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Potentially Toxic Elements in Solid Waste Streams: Fate and Management Approaches

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Abstract:

Solid wastes containing potentially toxic elements (PTEs) are widely generated around the globe. Critical concerns have been raised over their impacts on human health and the environment, especially for the exposure to PTEs during the transfer and disposal of the wastes. It is important to devise highly-efficient and cost-effective treatment technologies for the removal or immobilisation of PTEs in solid wastes. However, there is an inadequate overview of the global flow of PTEs-contaminated solid wastes in terms of geographical distribution patterns, which is vital information for decision making in sustainable waste management. Moreover, in view of the scarcity of resources and the call for a circular economy, there is a pressing need to recover materials (e.g., precious metals and rare earth elements) from waste streams and this is a more sustainable and environmentally friendly practice compared with ore mining. Therefore, this paper aims to give a thorough overview to the global flow of PTEs and the recovery of waste materials. This review first summarises PTEs content in various types of solid wastes. Then, with respect to their patterns of transport transformation and risks in the changing environment, toxic metal(loid)s, radioactive elements, and rare earth elements are critically reviewed. Different treatments for the management of these contaminated solid wastes are discussed. Based on an improved understanding of the dynamics of metal(loid) fates and a review of existing management options, new scientific insights are provided for future research in the development of high-performance and sustainable treatment technologies for PTEs in solid wastes.

Keywords: metals/metalloids; rare earth elements; radioactive elements; sustainable waste management; environmental and human health risk; green and sustainable remediation.

Table of Contents

List of Abbreviations

APCDs	Air Pollution Control Devices
CCC	Chronic Criterion Concentration
CCRs	Coal Combustion Residues
EPA	United States Environmental Protection Agency
ESP	Electrostatic Precipitators
E-waste	Electronic Wastes
FGD	Flue Gas Desulfurization
HREE	Heavy Rare Earth Elements
IAEA	International Atomic Energy Agency
LREE	Light Rare Earth Elements
L/S	Liquid to Solid
MSW	Municipal Solid Wastes
PRB	Permeable Reactive Barrier
REEs	Rare Earth Elements
S/S	Stabilization/Solidification
TCLP	Toxicity Characteristic Leaching Procedure
TENORM	Technologically Enhanced Naturally Occurring Radioactive Materials
WEEE	Wastes of Electrical and Electronic Equipment

1. Introduction

Solid wastes, including municipal, electronic, and industrial wastes, lead to global environmental problems. Metal(loid)s (e.g., chromium, arsenic, selenium, cadmium, mercury, and lead) are widely used in industry and present a major concern in solid waste management in view of their acute and chronic risks to human health and ecosystems (Song and Li, 2014; Marra et al., 2017). Similarly, radioactive elements (e.g., uranium, radium, and thorium) in wastes generated from nuclear applications for energy, research, and clinic purposes should be cautiously treated to avoid leaching and subsequent environmental risks. For rare earth elements (e.g., yttrium and scandium), there are economic incentives for people to recover these precious resources from solid wastes due to their limited availability in the Earth's crust (Binnemans et al., 2013; Tunsu et al., 2015; Işıldar et al., 2019). Studying the fate and transportation of these toxic elements in solid wastes are of great significance to better understand the pathways of contaminants and their interactions with the environment. This is essential for the design of effective treatment processes based on sound scientific understanding.

In recent years, increasing attention has been drawn to the environmental fate of potentially toxic elements (PTEs) from wastes. PTEs refer to chemical elements including both metals and non-metals that may potentially cause harmful effects on the organisms in the environment if present in high concentrations (Antoniadis et al., 2019). It has been recognized that municipal solid wastes are a source of toxic elements produced in waste-to-energy incineration and found in flue gas and fly ash (Hu et al., 2018). The leachability and mobility of PTEs in flue gas and fly ash show sensitivity to pH (Izquierdo and Querol, 2012; Chen et al., 2019). In addition, the PTEs in wastewater accumulate in sewage sludge. Upon thermal treatment of sewage sludge, PTEs are distributed in different fractions (i.e., ashes, chars, and bio-oils), largely depending on the conditions of temperature and humidity (Fang et al., 2018; Udayanga et al., 2018). The ultimate destination of PTEs is accumulation in soil, sediment, and surface water or groundwater systems, depending on various physical and chemical characteristics of the wastes (Tabelin et al., 2018). Previous reviews have only considered the association of PTEs with certain kinds of solid waste streams such as fly ash. However, the quality of solid wastes derived from different industries and under different regulatory contexts can have considerable variations, and few studies consider that.

In management of hazardous waste, stabilisation/solidification (S/S) using cementitious materials or other stabilisers is the most common approach, which covers approximately 40% of the commercial treatment market (Wang et al., 2015; Wang et al., 2018a; Chen et al., 2019). Recent research efforts have explored innovative treatment technologies for PTEs in solid wastes, such as biohydrometallurgy (Sethurajan et al., 2018), electrodialytic treatment (Kirkelund et al., 2018), and sorbent capture (Ke et al., 2019). There is an emerging perspective for solid waste management, which is to valorise the wastes for different applications, e.g., dredged sediment as construction materials (Wang et al., 2015), blast furnace slag as cementitious materials (Wang et al., 2018a), red mud or calcined alum sludge as a metal sorbent (Ahmed and Ahmaruzzaman, 2016; Jeon et al., 2018), and incinerator sludge ash for zeolite catalyst synthesis (Feng et al., 2018; Zhang et al., 2018a).

In contrast to previous reviews on PTEs which tend to focus on specific waste streams (Hu et al., 2018; Udayanga et al., 2018; Işıldar et al., 2019), this review aims to provide a PTEs-oriented and comprehensive overview of the fate of metal(loid)s, radioactive elements, and rare earth elements in a wide range of solid wastes to elucidate their potential environmental and human risks, and to identify the opportunities for new developments in solid wastes management including recycling and valorisation. State-of-the-art treatment technologies are discussed to assess their feasibility on the field scale and facilitate decision making by industrial practitioners. Moreover, this review identifies existing managerial and technological limitations, which should shed light on the directions of future research for the scientific community.

2. Flow of metal-containing solid wastes

Solid waste is a summative term covering all the discarded materials in the solid or semi-solid forms which come from municipal, industrial, commercial, mining, and agricultural activities (McDougall et al., 2001). Metal-containing solid wastes, encompassing those from municipal to industrial sources, have been extensively concerned all over the world (Table 1) and their fates are summarised in Figure 1. Derived from household activities, municipal solid waste (MSW) is one of the largest components of wastes streams and includes food wastes, paper, textile, rubber, plastic, glass, metals, wood, and miscellaneous inorganic wastes. The composition of MSW varies from district to district, which influences the PTEs composition in the waste stream. MSW is predominately treated by incineration, landfill, and sometimes, composting. MSW incineration has grown steadily in recent years worldwide. For example, the amount of incinerated municipal solid wastes in the 27 EU Member States has risen by 32 million tonnes in ten years, reaching 64 million tonnes in 2014 (Abramov et al., 2018). In China, approximately 40% of MSW in metropolitan areas is incinerated, and this figure is expected to reach 60% by 2020. As a result, the generation of incinerator fly ash in China exceeded 5 million tonnes in 2016 (Du et al., 2018).

During MSW incineration, organometallic compounds volatilise along with volatile metals (e.g., Hg and Cd). As temperature decreases in the downstream units, volatilised metallic compounds experience condensation and nucleation reactions to form submicron particles. Most organic-based materials can be converted to carbon dioxide and water vapour by complete oxidation at high temperature, whereas the metal fractions still remain in the ash particles. Ashes consisting of metals, salts, and organic contaminants either aggregate in the heat recovery phase (coarse particles) or are removed from the flue gas by fabric filters or electrostatic precipitators (ESPs) (fine particles)

(Weibel et al., 2017). Boiler and ESP ashes constitute the fly ash, while the incombustible residue that accumulates at the bottom of incinerators is the bottom ash. These incineration solid residues have gained increasing concern due to their leaching potential to soil and water after disposal in landfill sites. Volatile components are separated from the flue gas via a flue gas control process; metals can be present in flue gas (e.g., Fe-oxides and brass) as well as mineral aggregates (e.g., quartz, feldspar, and glass), or may be vaporised and condensed as water-soluble chlorides (K_2ZnCl_4) or sulphates ($PbSO_4$). Metal mobilisation and leaching are significantly dependent on these metal associations (Weibel et al., 2017). As for disposal of hazardous wastes such as incineration fly ash, significant effort should be made in developing metal stabilisation technologies (e.g., S/S, vitrification, sintering, and thermocatalytic and hydrothermal treatment). On the other hand, it is estimated that the value of precious metals, light and heavy rare earth elements (REEs), and other technology-critical elements including Cu in incinerated resides can reach $1,916 per ton of material (Abramov et al., 2018). Therefore, the recycling of metals from incineration ash is a promising future approach.

The deposition and accumulation of PTEs during MSW disposal in landfill sites is a major concern; PTEs have complex pathways into the environment and potentially pose high risks to living organisms surrounding landfill areas. The content of major metals in landfill material is reflected by waste precursors, which are the solid wastes formed from heterogeneous interactions between disposed wastes, climatic conditions, and the management practices of the landfill (Adelopo et al., 2018). Waste degradation reduces the weight of landfilled waste, but metal concentrations increase. Unlike organic pollutants, metals do not degrade in landfills and their residence time is approximately 150 years (EU, 2002). Metal leachability from landfills results in the pollution of farmlands, surface waters, and underground waters. Higher levels of metal contamination have been found in closed landfills than active ones due to on-going degradation. However, the metal bioavailability might reduce with ageing processes as a result of the stronger binding formed with soil components (Tye et al., 2003).

