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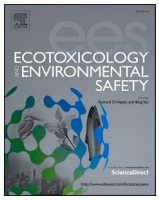

Review

A review on disposal and utilization of phytoremediation plants containing heavy metals

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ABSTRACT

The reasonable disposal of plant biomass containing heavy metals (HMs) is a difficult problem for the phytoremediation technology. This review summarizes current literature that introduces various disposal and utilization methods (heat treatment, extraction treatment, microbial treatment, compression landfill, and synthesis of nanomaterials) for phytoremediation plants with HMs. The operation process and technical parameters of each disposal method are different. HMs can migrate and transform in different disposal processes. Some disposal and utilization methods can get some by-products. The main purpose of this paper is to provide reference for technical parameters and characteristics of various disposal and utilization methods, so as to choose and use the appropriate method for the treatment of plant biomass containing HMs after phytoremediation.

1. Introduction

Heavy metal (HM) pollution of soil and water is a worldwide concern because of its harmful effect on human health. Currently, remediation technologies based on physical, chemical and biological approaches are widely used to remove HMs from contaminated soil and water (Hu and Zhang, 2012; Khalid et al., 2017). Phytoremediation, which uses green plants to remove HMs from soil, sediments and water, attracted a lot of attention from scientists and engineers worldwide during recent years. The advantages of phytoremediation compared with traditional physical and chemical remediation methods are low cost, simplicity and environmental friendliness (Lee et al., 2017; Liu et al., 2020a, 2020b). Types of phytoremediation include phytoextraction, phytovolatilization, phytofiltration, phytostabilization, phytodegradation, and rhizosphere bioremediation (Yadav et al., 2018). The HMs and plant species often used in phytoremediation are listed in Table S1. Traditional phytoremediation techniques often require hyperaccumulators and increasing numbers of crop cycles, which is applicability for low/moderately contaminated sites (Khalid et al., 2017). In recent years, many researchers have utilized assisted measures such as adding plant growth agents and microorganisms, HM accelerators, and transgenic plants, which can significantly and positively enhance overall phytoremediation (Mesa et al., 2015; Kuan et al., 2016; Park et al., 2017; Kumar et al., 2017; Wang et al., 2017b, 2017c, 2017a; Liu et al., 2018a; Arnao and Hernández-Ruiz, 2018; Fasani et al., 2018; Uraguchi et al., 2019; Mousavi et al., 2021). Some field applications of phytoremediation technology have been implemented in the past 20 years. Yin et al. (2014) used *Solanum nigrum* L. in the phytoremediation of cadmium (Cd) contaminated field soils (Shenyang Zhangshi Cd sewage irrigation area in China) and the field experiment period was 2 years. *Solanum nigrum* L. had an obvious effect on the removal of Cd in the surface and subsurface of field soils, and it had a good effect on the remediation of light and moderate Cd contaminated field soils (Yin et al., 2014). Ma (2015) used *Ricinus communis* L. and *Helianthus annuus* L. in the phytoremediation of Cd and zinc (Zn) contaminated field soils (Shanghai Pudong New Area factory relocation site in China) (Ma, 2015). *Ricinus communis* L. and *Helianthus annuus* L. had an obvious effect on the removal of Cd and Zn in the field soils (Ma, 2015). Zhou (2016) used *Pennisetum giganteum* in the assisted phytoremediation of Cd and copper (Cu) contaminated field soils (surrounding area of Guixi Smelter of Jiangxi Cu Group in China) and the site area was about 1.3 km^2 (Zhou, 2016). *Pennisetum giganteum* could absorb and transfer 454.3 g Cu and 9.5 g Cd in about 666.7 m^2 soils each year (Zhou, 2016). Two field surveys (2-ha survey and 8-ha survey in Portugal and Russia)

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over 2 years were carried out by Kikuchi et al. (2011) in order to understand the multiple-metal (Cu, lead (Pb), nickel (Ni)) effect on phytoremediation. Both field tests in Portugal and Russia suggested that the efficiency of phytoremediation was high and the root system was more important than the leaf system in the evaluation of remediation efficiency (Kikuchi et al., 2011). Zhang (2019) used *Suaeda Salsa* in the phytoremediation of Cu, Zn, arsenic (As) and chromium (Cr) contaminated field soils (Qinhuangdao wetland in China). The content of extractable HMs (Cu, Zn, As and Cr) decreased in soils after the field experiment (Zhang, 2019). At present, there are few field applications of phytoremediation for mercury (Hg) pollution.

