

Evaluation and Risk Assessment of Toxic Heavy Metals in the East Grand Bahama and Dover Sound Mangrove Sediments of Grand Bahama

Antonio Cadet

<https://orcid.org/0009-0008-2905-0315>

Veronica Toppin

<https://orcid.org/0000-0002-3419-9818>

Tamia Hamilton

<https://orcid.org/0009-0007-0418-0861>

Carlton Watson

<https://orcid.org/0000-0002-2101-399X>

Williamson Gustave

<https://orcid.org/0000-0001-6934-5689>

University of The Bahamas

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Abstract

Mangrove ecosystems in The Bahamas provide critical coastal protection and ecological functions but may accumulate contaminants derived from anthropogenic activities. This study quantified concentrations of selected heavy metals, chromium (Cr), manganese (Mn), nickel (Ni), cadmium (Cd), antimony (Sb), lead (Pb), copper (Cu), arsenic (As), and zinc (Zn), in surface sediments collected from two coastal sites on Grand Bahama Island, Dover Sound (DS) and East Grand Bahama (EGB). Metal concentrations were determined using inductively coupled plasma mass spectrometry (ICP-MS), and contamination levels were evaluated using multiple ecological and human health risk indices. Mean concentrations exceeding sediment quality reference values at DS were observed for Cd (25.64 mg kg⁻¹), Sb (44.92 mg kg⁻¹), and Cu (195.33 mg kg⁻¹), whereas at EGB exceedances were recorded for Cd (442.58 mg kg⁻¹) and Sb (56.62 mg kg⁻¹). The pollution load index (PLI < 1) indicated that sediments at both sites were not classified as polluted overall. In contrast, the ecological risk index (RI > 600) indicated very high ecological risk at most sampling locations, except D2, D3, and D4, largely driven by elevated Cd concentrations. Contamination factor analysis identified moderate contamination by Cu ($1 \leq CF < 3$) and high contamination by Cd and Sb ($CF > 6$), while geo-accumulation index values ($I_{geo} > 1$) supported anthropogenic enrichment of Cd and Sb. Human health risk assessment using the U.S. EPA screening framework indicated that Cd may pose potential non-carcinogenic and carcinogenic risk, particularly for children. These findings demonstrate that metal-specific enrichment in small island coastal systems can generate disproportionate ecological and health implications and highlight the importance of continued monitoring and improved baseline characterization to inform environmental management strategies in The Bahamas.

Introduction

Mangroves are coastal wetlands dominated by salt-tolerant trees and shrubs that occupy intertidal zones of tropical and subtropical regions, typically between 25°N and 25°S latitude (Twilley, 2019). These ecosystems provide essential ecological services, including habitat provision, shoreline stabilization, and nutrient cycling within protected bays, estuaries, and lagoons, with influences that extend beyond the intertidal zone (Allard et al., 2020). Mangroves also contribute to the regulation of coastal water quality, as their dense root systems trap suspended particles and associated contaminants. Consequently, mangrove sediments function as natural sinks for heavy metals and serve as indicators of regional pollution and ecological risk (Mitra, 2019; Shi et al., 2019). The accumulation of metals within sediments can limit their bioavailability to plants, enabling mangrove ecosystems to persist under moderate contaminant loads (de Lacerda et al., 2022). Despite their ecological importance, mangroves have experienced significant global decline, with approximately 35% of global coverage lost since the mid-twentieth century and current annual loss rates estimated at 0.16–0.39% (Friess et al., 2019). Large-scale deforestation driven by aquaculture, agriculture, and urbanization remains a primary cause of decline. Additional degradation results from pollution and resource overexploitation, including chronic exposure to heavy metals derived from anthropogenic activities, which contribute to the vulnerability of mangroves as one of the most threatened tropical ecosystems (Deng et al., 2021; Friess et al., 2019).

Multiple anthropogenic sources contribute to heavy metal accumulation in mangrove

environments, including oil spills, agricultural runoff, untreated domestic sewage, industrial effluents, and waste generated by coastal communities. Heavy metals are transported to aquatic systems through stormwater runoff, wastewater discharges, and agricultural drainage, where they may ultimately accumulate in fine-grained sediments (Twilley, 2019; Uddin et al., 2019). In Grand Bahama, individuals have raised concerns regarding illegal and indiscriminate dumping in and around Freeport, which may introduce metal-containing debris into coastal and nearshore ecosystems (Hall, 2024; Maycock, 2023). Such debris, including electronic and electrical waste, can contain toxic elements such as cadmium, arsenic, and lead, which persist in the environment and pose ecological and human health risks (Numbere & George, 2019). Improper disposal of solid waste therefore represents a potential localized source of heavy metal contamination in Bahamian coastal systems.

Contamination of surface sediments with heavy metals can affect both aquatic organisms and human populations through trophic transfer. Mangrove wetlands contain heavy metals from various sources with long retention times, posing significant post-pollution remediation challenges. Once they enter the ecological cycle, they are rarely biodegradable as they disrupt normal physiological metabolism, enter the food chain through lower trophic levels, and eventually bioaccumulate, damaging both human health and the ecological environment (Yang et al., 2025). The World Health Organization estimates that approximately 420,000 deaths and 600 million illnesses annually are linked to the consumption of contaminated food worldwide (2026). Although certain metals such as zinc (Zn), manganese (Mn), iron

(Fe), and copper (Cu) are essential micronutrients required for physiological function, they become toxic when concentrations exceed biological thresholds (Al-Kahtany et al., 2023; Islam et al., 2015). Human exposure typically occurs through ingestion, inhalation of contaminated particulates, or dermal contact, and the severity of health effects depends on concentration, exposure frequency, and duration (Al-Kahtany et al., 2023). Among metals of major public health concern are arsenic (As), cadmium (Cd), lead (Pb), and mercury (Hg), which have been associated with carcinogenic, teratogenic, and systemic toxic effects (Guimarães-Ervilha et al., 2026; Numbere & George, 2019).

Ruiz-Hardy et al. (2025) conducted a comprehensive assessment of heavy metal contamination across several Caribbean nations; however, the Bahamas was not included in their analysis. Their review documented the widespread presence of potentially toxic elements in Caribbean coastal systems and highlighted the risk of bioaccumulation in marine organisms, with implications for food security and public health throughout the region. Given comparable anthropogenic pressures across Caribbean islands, similar contamination concerns may exist within Bahamian ecosystems. Studies conducted within The Bahamas have primarily focused on other environmental compartments, including cave sediments (Schwabe et al., 2010), coastal shark species (Shipley et al., 2021), and seagrass beds or marina environments in Abaco (Salas-Ortiz et al., 2025). To date, no published studies have quantified heavy metal contamination and associated ecological and human health risks in Bahamian mangrove sediments. Schwabe et al. (2010) further emphasized the need for expanded research and locally derived databases to support ecological risk assessment within the archipelago.

