Therefore, knowledge of sediment grain size is important for interpreting ecological dynamics and pollution potential.

In the study area, a sand dune runs parallel to the commercial line, separating the inner stations (S11, S12, S13, and C1) from the outer stations (S21, S22, S23, S31, S32, and S33). Notably, compared with both the inner and outer stations, the stations along the middle line (S21, S22, and S23) have lower silt contents. This could suggest that the middle area experiences less turbulence than the other two regions do.

The grain size distribution curves (Figure 4) for the four samples (stations C1, S12, S22, and S32) indicate the presence of at least three components—approximately 2, 10, and 100 microns—with the majority falling in the 100-micron range. The maximum detected grain size was less than 300 microns. The arithmetic mean

grain sizes for samples C1, S12, S22, and S32 were 89, 78, 106, and 79 microns, respectively. The approximate ratio of sand to silt to clay across all the samples was 62:26:12. Notably, the sand fraction in sample S22 was 69%, which is higher than the proportions observed in the other samples, indicating a distinct grain size composition at this station.

4) Organic carbon (OC) and nitrogen

The data on OC and nitrogen levels in seagrass sediment are presented in Figure 5. The highest OC percentage was observed in sample S12, whereas the lowest was found in sample S23. OC levels are crucial indicators of carbon storage, nutrient cycling, and overall ecosystem health, as they reflect the amount of carbon in organic compounds present in the sediment.

The carbon-to-nitrogen (C/N) ratio for this study ranged between 10 and 20, which falls within the range of mixed sources for organic matter in coastal sediments. This range suggests contributions from multiple organic matter sources rather than a single dominant source. This mixed-source interpretation is consistent with the coastal setting, where seagrass meadows receive organic matter from both autochthonous (in situ seagrass production) and allochthonous (terrestrial runoff, tidal import) sources (Kennedy et al., 2010). The absence of C/N ratios >20 suggests a limited dominance of fresh terrestrial vascular plant material, whereas the absence of ratios <10 indicates that marine phytoplankton are not the primary source. The intermediate values suggest that seagrass detritus is likely a significant contributor, but not the exclusive source of sedimentary OC. While the OC levels in coastal sediments can indeed be influenced by anthropogenic factors (such as urban waste), the findings of this study do not explicitly support a dominant anthropogenic source. The C/N ratio indicates a relatively more terrestrial or biological source of carbon, which aligns with the presence of seagrass ecosystems known for their carbon sequestration abilities (Fourqurean et al., 2012a).

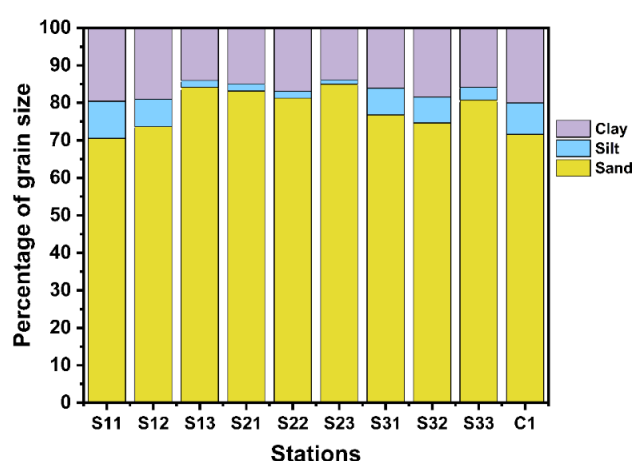


Figure 3 Percentage of grain size in the study area.