In comparison, composting is an alternative approach to treating the organic fraction of MSW (Godlewska et al., 2017), but this can also pose a threat to aquatic and edaphic systems. For example, open-air windrow composting piles has resulted in the uncontrolled emission of metal contaminants and considerable ecological risks in the surrounding watershed (Sharifi et al., 2016). Natural events such as rainfall also tend to accelerate the leaching of PTEs if inadequate control is in place to isolate open windows or stockpiled composting sites.

Electronic wastes have recently drawn extensive concern due to the short lifespan of electrical and electronic products nowadays. With the development of the electronics industry, a significant amount of electronic waste (e-waste) is released into the environment at the rate of 20–50 million tonnes per year. Common metals found at e-waste sites are both valuable (e.g., Cu and platinum groups) and hazardous (e.g., Pb, Sb, Hg, Cd, and Ni) (Beiyuan et al., 2016; Jiang et al., 2019). After e-waste disposal in landfill sites, PTEs are not reduced as other organic contaminants, but are accumulated, transported, and diffused into soil pore water, thereby threatening soil ecosystems. Furthermore, PTEs are released in the form of airborne, size-fractionated bounded particles. Research has found that metals with carcinogenic effects pose severe health risks to humans by means of inhalation, particularly for children in the vicinity of e-waste recycling areas (Huang et al., 2016).

Industrial wastes produced by numerous manufacturing and processing activities are also critical sources of PTEs. Among them, coal combustion is an essential means of electricity and heat generation, generating significant volumes of coal combustion residues (CCRs) and coal combustion products (CCPs). Globally, coal-based power generation produces over 750 million tonnes of coal ash per year (Izquierdo and Querol, 2012). It has been estimated that over 130 million tonnes of

CCRs are generated by US power plants every year (Ruhl et al., 2012). Composed of fly ash, bottom ash, and flue gas desulphurisation (FGD) materials from coal combustion, CCRs contain numerous crystalline oxide phases of silicon, aluminium, calcium, and iron as well as metals and trace elements (Schwartz et al., 2018). Similar to MSW incineration fly ash, the fly ash from coal combustion can be collected from flue gas by ESP or bag filters. Coal ash is regarded as a major potential source of PTEs in the environment. Hence, its disposal has drawn great attention. The characteristics of the coal source significantly influence the variations in PTEs content. For example, it is estimated that coal-combusted fly ash contains 2.3 to 6,300 mg/kg arsenic, 0.02–0.36 mg/kg mercury, 1.2–236 mg/kg molybdenum, and 0.2–134 mg/kg selenium, and these concentrations are approximately 10 times higher than those of the original source (Feng et al., 2018). Due to the high content of organic and inorganic contaminants, EU directives (91/689/EEC) classify coal fly ash as a hazardous waste. In contrast, in the United States (Resource Conservation and Recovery Act), coal fly ash is considered a non-hazardous material if leaching rates pass the TCLP (toxicity characteristic leaching procedure) (Tsang et al., 2014; Kirkelund and Jensen, 2018). Furthermore, the water used for producing steam, cooling, and transportation—which may also be contaminated by CCRs—poses a threat to the natural water environment.

The generation of sewage sludge from wastewater treatment plants has been steadily increasing over time. It is estimated that America generates 71 million tonnes of sewage sludge per year, and in Europe, the average annual outputs of sewage sludge in Germany, England, and France are 22, 12, and 8.5 million tonnes, respectively. In China, over 20 million tonnes of sewage sludge are generated each year (Xu et al., 2014). Sewage sludge is a heterogeneous mixture of microorganisms and undigested organic (e.g., proteins, polysaccharides, and lipids) and inorganic (e.g., minerals and metals) components with high moisture content (Kwon et al., 2018; Yang et al., 2017). The disposal and treatment of sewage sludge is a global environmental challenge. Recently, fuel generation (i.e., heat, hydrochar, bio-oil, and syngas) from sewage sludge via thermal processes (i.e., incineration, gasification, pyrolysis, and hydrothermal treatment) has gained increasing attention (Kwon et al., 2018). However, the elevated concentrations of metals that result from thermal processes has presented environmental challenges to their increasing application (Udayanga et al., 2018).

3. Fate of toxic metal(loids) and their treatment

In the following subsections, the emission and transfer of commonly encountered toxic metal(loids) (i.e., chromium, arsenic, selenium, cadmium, mercury, and lead) from solid waste streams are explained (Section 3.1 to 3.6), in an order with respect to the periodic table, and relevant research is summarised in Table 2. Afterwards, their decontamination approaches are elaborated and discussed in Section 3.7.

3.1. Chromium

Chromium is a PTE that primarily exists in trivalent and hexavalent forms. Because of its strong oxidising and toxicity properties, hexavalent chromium compounds can easily contaminate soil, groundwater, and rivers through leaching, which leads to serious ecological problems (Rajapaksha et al., 2013; Rajapaksha et al., 2018). The toxicity of chromium varies depending on the valence state and its morphology. Hexavalent chromium is a toxic carcinogenic element, and its toxicity is one hundred times greater than that of trivalent chromium (Mandal et al., 2017). Metal chromium and chromium salts are regarded as important strategic resources, widely used in leather, electroplating, chemical, ceramics, advanced alloy materials, spices, pigments, printing and dyeing, anti-corrosion treatment, medicine, and other industries (Tsang and Yip, 2014). Chromium may derive from e-waste such as data tapes and floppy-disks and has a typical concentration of 9,900 mg/kg in e-waste with a global generation rate of 198,000 tonnes (Gaidajis et al, 2010). Landfill, as the most common approach for e-waste disposal, limits the recycling and reuse of Cr-containing materials from this waste stream (Ashraf et al., 2017).

Leather production enterprises are important Cr sources; Cr is produced from chromium powder, and chromium-containing wet blue and other pigments in the tanning process (Wu et al., 2015). The concentration of Cr predominantly depends on the production conditions, the types of raw materials used, and the level of production management. Tannery sludge is a toxic industrial solid waste produced during the physical treatment of tannery wastewater. By treating tannery sludge with ladle furnace slag and organoclay, chromium and dissolved organic carbon concentrations can be lower than the specified limits for non-hazardous waste for landfill disposal (Rajapaksha et al., 2013). In addition, bioleaching has been developed as a successful and cost-effective method for removing Cr(III) from tannery sludge. Mixed inoculations of primary iron sulphur-oxidising bacteria, Fe2+, and elemental sulphur are used, which accelerate the bio-oxidation of Fe2+ and elemental sulphur to produce acid and increases the redox potential, reaching a maximum removal rate at 95.6% (Zhou et al., 2006).

Electroplating production also generates large amounts of sludge containing 3% to 5% chromium, which mainly exists as hydroxide (Yuan et al., 2013; Yi et al., 2014). There are three main treatment methods for electroplating sludge: (1) stabilisation treatment; (2) thermochemical treatment; and (3) resource utilisation (Yuan et al., 2017). Cement and fly ash can be used to immobilise high chromium plating waste by S/S process (Dayananda et al., 2013). The MSW incineration fly ash-based Friedel matrix also can be used to stabilise industrial chromium-containing electroplating sludge. The formation of ettringite and Friedel plays an important role in the fixation of metals in electroplating sludge (Qian et al., 2009). Recently, the possibility of using stabilised and solidified sludge in the construction and building industry has been explored (Kanwan et al., 2017). During plasma in-fight treatment, hexavalent Cr is converted into trivalent, leading to a decrease in toxicity, especially at high temperatures. In addition, incineration can effectively enrich Cr in electroplating sludge, and 90% of the chromium is retained in the ash (Espinosa and Tenório, 2001). For resource utilisation, Cr in electroplating sludge can be recovered by alkaline oxidation of hypochlorite.

Hypochlorite can completely oxidise chromium(III) hydroxide in suspension to chromate under alkaline conditions, and can then pass sludge roasting and aqueous leaching (Matthew et al., 2000). The element can then, finally, be filtered to separate the chromate solution. The recovery efficiency of chromium can reach 92% under the optimal conditions (Silva et al., 2005). Furthermore, sulphidation and oxidation treatment can be used. Firstly, mixed electroplating sludge is simultaneously extracted and copper vulcanised at pH 1.5 to dissolve the metals, and Cu2+ precipitates in the form of CuS. Secondly, Zn is subjected to sulphurisation treatment at pH 4.5, and then at pH 10, and the remaining liquid is oxidised with H2O2. The oxidation rate of Cr increases when both the temperature and the amount of added H2O2 increase (Kuchar et al., 2010). Metal phosphate and chromium oxide nanocomposites have been prepared from Cr-III electroplating sludge using an extraction-precipitation-calcination process. Metal-phosphate/chromium oxide nanocomposite has a good potential to be used as a functional pigment in energy-efficient buildings (Li et al., 2015).

3.2. Arsenic

Arsenic widely exists in contaminated soil and tailings produced from metallurgical processes. Some sediments and bio-wastes also contain high concentrations of arsenic, which can cause significant harm to the environment. In recent years, due to irrigation and the use of a large amount of groundwater in industrial processes (such as mining, non-ferrous metal smelting, coal mining) as well as the use of arsenic-based pesticides and municipal sewage sludge, arsenic concentrations in soils have increased (Tsang et al., 2014; Niazi et al., 2018; Shakoor, et al., 2018). Groundwater contaminated with arsenic has been a serious problem. When arsenic in soil enters plant tissue, metabolism is affected; arsenate weakens phosphorylation in mitochondria. Arsenite inactivates many enzymes by reacting with sulfhydryl proteins, hindering the moisture uptake in the plant. It also affects the absorption of water and nutrients by plants and causes arsenic to accumulate in plant tissue, resulting in damage to chlorophyll. Arsenic-containing tailings often contaminate soil, groundwater, and surface water, causing further damage to plants, animals, and humans (Mandal and Suzuki, 2002).