Collectively, these studies reveal two critical gaps: (1) the absence of targeted heavy metal data for Bahamian mangrove sediments and (2) the lack of ecosystem-specific ecological and human health risk assessments. The present study addresses these gaps by quantifying heavy metal concentrations and evaluating associated ecological and human health risks in mangrove sediments of Grand Bahama. Specifically, this study applies pollution load index (PLI), contamination factor (CF), geo-accumulation index (I_{geo}), risk index (RI), hazard index (HI), and total carcinogenic risk (TCR) to Bahamian mangrove systems for the first time. By generating these data for East Grand Bahama and Dover Sound, this study establishes an empirical baseline to support local risk assessment, inform environmental management strategies, and contribute to national policy initiatives such as Vision 2040 (Bahamas National Development Plan, 2016) aimed at mitigating heavy metal contamination in Bahamian coastal ecosystems.

Dover Sound is situated near industrial and waste-handling facilities, whereas East Grand Bahama is located in a less industrialized region of the island. Following Hurricane Dorian in 2019, extensive flooding and storm surge may have mobilized industrial materials and redistributed contaminants across coastal and nearshore environments. These contrasting site characteristics provide an opportunity to evaluate spatial differences in heavy metal contamination. This study aims to quantify concentrations of selected toxic heavy metals in mangrove sediments from Dover Sound and East Grand Bahama and to assess associated ecological and human health risks. We hypothesize that elevated sediment metal concentrations will correspond to increased contamination indices, higher RI values, and greater HI and

TCR estimates. The findings are intended to inform environmental monitoring and management strategies in The Bahamas by identifying metals that may pose ecological and public health concerns.

Methods

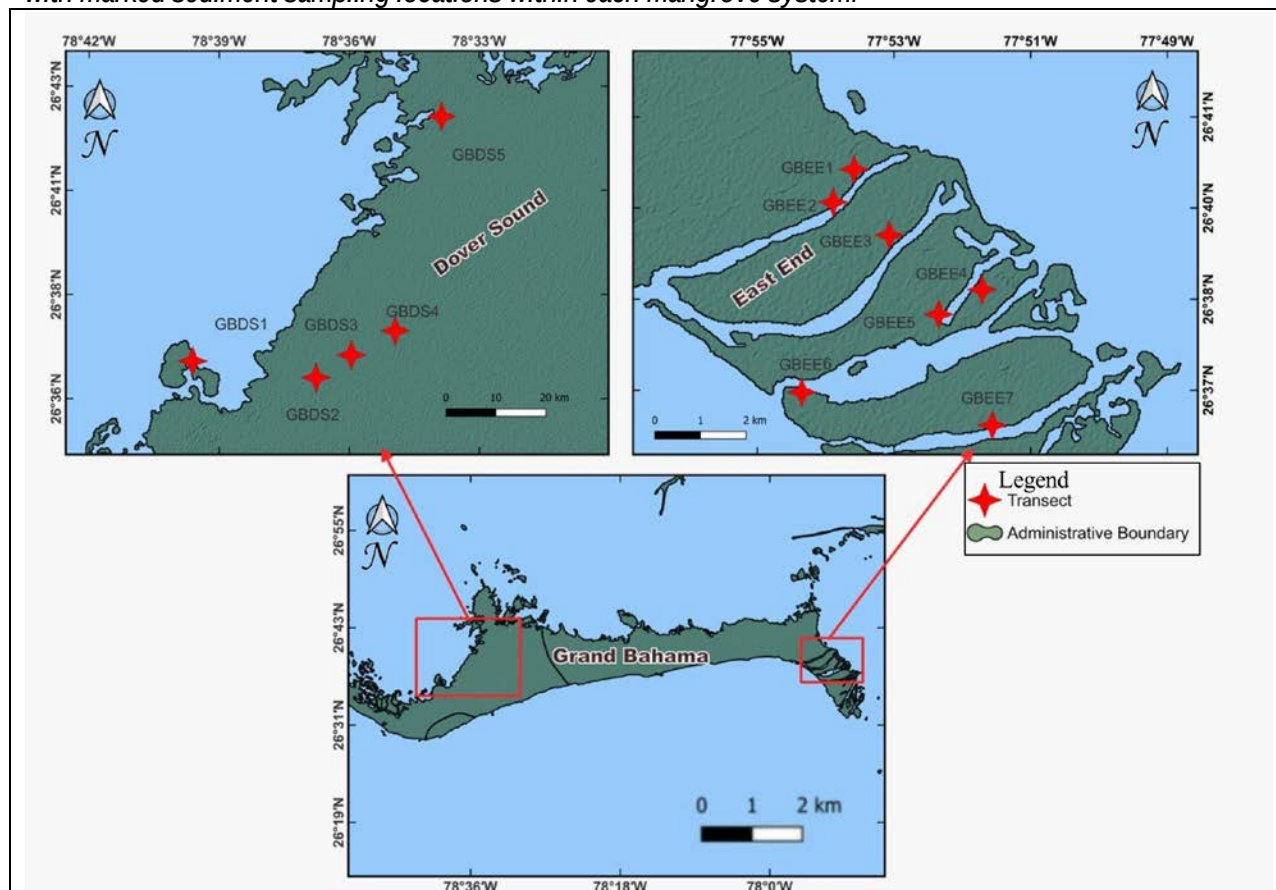
Study Area

The Bahamas archipelago is a low-lying limestone-based carbonate platform located in the Western Atlantic Ocean (Buchan, 2000). The islands are characterized by alkaline soils derived primarily from the weathering of calcium carbonate bedrock, with generally low organic carbon content and limited clay fractions (Currie et al., 2019; Chambers et al., 2023). These geochemical characteristics influence metal

mobility and retention within sediments. This study was conducted on Grand Bahama Island (26.6468°N, 78.5213°W), the second-most populous island in the archipelago prior to Hurricane Dorian (Lewis et al., 2025). Two mangrove-dominated coastal sites were selected for investigation: Dover Sound (26.6228°N, 78.5897°W) and East Grand Bahama (26.6727°N, 77.9120°W) (Figure 1). Dover Sound is situated near industrial and waste-handling facilities, whereas East Grand Bahama is located in a comparatively less industrialized region. These contrasting land-use characteristics provided an opportunity to evaluate spatial differences in heavy metal contamination.

Figure 1

Map of Grand Bahama Island illustrating the study sites at Dover Sound (DS) and East Grand Bahama (EGB), with marked sediment sampling locations within each mangrove system.



