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Occurrence of uranium, thorium and rare earth elements in the environment: A review

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Uranium, thorium, and rare earth elements (REEs) are important strategic elements in today's world with a range of applications in high and green technology and power generation. The expected increase in demand for U, Th, and REEs in the coming decades also raises a number of questions about their supply risks and potential environmental impacts. This review provides an overview of the current literature on the distribution of these elements in different environmental compartments. For example, the processes of extraction, use, and disposal of U-, Th-, and REE-containing materials have been reported to result in elevated concentrations of these elements in air, in some places even exceeding permissible limits. In natural waters, the above processes resulted in concentrations as high as 69.2, 2.5, and 24.8 mg L^{-1} for U, Th, and REE, respectively, while in soils and sediments they sometimes reach 542, 75, and 56.5 g kg^{-1}, respectively. While plants generally only take up small amounts of U, Th, and REE, some are known to be hyperaccumulators, containing up to 3.5 and 13.0 g kg^{-1} of U and REE, respectively. It appears that further research is needed to fully comprehend the fate and toxicological effects of U, Th, and REEs. Moreover, more emphasis should be placed on developing alternative methods and technologies for recovery of these elements from industrial and mining wastes.

KEYWORDS

rare earth elements, environment, secondary resource, uranium, thorium

1 Introduction

Uranium, thorium, and rare earth elements (REEs) are vital components of today's modern world. They play a critical role in power generation and in many emerging technologies such as magnetic resonance imaging, satellites, batteries, light-emitting diodes (LED), and solar cells, as well as in the production of advanced materials such as catalysts, high-temperature ceramics and welding electrodes.

According to the 2020 Joint Report of the Nuclear Energy Agency (NEA) and the International Atomic Energy Agency (IAEA) (OECD, 2020), sufficient uranium resources are available to support the long-term, sustainable use of nuclear energy for low-carbon electricity generation, as well as for other uses such as industrial heat applications and hydrogen production. However, some recent events, such as the COVID pandemic and its impact on the industry, as well as recent declines in uranium production and exploration, could affect available supplies. On the other hand, the limiting factor in REEs production is the extraction process itself. Despite their name, REEs are relatively abundant (Suli et al., 2017; Balaram, 2019). However, the process of extraction and conversion into usable materials is expensive and environmentally harmful. Currently, REE resources are obtained from various geological settings, including carbonatites, ion-absorbing clay deposits, and monazite xenotime placer deposits (OECD, 2020; PMF IS, 2020), with China being the leading producer. The main production of REEs in China comes from carbonatite deposits, where the main mineral is bastnaesite. Outside China, minerals such as monazite, xenotime, and apatite are the main sources. It should be emphasized that monazite is also the most abundant thorium mineral, while thorium in uranium deposits is concentrated in thorite and thorianite. In addition, high Th contents have been found in carbonatites where it is associated with REEs in bastnaesite.

While primary U, Th, and REE resources currently account for the majority of industrial production and use, secondary resources are becoming increasingly important from an environmental and resource conservation perspective (Amaral et al., 2018). Since these elements often occur in the same mineral assemblages and source rocks (Rhodes, 2011; Jonathan, 2012; Goodenough et al., 2016; Ramos et al., 2016; Schulz et al., 2017), e.g., uraniferous phosphates processed by the fertilizer industry or monazite processed to extract REEs (Barthel and Tulsidas, 2014; Suli et al., 2017), the residues of the above processes are the most obvious target. Although some of the tailings have been extensively explored for the above purposes, they have generally been quickly abandoned for economic reasons. To date, such unconventional resources account for just over 11% of historical uranium production (OECD 2020), and even less for REEs and thorium.

A major advantage of secondary resources is that the main costs of mining and processing are incurred in the extraction of primary resources, and only the additional costs of processing them need to be considered. A second major advantage in terms of U and Th comes from radiation protection requirements. Primary sources are associated with higher levels of radioactive daughter products due to their higher uranium and thorium content (IAEA 2018), resulting in stringent radiological safety standards for mining and processing personnel, industry, and the environment. Secondary raw materials, on the other hand, require few additional safety measures for source materials.

As demand for U, Th, and REEs increases, the amount of tailings, byproducts, and wastes generated in the production and use of these elements continues to grow, inevitably leading to increasing environmental impacts. Due to industrial emissions, leaching of minerals, soils, and industrial wastes (e.g., coal ash, red mud, cement, and metal wastes, e-waste, used catalysts, etc.), the natural concentrations of these elements are increasing in various environmental compartments (e.g., Neves et al., 2009; Erickson, 2018; Arome et al., 2019; Wang et al., 2019b; Li Z et al., 2019; Piarulli et al., 2021; Pereirada et al., 2022). This increase highlights the need to understand not only their sources, but also their fate, mobility, and toxicity in the environment, especially in contaminated areas. Further research is also needed to understand the environmental impacts of industry from a life-cycle perspective in the secondary resource sector.

The objective of this review is to discuss the occurrence of U, Th, and REE in the environment, the composition of their resource materials, the potential of industrial wastes and end-use materials for recovery purposes, and the major environmental issues (contamination, sources, fate, mobility, and toxicity) associated with their increasing anthropogenic emissions.

2 Results and discussion

2.1 Literature search and selection criteria

Regarding the occurrence, use and distribution of REEs, U and Th, as well as the extraction, separation and purification techniques, a literature search on these elements was performed in four major academic databases, namely Web of Science, ScienceDirect, PubMed, and Google Scholar. After screening the results of the literature search considering the mobility and fate of these elements in different environmental compartments, 197 references were selected, including research articles (151), reviews (13), monographs, reports or books (27), and conference proceedings (6) from 1960 to 2022. In general, the publication of research articles on these critical elements has greatly increased since 2015 due to their wide use and industrial application.

Of the 197 references, 56% addressed the distribution of these elements in primary and secondary resource materials (P/SRM), air, water, soil, sediments, and biota. The remaining 44% of

TABLE 1 Contents of U, Th and REEs in some representative rocks.

Country	Rock	Element	Concentration	References
India	Granite	U	36.7 mg kg^{-1}	Asokan et al. (2020)
		Th	74.6 mg kg^{-1}	
		REEs	3385 mg kg^{-1}	
	Granite	U	0.2%–2.7%	Gupta et al. (2008)
Spain	Granitic rock Slate	U	30 mg kg^{-1}	Santos–Francés et al. (2018)
			71.2 mg kg^{-1}	
Greece	Phosphatized limestones	U	3–647 mg kg^{-1}	Tzifas et al. (2014)
United States	Phosphate deposits	REEs	900–1800 mg kg^{-1}	Emsbo et al. (2015)
Saudi Arabia	Phosphate ores	U	46–85 mg kg^{-1}	Al–Eshaikh et al. (2016)
	Bauxite	U	11.4 mg kg^{-1}	Adams and Richardson (1960)
		Th	48.9 mg kg^{-1}	
Greece	Bauxite	REEs	369–471 mg kg^{-1}	Deady et al. (2016)
Italy			498 mg kg^{-1}	
Turkey			614–675 mg kg^{-1}	
China (Bayan Obo)	Carbonatites	REEs	6% RE_2O_3	Fan et al. (2016)
United States (Mountain Pass)	Carbonatites	REEs	8.9% RE_2O_3	Castor (2008)

citations addressed their use, sources, fate, mobility, recycling, and toxicity.