Arsenic is ubiquitous in the environment and highly toxic to all forms of life. Trivalent arsenic is more toxic its pentavalent form, and inorganic arsenic is more toxic than organic arsenic (Niazi et al., 2018). High levels of organic arsenic compounds such as arsenic betaine and arsenic choline are found in marine organisms, and they show good resistance to chemical degradation. Arsenic can accumulate in organisms and then slowly excreted. In general, organic arsenic compounds are excreted more quickly than inorganic arsenic, and pentavalent arsenic is removed from the body faster than trivalent arsenic (Mandal and Suzuki, 2002). Arsenic enters the body of organisms through water pollution, and through the consumption of As-containing plants, dairy products, and meat. Arsenic poisoning can cause melanosis, keratosis, hyperkeratosis, back edema, gangrene, skin cancer, and other diseases (Singh et al., 2015b). Arsenic in the human body distributes in hair, nails and teeth, generally in the order of 3-4 mg/kg, increases with age. The absorption of arsenic in the human body is high for anionic and soluble species, but very low for insoluble species (Singh et al., 2015b).

3.3. Selenium

Selenium (Se) is a non-metal with intermediate properties between sulphur and tellurium. Being prone to mobilisation and bioaccumulation in aquatic organisms, selenium is toxic at high levels, although it is an essential dietary micronutrient for humans and animals at trace concentrations. It also acts as a fertiliser additive to enrich selenium-poor soils for plant growth. The Se species partitioning plays a significant role in its mobility and bioavailability. Oxidised Se predominately exists as oxyanions dissolved in solution or adsorbed to mineral surfaces. Oxidised Se (i.e., selenite and selenate) are highly soluble and more mobile than the less soluble elemental Se0 (Liu et al., 2013).

Selenium originates from various industrial solid wastes, such as those resulting from glass manufacturing, selective oxidation catalysts, plating, rubber compound synthesis, and electrolytic manganese production. It is estimated that the world’s consumption of Se consists of 40% metallurgy, 25% glass manufacturing, 10% agriculture, 10% chemicals, 10% electronics, and 5% other uses (United States Geological Survey, 2018). Selenium can be recovered from the electrolytic refining of copper as well as from photocopiers and electronic materials. Coal is also a major source of selenium (0.5–12 mg/kg) (United States Geological Survey, 2018), which is 80–90 times higher than the average concentration in copper deposits. Selenium in coal is released during coal combustion, but it has a potential for recovery from coal fly ash. Selenium in coal ashes (including fly ash and bottom ash) is a mixture of elemental Se^0 and Se oxyanions, mostly existing as sulphide and organic associations, among which selenite is the major dissolved species, constituting approximately 75−94% of the total Se (Liu et al., 2013). Selenium associates with coal ash by the adsorption of Se oxyanions (selenite and selenite) on mineral surfaces such as Fe-, Al-, and Mn-(hydr)oxides. In addition, solid solutions can be formed with minerals such as gypsum ($CaSO_4$) and hydrocalumite ($Ca_4[Al(OH)_6]_2(OH)_2$ $6H_2O$). Selenium in fly ash also coprecipitates with multivalent cations including Ca^{2+}, Pb^{2+}, and Ba^{2+} (Liu et al., 2013). The weak bonding with the coal solid phase means that Se is easily released and almost completely volatilised during combustion (Izquierdo and Querol, 2012). As an important component of e-waste, Se enters the soil after industrial or municipal solid waste disposal. Its transport involves metal complexation and adsorption, which are closely related to landfill age (Adelopo et al., 2018), but does not appear to be influenced by depth (Huang et al., 2016).

Leaching potential is regarded as an important parameter when elucidating the mechanisms of Se migration and its impact on aquatic ecosystems. It has been reported that during leaching from coal ash samples, dissolved Se generally increases with a decreasing liquid-to-solid (L/S) ratio and increasing reaction time, as well as a pH-dependent pattern (Liu et al., 2013). In a fly ash impact assessment for groundwater quality, most elements including Se and Pb showed high leaching effects at a L/S ratio of 5 (Dandautiya, et al., 2018). In these cases, the concentrations of Se in groundwater samples from nearby the fly ash dumping site were up to 0.077 mg/L and higher than the World Health Organization’s desirable drinking water limit (0.04 mg/L). When pH is less than 2.5, the dominant species (H2SeO3) is not easily adsorbed on the surface of ash particles and so remains in solution. It has been reported that over 70% of Se can be removed from coal fly ash at pH < 1 considering its primary surface association (Izquierdo and Querol, 2012). However, as pH increases, the dominant anionic species HSeO3 − and SeO32− can be adsorbed onto the surface of ash particles, from which desorption of Se oxyanions plays a vital role in Se release (Izquierdo and Querol, 2012). Low proton levels under alkaline conditions promote the oxidation of elemental Se0, thus boosting desorption or dissolution of Se oxyanions into the leaching solution. The amount of leachable Se from coal fly ash increases with pH, and dissolved Se concentrations at pH of 13 are 13−81 times higher than at pH 5 (Liu et al., 2013). Therefore, high Se leaching under alkaline conditions becomes a major concern about combustion wastes, and dynamic variations in ecosystems (notably pH increases) can be conducive to higher rates of Se mobilisation.

It has been reported in northern California that CCR from electrical power plants has had an impact on the surrounding waterways (i.e., lakes and rivers) (Ruhl et al., 2012); in comparison to upstream, Se was enriched at outfalls with concentrations as high as 87.2 µg/L, i.e., 17 times Chronic Criterion Concentration (CCC) of Se for aquatic life of (5 µg/L) as identified by the US EPA. Seasonal thermal stratification (i.e., depletion of oxygen in bottom water during summer and overturning during autumn) also induces spatial and temporal changes in water chemistry, which can lead to Se adsorption onto and desorption from suspended solids in rivers and its accumulation in bottom sediments. Nonetheless, Se can be immobilised in sediment by forming elemental selenium and metal−selenium complexes and is not incorporated into pore water (Ruhl et al., 2012).

Other important factors for Se are the nature of the combustion facilities at the source as well as the chemistry of the receiving environment. Effluent from flue gas desulphurisation (FGD) systems, which is usually added to ash pond water for dilution, has a higher enrichment of contaminants compared to ash discharged at outfalls. FGD disposal systems effectively remove volatile elements associated with SOx, and significantly accumulate contaminants in solid wastes and wastewater discharged from power plants. Selenium concentrations are also higher at FGD outfalls compared to outfalls that discharge ash only (without FGD). Anoxic conditions in water lead to lower dissolved Se concentrations. The oxidised species selenate (Se(VI)) is less reactive to adsorption onto oxyhydroxides in aerobic conditions, while anoxic conditions result in the conversion of Se into reduced forms (e.g., elemental Se0 and FeSe) that possess strong adsorption affinity towards both oxyhydroxides and clay minerals (Ruhl et al., 2012). Flow rate also plays a significant role in the dilution effect and thereby influences the contamination level of the receiving waters. During periods of drought, the dilution factor is reduced, e.g., a five-fold reduction in water flow would raise the CCR contribution up to 22%, thus increasing the concentrations of contaminants in downstream river water (Ruhl et al., 2012). In addition, changes in the ambient conditions of lake sediments (e.g., pH and redox state) tend to result in the release of Se metalloids into pore water.

3.4. Cadmium

Cadmium (Cd) is considered an extremely toxic element for human and aquatic life. It is normally immobile under near neutral and alkaline conditions (Rizwan et al., 2016). Exposure to high concentrations of Cd can damage the reproductive system, the lungs, DNA, and the kidneys, and can cause problems for learning, cognition, behaviour, and neuromotor skills in children (Rizwan et al., 2018). Consumption of cadmium and its compounds is linked to the wide use of industrial materials such as alloys, coatings, batteries, pigments, and plastic stabilisers (United States Geological Survey, 2018). For Cd in sewage sludge, concentrations ranging from 0 to 110 mg/kg were identified in different sewage sludge around the world (Udayanga et al., 2018). An average Cd concentration of 2.13 mg/kg dry matter was reported across different seasons in sewage sludge from Greece (Spanos et al., 2014), while a concentration of 1.7 mg/kg was reported for sludge in a sewage treatment plant in France (Manon et al., 2019). In the latter case, a large proportion of Cd was associated with Fe-Mn oxides (52%) and oxidisable fractions (21%), with the rest being associated with the residual fraction. The pH-dependent leaching of Cd showed a bell-shaped relationship, which may be attributed to Cd desorption from Fe-Mn oxides the dissolution of Fe-oxide bearing phases, and the leaching of organic ligands from particulate organic matter (Manon et al., 2019).