Phytoremediation involves contaminant accumulation in the crop biomass during plant growth (Attinti et al., 2017). Harvesting these plants creates highly contaminated bio-waste (Gong et al., 2018), which could easily become a secondary pollution source if mishandled. The bio-waste is hazardous waste. Thus, appropriate disposal and utilization methods for such biowaste are required. Therefore, this paper reviews published reports discussing various disposal and utilization methods (heat treatment, extraction treatment, microbial treatment, compression landfill, and synthesis of nanomaterials) for phytoremediation plants with HMs. Our main goal is to provide a summary, which could be used for reference purposes and describes the advantages and disadvantages of various disposal and utilization steps following the phytoremediation process.

2. Heat treatment

Heat treatment is the most commonly method using heat energy to dispose plants contaminated by HMs and includes incineration, pyrolysis, and gasification, as well as hydrothermal carbonization, liquefaction and gasification (Cui et al., 2021; Fu et al., 2021). The technical parameters of these methods differ slightly, as shown in Table 1. The incineration, pyrolysis, and gasification temperatures are typically higher than hydrothermal carbonization, liquefaction, and gasification, which require a pressurized environment (Table 1). Hydrothermal reactions often require higher pressure compared to incineration, pyrolysis, and gasification (Table 1). Pyrolysis is conducted in an inert atmosphere, but incineration and gasification is performed under oxidative conditions (Cui et al., 2021). The medium of hydrothermal carbonization and hydrothermal liquefaction is subcritical water, however, The medium of hydrothermal gasification is supercritical water (Table 1). Hydrothermal carbonization usually needs longest heating time among all heat treatment methods (Table 1).

Table 1
Technical parameters of various heat treatment methods.

Heat treatment method	Plants with HMs	Device	Heating temperature (°C)	Heating time (min)	Medium	Gas flow (L/min)	References
Incineration	*Sedum plumbizincicola* (Zn, Pb, Cd)	Tube furnace	350–950	/	O_2	0.5–1.2	Zhong et al. (2015)
	Leersia hexandra Swartz (Cr)	Tube furnace	350–800	15–45	O_2	0.12	Wen et al. (2018)
	Ryegrass (Cd, Pb, Zn)	Tube furnace	600–900	10	O_2	3.33	Zhu et al. (2019)
	Pteris vittata L. (As)	Incinerator	850	/	O_2	/	Lei et al. (2019)
Pyrolysis	*Sedum plumbizincicola* (Zn, Cd, Pb)	Tube furnace	450–750	/	N_2	/	Zhong et al. (2016)
	Pteris vittata L. (As)	Tube furnace	400–900	30	N_2	0.3	Duan et al. (2017)
	Arundo donax (As, Cd, Pb)	Muffle oven	250–600	30–180	O_2-limited	/	Liu et al. (2017a), (2017b)
	Leersia hexandra Swartz (Cr)	Tube furnace	350–800	15–45	N_2	0.12	Wen et al. (2018)
	Brassica juncea (Zn, Pb, Cd)	Muffle oven	350–750	60	N_2	0.2	Huang et al. (2018)
	Ceratophyllum demersum L., *Myriophyllum verticillatum* L., *Hydrocotyle vulgaris*, *and Oenanthe javanica* (Bl.) DC. (Cu, Cd)	Muffle oven	350	60–180	N_2	0.2	Liu et al. (2019)
	Willow (Cu, Ni)	Vertical solar furnace	600–1600	5	Ar	9	Zeng et al. (2019)
Gasification	*Pteris vittata* L. (As)	Tube furnace	400–900	30	CO_2	0.3	Duan et al. (2017)
	Sedum alfredii (Zn, Cd, Pb)	Fixed bed gasifier	300–900	/	Air, CO_2, N_2	0.3	Cui et al. (2018)
	Sedum plumbizincicola (Zn, Cd, Pb), *Pteris vittata* L. (As, Pb)	Customized microwave oven	700–900	40	Steam	0.4×10^{-3}-10^{-3}	Zhang et al. (2019)
Hydrothermal carbonization	Wheat straw (with no HMs)	High pressure reactor	190–250	60–360	Subcritical water	Saturated pressure	Funke et al. (2013)
	Rice straw (with no HMs)	High pressure reactor	200	240–840	Subcritical water	Saturated pressure	Dai et al. (2014)
	Hydrocotyle verticillata, *Myriophyllum spicatum*, *Canna indica* (with no HMs)	High pressure reactor	200–260	120	Subcritical water	Saturated pressure	Cui et al. (2020)
	Helianthus annuus (Cd, Cu, Ni, Pb, Zn)	High pressure reactor	160–260	30	Subcritical water	Saturated pressure	Lee and Park (2021)
Hydrothermal liquefaction	*Phytolacca americana* L. (Mn, Cr, Zn)	High pressure reactor	320–380	30	Subcritical water	High pressure	Deng et al. (2014)
	Sedum plumbizincicola (Zn)	High pressure reactor	190–310	120	Subcritical water	High pressure	Qian et al. (2018)
Hydrothermal gasification	Timothy grass (with no HMs)	High pressure reactor	450–650	15–45	Supercritical water	High pressure	Nanda et al. (2016)
	Pteris vittata L. (As, Cd, Pb, Zn)	High pressure reactor	395–445	10–40	Supercritical water	High pressure	Li et al. (2018b)