Most studies (80%) were conducted in Asia and Europe, and only 20% involved other continents.

2.2 Occurrence of U, Th, and REEs

2.2.1 Uranium and thorium

Uranium gained importance with the development of the practical use of nuclear energy. In addition to its use in electricity generation, nuclear weapons, and fission reactors, it also found application in photography, lamps, X-ray equipment, etc. It is mainly extracted from minerals such as uraninite, pitchblende, carnotite, and brannerite (Lauf, 2016), and its occurrence is related to various geological conditions, including intrusive rocks, volcanic rocks, and polymetallic iron oxide breccia complexes. Economically mineable U mineral resources currently exist in many countries (PMF IS 2020), a large amount of uranium is also contained in seawater, given the relatively high U concentration of about 3 ng ml^{-1} (Dungan et al., 2017).

In geological matrices, uranium is found in a wide range of concentrations, e.g., from below 1–23 g kg^{-1} in igneous rocks, 2–5 mg kg^{-1} in metamorphic rocks, from 3 to 647 mg kg^{-1} in phosphatized limestones, from 46 to 85 mg kg^{-1} in phosphate ores, and around 71 and 11 mg kg^{-1} in slate and bauxites, respectively (Adams and Richardson, 1960; Godinez et al., 1997; Gupta et al., 2008; Tzifas et al., 2014; Wu et al., 2014; Al-Eshaikh et al., 2016; Santos-Francés et al., 2018; Asokan et al., 2020) (Table 1).

Thorium abundance is three times that of uranium in the Earth's crust (Adams and Richardson, 1960; Korna et al., 2014; Asokan et al., 2020), it is usually mined from monazite, a phosphate mineral, although it also occurs in minerals such as thorite or thorianite (Barthel and Tulsidas 2014). Thorium concentrations in the lithosphere range from 8 to 12 μg g^{-1}. It is a weakly radioactive metal, similar to uranium, and occurs naturally in various minerals, soils, and waters (Lauf, 2016). Thorium is not only critical for power generation, but also has applications in the production of alloys, arc welding, heat-resistant ceramics, ceramics, glass, etc.

2.2.2 Rare earth elements

The REEs are a group of 17 chemical elements in the periodic table, namely the 15 lanthanides (La, Ce, Pr, Nd, Pm, Sm, Eu, Gd, Tb, Dy, Ho, Er, Tm, Yb, and Lu) and the lighter scandium and yttrium. Scandium and yttrium are included among the REEs because they generally occur in the same ore deposits as the lanthanides and have similar physical and chemical properties. They represent a coherent group of elements typically found together in minerals, some of which are technologically critical elements: La, Ce, Gd, Nd, Pr, and Dy (Malhotra et al., 2020).

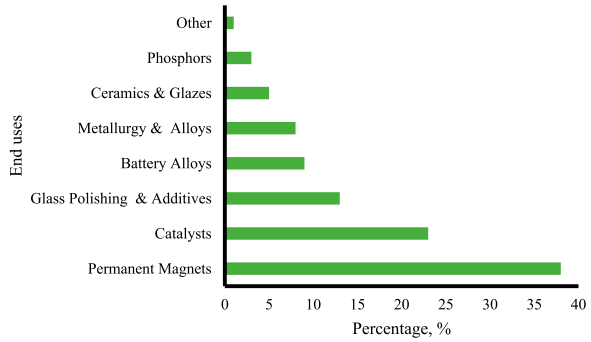

FIGURE 1
Major uses of REEs.

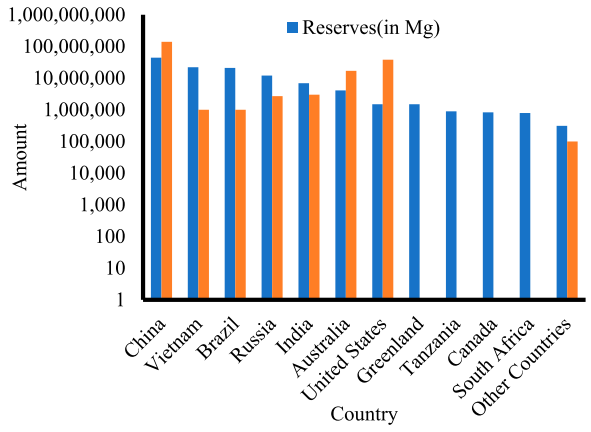

FIGURE 2
Current production and predicted reserves of REEs per country plotted based on the data by (LePan, 2021).

Precisely because of their exceptional physicochemical properties, REEs have found widespread application in industrial goods, which has led to a steady increase in global demand for these elements (Balaram, 2019). REEs are essential components of many technological devices, including electric, and hybrid vehicles, flat-screen displays, cell phones, and computer hard drives (Lucas et al., 2014), but they also have many other applications (Figure 1; LePan 2021).

For example, the pyrophoric alloy of Ce and Fe is used as "flint" in gas and cigarette lighters (Cotton, 1991); the mixed lanthanide alloys of Ce, La, Nd, and Pr are widely used in metallurgy (Sastri et al., 2003); many lanthanides are used as catalysts for petroleum refining (Kilbourn, 1986; Balaram, 2019) or in nuclear technology to absorb neutrons; Eu and Yb oxides are widely used in the color television industry; La_2O_3 is used to color ceramics and glasses; Ce_2O_3 is used to polish glass; Nd, Pr, and Dy have unusual magnetic properties; and other lanthanide compounds are used in lighting technology (Zhang et al., 2020). In addition, lanthanides have been used in agriculture for several decades, especially in China, as fertilizers to increase yields (Pang et al., 2002).

Despite their name, REEs are indeed relatively abundant in the Earth's crust, with a mean total concentration (∑REEs) in the Earth's crust of up to 240 mg kg^{-1} (Balaram, 2019), but are rarely found in concentrations that are economically mineable (Suli et al., 2017). Current resources are associated with several primary geologic settings: ion-absorption clay deposits, igneous systems including carbonatites which are the main source, and monazite-xenotime placer deposits. Ore production of REEs per country and projected mineral reserves are shown in Figure 2 (LePan, 2021).

As shown in Table 1, bauxite, bituminous coal, and lignite contain REEs in the range of 250–675 mg kg^{-1} (Mayfield and Lewis 2013; Deady et al., 2016; Hower et al., 2016; Montross et al., 2020). Much higher REEs concentrations (172–3385 mg kg^{-1}) have been found in granite, and phosphorus deposits (Hua et al., 2010; Emsbo et al., 2015; Asokan et al., 2020), while extremely high concentrations (17%–20%) have been recorded for carbonatites (Castor, 2008; RMREI, 2012; Fan et al., 2016).

