Rare earth elements in oysters and mussels collected from the Chinese coast: Bioaccumulation and human health risks

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Abstract

Rare earth elements (REEs) are increasingly used in various industries worldwide, resulting in their release into aquatic ecosystems. We evaluated the distribution and bioaccumulation of 14 REEs in marine sediments and biota (oysters and mussels) along the Chinese coasts. The total concentration of REEs (ΣREEs) in sediment samples was 41.65-170.94 mg/kg, where Ce concentration was the highest and Tm and Lu concentrations the lowest. The concentration of total light REEs (ΣLREEs) was higher than the concentration of total heavy REEs (ΣHREEs) at all study sites. The concentrations of ΣREEs were 1.97–4.77 mg/kg and 0.62–4.96 mg/kg dry mass (DM) for oysters and mussels, respectively. The bioaccumulation of ΣLREEs was higher than ΣHREEs in oysters and mussels. The bioaccumulation factor (BAF) for ΣREE was 0.34–1.49 and 0.25–1.10 for oysters and mussels, respectively, where the BAF was relatively higher in species collected from the south than the north. A positive relationship of REEs was found in bivalves, with a significantly higher correlation of HREEs than LREEs. The correlation between sediment and biotas was higher in mussels than in oysters, showing a good potential for being environmental indicators for REEs. The risk of REEs to humans via bivalve consumption could be negligible based on the estimated daily intake of REEs in oysters and mussels.

Keywords: Rare earth elements, Oysters, Mussels, Bioaccumulation, Health risk

1. Introduction

Rare earth elements (REEs) are emerging pollutants comprising lanthanide series elements (La, Ce, Pr, Nd, Pm, Sm, Eu, Gd, Tb, Dy, Ho, Er, Tm, Yb, and Lu), yttrium (Y), and scandium (Sc). They are used extensively in high technology areas, such as electronics (e.g., luminescent material of display), manufacturing (e.g., metal alloys), medicine (e.g., magnetic resonance imaging), clean energy (e.g., rechargeable batteries in hybrid cars), and agriculture (e.g., fertilizer) (Balaram, 2019). China has the largest reserve of rare earth element resources and the largest production chains in the world. Low-weight REE resources are mainly distributed in the Bayan Obo mining area in Baotou, Inner Mongolia, and their rare earth element reserves account for more than 83% of the country's total rare earth reserves. Compared with the northern region, the southern region is rich in high-weight REE resources (Ma et al., 2019).

The extensive industrial application of REEs has led to their release into aquatic and terrestrial ecosystems (Gu et al., 2020; Wang et al., 2022). Wang et al. (2022) measured the total concentrations of REEs (ΣREEs) that ranged from 1.02 to 178.55 μg/kg in 14 marine wild fish species from the northern coastal region of the South China Sea; a mean ΣREE of 0.35 mg kg^{-1} was reported in shellfish from the southern South China Sea, and a mean value of 0.12 mg kg^{-1} was reported in zooplankton from northwestern Italy (Li et al., 2016; Squadrone et al., 2019). However, REEs have been shown to reduce growth and nutritional quality and impair the metabolic functions of plants (Carpenter et al., 2015); produce genotoxicity and neurotoxicity for biotas (Blinova et al., 2018; Trifuoggi et al., 2017); bioaccumulate across food chains, are chronically and acutely toxic to soil organisms (Gardon et al., 2018); and cause nephrogenic systemic fibrosis, dysfunctional neurological disorder, fibrotic tissue injury, and male sterility in humans (Prince et al., 2008; Thomsen, 2017).

Marine filtering species, such as oysters and mussels, are exposed to various contaminants, including REEs, hence their wide use in the biomonitoring of environmental contaminants (Briant et al., 2021; Schaefer et al., 2022). As filter-feeding animals and primary consumers, oysters and mussels constitute major components of coastal trophic networks and ecosystem functioning (Briant et al., 2018). They can accumulate pollutants in their tissues at elevated levels related to pollutant availability in the marine environment (Beyer et al., 2017). On the other hand, oysters and mussels have high protein and amino acid contents consumed by humans worldwide (Venugopal and Gopakumar, 2017). Therefore, the accumulation of REEs in oysters and mussels poses a health risk to humans *via* dietary intake (Adeel et al., 2019). Studies on the accumulation of REEs in oysters and mussels and the corresponding human health risks posed by their consumption remain unclear. Therefore, this study was designed to (i) determine the REE concentrations in oysters and mussels along the Chinese coastline and determine the bioaccumulation factor of REEs from sediment to biological tissues; (ii) evaluate the possibility of using oysters and mussels as an indicator for REE concentrations in the local environment; and (iii) assess the possible human health risk of REEs via oyster and mussel intake.