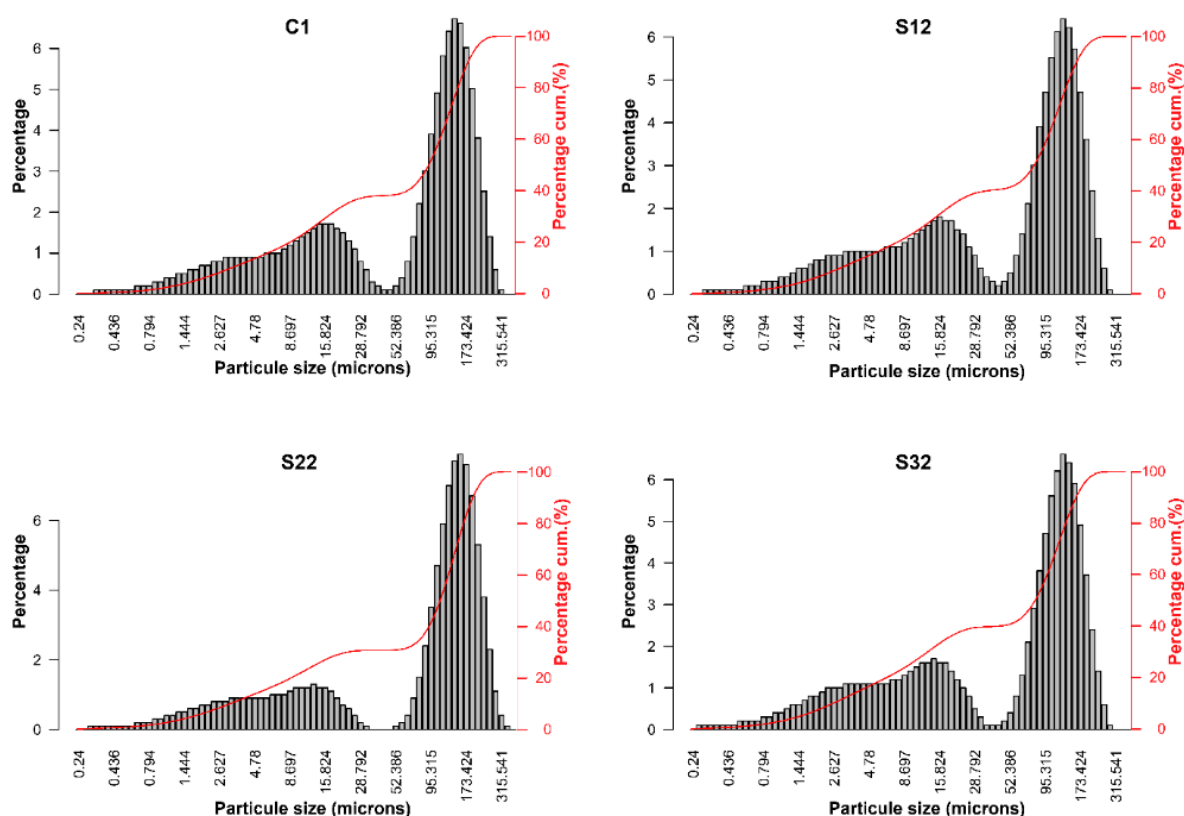


Figure 4 Grain size distribution curves.