Sampling Design

Sediment sampling was conducted at five locations within Dover Sound (DS) and seven locations within East Grand Bahama (EGB). At each location, four subsampling points were established using a quadrant-based layout, resulting in four surface sediment samples per location. In total, 48 sediment samples were collected (12 locations $\times$ 4 subsamples per location). Sampling locations were selected using a stratified random approach, whereby study areas were first defined based on site characteristics and accessibility, and sampling points were randomly assigned within each site to improve spatial representativeness while accounting for logistical constraints (Onjefu et al., 2020). The quadrant design allowed assessment of within-site variability, while spacing between locations was chosen to provide coverage across each mangrove system.

Sample Preparation and Instrumental Analysis

At each sampling location, surface sediments (0–20 cm) were collected from four quadrants established in a north–south and east–west orientation. Within each quadrant, one sediment sample was obtained using a pre-cleaned stainless-steel shovel. To minimize cross-contamination, tools were rinsed with deionized water between collections. Samples were placed in clean polyethylene bags, stored in coolers during transport, and transferred to the Soil and Complex Matrix Laboratory at the University of The Bahamas. In the laboratory, samples were air-dried at room temperature, homogenized, and sieved to less than 2 mm to remove coarse material. Approximately 300 ± 10 mg of each sample was digested using a microwave-assisted acid digestion procedure following Williams et al. (2009). Digestion was performed in

closed vessels using a 1:1 mixture of nitric acid (HNO_3) and hydrogen peroxide (H_2O_2), with sequential heating to 55 °C for 10 min, 75 °C for 10 min, and 95 °C for 30 min. Following digestion, solutions were filtered through 0.45 μm membrane filters and diluted with deionized water to a known volume. Heavy metal concentrations were determined using ICP-MS, and analytical accuracy was verified through triplicate analysis of a certified reference material (BCR-320R, channel sediment).

Data Collection and Analysis

Heavy metal concentrations in sediment digests were quantified using a NexION 300X inductively coupled plasma mass spectrometer (ICP-MS; PerkinElmer). Concentration data were used to calculate PLI, CF, Igeo, RI, HI, and TCR using established equations reported in the literature (Onjefu et al., 2020; Adimalla, 2020; Ayaz et al., 2023). Statistical comparisons between DS and EGB were performed using a two-sample t-test in Microsoft Excel (Microsoft 365).

Data Analysis

Pollution Load Index

PLI was calculated to assess the overall level of heavy metal contamination at each site following established methodology (Müller, 1969; Wang et al., 2010; Onjefu et al., 2020). CF for each metal was first determined as the ratio of the measured concentration (C_m) to the corresponding background value (CB_n). PLI was then calculated as:

$$\text{PLI} = (\text{CF}_1 \times \text{CF}_2 \times \text{CF}_3 \times \dots \times \text{CF}_n)^{1/n} \quad (1)$$

where CF represents the contamination factor for each metal and n is the number of metals included in the assessment.

Interpretation of PLI values followed standard classification criteria (provided in Supplementary Table S1), where $PLI = 1$ indicates baseline conditions, $PLI < 1$ indicates no pollution, and $PLI > 1$ indicates contamination.

Contamination Factor

CF was calculated as:

$$CF = C_m / CB_n \quad (2)$$

where C_m is the measured concentration of the metal in sediment (mg/kg) and CB_n represents the corresponding background concentration (quality reference value). Background values used in this study are provided in Table 1 (Alfaro et al., 2015).

Geo-accumulation Index

The geo-accumulation index (Igeo), originally proposed by Müller (1969), was used to assess the degree of metal enrichment in sediments relative to background concentrations. Igeo was calculated as:

$$Igeo = \log_2 \left(\frac{C_m}{1.5 \cdot CB_n} \right) \quad (3)$$

where C_m is the measured metal concentration (mg/kg) and CB_n is the corresponding background value. The constant 1.5 accounts for natural variability in background concentrations and minor lithogenic variations (Ayaz et al., 2023; El Turk et al., 2019). Classification of Igeo values is provided in Supplementary Table S1. Categories include: *unpolluted* ($Igeo < 0$); *unpolluted to moderately polluted* ($0 \leq Igeo < 1$); *moderately polluted* ($1 \leq Igeo < 2$); *moderately to strongly polluted* ($2 \leq Igeo < 3$); *strongly polluted* ($3 \leq Igeo < 4$); *strongly to very strongly polluted* ($4 \leq Igeo < 5$); and *very strongly polluted* ($Igeo \geq 5$).

Potential Ecological Risk Index

The potential ecological risk index method proposed by Hakanson (1980) was used to evaluate the ecological risk posed by heavy metals in sediments. This method incorporates contamination levels and the toxic response of individual metals. The ecological risk coefficient (E_r^i) and the integrated ecological risk index (RI) were calculated as follows:

$$CF = C_m / CB_n \quad (2)$$

$$E_r^i = T_r^i \times C_f^i \quad (4)$$

$$RI = \sum_{i=1}^n E_r^i \quad (5)$$

where CF is the contamination factor, T_r^i is the toxic response factor for each metal, and n is the number of metals evaluated. Toxic response factors used in this study were: Cr (2), Mn (1), Ni (5), Cd (30), Sb (7), Pb (5), Cu (5), As (10), and Zn (1). The toxic response factor for Sb was adopted from Wang et al. (2018). Classification criteria for E_r^i and RI are provided in Supplementary Table S2. Risk categories include: *low risk* ($RI < 150$); *moderate risk* ($150 \leq RI < 300$); *considerable risk* ($300 \leq RI < 600$); and *very high risk* ($RI \geq 600$).

Chronic Daily Dose

Human health risk was assessed using the U.S. Environmental Protection Agency (U.S. EPA) model for soil exposure (1989; 2002). Chronic daily dose (CDD, mg/kg/day) was estimated for three exposure pathways: ingestion, inhalation, and dermal contact. CDD for ingestion, inhalation, and dermal exposure were calculated as:

$$CDD_{\text{ingestion}} = (C \times IR_{\text{ing}} \times ED \times EF / BW \times AT) \times CF \quad (6)$$

$$CDD_{\text{inhalation}} = (C \times IR_{\text{inh}} \times ED \times EF/BW \times AT \times PEF) \quad (7)$$

$$CDD_{\text{dermal}} = (C \times SA \times SAF \times DAF \times ED \times EF/BW \times AT) \times CF \quad (8)$$

where C is the concentration of heavy metals in sediments (mg/kg); IR_{ing} is the ingestion rate (200 mg/day for children; 100 mg/day for adults); IR_{inh} is the inhalation rate (7.63 m³/day for children; 12.8 m³/day for adults); EF is the exposure frequency (365 days/year); ED is the exposure duration (6 years for children; 30 years for adults); CF is the conversion factor (1.0×10^{-6}); BW is body weight (20 kg for children; 70 kg for adults); AT is averaging time ($ED \times 365$ days for non-carcinogenic effects; 70×365 days for carcinogenic effects); PEF is the particle emission factor (1.36×10^9 m³/kg); SA is exposed skin surface area (1600 cm² for children; 4350 cm² for adults); SAF is the skin adherence factor (0.2 mg/cm²/day for children; 0.7 mg/cm²/day for adults); and DAF is the dermal absorption factor (0.001). Parameter values are provided in Supplementary Table S3.