2. Materials and Methods

2.1. Study site and sample collection

Sediment and bivalve samples were collected at six sites from four provinces (Liaoning, Jiangsu, Zhejiang and Guangdong) along the Chinese coastline between March and May 2021 (Fig. 1). Surface sediment samples were collected using a grab sampler, at about top 10 cm depth, the sediment was homogenized and divided into three portions for chemical analysis. The bivalve samples were collected with the help from local fishermen with fishing cages. Bivalves sample with similar sizes were chosen for the experiment, about five individuals were used for one mixed sample, and three mixed samples were used for chemical analysis. All collected samples were kept in clean polyethylene bags, sealed, and transported to the laboratory with ice. They were stored in the refrigerator at -20 °C in the laboratory.

Fig. 1. Study area and location of the sampling sites of the samples.

2.2. Chemicals and reagents

All reagents were purchased from Sigma Aldrich, Fisher Chemical, and Merck. Solutions were prepared in deionized water (resistance >18 MΩ·cm) using a Milli-Q water purification system (Millipore). The rare earth element mix for ICP (Trace CERT®, Sigma–Aldrich, Switzerland) was used as a calibration standard for total elemental analysis. Nitric acid (65% HNO_3, Trace Metal Grade, Fisher Chemical, Canada), hydrogen peroxide (30% H_2O_2, Merck, Germany), hydrochloric acid (37%

HCl, Merck), and hydrofluoric acid (30% HF, Merck, Germany) were used for sample treatments.

2.3. Sample preparation and determination

Each sample was washed with tap water and rinsed with ultrapure water. The edible tissue was harvested and freeze-dried for 96 hours. After freeze-drying, samples were ground to powder using a mortar and ball mill. Approximately 0.1 ± 0.01 g dry weight (dw) oyster/mussel powdery samples were digested with 3 ml HNO_3 and 1 ml H_2O_2 mixture in a Teflon® TFM-lined digestion vial using Mars XpressTM (CEM, USA). The temperature programmed in Mars Xpress was set to 20 min ramps to 190 °C and held for 20 min at 190 °C (Lin et al., 2021). Sediment samples were digested in 3 ml HNO_3, 1 ml HCl and 1 ml HF mixture in the digestion vessel using Mars Xpress^{TM} (CEM, USA). The temperature programmed in Mars Xpress was set to 20 min ramps to 210 °C and held for 20 min at 210 °C (Zhao et al., 2022). All the digested samples were filled to 50 ml in polyethylene centrifugation tubes with ultrapure water. The concentrations of REEs in sediments, oysters, and mussels were determined using ICP–MS/MS (iCAP TQ, Thermo Fisher, Germany) using a standard calibration curve and internal standards (10 ppb) Sc, In, Rh, and Ru for mass correction. The KED and O_2 modes were used for element analysis in the ICP–MS/MS, and the detailed operating and reaction modes are shown in the supplemental information (Table S1). All sediment, oyster, and mussel samples were run in batches with standards inserted every ten samples, method blanks, and certified reference materials. Analysis precision and accuracy were ensured using the National Center for Material Standards, near-shore marine sediment certified reference material (GBW 07314), GBW10024 (GSB-15, Scallop) from the Institute of geophysical and geochemical exploration, Chinese Academy of Geological Sciences (IGGE). The analyzed results were within ±5% of the certified value.

2.4. Accumulation of REEs in oysters and mussels

The REEs in sediments to bioaccumulate in oysters and mussels were determined using the bioaccumulation factor (BAF), which is the ratio of the concentration of REEs in oyster or mussel tissue to the concentration of the REEs in sediments assuming that the organism and the sediment are in equilibrium (Mackay et al., 2018).

$$BAF = \frac{C_{oyster\ or\ mussel}}{C_{sediment}}$$

2.5 Human dietary risk

The EDI of REEs through fish consumption was calculated using the following equation (Xu et al., 2020):

$$EDI=\frac{C \times IR}{BW}$$

where C is the concentration of REEs in oysters or mussels expressed as wet weight (mg/kg, ww) and IR is the average daily ingestion rate (g/day) of oysters or mussels. The IR was set at 25 g/day for children and 50 g/day for adults based on the questionnaire survey for traditional residents near Jiaozhou Bay, China (Zhang and Zhang, 2015). BW is the average human body weight (kg), which is considered to be 33 kg for children and 63 kg for adults (Xu et al., 2020).