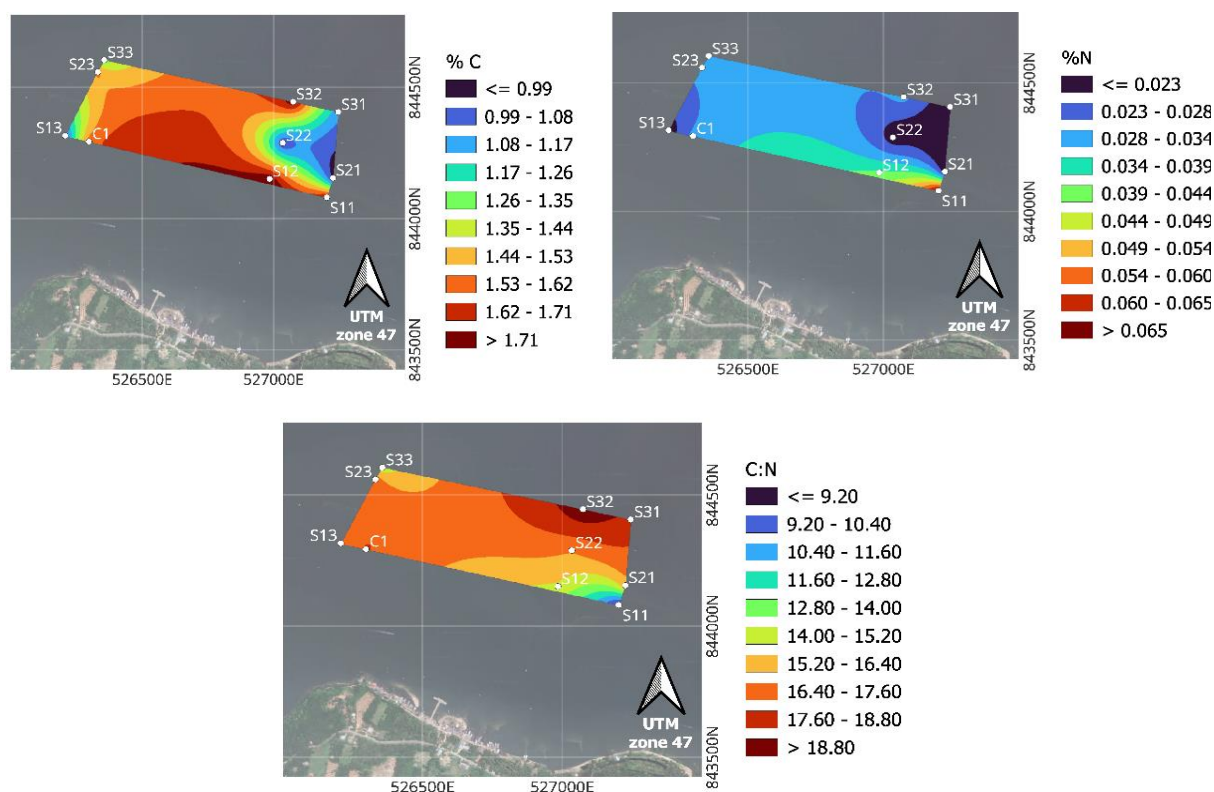


Figure 5 Distribution of %C, C:N, and %N in the seagrass area.

The OC concentrations reported here (0.32–0.60%) provide only a preliminary indication of surface carbon content. The observed OC values are within the lower range of global seagrass meadows (0.5–5% in surface sediments) (Fourqurean et al., 2012b). Without depth-integrated stocks, we cannot determine whether this

meadow functions as a significant long-term carbon sink. Maintaining optimal OC levels in seagrass sediments is critical for ensuring the ecosystem's role in carbon storage while minimizing the risks of environmental degradation.

Further studies should explore the long-term trends in OC storage and how different factors, such as sediment composition and hydrodynamics, affect the resilience of these unique ecosystems in terms of carbon sequestration.

5) Enrichment factor (EF), geoaccumulation index (I_{geo}), and ecological risk index (R_i)

The EF values for the heavy metals in this study ranged from 0.11–0.21 (mean±S.D. = 0.16±0.04) for Cu, and 0.33–0.62 (mean±S.D. = 0.50±0.10) for Pb (Table 4). The average EF values for Cu (0.16) and Pb (0.50) were less than 1, indicating depletion relative to the background reference rather than enrichment. These values fall within or near the range of 0.5–1.5, suggesting that the metals are derived primarily from crustal materials or natural processes rather than anthropogenic sources (Trajković et al., 2024).

The EF values < 1 observed in this study indicate that the Cu and Pb concentrations are lower than the background reference, suggesting the following: natural depletion due to differences in grain size (coarser sediments in the study area); dilution by carbonate or biogenic materials; and differences in mineralogy between intertidal and offshore environments. Importantly, values below 1 do not rule out natural sources; they simply indicate that metal concentrations are below the chosen reference baseline.

The I_{geo} values for the trace metals in this study ranged from 0.01–0.10 (mean±S.D. = 0.02±0.03) for Cu, and 0.02–0.25 (mean±S.D. = 0.05±0.07) for Pb. The I_{geo} results suggest that all the sampling stations fall into the “no pollution” to “moderately contaminated” categories, with I_{geo} values ranging from 0 to less than 1, indicating minimal to moderate contamination at the sites.