2.1. Incineration

Incineration involves the high-pressure oxidation and combustion reaction between forced air and phytoremediation residues containing HMs. The biomass of these plants will be concentrated in the incineration residue (ash, etc.) (Fig. 1). Thus, the biomass volume will be reduced (Samolada and Zabaniotou, 2014), which is very beneficial for the transportation and storage aspect of phytoremediation. Lei et al. (2019) reported that total weight loss of *Pteris vittata* L. after incarnation was over 94%. Wen et al. (2018) observed 90.6% weight loss after incinerating *Leersia hexandra Swartz*. Zhong et al. (2015) demonstrated 89.4% weight loss for incinerated *Sedum plumbizincicola*. All these data revealed that different kinds of plants containing HMs lose their weight differently upon incineration. Yet, incineration temperature also affects plant weight loss. The incineration of plants could be divided into three stages (Zhong et al., 2015; Fig. 1). During the first stage (which spans from room temperature to 185 °C), water evaporates from the plants completely (Zhong et al., 2015). The second stage is from 185° to 400°C and is characterized by the decomposition of cellulose and hemicellulose, which decreases the plant weight significantly (Zhong et al., 2015). During the third stage (typically in the 400–697 °C range), the carbon in the plants decomposes further; however, the weight loss at this stage slows down (Zhong et al., 2015). Additionally, the heat released during the plant incineration can be utilized for other applications (e.g. power generation) as well.

When phytoremediation residues are burned, fly ash containing HMs, CO, NO_x and other pollutants are discharged, which could cause secondary pollution if not caught or handled properly (Wang et al., 2021). HMs entering the combustion chamber could either (1) remain in the combustion chamber as the bottom ash, (2) form solid particles in the flue gas (also known as fly ash) (3) as well as in the exhaust, also known as flue gas (Kovacs and Szemmelveisz, 2017). Wen et al. (2018) reported that the content of Cr in the bottom ash was > 90% after *Leersia hexandra Swartz* containing Cr was incinerated at 350 °C. At the same time, Cr contents in the bottom ash, fly ash, and flue gas were equal to 77.6%, 11.2% and 11.2% after 45 min incineration at 800 °C according to Wen et al. (2018), who also suggested maintaining the burning temperature below 800 °C to avoid the Cr emission into the atmosphere. Zhong et al. (2015) found ~70% of initial Cd present in the *Sedum plumbizincicola* in the fly ash after burning it at 350 °C. Later, Zhu et al. (2019) demonstrated a 64.4% Cd recovery rate in bottom ash after the ryegrass was incinerated at 675 °C. Zhong et al. (2015) were able to volatilize Pb and Zn from *Sedum plumbizincicola* above 650 °C, followed by their condensation below 650 °C. Lei et al. (2019) found As in the bottom ash after burning *Pteris vittata* L. containing As at 850 °C. However, this amount accounted for only 9.0–23.5% of the initial As content in the biomass feedstock (Lei et al., 2019). Typically, the higher the temperature, the higher the HM vaporization rate is, and more metals concentrate in fly ash and flue gas, contents of which correlate with the HM volatilization behavior (Zhong et al., 2015). The metal contents in fly ash typically exceed legally allowable limits, making fly ash less suitable for the consequent reuse (Bonanno et al., 2013). The metals from the compounds remaining in the bottom ash could be easily leached, limiting the bottom ash reuse (Wang et al., 2021). In fact, weak mobility was observed for the Cd, Zn, and Pb left in the bottom (Zhong et al., 2015). Zhu et al. (2019) reported that higher incineration temperatures are beneficial to reduce Cd leaching capacity. Bonanno et al. (2013) reported that the metal contents in the bottom ash were 1.5–3 times higher than in the *Phragmites australis* and *Arundo donax* tissues. However, metal-enriched ash showed significantly lower metal contents than the legal limits in ash for its reutilization in agriculture and forestry. Thus, this biomass ash could be considered a fertilizer rather than hazardous waste (Bonanno et al., 2013).