Carcinogenic Risk Assessment

Carcinogenic risk (CR) was estimated to evaluate the probability of developing cancer over a lifetime due to exposure to carcinogenic metals. Lifetime carcinogenic risk for each metal was calculated as:

$$CR = CDD \times SF \quad (9)$$

$$TCR = \sum CR \quad (10)$$

where CDD is the chronic daily dose (mg/kg/day) and SF is the carcinogenic slope factor ((mg/kg/day)⁻¹). Total carcinogenic risk (TCR) represents the cumulative risk from all evaluated carcinogenic metals. Slope factor values used in this study are provided in

Supplementary Table S4 (U.S. EPA, 1989, 2002; Adimalla, 2020; Chen et al., 2022).

According to U.S. EPA guidelines, CR or TCR values less than 1×10^{-6} indicate negligible risk, values between 1×10^{-6} and 1×10^{-4} represent an acceptable risk range, and values greater than 1×10^{-4} suggest potential concern.

Non-Carcinogenic Risk Assessment

Non-carcinogenic risk was evaluated using the hazard quotient (HQ) and hazard index (HI) following U.S. EPA guidelines (U.S. EPA, 1989; Adimalla, 2020). HQ was calculated as:

$$HQ = CDD/RfD \quad (11)$$

where CDD is the chronic daily dose (mg/kg/day) and RfD is the reference dose (mg/kg/day) for each metal.

The hazard index (HI), representing the cumulative non-carcinogenic risk across exposure pathways for each metal, was calculated as:

$$HI = \sum HQ_i = \sum \frac{CDD}{RfD} \quad (12)$$

Reference dose values used in this study are provided in Supplementary Table S4. According to U.S. EPA guidelines, HI values less than 1 indicate no significant non-carcinogenic risk, whereas HI values greater than 1 suggest the potential for adverse health effects.

Results

Ecological risk: Concentration of Heavy Metals

Heavy metal concentrations varied between sites and among elements. In DS, the order of mean metal concentrations (mg/kg) was Mn > Cu > Ni > Cr > Sb > Cd > Pb > As,

whereas in EGB, the order was $Cd > Mn > Cu > Sb > Cr > Ni > Zn > Pb > As$. The highest mean concentration observed across both sites (Table 1) was Cd in EGB (442.58 mg/kg), followed by Mn in DS (269.85 mg/kg). Statistical comparison using a two-sample t-test of the sampling location means

($n = 57$) indicated no significant differences ($p > .05$) between DS and EGB for Cr, Sb, Pb, and As. However, Mn, Ni, Cd, and Cu showed statistically significant differences between the two sites ($p < .05$). Zinc was excluded from statistical comparison due to the absence of corresponding DS values.

Table 1

Mean Concentrations (mg/kg) of Heavy Metals in Mangrove Sediments from Dover Sound (DS) and East Grand Bahama (EGB) with Corresponding Quality Reference Values (QRV)

Heavy Metals	DS	EGB	QRV (mg/kg)
Cr	74.91 ± 17.24	43.94 ± 10.24	153
Mn	269.85 ± 86.91	88.70 ± 20.98	1947
Ni	82.48 ± 7.20	32.30 ± 11.59	170
Cd	25.64 ± 16.04*	442.58 ± 144.29*	0.6
Sb	44.92 ± 4.27*	56.62 ± 9.25*	6
Pb	11.33 ± 1.92	14.31 ± 3.59	50
Cu	195.33 ± 20.62*	67.44 ± 38.61	83
As	1.07 ± 0.24	3.09 ± 2.42	19
Zn	BDL	17.92 ± 9.22	86

Note: ±Standard error, BDL=Below detection level. Quality reference values were taken from Alfaro et al. (2015). An asterisk (*) represents heavy metal mean concentrations that exceeded their QRVs.

Pollution Load Index

Based on U.S. EPA classification criteria, PLI values at both DS and EGB were < 1 , indicating no overall pollution (Table 2).

Table 2

Pollution Load Index (PLI) Values of Dover Sound (DS) and East Grand Bahama (EGB)

Grand Bahama Mangrove Sediments	PLI
DS	0.87
EGB	0.80

Note: No Pollution (PLI < 1), Baseline (PLI = 1), Pollution (PLI > 1)

Contamination Factor

Contamination factor values indicated that Cr, Mn, Ni, Pb, and As were classified as low contamination in DS, while Cr, Mn, Ni, Pb, Cu, As, and Zn were classified as low contamination in EGB (Table 3). Moderate contamination was observed only for Cu in DS. The highest contamination levels were associated with Cd and Sb at both sites. Notably, CF for Cd in EGB was the highest recorded among all metals and sites, exceeding the high contamination threshold (CF > 6) by approximately 122-fold (Table 3).

Table 3
Contamination Factor (CF) Values for Heavy Metals in Mangrove Sediments from Dover Sound (DS) and East Grand Bahama (EGB)