2.6 QA/QC of rare earth elements determination

All the sample was prepared in the clean chamber in the inorganic chemistry lab, the certified reference material (CRM) for sediment and biological tissues were used for ICP-MS/MS determination verification. The detected CRM values were within ±5% of the certified value for the REEs. The LOD for the rare earth elements ranged 1.0×10^{-5} to 1.0×10^{-4} μg/L, and LOQ was calculated as three times concentration of LOD. All the detected REEs concentrations were above the minimum limits of quantification.

2.7 Statistical analysis

One-way ANOVA (analysis of variance) and nonparametric tests were performed to find significant variation between oysters and mussels at different sites using SPSS 25.0. Prior to analysis, all data were checked for normality using the Shapiro–Wilk test. Nonparametric tests such as Kruskal–Wallis were used for the data displaying nonnormality. A significance level of $p < 0.05$ was accepted for all statistical analyses.

3. Results and Discussion

3.1. Concentrations of REEs in sediments

The concentrations of REEs in sediment samples collected at different study sites along the Chinese coastline are shown in Table 1. The total concentration of REEs (ΣREEs) in sediment samples was 41.65-170.94 mg/kg. The concentration of Ce (15.02 ~ 80.59 mg/kg) was the highest, while that of Tm (0.07 ~ 0.34 mg/kg) and Lu (0.07 ~ 0.35 mg/kg) was the lowest. The ΣLREEs were higher than the ΣHREEs at all the study sites from the Bohai Sea to the South China Sea, while the ΣLREEs/ΣHREEs ratio ranged from 9.28 to 10.76, which indicated that LREEs were more easily enriched than HREEs in sediments. The REE content of Jiangsu in the North Yellow Sea was lower than that in other locations because the North Yellow Sea is dominated by Yellow River discharges carrying loess with a low REE content in the sediment (Jiang et al., 2008). The REE concentrations found in Liaoning and Zhejiang in the present study were similar to those obtained in a previous study, suggesting that there have been no changes in the discharge or removal of the REEs in this region (Mi et al. (2020)). The ΣREEs in Guangdong in this study were higher than those in the southern South China Sea (2.62-3.15 μg/kg) due to the differences in anthropogenic activities Li et al. (2016). Moreover, terrestrial weathering and anthropogenic activities will lead to a higher REE input to the local environment, where the concentration of REEs in coastal wetland sediment in the Vembanad estuary (29.55-229.67 mg/kg) was higher than that in our study (Manoj et al., 2016). Alhassan and Aljahdali (2021) found that the total concentration of REEs in sediment on the Red Sea coast was 12.2-108.98 mg/kg, which was lower than our study due to an underdeveloped high-tech industry and fewer human activities in Africa. Owing to rapid economic development, high-density industrial distribution, and abundant anthropogenic activities along the Chinese coastline, we are concerned that excessive contaminants in the sediments may elevate the bioaccumulation of such contaminants in the associated biota.

Table 1 Concentration of REEs in the sediments (In this study, n=3).

REEs	La mg/kg	Ce mg/kg	Pr mg/kg	Nd mg/kg	Sm mg/kg	Eu mg/kg	Gd mg/kg	Tb mg/kg	Dy mg/kg	Ho mg/kg	Er mg/kg	Tm mg/kg	Yb mg/kg	Lu mg/kg	L/H	∑REEs mg/kg
This study																
LN	23.36 ± 2.61	70.77 ± 7.92	7.97 ± 0.89	24.95 ± 2.79	5.05 ± 0.56	0.92 ± 0.10	4.19 ± 0.46	0.58 ± 0.06	3.41 ± 0.38	0.67 ± 0.07	1.91 ± 0.21	0.27 ± 0.03	1.47 ± 0.16	0.28 ± 0.03	10.41 ± 1.16	145.81 ± 16.33
JS	13.66 ± 1.57	15.02 ± 1.72	1.75 ± 0.20	5.90 ± 0.67	1.25 ± 0.14	0.44 ± 0.05	1.04 ± 0.12	0.14 ± 0.01	0.84 ± 0.09	0.16 ± 0.02	0.47 ± 0.05	0.07 ± 0.01	0.77 ± 0.08	0.07 ± 0.01	10.55 ± 1.21	41.65 ± 4.79
ZJ 1	25.67 ± 2.90	52.45 ± 5.92	6.08 ± 0.68	20.32 ± 2.29	4.19 ± 0.47	0.87 ± 0.09	3.37 ± 0.38	0.46 ± 0.05	2.61 ± 0.29	0.51 ± 0.05	1.46 ± 0.16	0.21 ± 0.02	1.31 ± 0.14	0.21 ± 0.02	10.76 ± 1.21	119.79 ± 13.53
ZJ 2	4.12 ± 0.37	46.08 ± 4.14	5.34 ± 0.48	17.93 ± 1.61	3.76 ± 0.33	0.13 ± 0.01	3.01 ± 0.27	0.43 ± 0.03	2.42 ± 0.21	0.48 ± 0.04	1.35 ± 0.12	0.19 ± 0.01	0.25 ± 0.02	0.19 ± 0.01	9.28 ± 0.83	85.72 ± 7.71
GD 1	28.08	80.59	8.99	30.57	6.37	0.97	4.94	0.72	4.22	0.80	2.19	0.34	1.75	0.35	10.14	170.94