2.3 Secondary resources of U, Th and REEs

Table 2 summarizes the concentrations of U, Th, and REEs found in some industrial wastes. As shown in Table 2, all these elements can be found in elevated concentrations in red mud, coal ash, and magnetic scrap. During the Bayer process, most REEs pass almost completely into the bauxite residue. Depending on the initial REE concentration in the bauxite ore, the bauxite residue may have a total REE concentration of up to 2500 mg kg^{-1}, as in the case of Jamaican red mud (Wagh and Pinnock, 1987). In addition to REEs, such residues may also be enriched in uranium and thorium. Gu et al. (2017) reported U, Th, and REEs ranging from 20 to 31, 72 to 159, and 468–1237 mg kg^{-1}, respectively, in Chinese red mud, while the resulting red mud after alumina extraction from Greek bauxites (497–674 mg kg^{-1} REEs) had REEs up to 1777 mg kg^{-1} (Binnemans et al., 2013; Deady et al., 2016). Coal combustion processes also lead to enrichment of REEs in ash by a factor of six to ten compared to coal (e.g., Seredin and Dai, 2012; Franus et al., 2015; Fiket et al., 2018a; Okeme et al., 2020). For example, in southwestern China (in Guizhou), bottom and fly ashes from five coal-fired power plants were found to be enriched in REEs and yttrium, with average concentrations of 658 ± 296 mg kg^{-1} and some elements reaching levels as high as 1257 mg kg^{-1} (Li C et al., 2019). The contents of REEs and critical REEs (Y, Nd, Eu, Tb, Dy, and Er) in various coal ashes reported by Kolker et al. (2017) were even higher, ranging from 192 to 1668 mg kg^{-1} and from 28.3% to 44.5%, respectively.

TABLE 2 Contents of U, Th and REEs in industrial wastes.

Country	Materials	Element	Concentration	References
Turkey	Bottom ash	U	70–94 Bq·kg^{-1}	Aytekin and Baldık (2012)
	Fly ash		64–85 Bq·kg^{-1}	
	Coal		32–55 Bq·kg^{-1}	
Nigeria	Fly ash	REEs	442–625 mg kg^{-1}	Okeme et al. (2020)
Poland	Fly ash	REEs	101–543 mg kg^{-1}	Franus et al. (2015)
United States	Coal ash	REEs	192–1668 mg kg^{-1}	Kolker et al. (2017)
Germany	NdFeB–based magnet scrap	Nd + Pr + Dy	1.2%–5.9%	Voßenkaul et al. (2013)
China	NdFeB, low REE magnet scraps	Nd + Pr + Dy	13.8–14.6%	Zhang et al. (2020)
	NdFeB, medium REE magnet scraps	Nd + Pr + Dy + Gd	24.4–29.5%	
	NdFeB, high REE magnet scraps	Nd + Pr + Dy + Sm	36.7%	
		Nd + Pr + Dy + La		
		Nd + Pr + Dy + Gd		
		Nd + Pr + Dy		
	Red mud	U	20–31 mg kg^{-1}	Gu et al. (2017)
		Th	72–159 mg kg^{-1}	
		REEs	468–1237 mg kg^{-1}	
Belgium	Red mud	REEs	1040 mg kg^{-1}	Binnemans et al. (2013)
Greece Turkey	Red mud	REEs	727–738 mg kg^{-1}	Deady et al. (2016)
			133–1777 mg kg^{-1}	
Italy	Mine tailing	REEs	300 mg kg^{-1}	Medas et al. (2013)
Egypt	Phosphogypsum	REEs, Y	481 mg kg^{-1}	Gasser et al. (2019)
Finland	Phosphogypsum	REEs	17 g kg^{-1}	Virolainen et al. (2019)
Brazil	Phosphogypsum	REEs	26 g kg^{-1}	Brückner et al. (2020)

Recently, Costis et al. (2021) presented a comprehensive review of the potential recovery of REE from abundant secondary sources such as mine drainage, phosphogypsum residues, and U mine waste. Such mining and industrial wastes contain a wide range of REEs, with concentrations reaching extremely high levels in certain locations. Moraes et al. (2017) reported values as high as 130 mg L^{-1} in the drainage of the Osamu-Utsumi mine in Brazil, while phosphogypsum extracted from sedimentary and especially igneous phosphate rock reaches even higher values. For example, 481 mg kg^{-1} of REE was measured in phosphogypsum derived from sedimentary rocks from Egypt (Gasser et al., 2019), while up to 26 g kg^{-1} REE was found for phosphogypsum derived from igneous rocks from Brazil (Brückner et al., 2020).

Nowadays, only some countries rely on recycling U, Th, and REEs from coal ash, red mud, pre-consumer scrap, industrial residues, mine waste, metallurgical slags, and e-waste (Kumar et al., 2014; Binnemans et al., 2015; IAEA, 2018). Among these secondary resources, some authors also include the so-called "unconventional sources" that have been exploited for some time, especially as uranium sources (e.g., uranium-bearing phosphate and black shale deposits). There are estimates that such unconventional uranium resources may contain up to 7.6 million tons of U (OECD, 2020). Although unconventional resources are currently uneconomic, recovery of uranium from these deposits could become profitable through improved extraction techniques or upgrading of byproducts.

Chemical and biological leaching methods, followed by solvent extraction or alkali smelting, are used to achieve substantial recovery of these metals from secondary sources (Maslov et al., 2010; Santos and Ladeira, 2011; Amaral et al., 2018; Dinal et al., 2019; Jyothi et al., 2020; Ahmed et al., 2021). The methods used for REE recycling of used electronic devices

also include crushing and grinding the devices into powder form, from which the essential components are extracted by further separating the elements using pyrometallurgical methods (Jowitt et al., 2018).

Some of the conventional methods involve the use of various acid mixtures, e.g., uranium (54.3 mg kg^{-1}) and REEs are often leached from coal ash with a mixture of HNO_3 and HF, using an anion exchanger to extract uranium from solution, followed by further purification (Maslov et al., 2010; Mondal et al., 2019). On the other hand, effective extraction and recovery of REEs in contaminated soil and other waste materials has been achieved by using reusable biosurfactants (Gao et al., 2012; Li C et al., 2019). Chour et al. (2018) developed a method to recover REEs from the hyperaccumulator plant *Dicranopteris dichotoma* by an improved ion-exchange leaching process, which resulted in a REE purity of 81.4% and a recovery of 78%. Santos and Ladeira (2011) extracted uranium (0.25%) from waste materials composed of Ca, S, Mn, and Al by using Na_2CO_3 and $NaHCO_3$ as sequestering agents, while other elements were crystallized as ettringite, gypsum, calcite, and bassanite. Recently, the macromolecule lanmodulin (binding protein) was proposed for the selective extraction of REEs from low-grade sources (Deblonde et al., 2020).

However, in order to optimize extraction processes, achieve much higher efficiency, and make them robust enough for different waste types, further efforts are needed, and not only at laboratory scale. This is especially true for the efficiency of extraction from technical waste. Most scrap magnets are based on REEs-Fe-B and generally consist of more than 30% neodymium, praseodymium and dysprosium. Since direct melting is not considered beneficial (Voßenkaul et al., 2013), a direct leaching step offers the possibility to recover REEs very easily and with lower energy consumption. Subsequent separation is also much easier compared to primary extraction, as only four of the seventeen REEs are in solution.