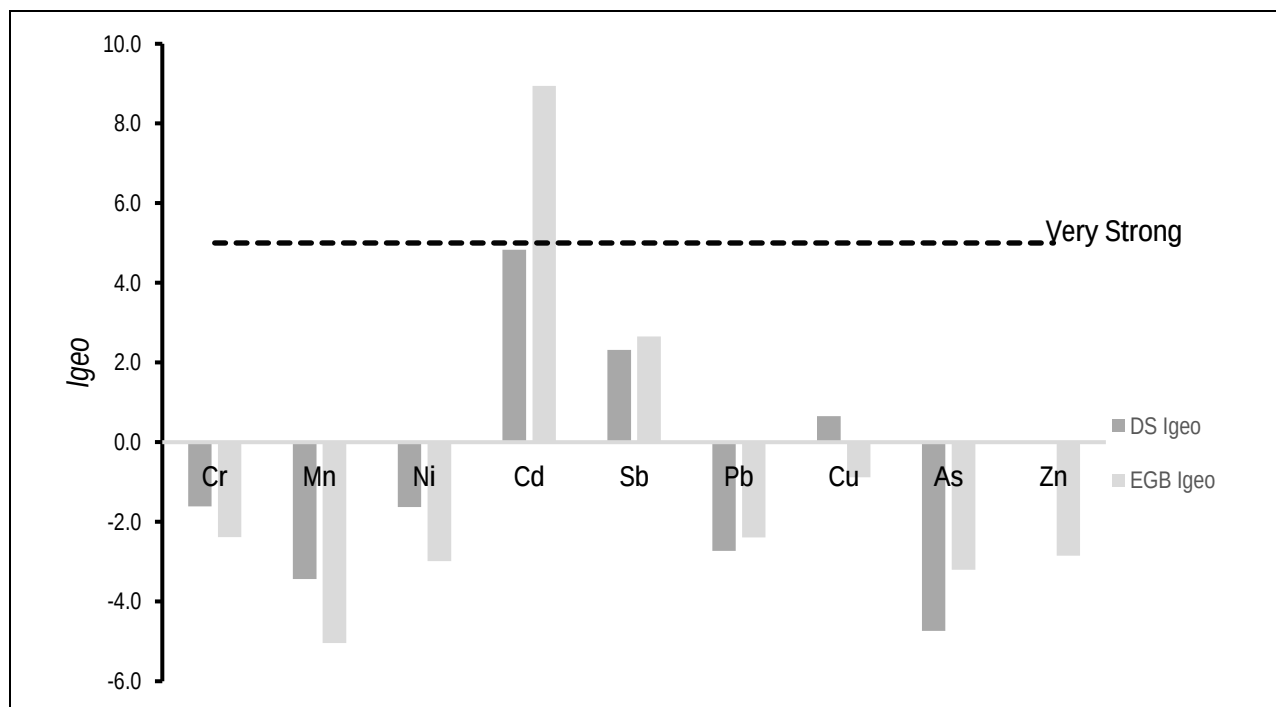
Heavy Metals	DS	EGB
Cr	0.49	0.29
Mn	0.14	0.05
Ni	0.49	0.19
Cd	42.47	737.64
Sb	7.49	9.44
Pb	0.23	0.29
Cu	2.35	0.81
As	0.06	0.16
Zn	ND	0.21

Note: ND = No available data. Low contamination ($CF < 1$), Moderate contamination ($1 \leq CF < 3$), Considerable contamination ($3 \leq CF < 6$), High contamination ($CF > 6$).

Geo-Accumulation Index

Based on Igeo classification criteria (provided in Supplementary Table S1), most metals were classified as unpolluted ($I_{geo} < 0$) at both DS and EGB, including Cr, Mn, Ni, Pb, As, and Zn (Figure 2). No Zn Igeo value was calculated for DS due to the absence of Zn concentration data at that site. Cu was classified as *unpolluted to moderately polluted* ($0 \leq I_{geo} < 1$) in DS and *unpolluted* ($I_{geo} < 0$) in EGB. Sb exhibited *moderate to strongly polluted conditions* ($2 \leq I_{geo} < 3$) at both sites. Cd showed the highest degree of enrichment, being classified as *strongly to very strongly polluted* ($4 \leq I_{geo} < 5$) in DS and *very strongly polluted* ($I_{geo} \geq 5$) in EGB. Overall, Igeo results indicate greater metal enrichment in EGB relative to DS, primarily driven by elevated Cd concentrations.

Figure 2
Geo-accumulation Index (I_{geo}) Classification of Heavy Metals in Sediments from DS and EGB



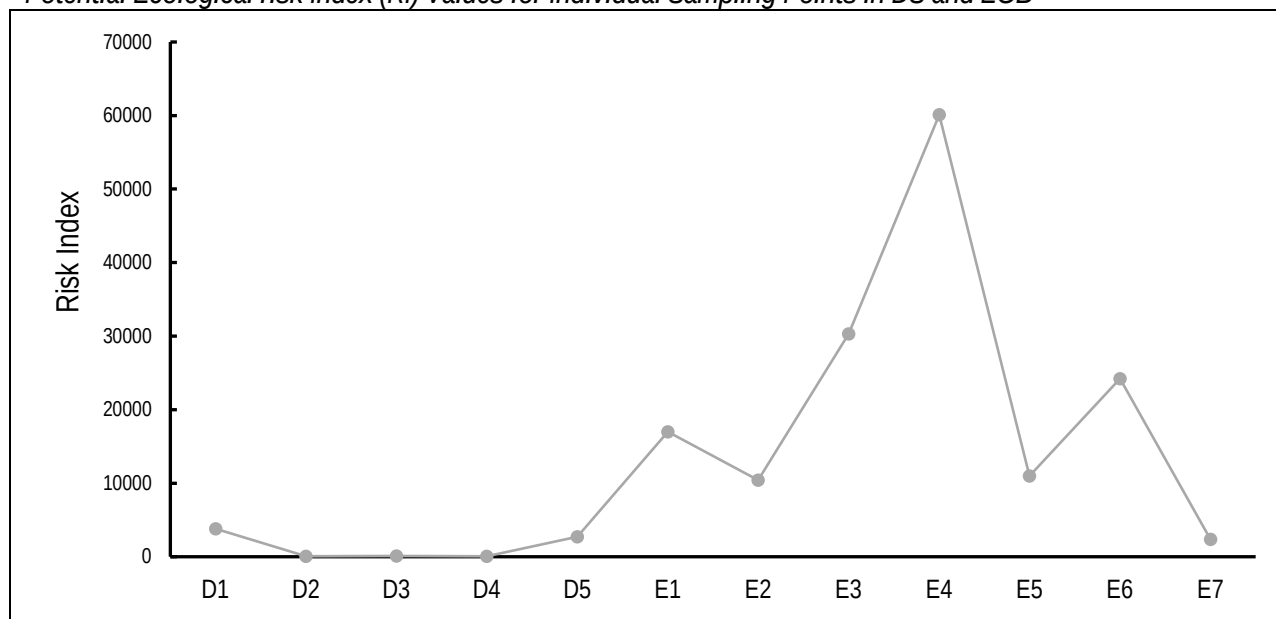
Potential Ecological Risk Index

The potential ecological risk coefficients (E_r^i) and integrated ecological risk index (RI) varied among sampling points. Sb exhibited moderate ecological risk ($40 \leq E_r^i < 80$) at most locations, with considerable risk ($80 \leq E_r^i < 160$) observed at E4 and E6 (Supplementary Figure S1e). Cd demonstrated the highest ecological risk, with very high risk ($E_r^i \geq 320$) at most sampling points. In contrast, D2, D3, and D4 showed E_r^i values of 0 for Cd due to non-detectable concentrations (Supplementary

Figure S1d). All other evaluated metals were classified as low ecological risk ($E_r^i < 40$) across sampling points (Supplementary Figure S1). The integrated ecological risk index (RI) was very high ($RI \geq 600$) at most sampling points, whereas D2, D3, and D4 were classified as low risk ($RI < 150$). The spatial distribution of RI closely corresponded to Cd E_r^i values, indicating that Cd was the dominant contributor to overall ecological risk at both sites (Figure 3).