	± 3.37	± 9.67	± 1.08	± 3.66	± 0.76	± 0.11	± 0.59	± 0.08	± 0.51	± 0.09	± 0.26	± 0.04	± 0.21	± 0.04	± 1.21	± 20.51
GD 2	17.35 ± 1.94	67.12 ± 7.51	7.55 ± 0.84	25.11 ± 2.81	5.29 ± 0.59	0.57 ± 0.06	4.26 ± 0.47	0.61 ± 0.06	3.53 ± 0.39	0.69 ± 0.07	1.99 ± 0.22	0.29 ± 0.03	1.08 ± 0.12	0.30 ± 0.03	9.61 ± 1.07	135.80 ± 15.21
Other studies																
The Southern South China Sea	0.73	0.62	0.13	0.60	0.14	0.03	0.17	0.03	0.16	0.03	0.10	0.01	0.08	0.01	3.72	2.88
The Central Red Sea	2.26 ~ 16.54	5.45 ~ 37.36	0.66 ~ 4.73	2.84 ~ 21.60	0.61 ~ 4.65	0.17 ~ 1.34	0.58 ~ 4.45	0.09 ~ 0.74	0.52 ~ 4.09	0.10 ~ 0.79	0.25 ~ 2.28	0.04 ~ 0.37	0.26 ~ 2.25	0.03 ~ 0.36	5.60 ~ 7.13	13.85 ~ 101.53
The Vembanad estuary	6.25 ~ 50.55	13.33 ~ 104.62	1.46 ~ 10.61	5.41 ~ 41.58	0.91 ~ 7.53	0.22 ~ 1.76	0.72 ~ 6.65	0.09 ~ 0.91	0.50 ~ 5.25	0.08 ~ 0.97	0.26 ~ 2.83	0.03 ~ 0.37	0.26 ~ 2.43	0.03 ~ 0.35	11.08 ~ 14.00	29.55 ~ 229.67

Note: LN: Liaoning, JS: Jiangsu, ZJ: Zhejiang, GD: Guangdong

3.2. Concentrations of REEs in oysters and mussels

Marine filtering species have an abundant capacity to filter pollutants from the surrounding seawater (Briant et al., 2021; Schaefer et al., 2022). In the present study, the ΣREE concentrations were 1.97 to 4.77 mg/kg dry mass (DM) and 0.62 to 4.96 mg/kg DM for oysters and mussels, respectively (Fig 2, Table S2). A few studies reported the REE concentrations in various marine biota, where the ∑REEs ranged from 0.16-9.10 mg/kg DM for oysters and 0.16-5.00 mg/kg for mussels in French Metropolitan Coasts (Briant et al., 2021), Canadian Artic (MacMillan et al., 2017), Ligurian Sea, Italia (Squadrone et al., 2019), Zhuhai, China (Briant et al., 2021; Ma et al., 2019). Habitats may have a great influence on the REE concentrations in oysters and mussels. Oysters (*Crassostrea gigas*) collected in the Gironde River mouth estuary had a higher concentration of ΣREEs (10.94 mg/kg) than Persuel Bay (0.29 mg/kg) in French Metropolitan Coasts. The ΣREE concentration in mussels (*Mytilus edulis* and *Mytilus galloprovincialis*) collected in the littoral zone was only 0.18-2.33 mg/kg from the same region (Briant et al., 2021). For filtering species, oysters and mussels tend to have higher capacities for accumulating pollutants in their tissues than fish and crustaceans. Li et al. (2016) found concentrations of ΣREEs in fishes (0.004-0.045 mg/kg) and crustaceans (0.10-1.95 mg/kg) from the southern South China Sea. Wang et al. (2022) reported that the average concentration of ΣREEs in 14 fish species from the northern South China Sea was 0.003-0.093 mg/kg, which was significantly lower than the REE concentrations in the filtering species.