Disposal of e-waste, such as batteries and paint residual, is an important source of Cd in the aquatic and soil environment in the vicinity of municipal landfill areas. Cadmium is reported as a potential contaminant in both active and closed landfills, presenting the highest health risk potential index rating among other metals including Cu, Pb, Zn, and Ag (Adelopo et al., 2018). Depending on their mobility, these metals can penetrate through the soils and disperse vertically/horizontally across subsurface layers. A reduction in pH decreases Cd adsorption and increases its mobility through the enhanced competition for negative surfaces between H+ and dissolved metals (Rizwan et al., 2017). Metal concentrations generally decrease with increasing depth, because more sorption sites are provided by higher organic matter content in surface soils and higher pH conditions reduce metal mobility in deeper soil profiles (Wu et al., 2015). As Cd mainly exists in an extractable form, it shows a relatively weak vertical mobility compared with other metals (Jiang et al., 2019). It was found that illegal e-waste recycling activity in south China caused severe contamination of surrounding surface soils (Cd > 0.39 mg/kg, exceeding the guideline level of 0.3 mg/kg) and pond water (Wu et al., 2015; Zhu et al., 2018). Furthermore, contaminated pond water has often been used for farmland irrigation, resulting in the metal contamination of paddy soil and significant environmental damage (Ok et al., 2011). It has been shown that closed landfill sites suffer more severely from metal contamination than active landfill sites. In addition to landfill age, metal concentrations also increase with a decrease in particle size, which corresponds to the degree of waste degradation. As discussed in Section 2, metals are not reduced but are redistributed or released during waste degradation in landfill sites. E-waste recycling/disposal also promotes the release of Cd into the atmosphere as well as the formation of airborne particle-bound metals that can be deposited in the respiratory systems of humans living in the vicinity of recycling sites. For example, airborne particle-bound Cd with an average concentration of 6.1 ng/m3 displayed a spatial distribution with peaks at a distance of 0.56–1.0 m and 1.0–1.8 m from a recycling site, resulting in an incremental lifetime cancer risk that was higher than the acceptable cancer risk threshold (Huang et al., 2016).

Different from e-waste recycling zones, general MSW landfill appears to have less of an impact, with typically lower Cd contents. In a life cycle assessment of MSW management system in Turkey, Cd derived from industrial hazardous waste or household solid waste was found to be negligible (≤ 0.05 mg/kg oven dry waste) (Yay et al., 2016). One study detected 1.0–3.0 mg/kg Cd in the fine fraction (< 10 mm) of landfilled wastes, where Cd dominantly existed in its reduced form, but was found in relatively low concentrations along with other potentially toxic elements such as Pb and Cr (Burlakovs et al., 2018). These metals were mostly bound in environmentally stable compounds, meaning there was potential to recover materials from the landfilled wastes after appropriate safety evaluations. It was found that Cd had little effect on bacterial communities (microbial distributions) in a landfill area, as was the case for other sulphurophile elements (Cu and Zn) (Liu et al., 2018). Metals such as Cd are suggested to have a limited influence on soil bacterial composition and diversity, even at e-waste sites (Jiang et al., 2018). It appears that soil bacterial taxonomic composition and microbial diversity are more influenced by soil texture, organic carbon content, and pH conditions (Sun et al., 2016; Jiang et al., 2019).

Volatile Cd species are largely transferred from the MSW sources to fly ash during the incineration process. MSW and coal combustion releases Cd to flue gas and fly ash; the most of gaseous forms are condensed to or adsorbed by particles and thus retained in the fly ash (Zhang et al., 2018b). Cadmium accounts for approximately 20% of the metal availability in fly ash and mostly exists in the non-mineral phase, resulting in its higher mobility and release than other metals (Du et al., 2018). Cadmium mostly exists as particulate matter in flue gas, which is collected in the fabric bag or electrostatic precipitator as fly ash (Ahmad et al., 2018). Metals with lower boiling points, such as Hg (357 °C) and Cd (767 °C), are distributed more commonly in fine fly ash particles. In addition to incineration, other thermal processes (such as pyrolysis, gasification, and hydrothermal treatment) are also employed for the treatment of sewage sludge. Cadmium in sewage sludge was reported to vary from below detection limits to 110 mg/kg and was more commonly found in acid-soluble and Fe-Mn oxides fractions (Udayanga et al., 2018). Cadmium has a greater tendency for volatilisation during incineration and gasification but exhibits moderate volatilisation during pyrolysis and hydrothermal process.

Incineration temperature has an important influence on the metal composition of fly ash. For example, it was found that higher incineration temperatures (between 1003 °C and 1036 °C) led to 235% higher Cd concentrations in the fly ash, as the transfer rates of volatile metals were increased (Shi et al., 2018). Cadmium exhibits a pH-dependent leaching pattern in fly ash, which is soluble in acidic conditions but becomes immobile towards near-neutral conditions (Izquierdo and Querol, 2010). Since Fe is oxidised to form (hydr)oxides in the early stage of combustion, the resulting alkalinity of the solid residue delays Cd mobilisation. However, with ageing, neutralisation of alkalinity and dissolution of carbonates and oxides occur and cause a reduction in pH and the subsequent mobilisation of metals. Therefore, industrial sources (e.g., wastewaters and uncontrolled chemical pollution), acid rain, and contact with soil solutions that lead to a lowering of pH may pose the threat of metal release during the storage or disposal of fly ash (Abramov et al., 2018). In addition, chloride may enhance Cd solubility via complexation and shift the solubility curve to a higher pH. During aging, fly ash forms a high carbonate content, which effectively immobilises Cd in the form of otavite (CdCO3), and the formed calcite and hydrous Fe/Al oxides also present strong adsorption, thus controlling the release of Cd (Du et al., 2018). Such adsorption also occurs in the case of other mineral phases, such as Fe and Al (oxy)(hydr)oxides and amorphous aluminosilicates. Particle size is another factor that effects the leaching behaviour. For example, fine particles (< 2 mm) of fly ash were found to contain 77% of the total Cd in all fractions below 16 mm, and thus leached more extractable Cd and other metals (Abramov et al., 2018).

3.5 Mercury

Mercury (Hg) is a heavy metallic element that exists in liquid form under standard conditions. The most common forms of mercury include the less reactive volatile Hg0 species as well as soluble and reactive Hg(II) forms, the latter tends to form soluble complexes with numerous ligands and surface complexes on solids such as HgS and HgSe (Bartov et al., 2013). With toxic, persistent, and bio-accumulative properties, mercury emission may pose a severe risk to the environment and human health (Beckers et al., 2019). Mercury is widely used in various household items, such as alkaline batteries, button cells, fluorescent lamps, thermometers, and sphygmomanometers, as well as in some dental, electronics, and fluorescent-lighting manufacturing industries (Hu et al., 2018; United States Geological Survey, 2018). Mercury released into the environment derives from various sources, including mercury-containing car switches, coal-fired power plant emissions, and incineration of mercury-containing medical devices (United States Geological Survey, 2018). The amount of mercury found in sewage sludge is relatively small. For example, less than 0.2 mg/kg Hg was measured in several different sources of sludge (Spanos et al., 2014). However, direct mercury emissions through gas or water from incineration or landfill of sewage sludge have significantly contributed to environmental toxicity. Hence, integrated sewage sludge treatment is environmentally and economically recommended (Xu et al., 2014)

Waste-to-energy (WTE) incineration is one of the major ways in which mercury in MSW is released into the environment. In China, incineration constitutes 21% of the anthropogenic mercury emissions in the Pearl River Delta district, while in north China, much more mercury derived from the use of coal (Hu et al., 2018). Mercury in flue gas is mostly present in its oxidised form (75–85% Hg2+), with only a small fraction existing in its elemental form (1–20 % Hg0) (Hu et al., 2018). During incineration, the high temperatures in combustion chambers lead to mercury emission from the waste as well as the supplementary fuel (i.e., coal) (O'Connor et al., 2019). Other elements present in the MSW, such as chloride, cause mercury oxidation and promote the formation of chloromercury compounds in the flue gas. After cooling, the reactive and oxidised form (Hg2+) in flue gas can be adsorbed onto unburnt carbon particulates and subsequently removed by particulate matter control devices or captured by wet scrubbers. In contrast, volatile, chemically inert, and poorly water-soluble Hg0 cannot be removed effectively from flue gas but can be oxidised to Hg2+ by selective catalytic reduction devices (Hu et al., 2018). Devices for capturing particulate matter, primarily fabric filters and electrostatic precipitators, perform well simultaneously removing mercury bound to particulates. Activated carbon is often used as an effective adsorbent for mercury-specific control in incineration flue gas in combination with other devices (Hu et al., 2018).

Similarly, Hg derived from coal combustion in coal-fired power plants involves a similar process to MSW incineration, and the resulting concentrations in fly ash and the flue gas desulphurisation products are also similar (O'Connor et al., 2019). Mercury, ranging from 0.001 to 10 mg/kg in CCRs, is mainly bonded to sulphur, chlorine, and carbon as Hg(I) and Hg(II) species (Deonarine et al., 2013). Mercury capture increases with a decrease in flue gas temperature and increases with an increase in the amount of activated carbon (Hood et al., 2007). Improper disposal of these CCRs would potentially transfer mercury into ecosystems such as waterways. For example, substrates in ash (such as sulphate) could stimulate the biomethylation of mercury to form methylmercury

(MeHg) via anaerobic bacteria in redox conditions. As a result, sediments in surrounding aquatic ecosystems can be damaged as a result of this bioaccumulative and neurotoxic organomercury compound. Research has shown that coal ash disposal has led to elevated concentrations of total mercury and methylmercury in the sediments of receiving rivers compared to upstream sediments (Deonarine et al., 2013).

3.6. Lead

Lead (Pb) is a heavy metal with a high density and low melting point. Pb and its compounds are poisonous, which may cause atmospheric, aquatic, and soil pollution, and have a significant effect on animals and plants. Industrial wastes, commercial wastes, institutional wastes, and residential wastes disperse Pb-bearing waste into the environment (Igalavithana et al., 2019; Li et al., 2019). Automobile exhaust emissions are the most important sources of Pb in the dust collected from the roadside, while non-exhaust emissions are also important sources (Li et al., 2018a). Mine tailings, coal ash, smelting waste, and clinker are the main five solid waste. Of these, mine tailings are the main source of solid wastes but have the lowest utilisation (Xu, 2018).