Some additives can prevent HMs in the phytoremediation residues from being transferred into the gas phase (Wu et al., 2013; Wen et al., 2018; Zhu et al., 2019; Lei et al., 2019). Wu et al. (2013) reported that 91.2% of Cd and 88.1% of Zn were removed by kaolin from the gas phase while 99.1% Zn, 97.6% Cd and over 99% of polycyclic aromatic hydrocarbons were removed from the flue gas by activated carbon. The kaolin presence can also reduce NO_x content in the flue gas to the levels established by the Chinese emission standards (Wu et al., 2013). Wen et al. (2018) added zeolite and CaO to the *Leersia hexandra Swartz* and

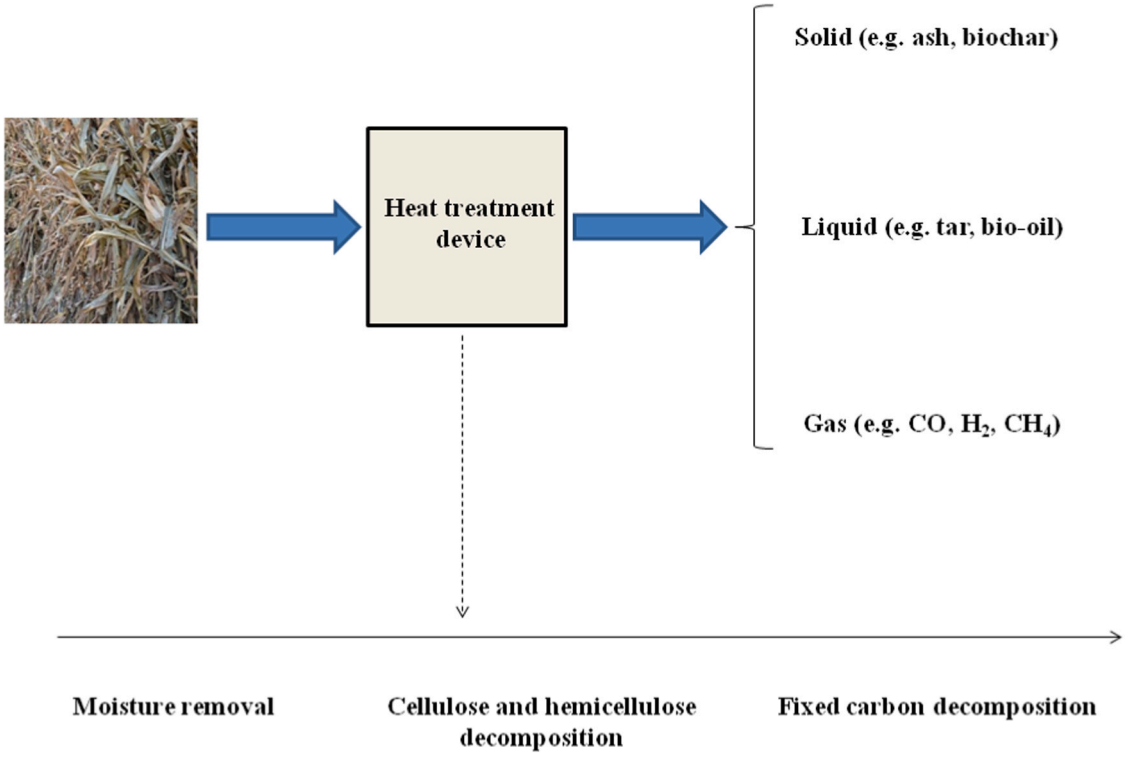

Fig. 1. Heat treatment process.