The chondrite-normalized patterns (REE_{CN}) of oysters and mussels from different study sites were similar, featuring undulating shapes with a maximum at La and a minimum at Ce (Fig. 3). The chondrite-normalized patterns (REE_{CN}) of oysters in the present study were different from those of French *Crassostrea gigas* characterized by a constant increase from La to Lu with slight Ce depletion (Briant et al., 2021). We implied that the difference was attributed to the seawater environment and differences in REE availability between sites because this pattern was similar to the chondrite-normalized plot in oysters in the Pearl River Estuary except Ce (Ma et al., 2019). The REE_{CN} of mussels in this study was different from that of mussels from the Peace Lagoon on the north beach of Eilat (Benaltabet et al., 2021). Filtering organisms feed on suspended particles and microalgae; thus, the REEs concentration in those species may correspond to the surrounding environment. Although the REE concentrations varied among the different study sites, the individual REEs showed a similar enrichment pattern.

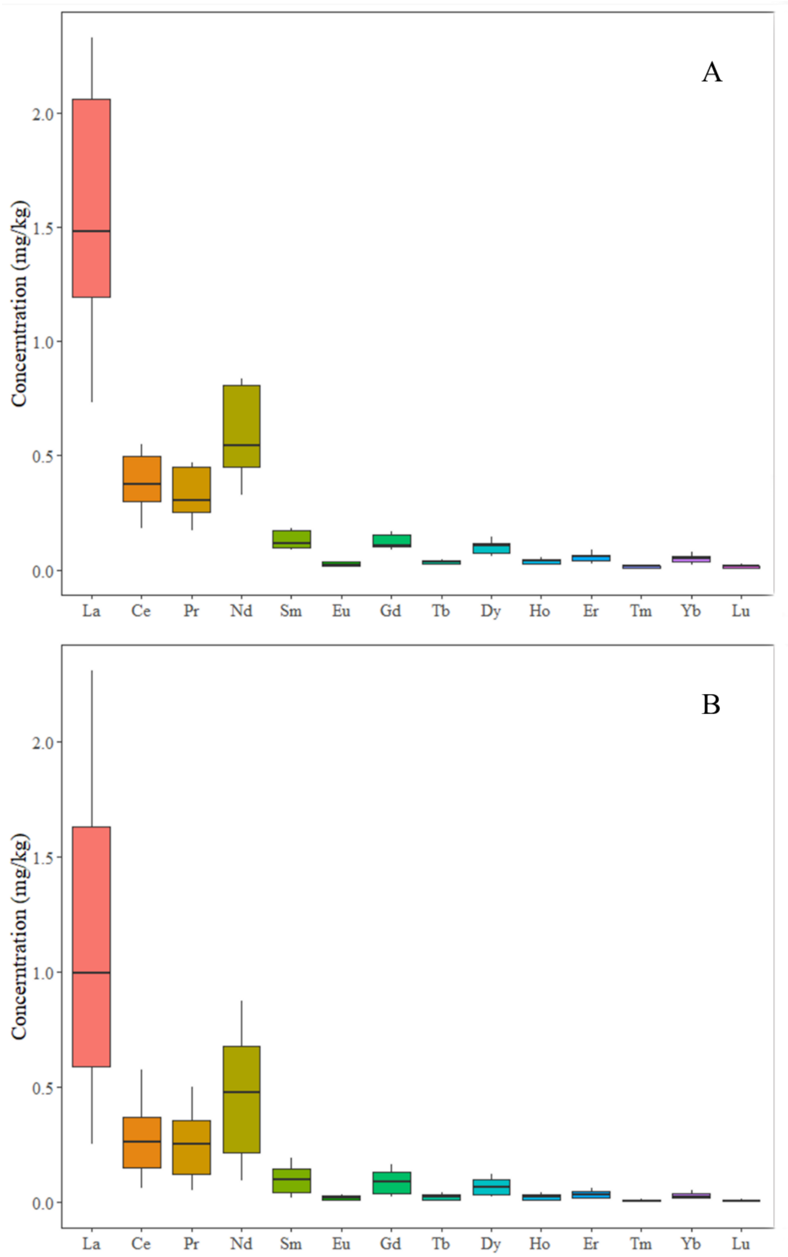

Fig. 2. Concentration of REEs in the oysters (A) and mussels (B).