Lead smelting slag is mainly derived from pyrometallurgy (De Angelis and Medici, 2012). At present, research on the resource utilisation of lead-smelting slag has mainly focussed on the recovery of valuable elements, the use of building materials, and the preparation of mine filling. Waste lead-acid batteries with lead grids and battery paste are a significant source for Pb recovery. Lead grids generally have relatively stable components and can be directly recovered under low-temperature smelting, but battery paste needs a more complex recovery process. Pre-treatment involves direct and indirect melting, where the battery paste of waste lead-acid batteries is melted at 1000 °C, or the sulphur in waste lead-acid batteries is removed by adding desulphuriser at relatively low temperatures before smelting. The lead sludge or lead paste from desulphurisation is then put into the furnace. Lead-contaminated gas, dust emissions, and other forms of pollution may occur if this material is not properly handled (Zhang et al., 2019). Typical hydrometallurgy includes leaching or desulphurisation, electrowinning or cementation, and subsequent smelting and casting. In the leaching process, inorganic acids (such as hydrochloric acid and nitric acid) are used to dissolve lead-containing substances or for desulphurisation, after which electrolysis is carried out. After electrolysis, more than 99.9% of lead can be obtained on the cathode. Recovery of lead oxide from waste lead-acid batteries can be achieved as follows: inhomogeneous surface of active material crystals is first removed by oxidation or reduction, and then desulphurised; lead oxide is the produced by calcining at 400–500 °C; hydrogen chloride can then be used to recover battery materials, while sodium chloride can promote the dissolution of lead sulphate. The leaching rate of this method can reach 99%, but its corrosiveness to equipment and impurity removal remains a concern.

Lead-containing solid wastes in some developing areas are still accumulating, which results in excessive occupation of land (Zhang et al., 2019). Open air stacking of fine Pb-bearing tailings and soot particles is likely to release dust and cause air pollution. Piled Pb may be eroded by rainfall into the surrounding soils. Even after MSW deposition in the landfill, airborne particulates may contain Pb depending on meteorological factors, landfill activity, and waste degradation (Bodihardjo et al., 2018). After erosion by rainfall and surface runoff, discharge and dispersion of contaminated water can cause soil contamination and subsequent damage to local ecological systems. The Pb-laden particulates in the air can cause widespread contamination through atmospheric deposition (Zhou et al., 2018). Lead has a highly negative charge and can easily form covalent bonds with iron, aluminium, manganese and other oxides, organic matter, and carbonates. When Pb enters plants, it firstly affects cell membranes and then influences a series of physiological and biochemical processes in cells, which all lead to metabolic disorders. Factors affecting the absorption of Pb by plants are diverse and determined by the ecological characteristics and external environment (e.g., pH, oxidation-reduction potential, and chemical interactions). In addition, Pb-entrained particulates in the air can enter plants via deposition (Zhou et al., 2018). Human may intake Pb via the respiratory tract, digestive tract, and skin. Children are particularly susceptible to atmospheric Pb. Lead causes damage to all organs, especially during the childhood growth period. The effects of lead on tissues and organs are permanent and irreversible even if environmental pollution is rectified.

3.7 Treatment of metal(loid)-containing wastes

Conventional waste treatment such as incineration and landfill would cause diverse adverse impacts on human health as well as freshwater, marine, and terrestrial ecotoxicity due to the release of metals in the waste streams. To address these problems, advancement has been made based on conventional treatment facilities. Air pollution control devices (APCDs) are installed to mitigate metal pollution from MSW incineration process. Electrostatic precipitators, bag filters, and wet scrubbers are used to capture fine fly ash (0.1–1.0 μm) and toxic gaseous compounds in the exhaust, while the coarse ash particles (1–20 μm) present in flue gas stream are generally captured by cyclones (Udayanga et al., 2018). Fabric filter shows high efficiency in capturing most Cd, Cr, Cu, Ni, Pb, and Zn (Ahmad et al., 2018). As the majority of Hg in flue gas passed through the fabric filter to the upstream of the wet scrubber and existed in an oxidized form, the wet scrubber could efficiently remove oxidized Hg by reduction to elemental Hg, achieving 75-93.2% Hg removal with injection of activated carbon (Ahmad et al., 2018; Hu et al., 2018). Nano-structured zinc sulphide as a novel adsorbent also effectively captured Hg from flue gas to fly ash with less than 0.01% leaching (Li et al., 2018b).

The addition of absorbent materials during combustion process effectively reduces the metal content in the solid residues. For example, the calcium-based adsorbents (i.e., CaO and CaCO3) accomplished volatilization rate of Cr and Zn up to 35.7% and 84.0% for food waste combustion (Ke et al., 2019). During the process, the CO2/O2 ratio, furnace temperature, and Ca type make a crucial influence on the metal removal. Metal recovery from flue gas scrubber sludge by hydrometallurgical process obtained recovery rate of Pb, Hg, and Se of 99.6%, 98.9%, and 95.5%, respectively (Xing et al., 2018). As for dumping sites with buried wastes, rapid and high contamination in the surrounding aquifer is often observed. The installation of permeable reactive barriers can relieve the groundwater contamination by adsorption and precipitation, which largely reduced metal concentrations in the outflow (Kumarasinghe et al., 2017).

Emerging mitigation and treatment approaches to overcome these problems are summarized in Table 3. From the perspective of sustainable waste management, waste separation and metal recovery are important to alleviate the negative effects from metals in the wastes. Pre-sorting prior to incineration can effectively decrease the metal content in source materials and increase the combustion efficiency as well as heat values for incineration. A system of source-classified collection of MSW was proved to decrease metal content from 1.22% to 0.17% and substantially reduced the leachate toxicity of bottom ash (Shi et al., 2018). Policies and regulations contribute significantly to the mitigation of PTEs in waste streams. The global Minamata Convention on Mercury since 2013 has facilitated the reduction of anthropogenic emissions and releases of mercury over the 140 countries in the world. For example, China has implemented several national plans for gradually phasing out the mercury-containing products from the markets by limiting mining activities and usage of mercury industrial productions (Hu et al., 2018).

In addition to mitigate PTEs contamination in the system design level, metal immobilization is also an essential solution if there is further reuse of such waste materials. Stabilisation (e.g., carbonation and chemical stabilisation) (Li et al., 2017) and extraction (e.g., acid washing and chemical extraction) are frequently used (Beiyuan et al., 2017a; Kirkelund and Jensen, 2018). As an example, for chemical extraction, As can be extracted from the solid waste by adding a weak acid salt such as ammonium acetate or oxalate or alkaline solutions (Beiyuan et al., 2017b; Kim et al., 2016). High liquid-to-solid ratio, pH, and oxidizing conditions are effective for facilitating metal depletion from fly ash (Weibel et al., 2017). Chemical leaching and metal recovery can be combined with waste-to-energy utilization. Metal desorption via electrodialytic treatment is also used for PTEs removal from incinerator fly ash, which employed ion exchange membrane to remove anions and cations from the ash suspension to the concentrate by the applied current (Kirkelund and Jensen, 2018). This process was pH dependent, where Cd was only desorbed at acidic conditions. Besides, biosorption involving chelation, ion exchange, complexion, adsorption, absorption or micro-precipitation can also act as a potential way for remediation. Indigenous fungi achieved a cadmium removal of 72.4% from municipal solid wastes leachate at spore concentration of 10-5, pH of 6.0 and temperature of 45 oC (Awasthi et al., 2017). Bioleaching or microbial reduction can be applied for As treatment using exogeneous bacteria, but the treatment cycle is long (Lee et al., 2015; Yamamura et al., 2008).

In recent years, the concept of green remediation is being extensively explored through stabilization/solidification, where contaminated wastes (such as ashes) can be utilized and recycled for secondary uses (Wang et al., 2018b; Wang et al., 2019). These practices effectively minimize the cost of waste disposal and resource extraction, thus sustainably reducing the environmental footprints and closing the waste loop. For example, synthesis of zeolite from coal combustion residues or incinerator sludge ash is considered as a safe application to immobilize toxic elements (Feng et al., 2018; Zhang et al., 2018a). Besides, diverse combustion ashes such as coal or incinerated ash as well as sewage sludge ash, can be utilized for blended cement or concrete block production (Wang et al., 2018c; Yin et al., 2018). Cement-solidified fly ash via natural ageing forms the high contents of carbonates and iron/aluminium hydroxides, which significantly controlled leaching of Cd, Cu, and Zn (Du et al., 2018). Recent research studied contaminated sediment and fly ash for producing non-load-bearing masonry units or fill materials for construction uses (Wang et al., 2015; Wang et al., 2018b). The microporous structure and high surface area of sediment blocks has effectively immobilized metals (such as Ni, Pb, Cu) by the C–S–H gel and hydration products with minimal leachability. However, in some cases, mobility of Cr, Cu, Pb and Zn would increase after cement stabilization due to dissolution of hydroxides in alkaline condition and presence of chloride, sulphate, and organic fractions (Yin et al., 2018). Cement-based materials can greatly stabilize the Pb-smelting slag, however, Pb may be released under weak acidic conditions (Angelis et al., 2012). When Pb-smelting slag is used as fine aggregate of road material, the mixture of asphalt and slag is formed and the shear resistance, drainage capacity, and percolation characteristics basically meet the requirements of environmental protection (Saikia et al., 2012).