2.4 Environmental impact of U, Th, and REEs

Intensive REE mining and production have resulted in significant environmental and health impacts. Mining and extraction releases REE-containing dust, other toxic metals, and chemicals into the air and surrounding waters, which can affect not only humans but also soil, wildlife, and plants. In addition, landfills exposed to weathering have the potential to further pollute air, soil, and water, resulting in further environmental degradation and human health hazards (Wang and Liang, 2015; Balaram, 2019). Because REE minerals often contain significant amounts of uranium and thorium, mining residues and their weathering can contaminate air, water, soil, and groundwater with radioactive elements. In addition, the extensive use of REEs in modern technologies, as well as the burning of biomass and coal, fertilizers, livestock feed, medical facilities, electronic waste, recycling facilities, and petroleum refining, further increases their concentration in the environment (Ganguli and Cook, 2018; Gwenzi et al., 2018; Shao et al., 2020). In the following text, the levels of U, Th, and REE in different environmental compartments (Tables 3, 4, 5) are discussed to provide an overview of their levels in natural and contaminated environments.

2.4.1 Air

As mentioned earlier, uranium, thorium, and REEs can be released and mobilized from primary and secondary deposits and tailings piles through natural weathering and erosion, as well as through processes of coal combustion and biomass burning, and can be further transported to various environmental compartments (Erickson, 2018; Li C et al., 2019; Piarulli et al., 2021). Consequently, the processes of extraction, use, and disposal of U-, Th-, and REE-containing materials lead to an increase in their concentrations in the ambient air as aerosols or particulates above the permissible or prescribed levels.

Background concentrations of uranium in ambient air vary widely from 0.02 to 0.40 ng m^{-3}, with an average value of about 0.1 ng m^{-3} or 1 μBq m^{-3} in radioactivity units (Bem and Bou-Rabee, 2004). In Canada, the recommended limit for U to protect human health is 30 ng m^{-3} (AAQCs 2012), while in India, the prescribed annual dose limit for Th (weighted sum of Ra, Th, and K activities) is 30 mSv for the population (Parthasarathy, 2018). The average maximum allowable concentrations (MAC) of REEs fluoride compounds (leading to pneumoconiosis) are 0.5 mg m^{-3} (Rim et al., 2013).

Concentrations exceeding the recommended limits set by AAQCs (2012) are often found near mining and industrial areas (Tracy and Meyerhof, 1967; Shaltout et al., 2013; Wang et al., 2014; Dai et al., 2016; Wang et al., 2016; Hussein et al., 2018; Fesenko and Emlutina 2021; Kolawole et al., 2021), Table 3. For example, U concentrations of 2–200 ng m^{-3} have been measured within 2 km of a refinery (Tracy and Meyerhof, 1967), which exceeds the recommended limits by a factor of 6. In addition, thorium and REE concentrations of up to 1×10^{-4} Bq·m^{-3} and 173–298 ng m^{-3}, respectively, have been detected in atmospheric particles from Chinese cities (Wang et al., 2014; Wang et al., 2016; Fesenko and Emlutina 2021). The average total concentration of REEs in total PM, PM_{10} (coarse dust), and $PM_{2.5}$ (fine dust) measured in the Yangtze River Delta region of China was 12.0, 9.4, and 2.2 ng m^{-3}, respectively (Dai et al., 2016), with particle size distribution indicating significant fractionation in the coarse fraction (Dai et al., 2016; Hussein et al., 2018). REE concentrations in collected dust samples from Greater Cairo (Egypt) ranged from 1 to 60 μg g^{-1}, with the highest concentrations found in dust samples outside the city, indicating a possible natural source of REE (Shaltout et al., 2013). Even higher REE levels were found in atmospheric dust from the city of Ibadan (Nigeria), ranging from 42.5 μg g^{-1}

TABLE 3 Contents of U, Th and REEs in ambient air.

Country	Source	Element	Concentration	References
China	TSP	REEs	149.6–239.6 ng m^{-3}	Wang et al. (2014)
	PM_{10}		42.8–68.9 ng m^{-3}	
	TSP	Th	820–39720 μBq·m^{-3}	Wang et al. (2016)
	coal–fired power plant	REEs	147.2–468.6 mg kg^{-1}	Li C et al. (2019)
Global	Air	Th	0.1 to 1 × 10^{-4} Bq·m^{-3}	Fesenko and Emlutina (2021)
Nigeria	Industrial/traffic dust	REEs		
		Ce, Dy, Er, Eu, Gd, La, Nd, Pr, Sm, Ho, Lu, Yb	331, 2.12, 0.89 0.86, 4.75, 251, 8.19, 4.94, 0.50, 0.36, 0.14, 0.89 mg kg^{-1}	Kolawole et al. (2021)

TABLE 4 Contents of U, Th and REEs in water.

Location	Water	Element	Concentration	References
Bathinda and Mansa, Faridkot and Muktsar, Mansa, Bathinda and Faridkot and Southwest Punjab, India	GW	U	2.3–645 μg L^{-1}	CGWB (2020), Pant et al. (2017), Rishi et al. (2017), Sharma et al. (2017), Saini and Bajwa (2016)
Kolar district, Karnataka, India	GW	U	0.3–1443 μg L^{-1}	Babu et al. (2008)
Central Tamil Nadu, India	GW	U	1.9–16.8 μg L^{-1}	Thivya et al. (2014)
Nalgonda, Andhra Pradesh, India	GW	U	0.2–521 μg L^{-1}	Raghavendra et al. (2014)
Nalgonda, Andhra Pradesh, India	BW, DW, HP	U	3–370 μg L^{-1}	Keesari et al. (2014), Brindha and Elango (2013)
Karnataka and Goa, Tamil Nadu, Uttar Pradesh, Madhya Pradesh and Chhattisgarh, India	GW	U	30–302 μg L^{-1}	CGWB (2020), Coyte et al. (2018), Sahu et al. (2020)
Haryana, India	GW	U	14.4–57.43 μg L^{-1}	Tanwer et al. (2022)
Datong basin, northern China	GW, SW	U	0.02–288 μg L^{-1}	Wu et al. (2014)
South Korea	SW	U	0.6–5.6 μg L^{-1}	Cordeiro et al. (2016)
Singida Urban District, Tanzania	SW, GW	U	0.09–1.10 mg L^{-1}	Kaishwa et al. (2018)
Saudi Arabia	Water	U	140–1140 μg L^{-1}	Aleissa et al. (2004)
United Arab Emirates	GW	^{235}U	0.1–508 ng L^{-1}	Ahmed et al. (2014)
		^{238}U	69.2 mg L^{-1}	
		^{232}Th	0.2–2529 ng L^{-1}	
China	GW, WW	REEs	0.0820–118 μg L^{-1}	Tang et al. (2020), Tian et al. (2020)
Sardinia, Italy	Water	REEs	1.4 μg L^{-1}	Medas et al. (2013)
Southwest England, United Kingdom	GW	REEs	Up to 229 μg L^{-1}	Smedley, (1991)
Poland	Mine drainage	REEs	Up to 24.8 mg L^{-1}	Migaszewski et al. (2019)
New Hampshire, United States	GW	U	<1–270 μg L^{-1}	Flanagan et al. (2014)

GW, groundwater; SW, surface water; PW, pond water; HP, hand pump.