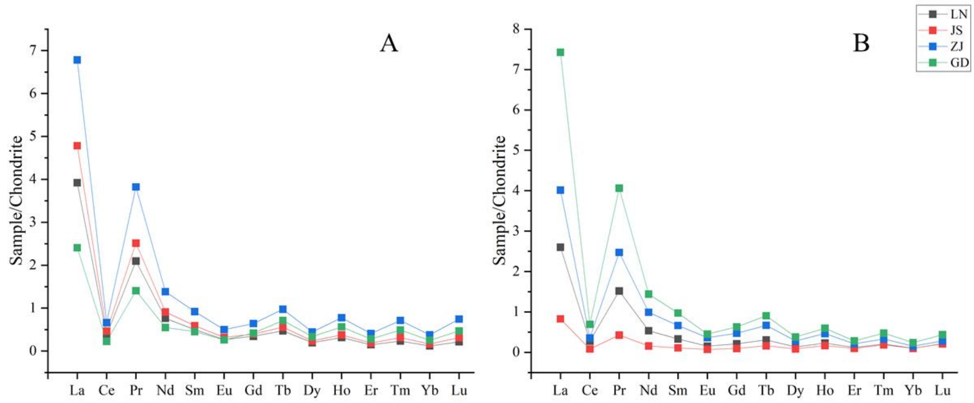

Fig. 3. Chondrite-normalized plots for the REEs in the soft tissue of oysters (A) and mussels (B). The chondrite values are from (Boynton, 1984).

3.3. Bioaccumulation factors of REEs in oysters and mussels.

In general, the bioaccumulation factor (BAF) of ΣREEs was less than one, except the BAF for the oysters (JS) and mussels (ZJ) were 1.49 and 1.10 (Fig. 4, Table S3). The BAF of ΣREEs from sediment to oysters and mussels varied among the sampling sites. The lowest BAF of ΣREEs for oyster and mussels was in Liaoning (LN) at the temperate zone, whereas a trend of increased BAF was found in other sampling sites with lower latitude, except the BAF for oyster in Guangdong (GD). The accumulation characteristics were similar for both species, except a higher BAF of Tb was found, possibly owing to the low Tb concentration in the sediment (Table 1, Table S3).

The total bioaccumulation factor of REEs was lower than one, except for oysters (JS) and mussels (ZJ), which were 1.49 and 1.10, respectively. The bioaccumulation factor of ΣREEs from sediment to oysters and mussels varied among the species and site locations, where the total BAF was higher in the oysters than in mussels (Fig 4). The total BAFs for the oysters and mussels ranged from 0.34 to 1.49 and 0.25 to 1.10, respectively. The BAF was element specific and site specific. The total BAFs of LREEs (LBAF) and the total BAFs of HREEs (HBAF) were higher in Jiangsu and lower in Guangdong in oysters. LBAF and HBAF were higher in Zhejiang and were lower in Liaoning in mussels. LBAF was lower than HBAF in oysters and mussels, except

LBAF was similar to HBAF in Zhejiang in mussels. The accumulation characteristics were different among species, where the Tb was highest in oysters and La was the highest in mussels. Interestingly, a previous study showed that La was more toxic to mussel embryos than other REEs (Mestre et al., 2019).

Marine biota, such as bivalves and fish, may constitute a pathway for human or animal dietary exposure (Squadrone et al., 2019). REE bioaccumulation patterns appear to be species- and tissue specific, and analysis of variance between taxa from the same ecosystem showed that biota at the base of the food web (vegetation, invertebrates) had significantly higher ∑REE concentrations than vertebrate muscle samples from the same ecosystem (MacMillan et al., 2017). Previous studies on aquatic vertebrates have shown that REEs are more concentrated in internal organs (liver, kidney, intestine, gills) than in muscle (Amyot et al., 2017; Copetti et al., 2016). Amyot et al. (2017) found that fish muscles are the edible parts consumed by human beings, and the bioaccumulation of REEs is very low. Although relatively higher bioaccumulation of REEs was found in bivalves, the potential bioaccumulation for REEs was limited (0.002-0.507) in oysters and mussels in the present study.