The interpretation of EFs, I_{geo}, and R_i in this study is constrained by the use of sediments from the offshore Gulf of Thailand (Shazili et al., 1999) as a geochemical background. Offshore and intertidal environments differ in several key aspects that influence natural metal concentrations. These differences mean that observed EF values < 1 (indicating depletion relative to the offshore background) may not necessarily reflect true depletion relative to natural intertidal conditions. Therefore, while the calculated indices suggest low enrichment and ecological risk, these findings should be considered preliminary until site-specific background values can be established. The relative patterns (e.g., higher concentrations in the inner zone) are likely robust, but the absolute classification of enrichment levels may be biased.

The potential R_i was calculated for Cu and Pb, as these were the only trace metals consistently detected in the sediment samples above the method detection limits. Cadmium (Cd) and chromium (Cr) were below the detection limits in all the samples and were therefore excluded from the R_i calculation, as risk assessment requires measurable concentrations. Iron (Fe) was excluded because it serves as a normalization element for enrichment factor analysis and is not typically considered a priority pollutant in ecological risk assessments (Hakanson, 1980).

The R_i values for Cu (21.54) and Pb (66.68) were both less than 150, indicating low ecological risk according to Hakanson's (1980) classification (R_i < 150 = low risk). The combined R_i for both metals (88.22) also falls within the low-risk category. While Cd and Cr were not detected, their absence suggests minimal contributions to ecological risk in this system. However, nondetected metals do not necessarily indicate zero risk, as they may accumulate over time or pose risks at very low concentrations through chronic exposure.

Table 4 Enrichment factor (EF) and geoaccumulation index (I_{geo}) of the sediment

Stations	EF		I _{geo}	
	Cu	Pb	Cu	Pb
S11	0.20	0.57	0.04	0.12
S12	0.21	0.46	0.04	0.09
S13	0.11	0.33	0.02	0.07
S21	0.14	0.49	0.03	0.10
S22	0.14	0.54	0.03	0.11
S23	0.13	0.41	0.03	0.08
S31	0.16	0.62	0.03	0.12
S32	0.18	0.59	0.04	0.12
S33	0.12	0.38	0.10	0.05
C1	0.20	0.60	0.03	0.25
Average	0.16±0.04	0.50±0.10	0.02±0.03	0.05±0.07
Min	0.11	0.33	0.01	0.02
Max	0.21	0.62	0.10	0.25

Note: EF interpretation after Trajković et al. (2024); I_{geo} classes after Müller (1969).

6) Correlations between heavy metals and grain size in surface sediment

Prior to analysis, the normality of the data distribution was tested using the Shapiro–Wilk method. The results indicated that %N, the C:N ratio, and Cu were significantly nonnormally distributed ($p < 0.05$). Therefore, Spearman's rank correlation, a nonparametric test, was employed to examine the relationships among all the parameters. The correlation matrix, based on 10 sediment samples ($n = 10$), is presented in Table 5.

Silt and clay are strongly positively correlated with Cu, suggesting that these metals share a common source, likely originating from terrestrial sources and being bound to organic matter (Gopal et al., 2023). The organic matter in the sediment plays a key role in controlling the mobility and leaching of heavy metals (Kou et al., 2024). The metal adsorption capacity increases as the grain size decreases (sand < silt < clay) (Buyang et al., 2019), indicating that finer sediments, such as clay, tend to accumulate more heavy metals than coarser sediments do.

A power analysis was conducted for the correlation between %C and Cu to assess the robustness of the findings. With a sample size of $n = 10$, a correlation coefficient (r_s) of 0.879, and a significance level of $\alpha = 0.05$, the statistical power was calculated to be 0.963. This indicates a 96.3% probability of correctly detecting a true correlation of this magnitude, confirming that the strong relationships observed in the sediment dataset are reliable despite the modest sample size.

The strong correlations observed between Cu and fine-grained sediments (silt and clay) are consistent with dry season conditions, when low-energy environments favor the deposition of metal-associated fine particles. During wet seasons, increased riverine input and resuspension events might alter these relationships, potentially introducing additional metal sources or redistributing sediments.

7) Bioconcentration and translocation of heavy metals in seagrass

The bioconcentration factors (BCFs) of heavy metals from sediments to seagrass leaves and roots are summarized in Figure 6. Seagrass can bioaccumulate essential metals (Cu) and nonessential metals (Pb) from the water column and sediment pore water, as both leaves and roots are sites of ionic uptake (Bonanno and Di Martino, 2017).