(residential) to 785 μg g^{-1} (industrial), with mean ΣREE levels in industrial, transportation, landfill, residential, remote areas, and local soil of 638, 283, 130, 163, 96.0, and 144 μg g^{-1}. The average ΣREE content in dust in industrial and transportation areas was about 4.5 and 2.0 times higher than the average ΣREE content in soil, respectively (Kolawole et al., 2021). The main sources of

TABLE 5 Distribution of U, Th and REEs in soil and sediment.

Location	Sample	Element	Concentration	Reference
China	Soil	U	4–207 mg·kg^{-1}	Wang et al. (2019a)
Singida Urban District, Tanzania	Soil	U	20–31.6 mg·kg^{-1}	Kaishwa et al. (2018)
Nigeria	Soil	U	44.3 Bq·kg^{-1} (2.6 mg·kg^{-1})	Arome et al. (2019)
Buhovo Region, Bulgaria	Soil	U	45–65 mg·kg^{-1}	Mihaylova et al. (2013)
		Th	75 mg·kg^{-1}	
Spain	Soil	U	207–542 mg·kg^{-1}	Santos–Francés et al. (2018)
Portugal	Soil	U	10–362 mg·kg^{-1}	Neves et al. (2009)
	Sediments	U	10–42 μg·g^{-1}	Bergmann and Graça (2020)
USA	Soil/sediment	U	630 mg kg^{-1}	Caldwell et al. (2012)
Egypt	Sediment	Th	2–34 Bq·kg^{-1}	El-Taher et al. (2018)
	Red Sea sediment	U	2.2 mg·kg^{-1}	El-Taher et al. (2019)
		Th	2.2 mg·kg^{-1}	
		ΣREE	47.6 mg·kg^{-1}	
China	Surface soil	REEs	156 mg·kg^{-1}	Wang and Liang, (2015)
			–56 g·kg^{-1}	
	Soil	REEs	181 mg·kg^{-1}	Han et al. (2017)
China	Soil	REEs	146.52–158.76 mg·kg^{-1}	Wang et al. (2022)
Serbia	Surface soil	REEs	162–319 mg·kg^{-1}	Fiket et al. (2016)
Sweden	Surface soil	REEs	25.3 mg·kg^{-1}	Sadeghi et al. (2013)
	Subsoil		53.6 mg·kg^{-1}	
	Sediment		23.9 mg·kg^{-1}	
Europe and USA	Surface soil	REEs	34–346 mg·kg^{-1}	Ramos et al. (2016)
Brazil	Mangrive soil	REEs	161–183 mg·kg^{-1}	Freitas Rodrigo et al. (2021)
South Korea	Sediment	U	124–23910 μg·kg^{-1}	Cordeiro et al. (2016)
Thailand	Sediment	U	1.9 mg·kg^{-1}	Kritsananuwat et al. (2014)
		Th	12 mg·kg^{-1}	
		REEs	102 mg·kg^{-1}	
	Marine sediment	REEs	9.4–206 mg·kg^{-1}	
Southeast Asia	Western Sunda Shelf sediment	U	7–193 mg·kg^{-1}	Wu et al. (2020)
Malaysia	Sediment	U	19–126 mg·kg^{-1}	Antonina et al. (2013)
China	Sediment	REEs	45–316 mg·kg^{-1}	Zhu et al. (1997)
Gulf of Thailand	Marine sediment	REEs	9–206 μg·g^{-1}	Kritsananuwat et al. (2014)
Japan	North Pacific Ocean sediment	REEs	2–4 g·kg^{-1}	Tanaka et al. (2020)
India	Tropical estuary sediment	ΣREEs	62–230 mg·kg^{-1}	Deepulal et al. (2012)
India	Lake sediment	REEs	0.4–49 mg·kg^{-1}	Sivasamandy et al. (2019)
Ambagarh Chouki, CG, India	Sediment	U	6.1–25 mg·kg^{-1}	Sahu et al. (2016)
		Th	0.5–1.9 mg·kg^{-1}	
		REEs	176 mg·kg^{-1}	

(Continued on following page)

TABLE 5 (*Continued*) Distribution of U, Th and REEs in soil and sediment.

Location	Sample	Element	Concentration	Reference
Croatia	Lake sediment	REEs	56–85 mg·kg^{-1}	Fiket et al. (2016)
Portugal	Tagus estuary	ΣREEs	18–210 mg·kg^{-1}	Brito et al. (2018)
Italy	western Mediterranean Sea sediment	ΣREEs	163 mg·kg^{-1}	Tranchida et al. (2011)
Spain	Galician Rias	ΣREEs	3–233 mg·kg^{-1}	Borrego et al. (2005), Prego et al. (2009)
Australia	North Australian river-estuary systems	ΣREEs	77–263 mg·kg^{-1}	Munksgaard et al. (2003)
Mexico	Marabasco river-estuary system	ΣREEs	27–157 mg·kg^{-1}	Marmolejo-Rodríguez et al. (2007)
South Korea	Northeastern Pacific	ΣREEs	700 mg·kg^{-1}	Kim et al. (2022)
Cameroon	River sediment	ΣREEs	282–727 mg·kg^{-1}	Ndjama et al. (2022)

airborne thorium are believed to be the sites of processing and extraction of thorium, uranium, and radium from ores and concentrates, but data on the fate and transport of airborne thorium are rather limited.

2.4.2 Water

Major anthropogenic sources of U, Th, and REE include wastes and effluents from medical facilities, mines, industrial mineral processing plants, nuclear power plants, the electronics industry, oil refineries, fertilizer and feed mills, etc. (e.g., Harmsen and Haan, 1980; Brindha and Elango, 2013; Keesari et al., 2014; Coyte et al., 2018; Gwenzi al, 2018; CGWB, 2020; etc.). Although the concentrations of U, Th, and REEs in natural waters are relatively low, about 3.3, 0.006, and 0.004–0.024 μg L^{-1}, respectively (Deng et al., 2017; Degueldre and Joyce, 2020), numerous processes, such as mining, extraction, weathering, and disposal of these elements, can affect their natural levels (Table 4).

Maximum permissible levels (MPLs) for U and Th in water are set at 30 μg L^{-1} (0.75 ng m^{-3} or 20.3 pCi L^{-1}) and 15 pCi L^{-1} (0.555 Bq·L^{-1}), respectively (Sneller et al., 2000; ATSDR, 2009). MPLs for fresh and (marine) water for Eu, La, Ce, Pr, Nd. Sm, Gd, and Dy are 62 (0.72), 10 (1.0), 22 (0.15), 9 (0.92), 1.4 (0.85), 7.6 (0.42), 6.8 (0.52), and 9.1 (0.16) μg L^{-1}, respectively (Sneller et al., 2000).