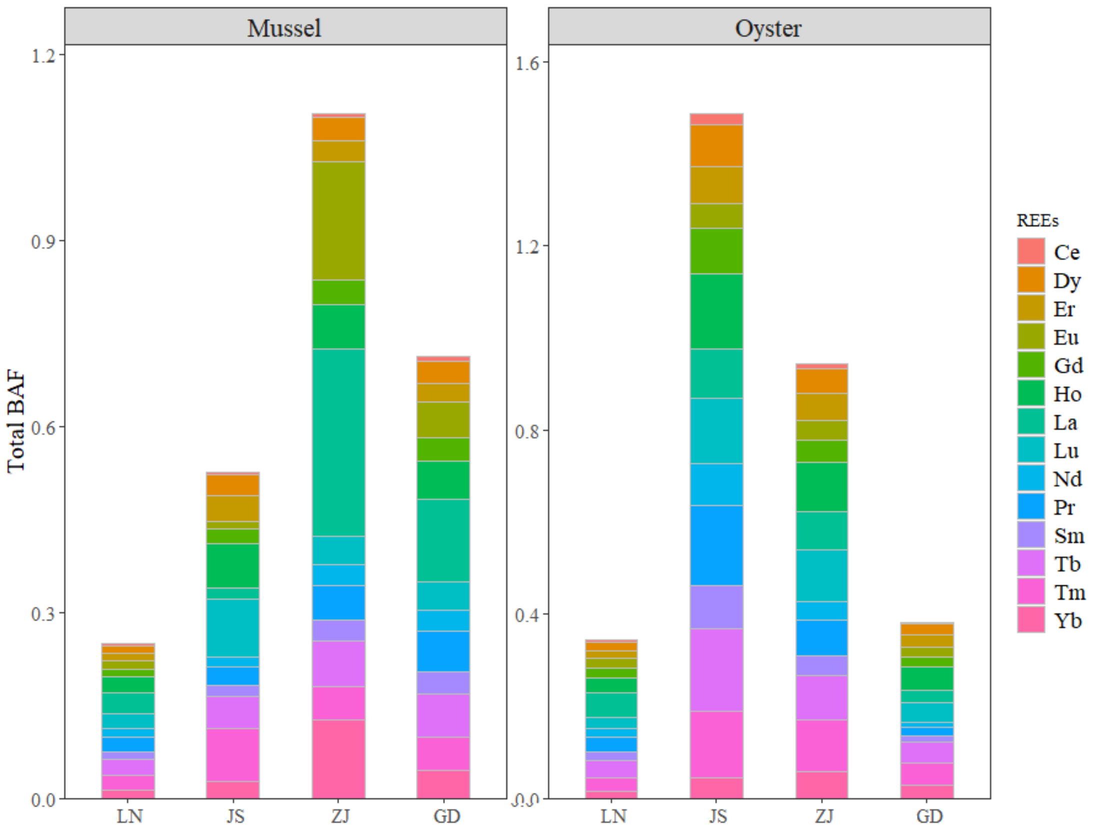

Fig. 4. BAF of REEs in oysters and mussels.

3.4. Relationship of the REEs between biotas and sediments

REEs are usually divided into LREEs and HREEs for analysis because their atomic, physical, and chemical properties vary gradually along the series. In this study, we investigated the relationship of LREEs and HREEs between sediment and biotas (Fig 5). The relationships presented a positive correlation, and the trend of HREEs showed a higher positive correlation. We deduced that this phenomenon was linked with the preferential scavenging of LREEs in waters and stronger complexation of HREEs in seawater (Deng et al., 2017; Elderfield and Greaves, 1982), so the filtering biotas could better absorb HREEs and present a more positive correlation with HREEs. In addition, we found the trend of mussels was more positive than oysters, and we thought it might be attributed to biological specific within the oysters and mussels.