As an alternative, recycled solid wastes can serve as sorbent material for metal sequestration in wastewater treatment. To utilize the high affinity of PTEs to organic matter, a study explored to use MSW organic windrow compost to remediate contaminated soil and achieved Cd (89.8%) and Zn (64.0%) adsorption capacities (Lima et al., 2018). Industrial wastes such as blast furnace slag, fly ash, and red mud are also studied for metal removal from aqueous solutions (Ahmed and Ahmaruzzaman, 2016). Biochar, as a waste biomass-derived material, is emerging as a low-cost adsorbent for removal of potentially toxic elements such as Cr, Cu, As, and Pb from wastewater and stormwater (Zhou et al., 2017; Mohanty et al., 2018; Rajapaksha et al., 2018). In addition to sludge-derived biochar which proved to stabilize metal-contaminated soils (Fang et al., 2016; Beiyuan et al., 2017c; Yoo et al., 2018), recent advances on tailoring biochar into functional carbon composites (Xiong et al., 2017; Sun et al., 2019) shed light on synthesis of novel and sustainable materials for environmental remediation.

4. Fate of radioactive elements and their treatment

4.1 Uranium

With the development and utilisation of nuclear energy, the uranium mining and metallurgy industries have developed rapidly. The mining and smelting of uranium have produced a large amount of uranium-containing waste rock and tailings, which have become the most serious source of uranium pollution. Currently, the world's accumulated more than 20 billion tonnes of uranium tailings and more than 40 billion tonnes of uranium waste rock. Uranium has caused relatively complicated radioactive pollution; as the grade of uranium ore continues to decrease, the amount of waste rock and tailings produced will increase (Bowker et al., 2007). Uranium tailings ponds are sites of recovery for slag and other low-level radioactive wastes, which contain a considerable amount of radionuclide uranium (Table 4). Thus far, 22 countries in the world are engaged in uranium mining and metallurgy production, and there are more than 4,000 uranium tailing ponds. Large uranium tailing ponds can store tens of millions of tonnes of uranium tailings, whereas small ponds can also store from dozens to millions of tonnes of uranium tailings. The country with the largest volume of uranium tailings in the world is Kazakhstan, accounting for 23% of the global total, followed by Germany (17%), Ukraine (14%), and the United States (13%) (Bourva et al., 2014). In China, uranium resources are mainly found in East, Central, and South China, accounting for approximately 80% of the total uranium production. Based on this distribution, 70%–80% of uranium resources are in humid and rainy hilly areas. Most uranium deposits are typically located in temperate monsoon and subtropical monsoon climates (Déjeant et al., 2014).

Solid waste containing uranium mainly exists in the form of uranium waste rock and uranium tailings, which are the main sources of uranium pollution. According to the pollution potential of the uranium tailings reservoir and the release potential of uranium from other pollution sources, uranium can be divided into the following six fractions: (1) exchangeable (water-soluble); (2) carbonate-bound; (3) organic matter-bound; (4) amorphous iron-manganese oxide/hydroxide-bound; (5) crystalline iron-manganese oxide/hydroxide-bound; and (6) residual (Liang et al., 2014). The exchangeable fraction of uranium refers to non-specifically adsorbed uranium on the surface of solid media by outer-sphere weak coordination. This form of uranium is most likely to be released into the environment through the effects of rainfall and surface runoff, presenting the greatest potential threat to the environment. The carbonate-bound fraction refers to uranium that is adsorbed on the surface of carbonates or that is co-precipitated. This form of uranium is highly susceptible to pH change and release into the environment, also posing a potential threat to the environment. Organic matter-bound uranium is chelated with organic matter, which can be released following the oxidative decomposition of organic matter. Amorphous iron-manganese oxide/hydroxide-bound fraction refers to uranium that is adsorbed or precipitated by an amorphous iron-manganese oxide/hydroxide component, which has poor stability and is prone to release under reducing conditions. Crystalline iron-manganese oxide/hydroxide-bound uranium refers to uranium that is adsorbed or co-precipitated by a crystalline iron-manganese oxide/hydroxide component, which is released only under strongly reducing conditions. Residual fraction refers to uranium that is firmly bound to primary or secondary silicate minerals. This form of uranium is not easily released under natural weathering conditions and can be stabilised for a long period of time without any significant impact on the environment.

A large number of uranium tailings may release pollution via rupturing of the conveying pipe, leakage from the tailings pond, and flooding of the tailings mine dam by heavy rain (Zhijie et al., 2014). In addition to the toxic effects of heavy metals, uranium also produces a certain amount of radioactivity. Since uranium tailings contain all the decaying bodies and uranium remaining after smelting, most 230Th and 226Ra is concentrated in the tailings. The radioactivity of uranium itself is not strong, but its half-life is long and produces decaying daughter products with high radioactivity. Therefore, the radioactive hazard persists for a considerable period of time after decommissioning of uranium tailings. The radioactive contamination of uranium tailings is not as easily detected as other types of pollution, and the sensitivity of different organisms to radioactivity varies widely. Singh et al. (2015a) studied the accumulation of major pollutants in a uranium tailings pond and the affected areas. Different plants grew under the same environment, while the biological concentrations and elemental accumulation in some plant species were more obvious than in others (Singh et al., 2015a). In general, human exposed to the same levels of radioactivity will be much more affected than other organisms. The latency of the carcinogenic effect of radiation is less than three years but can be more than 20 years. The genetic effects of radiation can also be delayed.

Uranium from the tailings migrates into the soil under natural conditions, causing its accumulation in the soil. The concentration of uranium ions in the soil decreases with increasing depth, and the affected depth does not generally exceed 30 cm. Uranium is transferred to plants via microbial activity. Different species of plants show large differences in their accumulation of radionuclides and heavy metals. There are also large differences in the enrichment of uranium in different parts of plants. For example, stems and leaves of plants are more enriched than seeds. When uranium accumulates to a certain concentration in plants, it will affect normal growth and can result in eventual death. More importantly, uranium accumulated in plants enters the human body through the food chain. In an experiment, three plants (corn, sunflower, and green pea) were grown in four matrix variants for 40 days (Santos et al., 2011). The results showed that the concentrations of U in the plants were dependent on plant growth; the concentrations of uranium in the substrates were important, and the uranium contents in the upper part and root of sunflower, corn, and pea were different. The concentrations of uranium in roots were significantly higher than in the above-ground biomass, and the uranium content in sunflower was higher than in corn and pea; the dry matter quantity of sunflower was the highest, followed by corn and green pea (Stojanovic et al., 2009). The uranyl ion (UO22+) that enters the human body can bind to the kidneys and other organs in combination with biomolecules, accumulates, and causes necrotic glomerular necrosis, tubular wall atrophy, and malignant bone and kidney lesions (i.e., tumours).

4.2. Radium

Naturally occurring radioactive radium may become concentrated in residues, waste materials, and end products of non-nuclear industrial activities such as fossil fuel burning. In particular, energy production from fossil fuels such as coal may lead to the enrichment of radium, which significantly increases human exposure to radiation. Coal-fired power plants generate large amounts of ash with naturally occurring radioactive radium (Figure 2). Radium-226 and Radium-228 are the main sources of low-dose ionizing radiation exposure in coal ash. Radium in coal ash would migrate to the soil under natural conditions, resulting in accumulation in the soil and subsequent absorption in plants. As an alkali-earth element with oxidative state of +2, radium has a higher solubility and is more biologically available to plants. Coal fly ash is widely used not only as supplementary material in concrete construction but also in road construction and asphalt, leading to a potential risk of radioactive contamination. All isotopes of radium are highly radioactive, with the most stable isotope being radium-226, which has a half-life of 1,600 years. Radium-228 is also a stable isotope, with a half-life of 5.75 years. It was estimated that over 130 million tonnes of coal combustion residues are generated in US power plants every year. Globally, coal-based power generation produces over 750 million tonnes of coal ashes per year. It was estimated that all forms of released radium-226 amount to approximately 1.55 TBq y-1, with gross particulate dispersion rates of 7.4 GBq y-1 (18.5 MBq d-1) (Laura et al., 2009).

For coal ashes, based on the modified sequential extraction method, radium can be divided into the following six fractions: (1) water-soluble; (2) ion-exchangeable; (3) carbonate-bound; (4) free oxide; (5) sulphide-bound; and (6) residual. Approximately 74% of Radium-226 exists in the form of residues, a certain amount exists in the carbonate-bound fraction, a small amount in the water-soluble, free oxide, sulphide-bound fractions, and almost none in ion-exchangeable fraction. Radium-228 also primarily exists in the form of residue (approximately 48%). There is a moderate amount in sulphide-bound and free oxide fractions, a small amount in carbonate-bound fraction, but none in water-soluble and ion-exchangeable fractions. Although some fractions of the isotopes also occur in association with vertical migration and root uptake, these nuclides are generally associated with the soil material. Radioactivity decreases with an increase in soil pH value. The highest activities of natural 226Ra have been found in soils where carbonates have been leached, while lower radioactivity has been found in soils with the highest content of carbonates (Xin et al., 2015).

The release of coal ashes with high levels of radioactivity into the environment has the potential to generate resuspended ambient fine particles (< 10 µm). Inhalation of fly ash emitted from the stacks and through the ingestion of foodstuffs receiving atmospheric deposition are the main pathways of human exposure to radiation. Leaching of contaminants from coal ashes causes heavy contamination of surface waters in areas of restricted water exchange, but only trace levels are found downstream due to dilution. However, the accumulation of coal ash in river sediments has a potential impact on downstream ecology through fish poisoning in anaerobic river sediments. Similarly, the accumulation of radioactive material in the ground near to coal-fired plants also results in higher levels of radioactivity in soils compared to soil located further away. This affects plant growth and possibly further enhances human uptake through the food chain. Emissions from coal-fired power plants in gaseous and particulate forms containing nuclides probably accumulate in the surrounding environment, thus exposing local populations to radiation (Bozena et al., 2017).