then incinerated it for 30 min at 800 °C. The Cr recovery rate in bottom ash increased by 12.2% and 8.8%, respectively, relative to the plant without additives. The addition of kaolin, montmorillonite, and γ-Al_2O_3 provides active Si- or Al-containing groups capable of binding Cd and, thus, preventing its evaporation (Zhu et al., 2019), which also reduces Cd concentration in the leaching solution. Montmorillonite was found to be more efficient in Cd retention than kaolin above 900 °C. The combustion trials with CaO addition demonstrated that the retained As contents in the bottom ash increased by 21.6–39.2% (Lei et al., 2019). Jagodzińska et al. (2019) reported that ammonium sulfate was effective in Cr, Cu and Hg capture while halloysite bonded with Cd, cobalt (Co), vanadium (V) and manganese (Mn). Kaolinite was the most effective in binding with Pb.

The primary mechanism of these additives in reducing fly ash amounts and inhibiting HM volatilization during incineration is governed by the physical and chemical adsorption of the HMs (Wen et al., 2018). At high temperatures, HMs react with these additives forming nonvolatile substances (Wang et al., 2012). For example, zeolite bonded Zn and Cr into nonvolatile silicates and aluminosilicates (Wang et al., 2012). A eutectic melting on the additive surface allows the HMs to diffuse into the structure of the additives (Gale and Wendt, 2005). Interaction of As with CaO occurs through physical adsorption; however, it results in an irreversible chemical reaction that yields Ca–As–O products, including $Ca_3(AsO_4)_2$ and $Ca_2As_2O_7$ (Sterling and Helble, 2003; Li et al., 2007). During real-life applications, the fly ash from incineration could easily block and corrode the flue gas collection system (Wang et al., 2021). At the same time, the bottom ash remaining at the furnace bottom could cause its corrosion (Wang et al., 2021). Thus, these drawbacks of incineration biowaste as a utilization method need to be solved in the future.

2.2. Pyrolysis

Pyrolysis involves the anaerobic decomposition of biomass at moderately elevated temperatures. The yields of the resulting products, which are typically biochar, bio-oil, and pyrolysis gases (Cui et al., 2021; Fig. 1), vary as a function of pyrolysis conditions (e.g., temperature, ramp rate, time). There are four types of pyrolysis recognized by scientists and engineers: slow, intermediate, fast, and flash (Ahmad et al., 2014; Mohan et al., 2014; Liu et al., 2017a, 2017b, 2020a, 2020b; Table S2). Slow pyrolysis is characterized by its moderate temperatures (300–700 °C), long heating times (5–720 min) and biochar production (Mohan et al., 2014; Table S2). Fast pyrolysis, typically performed at very high temperatures and times below 5 s, generally results in the formation of bio-oils (which account for 60–70% of the total final products formed, (Mohan et al., 2014; Table S2). Because of the necessary control over the anaerobic conditions, pyrolysis is typically performed in close containers, which, combined with moderate temperatures, confines the generated airborne pollutants (Conesa et al., 2009). Wen et al. (2018) and Zhong et al. (2015) observed maximum mass losses of *Leersia hexandra Swartz* and *Sedum plumbizincicola* after their pyrolysis to be equal to ~80% and 74%, respectively, which were lower than after their incinerated analoges. Similar to incineration, the plant pyrolysis undergoes three stages (Zhong et al., 2015; Fig. 1) with the following temperatures ranges: (1) 170 °C and below, (2) 170–400 °C, and (3) above 400 °C (Zhong et al., 2015).