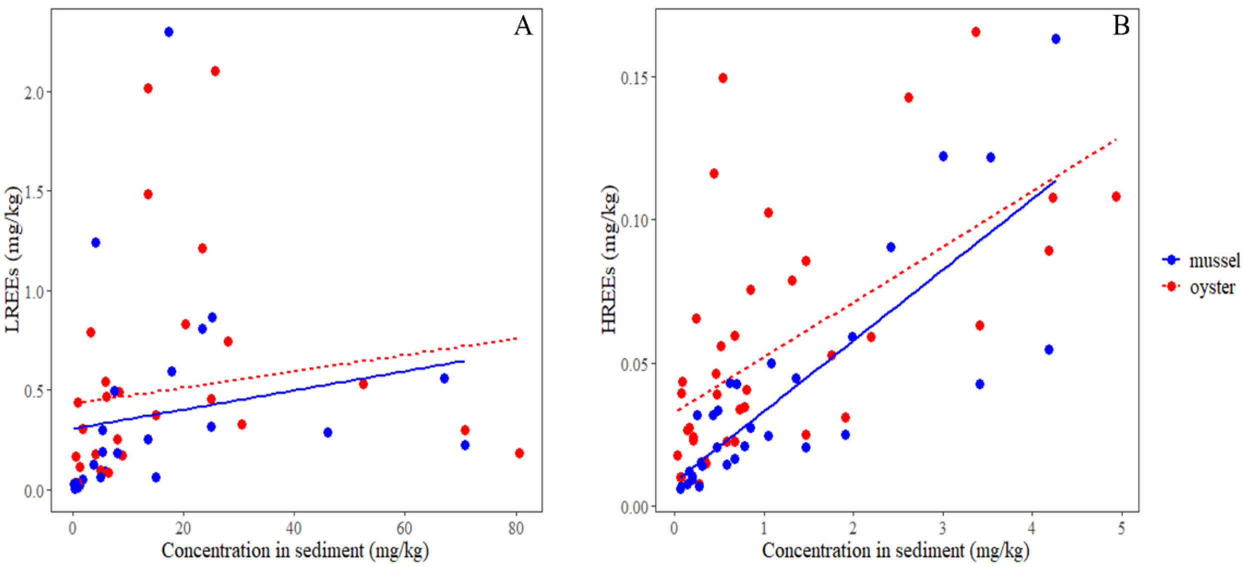

Fig. 5. Relationship of LREEs (A) and HREEs (B) between sediment and biotas.

3.5. Human dietary risk

REEs accumulate in marine sediments and biotas and ultimately enter the food chain. The risks, impacts, and chronic toxicity of widespread REE sediments are of concern due to their environmental persistence (Charalampides et al., 2016). Exposure to REEs may cause dysfunctional neurological disorders such as a reduced intelligence quotient (IQ) in children (Gwenzi et al., 2018). Bone alteration, genotoxicity, and fibrotic tissue injury were reported to be associated with several REEs (Chen and Zhu, 2008; Jenkins et al., 2011). Moreover, Marzec-Wroblewska (2015) showed anti-testicular effects and male sterility after REE exposure. The recommended threshold level of REEs is 70 μg/kg/day, based on an extensive human health survey in REE mining areas together with animal experiments (Zhu et al., 1997). In the present study, we tested the ∑REEs of oysters and mussels from different sites and calculated the EDI, which was divided between adults and children (Table 2). The EDIs of ΣREEs from oyster and mussel consumption were significantly lower than the recommended EDI threshold proposed by Zhu et al. (1997), indicating that the risk of REE exposure to humans via oyster and mussel consumption was negligible. However, populations living along the coast generally consume more seafood than those living inland, and the consumption of seafood may be significantly higher, thus leading to increased exposure to REEs (Wang et al., 2022).

Table 2 Estimated daily intake (EDI) of ΣREEs through oyster and mussel consumption by the general population of China.

		∑REEs	IR (g/day)		BW (kg)		EDI (μg/kg/day)	
			adults	children	adults	children	adults	children
Oyster	LN	0.654	50	25.1	63	33	0.52	0.50
	JS	0.793	50	25.1	63	33	0.63	0.60
	ZJ	1.193	50	25.1	63	33	0.95	0.91
	GD	0.493	50	25.1	63	33	0.39	0.37
Mussel	LN	0.450	50	25.1	63	33	0.36	0.34
	JS	0.155	50	25.1	63	33	0.12	0.12
	ZJ	0.740	50	25.1	63	33	0.59	0.56
	GD	1.239	50	25.1	63	33	0.98	0.94

Conclusion

By investigating the distribution of REEs in marine sediments and marine biota, this study expands our understanding of the sources, bioaccumulation, and food safety risks of REEs in marine ecosystems. It also provided critical data useful for assessing effectiveness of mitigation efforts, comparing REE occurrences with other regions, and evaluating human and environmental health effects in future. Sediment analysis confirmed that LREEs are preferentially sequestrated into marine sediments than HREEs. Ce and Nd concentrations in sediment were an order or two orders of magnitude higher than the other 11 REES. The ΣREEs in marine sediments in the Chinese coasts were relatively higher than those reported in other regions probably due to the extensive industrial development and urbanization in China. However, additional studies are required to better understand the role of biogeochemical characteristics of the sediments on the REEs sequestration. Our findings showed that while the bioaccumulation of HREEs in mussels and oysters was positively correlated to marine sediment concentrations, no significant relationship was observed for LREEs. Determining the BAF of REEs in mussels and oysters revealed that REEs bioaccumulation was element-, species-, and site-specific. For example, oysters and mussels, generally, had higher HBAFs than LBAFs but HBAFs and LBAFs in oysters from Jiangsu where higher than those from Guangdong. At elemental level, Tb had the highest BAF in oysters while La had the highest in mussels. This is concerning because La has been shown to be highly toxic to developing mussel embryos than the other REEs. However, studies on the toxicity of REEs to different marine organisms remain scarce. Although the EDIs of ΣREEs from oyster and mussel consumption were significantly lower than the recommended EDI threshold value, populations frequently consuming oysters and mussels may be exposed to substantially higher levels of REEs. There is need for further investigation on the trophic transfer of REEs in various organisms along the marine food web, and at varying temporal and spatial scales to better understand their ecological and human health risks.

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