4.3. Potassium

Potassium is an active reducing agent, which is easily oxidized to +1 valence of stable state. Since potassium has a high hydration energy, most potassium compounds are soluble in water. Naturally occurring potassium (K) is composed of three isotopes, namely K-39 (93.258%), K-40 (0.012%), K-41 (6.730%), of which K-40 is radioactive (Meija et al., 2016). Plants cannot distinguish isotopes similar in chemical properties when absorbing essential nutrients, so K-40 will be untaken by the plant's roots along with non-radioactive potassium and distributed throughout the plant (James et al., 2011). The soil-to-plant transfer factor of potassium in leaves was in the range of 0.49-5.6 with a mean of 1.4 (IAEA, 2010). Leached radioactive material may directly or indirectly enter human body through drinking water and food, and result in radiation contamination of terrestrial and freshwater ecosystems (Aliyu et al., 2015). In fact, potassium is an essential element for human and its average concentration in a male adult reaches approximately 2 g per kg of body weight (Gautam et al., 2015). Most of the internal radiation dose received by human is attributed to the consumption of food contaminated with different radionuclides (Khan et al., 2010). The total ingestion dose from natural radioactivity is about 0.21 mSv/y, of which more than 90% is attributed to K-40 (Verma, 1999). Potassium-40 is uniformly distributed throughout the tissues and organs of the human body, with a biological half-life of 30 days (Rabitsch and Pichl, 2008). The potassium content of the body is constant and is not affected by variations in environmental levels due to homeostatic control.

With a half-life of 1.248 × 109 years, K-40 can stably exist in the natural environment. It is, therefore, a major source of natural background radiation. There is no commercial or medical use of potassium radionuclides, however, K-40 is widely present in various types of solid wastes, mainly due to industrial production and construction activities (Sas et al., 2017; Schroeyers et al., 2018). Only a very small proportion of solid waste has a concentration of K-40 that exceeds the suggested limit of 10,000 Bq/kg (IAEA, 2014). This portion of solid waste is categorised as “technologically enhanced naturally occurring radioactive material” (TENORM), i.e., naturally occurring radioactive materials that have been concentrated or exposed to the environment as a result of human activities such as manufacturing, mineral extraction, and water processing.

Solid wastes containing K-40 are mainly derived from two sources — oil and gas resources exploitation and thermal power generation. In the process of oil and gas resource exploitation, radioactive potassium is gradually released from the rock matrix and is pumped to the surface with water, this finally precipitates as scale and sludge (Gazineu and Hazin, 2008). The largest volumes of scale occur in three areas: (1) water lines associated with separators; (2) heater treaters and gas dehydrators; and (3) oil stock and water storage tanks. Sludge mainly occurs in tanks, oil and gas separators, dehydrators, storage tanks, and at the bottom of slush pits. According to the US EPA, oil production processes used in conventional drilling generate an estimated 230,000 MT or 141 m3 of TENORM sludge each year. The global average activity concentration of K-40 in soil is 412 Bq/kg, and in the Earth’s crust is 750 Bq/kg (UNSCEAR, 2008). The activity concentrations of solid waste generated by the oil and gas industry fluctuate depending on geological factors. For example, in Brazil, the maximum radioactivity of K-40 in scales and sludge reached 144.6 kBq/kg and 53.8 kBq/kg, respectively, while the lowest values could not be detected in the states of Sergipe and Alagoas (Gazineu and Hazin, 2008).

4.4. Treatment of radioactive elements

In summary, the main technologies for controlling radioactive elements are S/S method, backfilling method, containment method, biochemical repair method, and integration of the above. For S/S method, cementitious materials could be employed as agents for the immobilisation of radioactive elements. Supplementary cementitious materials such as fly ash, silica fume, calcined clay were used to reduce carbon footprint and improve the S/S efficiency for radioactive wastes. Utilization of tailings or waste rock containing radioactive elements as backfilling material is also an option. It can not only reduce the cost of using other materials, but also reduce the accumulation of radioactive waste on the ground which expose harms to people. Nevertheless, backfilling method is more likely to contaminate the groundwater body than other treatment methods (Mishra et al., 2014). Containment by covering the tailings can also prevent pollution. It has a wide application range and is suitable for long-term treatment of radioactive tailings (Tan et al., 2012). However, there are problems associated with ecological restoration of the surface and potential harm to animals and plants. Biochemical repair of radioactive waste is a method of adding microorganisms and chemicals (e.g., carbonate/bicarbonate) or planting plants in the tailings (Santos et al., 2011). Choosing suitable plants and harvesting the above-ground parts with excessive accumulation characteristics of specific nuclide relieve the radioactive pollution in the soil (Li et al., 2011). Unlike the backfilling method and containment method, the biochemical repair method removes the contaminant instead of isolating it. In an integrated approach, the synergistic effects of different methods improve the ability to remove/stabilise harmful substances and reduce long-term hazard.

5. Fate of rare earth elements (REEs) and their treatment

5. 1. Natural existence of REEs in mineral deposits

Rare earth elements (REEs), as defined by the International Union for Pure and Applied Chemistry (IUPAC), incorporate a group of 17 elements consisting of 15 lanthanides as well as yttrium and scandium, covering Ce, Dy, Er, Eu, Gd, Ho, La, Lu, Nd, Pr, Pm, Sm, Sc, Tb, Lu, Tm, Yb, and Y. The term “REY” is sometimes used to denote rare earth elements plus yttrium. REEs are usually divided into the light rare earth elements (LREEs), including La through Sm, and heavy rare earth elements (HREEs), including Eu through Lu. LREEs are abundant in bastnäsite [(Ce,La)CO3(F,OH)] and monazite [(Ce,La,Nd,Th)PO4], while HREEs are usually present in apatite, cheralite, eudialyte, loparite, phosphorites, rare-earth-bearing (ion adsorption) clays, secondary monazite, spent uranium solutions, and xenotime (Duta et al., 2016). Their states vary from REY-bearing minerals and glass to fine nanoparticles embedded within the amorphous carbon surrounding the glassy particles (Hood et al., 2017). HREEs are preferentially desorbed by acidic solutions, have a stronger organic affinity, and present more stability in their complex forms than LREEs (Hower et al., 2016). Furthermore, based on the abundance and industrial demand, REEs can be classified as critical (Y, Nd, Eu, Tb, Dy, and Er), uncritical (La, Pr, Sm, and Gd), and excessive (Ce, Ho, Tm, Yb, and Lu) (Hower et al., 2016).

Their unique structures (4f orbitals) and distinctive properties render REEs critical for various industrial applications, including uses in superconductors, photonic device permanent magnet materials, etc. (Chen et al., 2019). Moreover, REEs show great prospects for sustainable energy fields such as wind power, electric vehicles, and energy-efficient lighting (Tan et al., 2017). REEs show a scattered geological distribution with generally low abundance, unevenly distributed around the world. It is estimated that the total demand for REEs will grow by over 50% by 2020 (UNEP, 2009). As the largest export source of REEs, China is meeting more than 90% of the global REE demand, and this monopoly is prompting other countries to develop techniques for identifying and recovering REEs. Recent studies on the fate of REEs in solid wastes are summarised in Table 6.

Conventional mining includes ore deposits, grinding, and cracking of minerals to produce mixed rare earth oxide concentrates, separation and purification of oxides, and refining to produce industrial grade REEs (Duta et al., 2016). Conventional REEs production requires high temperature and stringent treatment conditions, consuming a high amount of energy as well as generating significant volumes of toxic wastes and acidic wastewater. Moreover, a high content of thorium and uranium in REEs-bearing ores (up to 10%) may result in the emission of radioactive wastes during REEs mining. This appeals for comprehensive disposal regulations (Fathollahzadeh et al., 2018a). Exploitation for REEs is challenging as REE-containing minerals seldom occur in concentrated forms but are usually present together in the same deposits (Tunsu et al., 2015). Similar chemical properties of REEs further add to difficulties in their separation and purification.

That said, the chemical behaviours of sub-structures, known as the tetrad-effect, are utilised in the recovery of REEs in groups of adjacent elements. This is because the stabilisation energy related to the inter-electron repulsion energy of the electrons can cause the ionic radii to decrease within the lanthanide series, thus resulting in variation in the strength of the complexes that REEs form in aqueous solutions. Hence, heavier REEs create stronger complexes with extractant molecules compared to lighter REEs, while yttrium complexes have an ionic radius comparable to heavy lanthanoids. Nevertheless, it is still challenging to accomplish complete separation of adjacent elements on a large scale (Tunsu et al., 2015). Biotechnology is currently emerging as an environmental alternative for REEs extraction from minerals, with the release of less hazardous waste. Bioleaching of monazite has recently been reported using a combination of heterotrophic and autotrophic acidophilic microorganisms, which solubilised REEs (Ce, La, Nd, Pr, and Y) up to a final concentration of 40 mg/L (Fathollahzadeh et al., 2018b).