However, uranium concentrations in surface and groundwater exceeding 6 μg L^{-1} have been reported in several countries (notably China, India, UAE, Saudi Arabia, Korea, Nigeria, and Tanzania), with concentrations in the waters of many countries exceeding the ATSDR (2009) permissible limit of 30 μg L^{-1} (Aleissa et al., 2004; Ahmed et al., 2014; Flanagan et al., 2014; Wu et al., 2014; Cordeiro et al., 2016; Shin et al., 2016; Pant et al., 2017; Kaishwa et al., 2018; Bergmann and Graca, 2020; CGWB, 2020). In India, for example, U concentrations in groundwater vary from 0.6 μg L^{-1}–1443 μg L^{-1} and often exceed the above limit of 30 μg L^{-1}, especially in areas of uranium mineralization that have a natural anomaly of U (Babu et al., 2008; Brindha and Elango, 2013; Keesari et al., 2014; Raghavendra et al., 2014; Thivya et al., 2014; Saini and Bajwa, 2016; Pant et al., 2017; Rishi et al., 2017; Sharma et al., 2017; Coyte et al., 2018; CGWB, 2020; Sahu et al., 2020; Tanwer, et al., 2022) or near industries such as cement factories, fertilizer plants, chemical factories, and coal-fired thermal power plants (Saini and Bajwa, 2016; CGWB, 2020).

In fact, U concentrations in water were found to be many times above the permissible limit in 14 Indian states: Panjab, Haryana, Telangana, Delhi, Rajasthan, Andhra Pradesh, Uttar Pradesh, Karnataka, Madhya Pradesh, Tamilnadu, Jharkhand, Chhattisgarh, Gujrat, Himachal Pradesh, Maharashtra, Odisha, West Bengal, and Bihar (CGWB, 2020). Even unexploited uranium ores have been reported to contribute to uranium levels in local waters (Cordeiro et al., 2016). For example, uranium contamination of groundwater by naturally occurring uranium was observed in the Horta da Vilariça region of Portugal, with U concentrations in groundwater as high as 3.5 mg L^{-1} (Cordeiro et al., 2016).

Ahmed et al. (2014) reported thorium concentrations in groundwater ranging from 0.2–2.5 μg L^{-1}. As for REEs, concentrations in surface water, groundwater, and wastewater were found to range from 1.4 to 2.9, 0.1 to 229, and 4.5–118 g L^{-1}, respectively, in different countries (Smedley, 1991; Li et al., 2013; Medas et al., 2013; Tang et al., 2020; Tian et al., 2020). Extremely high REE water contents are usually associated with mining sites or associated mine drainages, as in the Wiśniówka waters in Poland, where 8.5 and 17.9 mg L^{-1} total REE associated with As-rich pyrite mineralization were detected in ponds and drainage ditches of the Wiśniówka mining area (Migaszewski et al., 2019). Moreover, the increasing use of REEs, especially Gd, for medicinal purposes in recent decades has led to an increase in their concentrations not only in wastewater but also in streams (e.g., Bau and Dulski, 1996; Bau et al., 2006; Kulaksız and Bau, 2013; Klaver et al., 2014), up to two orders of magnitude higher than natural concentrations.

Regardless of their origin, aquatic environments play a dual role with respect to these elements: either as temporary or long-

term storage (Gwenzi et al., 2018; Gwenzi et al., 2021). Because REEs readily bind to colloidal particles and suspended solids or are complexed at the surface with inorganic and/or organic ligands (Gwenzi et al., 2018), they are eventually removed from the water column into the sediment. Their co-precipitation with solids and removal in sediments is generally dominated by Fe and Mn oxyhydroxides, which are characterized by a large surface area and high adsorption capacity, while their return to solution depends on changing environmental conditions. Whether they are in sediment, sediment pore water, or water itself, elevated concentrations of U, Th, and REEs can cause a wide range of toxicities in aquatic organisms (ATSDR, 2009; Briner 2010; Tai et al., 2010; Pagano et al., 2015; Gwenzi et al., 2018; Malhotra et al., 2020).

2.4.3 Soil and sediments

Because the elements U, Th, and REE are ubiquitous in the Earth's crust (Kasar et al., 2019), their main sources in surface soils and sediments are predominantly geogenic; however, inputs from industrial and mining processes and atmospheric deposition should not be ignored when interpreting their distribution in the environment (Duplay et al., 2014; Kritsananuwat et al., 2014; Gu et al., 2020). While the mobility of U in the ecosystem depends on redox conditions (IRNS, 2012) and complexation by organic matter (Bone et al., 2020), with the most stable and mobile form being U(VI), the mobility of Th in soil is very limited due to strong sorption to soil particles (Torstenfelt, 1986). In contrast, the content of REEs in soil is related to redox potential and pH as well as to the presence of clay, carbonate, organic matter, and Al, Fe, and Mn oxides and oxyhydroxides (Aide and Aide, 2012; Felipe-Sotelo et al., 2017; Mihajlovic and Rinklebe, 2018; Aide, 2019; Tang et al., 2020; Ogawa et al., 2021). Their content generally decreases with the decrease of pH, redox potentials, and Al-Mn-Fe contents (Aide and Aide, 2012; Mihajlovic and Rinklebe, 2018; Aide, 2019). At the same time, their residence time in sediments increases under less oxidative and alkaline conditions in an aquatic reservoir (Felipe-Sotelo et al., 2017; Ogawa et al., 2021).

The average concentrations of U and Th in soils reported by Harmsen and Hann (1980) are 1–2 and 6 mg kg^{-1}, respectively, while Sneller et al. (2000) reported MPLs at least 10-fold higher for Y, La, Ce, Pr, Nd, Sm, Gd, and Dy in freshwater sediments than in marine sediments: 1.4(0.18), 4.7(0.51), 18.8(0.22), 5.8(0.61), 7.5(0.48), 2.5(0.15), 1.8(0.14), and 2.2 (0.89) g·kg^{-1} (dry weight).

However, uranium, thorium, and REE levels of 630, 75, and 56500 mg kg^{-1}, respectively, have been detected in various surface soils, with the highest levels measured in some areas of the United States of America, Bulgaria, and China (Neves et al., 2009; Caldwell et al., 2012; Mihaylova et al., 2013; Sadeghi et al., 2013; Wang and Liang, 2015; Fiket et al., 2016; Ramos et al., 2016; Han et al., 2017; Brito et al., 2018; Kaishwa et al., 2018; Santos-Francés et al., 2018; Arome et al., 2019; Wang et al., 2019a; Freitas Rodrigo et al., 2021; Wang et al., 2022; Table 5). In sediments, U, Th, and REE values of up to 126, 12, and 4000 mg kg^{-1}, respectively, were detected, with the highest values measured in some areas of Malaysia, Thailand, and Japan (Zhu et al., 1997; Munksgaard et al., 2003; Borrego et al., 2005; Marmolejo-Rodríguez et al., 2007; Prego et al., 2009; Tranchida et al., 2011; Deepulal et al., 2012; Antonina et al., 2013; Kritsananuwat et al., 2014; Cordeiro et al., 2016; Sahu et al., 2016; Fiket et al., 2018b; Brito et al., 2018; El-Taher et al., 2018; El-Taher et al., 2019; Elias et al., 2019; Nascimento et al., 2019; Sivasamandy et al., 2019; Bergmann and Graça, 2020; Tanaka et al., 2020; Wu et al., 2020; Freitas Rodrigo et al., 2021; Kim et al., 2022; Ndjama et al., 2022).