Figure 3

Potential Ecological risk Index (RI) Values for Individual Sampling Points in DS and EGB



Human Health Risk :Non-Carcinogenic and Carcinogenic Health Risk

The HI and TCR were used to evaluate potential human health risks associated with heavy metal exposure. According to U.S. EPA guidelines (1989), HI less than 1 indicates no significant non-carcinogenic risk, whereas HI greater than 1 suggests potential adverse health effects. The maximum HI values for Cd exceeded 1 for both adults and children, and the mean HI

value for children also exceeded 1 (Table 4), indicating potential non-carcinogenic risk associated with Cd exposure. In contrast, HI values for all other evaluated metals were below 1 for both age groups. The data obtained for Zn were calculated from EGB sampling points only. Across metals, the maximum, minimum, and mean HI values followed the trend children greater than adults, reflecting greater susceptibility due to lower body weight and higher exposure rates.

Table 4
Results of Non-Carcinogenic Risks for Adults & Children by Ingestion, Inhalation, & Dermal Pathways.

Results of Non-Carcinogenic Hazard for Adults and Children from Ingestion, Inhalation, and Dermal Contact									
Adults					Children				
Non-Carcinogenic	HQ _{Ingestion}	HQ _{Inhalation}	HQ _{Dermal}	HI	HQ _{Ingestion}	HQ _{Inhalation}	HQ _{Dermal}	HI	
Cr	Max	6.59E-02	6.50E-04	2.01E-03	6.85E-02	4.61E-01	1.36E-03	7.38E-04	4.63E-01
	Min	8.23E-03	8.12E-05	2.50E-04	8.56E-03	5.76E-02	1.69E-04	9.21E-05	5.78E-02
	Mean	2.71E-02	2.67E-04	8.24E-04	2.82E-02	1.89E-01	5.58E-04	3.03E-04	1.90E-01
Ni	Max	7.51E-03	6.86E-07	8.47E-04	8.35E-03	5.25E-02	1.43E-06	3.11E-04	5.29E-02
	Min	4.19E-04	3.83E-08	4.73E-05	4.66E-04	2.93E-03	7.99E-08	1.74E-05	2.95E-03
	Mean	3.80E-03	3.47E-07	4.29E-04	4.23E-03	2.66E-02	7.25E-07	1.58E-04	2.68E-02
Cd	Max	1.71E+00	1.61E-04	2.09E+00	3.80E+00	1.20E+01	3.37E-04	7.68E-01	1.28E+01
	Min	6.56E-02	6.17E-06	7.99E-02	1.46E-01	4.59E-01	1.29E-05	2.94E-02	4.89E-01
	Mean	3.84E-01	3.61E-05	4.68E-01	8.52E-01	2.69E+00	7.54E-05	1.72E-01	2.86E+00
Pb	Max	2.65E-02	9.91E-07	2.15E-03	2.86E-02	1.85E-01	2.07E-06	7.92E-04	1.86E-01
	Min	3.25E-03	1.22E-07	2.64E-04	3.51E-03	2.27E-02	2.54E-07	9.72E-05	2.28E-02
	Mean	1.33E-02	4.99E-07	1.09E-03	1.44E-02	9.34E-02	1.04E-06	3.99E-04	9.38E-02
Cu	Max	9.85E-03	9.27E-07	1.00E-03	1.08E-02	6.89E-02	1.93E-06	3.68E-04	6.93E-02
	Min	1.25E-05	1.18E-09	1.27E-06	1.38E-05	8.76E-05	2.46E-09	4.67E-07	8.80E-05
	Mean	4.31E-03	4.06E-07	4.38E-04	4.75E-03	3.02E-02	8.47E-07	1.61E-04	3.03E-02
As	Max	8.32E-02	1.91E-05	6.18E-03	8.94E-02	5.82E-01	3.98E-05	2.27E-03	5.84E-01
	Min	1.78E-04	4.08E-08	1.32E-05	1.91E-04	1.24E-03	8.51E-08	4.86E-06	1.25E-03
	Mean	1.07E-02	2.46E-06	7.95E-04	1.15E-02	7.50E-02	5.13E-06	2.93E-04	7.53E-02
Zn	Max	2.89E-04	2.72E-08	4.40E-05	3.33E-04	2.02E-03	5.68E-08	1.62E-05	2.04E-03
	Min	1.33E-04	1.26E-08	2.03E-05	1.54E-04	9.34E-04	2.62E-08	7.47E-06	9.42E-04
	Mean	8.53E-05	8.03E-09	1.30E-05	9.83E-05	5.97E-04	1.68E-08	4.78E-06	6.02E-04

Note: Hazard Quotient (HQ). Hazard Index (HI). No health risk (HI < 1 or 1.00E+00), Health risk (HI > 1 or 1.00E+00).

For carcinogenic risk, U.S. EPA (1989) defines CR or TCR values less than 1×10^{-6} as negligible, values between 1×10^{-6} and 1×10^{-4} as within the acceptable risk range, and values greater than 1×10^{-4} as indicative of potential concern. TCR values for Cd, including maximum, minimum, and mean estimates, exceeded 1×10^{-4} for both adults

and children (Table 5), suggesting an elevated carcinogenic risk associated with Cd exposure. Carcinogenic risk was not calculated for Mn, Ni, Sb, Cu, and Zn due to the absence of U.S. EPA slope factor values for these metals.

Table 5

Results of Carcinogenic Risks for Adults & Children by Ingestion, Inhalation, & Dermal Pathways.

		Adults				Children			
Carcinogenic		CR _{Ingestion}	CR _{Inhalation}	CR _{Dermal}	TCR	CR _{Ingestion}	CR _{Inhalation}	CR _{Dermal}	TCR
C	Max	4.24E-05	3.35E-07	5.16E-05	9.43E-05	5.94E-05	1.40E-07	3.79E-06	6.32E-05
	Min	5.30E-06	4.18E-08	6.44E-06	1.18E-05	7.42E-06	1.74E-08	4.74E-07	7.91E-06
	Mean	1.74E-05	1.38E-07	2.12E-05	3.88E-05	2.44E-05	5.74E-08	1.56E-06	2.60E-05
Cd	Max	4.48E-03	4.36E-07	5.46E-03	9.94E-03	6.27E-03	1.82E-07	4.02E-04	6.68E-03
	Min	1.71E-04	1.67E-08	2.09E-04	3.80E-04	2.40E-04	6.96E-09	1.54E-05	2.55E-04
	Mean	1.00E-03	9.76E-08	1.22E-03	2.23E-03	1.41E-03	4.07E-08	9.00E-05	1.50E-03
Pb	Max	1.35E-07	6.28E-11	4.11E-09	1.39E-07	1.89E-07	2.62E-11	3.02E-10	1.89E-07
	Min	1.66E-08	7.71E-12	5.05E-10	1.71E-08	2.32E-08	3.22E-12	3.71E-11	2.32E-08
	Mean	6.80E-08	3.16E-11	2.07E-09	7.01E-08	9.52E-08	1.32E-11	1.52E-10	9.54E-08
As	Max	1.60E-05	4.33E-12	1.19E-06	1.72E-05	2.25E-05	1.81E-12	8.77E-08	2.25E-05
	Min	3.43E-08	9.25E-15	2.55E-09	3.68E-08	4.80E-08	3.86E-15	1.87E-10	4.82E-08
	Mean	2.07E-06	5.57E-13	1.53E-07	2.22E-06	2.89E-06	2.33E-13	1.13E-08	2.90E-06

Note: Carcinogenic risk (CR). Total carcinogenic risk (TCR). No health risk (TCR < 1.00E-04), Health risk (TCR > 1.00E-04).