5. 2 Recycling from coal combustion residues

As REEs ore deposits are often laced with radioactive elements such as uranium and thorium, extraction would cause radioactive stockpiles. Other environmental impacts include soil erosion, biodiversity loss, land use change, flooding; pollution of air, soil, and water; and crop uptake of REEs. All pose severe threats to human health. As a result, waste-derived sources of REEs are attracting attention as an alternative approach. Coal fly ash and other combustion/incineration residues serve as promising secondary sources for REEs recovery, showing both economic feasibility and environmental sustainability. The content depends on the characteristics of coal. For example, more than 1,000 mg/kg of REEs has been reported in heating plant stoker ash from Kentucky State University, while up to 1,220 mg/kg of REEs was found in stoker ash from a Kentucky power plant burning Appalachian Basin coal (Taggart et al., 2016; Hower et al., 2018). Nitric acid digestion is generally used for REE extraction. Research has shown that pretreatment with an alkaline agent (e.g., lime roasting) tended to liberate REEs without resorting to hydrogen fluoride digestion, which mitigates environmental hazard as well as increasing REEs extractability (Taggart et al., 2016). Bioleaching using H2SO4 and mixed culture of acidophilic bacteria achieved recovery rates of 76% Nd, 52% Sb, and 48% Ce from MSW incineration fly ash (Funari et al., 2016).

5.3. Recycling from other end-of-life products

Urban mining is defined as the "systematic management of anthropogenic resources stocks and waste (products and buildings), in the view of long-term environmental protection, resource conservation, and economic benefits” (Cossu, 2013). As such, urban mining is emerging as an approach to REEs recycling from e-waste and other wastes streams including construction and demolition wastes, combustion residues, sludge, exhausted oil, previously landfilled wastes, and industrial residues among others (Tunsu et al., 2015).

Notably high concentrations of REEs were detected in the surface waters of the San Francisco Bay area, presumably due to the discharge of refractory magnetic resonance imaging contrast agents (used in hospitals and medical research centres) via wastewater treatment plants. Urban mining might be a useful means of tackling some of the undesired effects of REEs discharge. Unlike the primary REEs ores, recycled REEs material does not contain radioactive substances, implying that health problems can be partially mitigated (Duta et al., 2016). In construction and demolition waste management, up to 100% metal recycling performance is expected for the potential closed-loop waste lifecycle (Kucukvar et al., 2016). Urban mining recovery of technology-critical elements from waste sources such as spent batteries, mobile phones, and magnet motors is a promising strategy to relieve pressure on both resource depletion and environmental pollution (Zuo et al., 2019). Future perspectives may focus on overcoming disadvantages such as collection and dismantling requirements, difficulty of separation by conventional milling operations, and matrix heterogeneity (Tunsu et al., 2015). Furthermore, the adoption of geographic information systems for a comprehensive overview may offer a potential means of developing urban mining (Koutamanis et al., 2018).

Currently, some chemical and electrical companies (e.g., Umicore and Solvay Rhodia, Mitsubishi Electric, and Veolia Environmental Services) are making efforts to develop REEs waste recycling processes (Duta et al., 2016). Permanent magnets are an important means of recycling REEs. Conventional hydrometallurgy employs chemicals such as strong mineral acids, where magnet alloys are dissolved and the REEs are selectively precipitated with sulphates, oxalates, or fluorides. In contrast, a pyrometallurgical approach requires high-temperature routes and includes electroslag refining, liquid metal extraction, glass slag method, direct melting, and gas-phase extraction methods

(Binnemans et al., 2015). It is estimated that REEs constitute over 23 wt% of phosphors in fluorescent lamps (Tan et al., 2017). Innovative use of mechanical activation as a pretreatment for the recovery of REEs from waste phosphors has increased the leaching rates of terbium (Tb), lanthanum (La), and cerium (Ce), achieving a significant increase to approximately 90% (Tan et al., 2017). Recovery of REEs from other wastes, such as nickel-metal-hydride batteries and lamp phosphors, is technically feasible (Binnemans et al., 2013). For example, ceria can be separated and extracted from glass polishing powder wastes; surfactant collector (sodium oleate) at a neutral pH recovered up to 97.1% from a ceria-based abrasive (Wang et al., 2018b).

In addition to conventional hydrometallurgical or pyrometallurgical processes, biotechnologies are emerging as promising alternatives that have less environmental impacts, greater practicality with respect to operation, and higher cost-effectiveness (Işıldar et al., 2019). Biologically induced leaching (bioleaching) from various matrices, biomass-induced sorption (biosorption), and bioelectrochemical systems are being explored for REEs recovery from WEEEs. For example, a bioleaching process employing acidophilic and cyanogenic bacterial strains achieved lanthanum and yttrium extraction at a yield of 80% from shredded WEEE material (Marra et al., 2017).

6. Prospects and future research

As improper storage and disposal of potentially toxic elements-containing solid wastes could lead to severe degradation of the environment and deleterious consequences to human health, strengthening waste management systems, as well as developing alternative techniques in recycling valuable resources, are required to tackle this problem. With regard to waste streams, measures could be implemented from source reduction, contaminant isolation, resource recycling, and waste stabilization. Therefore, an integrated and sustainable waste management system is recommended, to maximize resource utilization efficiency and minimize the impact on the environment. Research into more effective contamination mitigation measures during waste treatment (such as air pollution control devices for incineration and permeable reactive barrier for landfill) are encouraged, while more advanced techniques are urged to stabilize or safely dispose of these secondary products (e.g., incineration ashes and combustion residues).

Although numerous treatment methods have emerged in recent years, environmental impact and economical cost may still be the major concerns. The prospect for future development should focus on green and sustainable remediation, where implementation is incorporated into sustainable strategies, to restore contaminated solid wastes for productive use, with reduced associated costs and environmental burdens. Green and sustainable remediation also help formulate strategies in exploring the relationships among regulation, economics, and technology. Besides, prior to final waste disposal, the technology-critical, radioactive, and rare earth elements should be prioritised for resource recovery. The development in both technology and management/policy aspects are necessary for meeting the United Nation’s Sustainable Development Goals.

In addition, the valorisation or re-exploration of valuable resources from existing waste is attracting more attention. Some innovative waste recycling approaches, such as landfill mining (Burlakovs et al., 2018) and sustainable biorefinery from waste biomass (Chen et al., 2017; Cao et al., 2018; Yu et al,. 2019), are now emerging to substitute conventional resource exploration and chemical production. In this regard, prudent safety assessment and measures for landfill mining and industrial operation are in need to avoid possible potentially toxic elements leaching to water streams or aerosols during excavation, sorting/separation, and extraction/conversion. Life cycle assessment and techno-economic investigations into various approaches are needed to reduce economic cost, decrease environmental impacts, and devise sustainable strategies for the management of potentially toxic elements-containing solid wastes.

7. Conclusions

This paper critically reviews the fate, risks, and management approaches of potentially toxic elements in solid waste streams. It summarises the flows and contents of potentially toxic elements in three typical solid wastes, specifically municipal solid wastes, electronic wastes, and industrial solid wastes. In particular, toxic metal(loid)s, radioactive elements, and rare earth elements are critically reviewed with respect to their patterns of transport, transformation, risks in a changing environment, as well as their corresponding management approaches.

It should be recognised that municipal solid waste is one of the largest components of waste streams. Incineration and landfill are two major treatment methods for municipal solid waste. For incineration of municipal solid waste, the potentially toxic elements in the waste are concentrated in fly ash and bottom ash, whereas for landfill method, the concentrations of the toxic materials also increased in landfill sites after waste degradation, especially in closed landfill sites. Electronic waste has drawn increasing attention due to the short lifespan of electrical and electronic products. The potentially toxic elements in e-waste are not reduced with other organic wastes during landfilling, but instead, are accumulated, transported, and diffused into soil pore water and thereby threaten the soil ecosystem. Industrial solid wastes derived from numerous manufacturing and processing activities are also critical sources of potentially toxic elements. Industrial solid wastes usually contain more homogeneous forms of potentially toxic elements albeit at high concentrations. Therefore, specific treatment methods should be designed for different industrial solid wastes.

The commonly encountered toxic metals, chromium, arsenic, selenium, cadmium, mercury, and lead are comprehensively reviewed in this paper. These elements have similar but distinctive physical, chemical, and toxic characteristics. Different treatments and recycling methods for toxic metals are discussed, including metal immobilisation, chemical extraction, electrochemical treatment, biosorption, and transformation into valuable materials. In addition, radioactive wastes including uranium, radium, thorium, and potassium have become one of the most serious sources of pollution for ecosystems. To tackle the grave environmental risks of radioactive wastes, stabilisation/solidification, backfilling, containment, and biotechnology methods as mainstream technologies are used, whereas recycling methods are not yet mature. For radioactive tailings with relatively low risks, biochemical repair, phytoremediation methods, and other comprehensive methods can be potential treatment technologies. Rare earth elements are mainly produced from natural rare earth ore and industrial wastes (e.g., coal combustion residues and industrial sewage sludge). However, rare earth element mining and extraction processes are usually accompanied by emission of radioactive wastes, which leads to potential environmental risks. As rare earth elements are valuable materials, urban mining is emerging as a means of recycling rare earth elements from solid wastes rather than disposing of them.

Based on a holistic understanding of the dynamic fate of potentially toxic elements and a review of existing management options, the development of an integrated and sustainable waste management system is required for resource recovery, waste minimisation, and environmental decontamination. It is imperative to develop novel and advanced treatment technologies for achieving high-efficiency and cost-effective potentially toxic elements removal, recycling, and/or immobilisation methods for different solid wastes. Furthermore, life-cycle analysis and cost-benefit assessment can advise further improvements in energy consumption and the associated carbon footprint. Sustainable remediation, low-carbon treatments, and resource recovery of solid wastes are recommended for meeting the sustainable development goals.

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