The average root BCF values for the heavy metals followed the order Cu (0.98) > Pb (0.23), while the average leaf BCF values were Cu (0.82) > Pb (0.27). Copper exhibited the greatest enrichment capacity for transfer from sediment to roots and leaves, likely because of its high bioavailability in the sediment (Chitimus et al.,

2023). Copper is known to be an essential metal for photosynthesis and growth in seagrass (Llagostera et al., 2016). The BCF values of lead were greater in the roots than in the leaves, whereas the BCF of Cu was greater in the leaves than in the roots. Metals are typically absorbed by the roots in the rhizosphere, accumulate in the roots, and are then transported to the stems and leaves, depending on the specific plant species (Vasilachi et al., 2023).

In this study, low BCF values (< 1) were observed for Cu and Pb in both leaves and roots. This suggests that seagrass does not have the capacity for metal hyperaccumulation but instead absorbs these metals in proportion to their availability in the sediment (Mng'ong'o et al., 2021). The translocation factor (TF) calculations revealed values generally less than 1, except for Cu, whose TF was greater than 1. These findings indicate that Pb is stored mainly in the roots, whereas Cu is translocated into the green parts of the plant (Zojaji et al., 2015). However, the TF values were not statistically significant, suggesting a minimal ability for phytoextraction in seagrass.

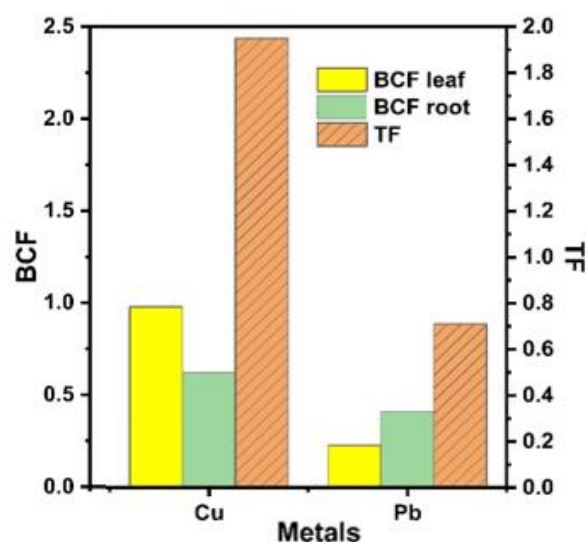


Figure 6 Bioconcentration factor (BCF) in leaves and roots and translocation factor (TF) of seagrass.

Limitations

This study has several limitations that should be considered when interpreting the results:

1) Temporal limitations

Sampling was conducted only during the dry season (May 2024). Given that metal distribution, sediment dynamics, and organic matter accumulation in coastal environments vary seasonally because of changes in rainfall, river discharge, and hydrodynamic conditions, the findings represent only a snapshot of the dry season. Future studies should incorporate both dry and wet season sampling to capture seasonal variability.

Table 5 Spearman's rank correlation matrix (ρ) among physicochemical parameters and heavy metals in surface sediments ($n = 10$)

Parameters	% C	%N	C:N	Cu	Pb	Clay	Silt	Sand
%C	1							
%N	0.879**	1						
C:N	0.018	-0.388	1					
Cu	0.879**	0.769**	0.018	1				
Pb	0.576	0.478	0.176	0.770*	1			
Clay	0.600	0.550	-0.067	0.794*	0.830**	1		
Silt	0.600	0.595	-0.164	0.770*	0.806**	0.890	1	
Sand	-0.673*	-0.647*	0.139	-0.818*	-0.842**	-0.952**	-0.976**	1

Note: *Correlation is significant at the 0.05 level (2-tailed). **Correlation is significant at the 0.01 level (2-tailed).

Spearman's rank correlation was used because of the nonnormal distribution of several variables. $n = 10$ sediment samples.

2) Hydrodynamic data limitations

This study lacks hydrodynamic measurements (e.g., current velocity, water residence time, and flow patterns) to confirm the mechanisms by which the sandbar influences metal accumulation. The proposed roles of reduced energy and increased detention time are inferred from spatial patterns and the literature but remain speculative. Future studies should incorporate hydrodynamic measurements to validate these mechanisms and quantify their relative importance.