Discussion

Ecological Risk: Concentration of Heavy Metals

Quality reference values (QRVs) represent baseline geochemical concentrations against which anthropogenic enrichment can be evaluated (Alfaro et al., 2015). The study's background data come from Cuban soils, which have strong geochemical relationships with iron and naturally high amounts of heavy metals. Because of their larger surface area and cation-exchange capability, Cuban's fine-grained, clay-rich sediments usually have higher adsorption capacities (Alfaro et al., 2015). On the other hand, carbonate-dominated sediments, like those found in The Bahamas, typically contain less clay and organic matter, which reduces their ability to retain metals. The Bahamas has carbonate sediments, which are expected

to retain metals less efficiently than the clay-rich Cuban soils from which the reference values were derived; true local baselines are likely lower than the values applied here. Consequently, the contamination factors and geo-accumulation indices reported here are likely conservative, and the actual degree of enrichment, particularly for Cd, may be greater than indicated. In this study, mean concentrations of most heavy metals were below their respective QRVs at both DS and EGB (Table 1), indicating limited enrichment for the majority of elements. However, Cd and Sb exceeded QRVs at both sites, and Cu exceeded its QRV at DS. Among these, Cd exhibited the greatest degree of enrichment. Cd concentrations were substantially higher than values reported in comparable coastal and mangrove studies (Ahmad et al., 2021; Onjefu et al., 2020; Siddique et al., 2025;

Wu et al., 2014). Moreover, Cd levels exceeded the World Health Organization guideline value (0.8 mg/kg) and the Pakistan-EPA standard (3 mg/kg), contributing to elevated CF and Igeo classifications (Ayaz et al., 2023). These findings suggest potential anthropogenic inputs, which may include industrial discharge, improper waste disposal, stormwater transport, or storm-driven redistribution of contaminants. Since no pre-Dorian sediment data are available for these sites, any contribution of Hurricane Dorian in 2019 to contaminant redistribution remains hypothesized rather than demonstrated and warrants direct investigation in future studies. Also, it can be noted that the root systems of mangroves can trap toxic metals in sediments, which may have increased heavy metal concentrations to levels that exceeded their QRVs (de Lacerda et al., 2022).

Mn, Ni, Cd, and Cu differed significantly between DS and EGB ($p < .05$), indicating heterogeneity in metal distribution likely influenced by differences in land use, hydrodynamic conditions, or proximity to contamination sources. In contrast, Cr, Sb, Pb, and As showed no significant differences between sites ($p > .05$), suggesting that their distribution may be more strongly influenced by natural geochemical background conditions (Siddique et al., 2025). Overall, Cd emerged as the dominant contaminant across both mangrove systems and represents the primary contributor to ecological risk in this study.

Pollution Load Index

Pollution Load Index (PLI) values for both DS and EGB were less than 1 (Table 2), indicating that overall sediment contamination did not exceed the pollution threshold. However, PLI values were close

to unity (0.80–0.87), and under the applied classification criteria, this still indicates no overall pollution. Because PLI is calculated as the geometric mean of all CF values, it integrates contamination across all evaluated metals. This averaging effect may obscure the influence of metals with disproportionately high contamination levels, such as Cd (Table 3). Consequently, reliance solely on PLI may underestimate localized or metal-specific ecological risk. Interpretation of sediment quality, therefore, requires consideration of individual contamination indices (CF, Igeo) in addition to composite measures such as PLI.

Contamination Factor

Contamination Factor analysis confirmed substantial enrichment of Cd and Sb at both DS and EGB (Table 3), with Cd exhibiting the highest degree of contamination. Notably, the CF for Cd in EGB exceeded the high contamination threshold ($CF > 6$) by approximately 122-fold, indicating extreme enrichment relative to background levels. Such elevated Cd concentrations may reflect anthropogenic inputs, including industrial activities, improper waste disposal, atmospheric deposition, or storm-driven redistribution of contaminated materials. Cd can enter coastal systems through both natural weathering and human-induced processes such as metal processing and emissions, and its persistence facilitates transport via water and sediment pathways (Ayaz et al., 2023). Sb contamination at both sites may similarly reflect anthropogenic sources, including industrial emissions and waste-related inputs (Li et al., 2023). In contrast, Cu exhibited moderate contamination in DS, consistent with findings from comparable coastal sediment studies (Hossain et al., 2022). Overall, CF results reinforce that Cd is the primary driver of contamination in these mangrove systems.

Geo-Accumulation Index

The Geo-Accumulation Index was used to evaluate sediment metal enrichment relative to background concentrations (El Turk et al., 2018). In the present study, Cu at DS was classified as unpolluted to moderately polluted, consistent with findings from mangrove systems reported by Hossain et al. (2022). Cd exhibited the highest enrichment, with classifications ranging from strongly to very strongly polluted in DS and very strongly polluted in EGB, while Sb was classified as moderately to strongly polluted (Figure 2). These elevated Igeo values reflect substantial deviation from background concentrations and indicate anthropogenic influence.

A consistent relationship was observed between CF and Igeo classifications, whereby metals exhibiting high CF values also showed elevated Igeo classes. This correspondence strengthens confidence in the contamination assessment, as both indices independently identified Cd as the dominant contributor to sediment enrichment. Although previous studies have reported variable Igeo ranges for Cd, often linked to anthropogenic activities (Adimalla, 2020), the magnitude of Cd enrichment in this study was considerably greater. Research on Sb in mangrove sediments remains comparatively limited outside mining contexts (Li et al., 2023), making regional comparisons challenging. Overall, Igeo results confirm significant enrichment of Cd and moderate enrichment of Sb and Cu, suggesting localized anthropogenic inputs influencing sediment quality in these mangrove systems.