Although some sources are associated with nearby ore deposits (e.g., Neves et al., 2009; Mihaylova et al., 2013; Kaishwa et al., 2018; Wang et al., 2019a) or secondary sources of these elements, such as the remains of the phosphorus fertilizer industry (Brito et al., 2018) or coal ash deposits (Fiket et al., 2018a), elevated U, Th, or REE concentrations in soils and sediments can also be attributed to other influences, such as the impact of steel mills (Arome et al., 2019), different agricultural practices (Han et al., 2017), or mining areas for other commodities (Pereirada et al., 2022). For example, soils in the gold mining region of the Amazon were found to be contaminated with REEs, with Y being most abundant in urban, agricultural, and mining areas and enriched in soil by a factor of 18.2, 39.0, and 44.4, respectively (Pereirada et al., 2022).

2.4.4 Biota accumulation

Due to their widespread distribution in the environment, U, Th, and REE are also found in certain concentrations in biota. Current literature indicates their occurrence in macroalgae, benthic invertebrates, including bivalves, echinoderms and crustaceans, fish and plants (Hegazy and Emam, 2011; Kovaříková et al., 2019; Piarulli et al., 2021), and some species can be considered hyperaccumulators. The transfer of these elements from soil to plant is a complex process that depends on many factors: pH, redox potential, organic matter in plants, and presence of certain ions (Kasar et al., 2019; Kovaříková et al., 2019). In general, element content decreases from leaves to stems to fruits and seeds (Tyler 2004). It is related to the nutrient content of the soil (Tyler, 2004; Rodrı´guez et al., 2005; Oufni et al., 2011) and the content of clay and organic matter in the soil (Tyler, 2004; Hegazy and Emam, 2011). A selection of data on uranium and REEs in plants, including mosses, ferns, algae, macrophytes, and freshwater organisms, is presented in Table 6. However, the table does not include data for Th because such data are very limited or non-existent.

Plants and aquatic organisms have been reported to accumulate U from soils and water in the range of 0.08–3500 mg kg^{-1} (Huang et al., 1998; Dienemann et al.,

TABLE 6 Contents of U and REEs in biota.

Location	Plant	Element	Concentration	Reference
China	*Nymphaea tetragona*	U	3446 ± 155 mg·kg^{-1}	Li C et al. (2019)
Japan	*Dictyota dichotoma*	U	460–500 μg·kg^{-1}	Sakamoto et al. (2008)
	Undaria pinnatifida		490–820 μg·kg^{-1}	
South Korea	*Brassica napus* var. *napus*	U	3500 mg·kg^{-1}	Chang et al. (2005)
	Scorpiurium deflexifolium		49639 mg·kg^{-1}	
South Korea	*Fontinalis antipyretica*	U	35771 μg·kg^{-1}	Cordeiro et al. (2016)
	Rorippa sylvestris		33837 μg·kg^{-1}	
	Oenanthe crocata		17807 μg·kg^{-1}	
	Nasturtium officinale		10995 μg·kg^{-1}	
Poland	*Pleurozium schreberi*	U	80.3–95.7 μg·g^{-1}	Boryło et al. (2017)
	Dicranum scoparium		89.7–104.7 μg·kg^{-1}	
Portugal	Macrophytes	U	47.6 ± 6.9 μg·g^{-1}	Bergmann and Graça (2020)
	Scrapers		14.3 ± 1.0 μg·g^{-1}	
	Shredders		13.0 ± 0.8 μg·g^{-1}	
	Predators		7.0 ± 1.3 μg·g^{-1}	
Portugal	*Fontinalis squamosa*	U	0.68–1.62 mg·kg^{-1}	Pratas et al. (2017)
	Brachythecium rivulare			
	Platyhypnidium riparioides			
	Thamnobryum alopecurum			
Germany	Epiphytic algae	U	300 mg·kg^{-1}	Dienemann et al. (2002)
Germany	*Arabidopsis helleri* roots	U	35 mg·kg^{-1}	Viehweger and Geipel (2010)
	Arabidopsis helleri shoots		17 mg·kg^{-1}	
USA	*Impatiens capensis* root	U	1030 mg·kg^{-1}	Caldwell et al. (2012)
USA	*Impatiens capensis* root	U	1030 mg·kg^{-1}	Caldwell et al. (2012)
China	*Phytolacca americana*	REEs	1040 mg·kg^{-1}	Yuan et al. (2018)
China	Wheat	REEs	109 μg·kg^{-1}	Zhuang et al. (2017)
	Maize		42.9 μg·kg^{-1}	
	Legume		95.1 μg·kg^{-1}	
China	Tea	REEs	2.95 μg·g^{-1}	Zhang et al. (2007)
China	Potato	REEs	0.12 μg·g^{-1}	Li et al. (2012)
China	Fruits /vegetables	REEs	12.9/62.4 μg·g^{-1}	Shi et al. (2022)
Malaysia	Algae (*Padina* sp.)	REEs	8.4–62.4 μg·g^{-1}	Mashitah et al. (2012)
France	*Phytolacca icosandra*	REEs	13000 mg·kg^{-1}	Grosjean et al. (2019)

(Continued on following page)

TABLE 6 (*Continued*) Contents of U and REEs in biota.

Location	Plant	Element	Concentration	Reference
Germany	Tree	REEs	1 μg·g^{-1}	Markert (1987)
Portugal	*Fontinalis squamosa*	REEs	0.02–12.2 mg·kg^{-1}	Pratas et al. (2017)
	Brachythecium rivulare			
	Platyhypnidium riparioides			
	Thamnobryum alopecurum			
Sweden	Moss	REEs	1.489 μg·g^{-1}	Tyler (2004)
Poland	Wild mushrooms	REEs	200 μg·g^{-1}	Mędyk and Falandysz (2022)
Japan	*Dictyota dichotoma*	La, Ce, Pr, Nd, Sm, Eu, Gd, Tb, Dy, Ho, Er, Tm, Yb, Lu, and U	35–38, 74–79, 12-13, 68–70, 28–29, 13–14, 55–62, 9.7–10, 77–80, 16–17, 50–53, 7.2–7.8, 48–53, 7–9 and 460–500 ng·g^{-1}	Sakamoto et al. (2008)
	Eckloniastolonifera	La, Ce, Pr, Nd, Sm, Eu, Gd, Tb, Dy, Ho, Er, Tm, Yb, Lu, and U	20, 92, 5.4, 27, 16, 64, 31, 23, 28, 24, 28, 25, 30, 28 and 270 ng·g^{-1}	
	Sargassum hemiphyllum	La, Ce, Pr, Nd, Sm, Eu,Gd, Tb, Dy, Ho, Er, Tm, Yb, Lu, and U	33–53, 71–100, 4.9–9.7, 40–51, 10–13, 2.7–3.9, 10–15, 1.4–2.4, 11–16, 1.9–3.2, 7.8–9.2, 0.59–0.91, 6.2–7.9, 0.9–1.6 and 290–480 ng·g^{-1}	
	Sargassum honeri	La, Ce, Pr, Nd, Sm, Eu, Gd, Tb, Dy, Ho, Er, Tm, Yb, Lu, and U	33–66, 77–95, 9.3–13, 34–49, 8.2–12, 2.6–4.8, 8–15, 1.3–2.2, 9.4–15, 2–3.5,2.9–4.8, 0.65–1.6, 6.11–7.3, 0.71–1.5 and 240–420 ng·g^{-1}	
	Undaria pinnatifida	La, Ce, Pr, Nd, Sm, Eu, Gd, Tb, Dy, Ho, Er, Tm, Yb, Lu, and U	30-48, 67-100, 5.3–11, 23–44, 3.6–12, 1.9–3.9, 3.8–12, 0.6–3.1, 3.4–9.9, 0.9–2.8, 3.8–7.4, 0.24–0.58, 3.8–8.5, 0.76–1.3 and 490–820 ng·g^{-1}	
Portugal	Litter	U	83.8 ± 5.4 μg·g^{-1}	Bergmann and Graça (2020)
	Macrophytes		47.6 ± 6.9 μg·g^{-1}	
	Scrapers		14.3 ± 1.0 μg·g^{-1}	
	Shredders		13.0 ± 0.8 μg·g^{-1}	
	Engulfer predators		7.0–1.3 μg·g^{-1}	
	Aquatic insects		0.25–2.9 μg·g^{-1}	
	Spiders		1.0–1.7 μg·g^{-1}	
Finland	Urchin entrails	U	178 ± 45 mg·kg^{-1}	Matara-aho (2020)
	Urchin mouth		50 ± 10 mg·kg^{-1}	
	Urchin shell		17 ± 4 mg·kg^{-1}	
USA	Marine biota (*Johnius dussumieri, Pseudorhombus malayanus, Arius maculates, Portunus pelagicus, Charybdis natator*)	U	1.06–1.26 mg·kg^{-1}	Abdullah et al. (2015)