3) Background value limitations

Enrichment factors were calculated using offshore Gulf of Thailand sediments (Shazili et al., 1999) as a background reference. Offshore and intertidal sediments represent different depositional settings with potentially different natural metal concentrations because of variations in grain size, organic matter content, and sediment sources. Therefore, enrichment factors should be interpreted as relative indicators rather than absolute measures of anthropogenic enrichment. Future studies should establish local background values using deep sediment cores from seagrass meadows.

4) Blue carbon assessment limitations

This study reports surface sediment OC concentrations (%) and C/N ratios but cannot quantify the actual blue carbon potential of the meadow. Assessment of blue carbon storage requires (i) sediment bulk density to convert %OC to carbon mass per volume; (ii) sediment depth profiles to calculate the total carbon inventory to a standardized depth (typically 1 m); and (iii) accumulation rates to determine carbon sequestration over time. Future research should incorporate sediment coring, bulk density measurements, and isotopic dating (e.g., ^{210}Pb) to enable comprehensive blue carbon assessment.

5) Ecological risk assessment limitations

The R_i was calculated using only two metals (Cu and Pb) because Cd and Cr were below the detection limits and could not be included, whereas Fe was excluded as a normalization element rather than a priority pollutant. A more comprehensive risk assessment would include a wider suite of metals and consider bioavailability and site-specific toxicity data.

6) Seagrass sample size limitations

The seagrass sample size was limited to six individuals (three leaves and three roots) because of conservation constraints within the protected marine area and the patchy distribution of the seagrass bed. Consequently, no inferential statistical analyses were performed on seagrass tissue data; the results are presented as descriptive statistics (means $\pm$ S.D.) to provide a preliminary baseline. These findings should be interpreted with caution, as they may not be representative of broader spatial or temporal patterns. Future studies should include larger sample sizes ($n \geq 30$) collected across multiple stations and seasons to enable robust statistical analysis and a more definitive assessment of heavy metal dynamics in seagrass tissues.

Conclusions

A study on heavy metal accumulation in sediments and seagrass around Kalase Bay during the dry season revealed that the average EF values for Cu and Pb were less than 1, suggesting that significant anthropogenic enrichment of these metals is unlikely at the sampling stations. Additionally, the BCFs for Cu and Pb in seagrass leaves and roots were less than 1, indicating that seagrass does not hyperaccumulate these metals. The presence of sand dunes in the area is unique to this study, potentially influencing the accumulation of both metals and organic carbon in the sediment. The results indicate that the organic carbon in seagrass sediments originates primarily from natural sources, highlighting the carbon storage potential of these ecosystems. Seagrasses continue to play a crucial role in blue carbon sequestration because of their ability to store significant amounts of carbon in sediment. Correlation analysis revealed that the fine sediment fraction is important for metal absorption, with the inner areas of the sand bar being more prone to accumulate heavy metals, indicating greater vulnerability to contamination in these regions. On the basis of these findings, a systematic and periodic monitoring program is strongly recommended to track heavy metal accumulation in the sediments and biota of Kalase Bay.

This should include the measurement of suspended sediments, which influence the distribution and bioavailability of metals. Regular monitoring will help detect contamination trends and support effective management and conservation efforts, ultimately protecting both marine ecosystems and human health. Moreover, for future studies, investigating the stock of organic carbon in sediment cores and examining other factors expected to influence blue carbon accumulation are recommended. Additionally, given the decline in seagrass coverage, methods to enhance its restoration through various approaches should be explored.

This study provides important baseline data on metal distribution in surface sediments during the dry season. However, given the potential for seasonal variability, these findings should be considered preliminary. Future research incorporating both dry and wet season sampling is essential to fully understand the dynamics of metal accumulation and carbon storage in this coastal ecosystem.

Data availability statement

Information and data used in the study will be disclosed upon request.

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Conflicts of interest

The authors declare that there are no conflicts of interest in competing financial or personal relationships

that could have appeared to influence the work reported in this work.

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