Potential Ecological Risk Index

Ecological risk assessment revealed that Cd and Sb were the primary contributors to elevated RI values across sampling sites. Cd exhibited very high E_r^i values at most

locations, consistent with previous studies identifying Cd as a metal of significant ecological concern in coastal sediments (Ahmad et al., 2021). In contrast, Islam et al. (2017) reported low Sb ecological risk coefficients and only considerable Cd risk in their study area, whereas the present study identified moderate to considerable E_r^i values for Sb and very high E_r^i values for Cd. These comparisons indicate that Cd enrichment in DS and EGB exceeds levels reported in several other coastal systems.

The integrated RI values were predominantly classified as very high, largely driven by the elevated Cd E_r^i values, consistent with findings reported by Wu et al. (2014), where Cd similarly dominated ecological risk assessments. The close correspondence between Cd E_r^i distribution (Supplement Figure S1d) and RI values (Figure 3) demonstrates the strong influence of Cd on overall ecological risk. At sampling points D2, D3, and D4, where Cd concentrations were absent or negligible, Sb contributed most to RI values. Among all locations, E3, E4, and E6 exhibited the highest ecological risk, with E4 recording the maximum RI value (60,096.12). These findings confirm that Cd, and to a lesser extent Sb, are the dominant drivers of ecological risk in the studied mangrove sediments.

Non-Carcinogenic and Carcinogenic Health Risk

Non-carcinogenic and carcinogenic risks were assessed for adults and children across three exposure pathways (ingestion, inhalation, and dermal contact) using the U.S. EPA human health risk assessment framework (1989; 2002). This model applies standardized exposure parameters and is widely used for screening-level risk evaluation; however, it represents conservative assumptions and may not

reflect site-specific behavioral variability. In the present study, Cd was the primary contributor to non-carcinogenic risk. Maximum HI values for Cd exceeded 1 for both adults and children, and the mean HI value for children also exceeded 1 (Table 4). According to U.S. EPA guidance, HI values greater than 1 indicate the potential for non-carcinogenic effects and warrant further evaluation. All other evaluated metals had HI values below 1. Consistent with previous studies (Adimalla, 2020), HI estimates were higher for children than adults, reflecting differences in body weight and exposure intake relative to body mass under the applied model assumptions.

Carcinogenic risk assessment similarly identified Cd as the dominant contributor to TCR. Maximum, minimum, and mean TCR values for Cd exceeded the U.S. EPA benchmark value of 1×10^{-4} (Table 5), which is generally considered indicative of potential concern under screening-level assessment. These elevated TCR values are consistent with the high sediment Cd concentrations and corresponding ecological risk indices observed in this study. It is important to note that carcinogenic risk estimates represent probabilistic lifetime risk projections based on standardized exposure assumptions and do not confirm actual health outcomes. Carcinogenic risk was not evaluated for Mn, Ni, Sb, Cu, and Zn due to the absence of U.S. EPA slope factor values for these metals.

Limitations

While the present study provides important insight into ecological and human health risks associated with heavy metal contamination in Bahamian mangrove sediments, several methodological constraints should be considered when interpreting the findings. First, The Bahamas currently lacks locally established

geochemical background concentrations for heavy metals in mangrove sediments. Consequently, background values applied in this study were referenced from geographically proximate regions with broadly similar environmental settings. However, given the carbonate-dominated geology of The Bahamas, particularly its limestone substrate, true baseline concentrations may differ from those used here. Therefore, contamination indices such as CF, Igeo, and RI should be interpreted within this context. Establishing site-specific background concentrations for Bahamian coastal sediments should be prioritized in future studies to enhance the accuracy of contamination and risk assessments.

Conclusion

This study provides the first integrated assessment of heavy metal contamination and associated ecological and human health risks in mangrove sediments of DS and EGB, Grand Bahama. Although PLI values indicated that overall sediment quality at both sites did not exceed the pollution threshold, metal-specific indices revealed substantial enrichment. CF and Igeo analyses identified Cd and Sb as the primary contaminants, with Cd exhibiting extreme enrichment, particularly in EGB. RI results further demonstrated that Cd was the dominant contributor to ecological risk across most sampling points, while Sb contributed to localized elevations in ecological risk where Cd concentrations were low.

Human health risk assessment using the U.S. EPA screening framework similarly identified Cd as the principal metal of concern. Non-carcinogenic HI risk estimates exceeded the benchmark value of 1 for Cd at several locations, particularly for children, and TCR exceeded the U.S. EPA threshold for potential concern. These findings indicate that, while overall composite

pollution indices suggest limited system-wide contamination, Cd enrichment represents a localized but significant ecological and potential human health concern under conservative, screening-level exposure assumptions within these mangrove environments.

The results underscore the need for continued monitoring of Cd and Sb in Bahamian coastal systems and highlight the importance of establishing site-specific geochemical baseline values for improved risk characterization. Given that the non-carcinogenic hazard indices for Cd exceeded unity for children at the most enriched locations (E3, E4, and E6), precautionary measures, including site-specific exposure assessment and consideration of public advisories at these sites, are warranted pending further study. Expanded regional assessments across additional islands, supported by robust background datasets, would strengthen national environmental management strategies and contribute to evidence-based coastal policy development, including long-term sustainability initiatives such as Vision 2040 (Bahamas National Development Plan, 2016).

Authors' contributions

WG, CW and VT contributed to the design and implementation of the research experiment. AC, WG and TH processed and conducted the data analysis. AC, TH, CW, VT and WG provided necessary feedback and insight into the organization of the paper and results. The draft of the manuscript was composed by AC, TH, VT, CW and WG.

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Abbreviations

As	Arsenic
AT	Average time
BW	Average body weight
C	Concentration of heavy metals
C _m	Measured concentration of heavy metals
CB _n	Quality reference value / Background value
Cd	Cadmium
CDD	Chronic daily dose
CF	Contamination factor
CF	Conversion factor
Cn	Metal Concentration
CR	Carcinogenic risk
Cr	Chromium
Cu	Copper
DAF	Dermal absorption factor
DS	Dover Sound
ED	Exposure duration
EF	Exposure frequency
EGB	East Grand Bahama
E _r	Risk coefficient
Fe	Iron
Hg	Mercury
HI	Hazard Index

HQ	Hazard quotient
ICP MS	Inductively Coupled-Plasma Mass Spectrometer
Igeo	Geo-accumulation index
IR	Ingestion rate
IR _{inhalation}	Inhalation rate
Mn	Manganese
Ni	Nickel
PEF	Particle emission factor
Pb	Lead
PLI	Pollution load index
QRV	Quality reference value
RfD	Reference dose
RI	Risk index
SA	Exposed skin surface area
SAF	Skin adherence factor
Sb	Antimony
SF	Slope factors
TCR	Total carcinogenic risk
T _r	Metal toxicity
U.S. EPA	United States Environmental Protection Agency
WHO	World Health Organization
Zn	Zinc

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