2002; Chang et al., 2005; Sakamoto et al., 2008; Viehweger and Geipel, 2010; Caldwell et al., 2012; Cordeiro et al., 2016; Boryło et al., 2017; Li C et al., 2019; Bergmann and Graca, 2020). Hyperaccumulation of U from contaminated sites has been reported for several plant species and aquatic organisms (Dienemann et al., 2002; Chang et al., 2005; Viehweger and Geipel, 2010; Caldwell et al., 2012; Li C et al., 2019; Bergmann and Graca, 2020). Following the above mentioned general rule, U is accumulated more in plant leaves (up to 100 μg kg^{-1}) than in stems, fruits, and seeds (Anke et al., 2009). For marine biota, U concentrations of up to 178 ± 45 mg kg^{-1} have been reported (Abdullah et al., 2015; Bergmann and Graça, 2020; Matara-aho, 2020).

Plants, mosses, fungi, tea trees, algae, and trees accumulate REEs at trace to low levels, ranging from 0.2 to 109 g kg^{-1} (Markert 1987; Zhang et al., 2007; Li et al., 2012; Mashitah

et al., 2012; Brioschi et al., 2013; Mahmood et al., 2015; Pratas et al., 2017; Zhuang et al., 2017; Mędyk and Falandysz, 2022; Shi et al., 2022), while *Phytolacca icosandra* has been found to hyperaccumulate REEs with concentrations up to 13 g kg^{-1} (Yuan et al., 2018; Grosjean et al., 2019). In general, light REEs are more readily taken up by plants than heavy REEs due to their higher mobility in soil. For example, citrus plants have been reported to act as bioaccumulators for light REEs (Ramos et al., 2016).

As part of research efforts aimed at recycling these elements in a stepwise and cost-effective manner (Sinha et al., 2016), studies on the use of plants (Sytar et al., 2016) or microorganisms (Jalali and Lebeau, 2021) for REE recycling are receiving increasing attention due to their ability to accumulate these elements in significant concentrations.

The positive effects of REEs on quality and growth have also been studied in crops and livestock (Vítová et al., 2018). In the former, positive effects on photosynthesis, biomass production, cytoplasmic membranes, nutrient metabolism, enzymes, water use efficiency, germination and growth, amount of sugars and other metabolites, and number of vitamins, among others, have been found (Kovaříková et al., 2019). In animals, REEs are used as food additives because they can improve the production of milk, eggs, and meat. However, some studies show just the opposite effect, with toxic effects associated with substitution of Mg, Ca, Fe, and other elements by REEs that disrupt bone integrity, cellular signaling, fluid peroxidation, and phosphate deficiency in a variety of animals and plants (Sneller et al., 2000). Such positive or negative effects depend on the REE dose. Therefore, further research on this topic is needed to accurately determine whether REEs are "essential" or merely support growth and development (Ramos et al., 2016).

Human activities have caused concentrations of U, Th, and REEs to increase above natural levels in many environmental compartments, making them accessible to humans through various pathways. According to the Agency for Toxic Substances and Disease Registry (ATSDR, 1990), the health effects of natural and depleted uranium are due to chemical effects rather than radiation. To primarily prevent kidney damage, emission limits for human body exposure to uranium have been established for ingestion of contaminated food (0.6–1.0 pCi d^{-1}, 1 μg = 0.72 pCi), drinking water (0.6–1.0 pCi d^{-1}), and inhalation of contaminated air (0.0007–0.007 pCi d^{-1}), respectively (USEPA, 1983; ATSDR, 1990). On the other hand, inhalation of thorium dust has been reported to increase the risk of lung and pancreatic cancer (USEPA, 2015). The health effects of REEs have been found to be associated with modulation of oxidative stress on Nuclear Factor Erythroid-2-Related Factor-2 (transcription factor) protein and endocrine effects on the hypothalamic-pituitary-thyroid axis (Pagano et al., 2015; Guo et al., 2020). The standard sanitary limit regulated by national standard of China for REEs in vegetables and fruits is 0.70 mg kg^{-1} (Jin et al., 2015).

3 Conclusion

The rapid development of today's society, accompanied by increasing demand for energy and environmentally friendly high technologies, requires an ever-increasing access to U, Th, and REEs. As the demand for these elements increases, so does the amount of waste, byproducts, and residues generated during the various steps of manufacturing and using products containing U, Th, and REEs. All this leads to an ever-increasing pollution of the environment and all its components. As a result, elevated concentrations of U, Th, and REE have been detected in air, water, and biota. However, there is still a lack of information on their fate in the environment, their toxicological effects on organisms, and the consequences for the ecosystem, prompting research on these issues.

In this context, secondary resources are gaining importance not only from the point of view of environmental friendliness and resource conservation, but also as a means of securing supply under conditions of an uncertain and unstable market. Recycling of U, Th, and REE from industrial wastes such as coal ash, red mud, magnets, catalysts, etc. Has been studied and applied for years, but not on a large scale. At the same time, alternative methods such as the use of hyperaccumulator plants, extraction from unconventional sources, or extraction with macromolecules or microorganisms are becoming increasingly important.

Author contributions

KP: Conceptualization, data curation, investigation, methodology, validation, visualization, writing—original draft. SS: Data curation, visualization, graphics drafting, review. JM: Investigation, validation, editing. PM-R: Data curation, graphic drafting, editing original draft. ŽF: Software, editing—original draft. PB: Review and editing. YZ, Discussion, review and editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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