The significant drawbacks of wider practical applications of pyrolysis are the need for sophisticated facilities and their operating difficulty (Samolada and Zabaniotou, 2014), all of which make pyrolysis very expensive to run. However, the cost could be reduced by the directed production and usage of high value-added byproducts such as biochar, bio-oil, and pyrolysis gas (Cui et al., 2021). Bio-oil and gases generated by biomass pyrolysis could be used to generate power or to synthesize valuable chemicals (Xiu and Shahbazi, 2012; Debalina et al., 2017), while biochar, a multi-functional material, is beneficial for applications where carbon sequestration, environmental remediation, and soil improvements are required (Biederman and Harpole, 2013; Ahmad et al., 2014; Cui et al., 2016). The main predicament of biochar utilization is the HM leaching risk, especially when the biochar contains an excess amount of HMs. An environmental safety assessment of the biochar should be implemented before it is used for the utilization. The poor yield and quality of the crude bio-oil obtained after biomass pyrolysis also introduce some challenges (Cui et al., 2021). It was recently demonstrated that HMs such as Zn, Pb, and Ni catalyze bio-oil production by promoting the corresponding hydrogenation process and organic acids formation (Wang et al., 2013; Yu et al., 2014). Zeng et al. (2019) applied solar flash pyrolysis to burn willow biomass to study how HMs affected biogas production and discovered that 14.8% and 34.5% more H_2 and CO were produced, respectively, during 5 min pyrolysis at 1200 °C. Ni presence in the hyperaccumulator modified the bio-oil composition by enhancing the formation of N-containing compounds such as triacetoneamine (Doroshenko et al., 2019). Thus, the properties of bio-oil derived from certain hyperaccumulators can be enhanced by adding Zn, Pb, and Ni, which would act as catalysts (Cui et al., 2021). Zhong et al. (2016), while studying how the pyrolysis temperature affected the compositions of bio-oils derived from *S. plumbizincicola*, obtained an increased bio-oil yield (from 22.0% to 31.7%) when the temperatures were increased from 450° to 650°C. However, this yield decreased to 15.9% at 750 °C due to the cracking effect (Zhong et al., 2016). At 450 °C, the major components of the produced bio-oil were acids, the yields of which declined as the pyrolysis temperature was increased. The highest alkene yield was observed at 650 °C for the pyrolyzed biomass containing a minimum of the oxygenated compounds (Zhong et al., 2016).

When phytoremediation residues are pyrolyzed, HMs enter biochar, bio-oil, and pyrolysis gas, and their distribution is very similar to the one obtained for incarcerated biomass, as it was in the case of Cr-containing pyrolyzed plants, which was also incinerated at 350 °C for comparison (Wen et al., 2018). As the pyrolysis temperature was increased, the Cr content in the bottom ash increased relative to its incinerated counterpart (Wen et al., 2018). However, Cr content in the gas during biomass pyrolysis was slightly lower than in the residue obtained by the incineration of the same biomass at 350 °C (Wen et al., 2018). Because of HM toxicity, the environmental reliability of the hyperaccumulator-derived biochar, bio-oil, and pyrolysis gas should be evaluated before their further applications. Several studies indicated that the residues of the pyrolyzed biomass possessed smaller risks of HM leaching risk (Wang et al., 2017c; Cui et al., 2018). Additionally, fewer HMs are typically observed in the hyperaccumulator-derived biochar, which is beneficial for its safe disposal (Wang et al., 2017c; Cui et al., 2018). To further control these risk factors of biochar, one needs to optimize temperature, time, additive and entrained flow during the pyrolysis (Liu et al., 2020a, 2020b). Just increasing the pyrolysis temperature would not minimize the leachable content of different HMs in biochar (Shi et al., 2017; Li et al., 2017; Gong et al., 2018; He et al., 2019). However, the optimized temperature is one of the key factors of decreasing the biochar potential risks. This optimized temperature could also vary for different plants and also with different HMs and their contents (Liu et al., 2020a, 2020b). Chami et al. (2014) recovered almost all Ni and Zn in the char during slow pyrolysis. However, during flash pyrolysis, they could only recover some metals from the char. In general, additives (such as calcium dihydrogen phosphate, NaOH, $CaCO_3$, Al_2O_3, $FeCl_3$) can reduce the HM leaching from biochar (Shi et al., 2017; Liu et al., 2017a, 2017b, 2019). As was mentioned earlier, some additives can react with HMs in the biomass and form stable compounds during pyrolysis (Liu et al., 2020a, 2020b). Recently, it was demonstrated that some biochar could also be used as additives during phytoremediation of HM-contaminated soil: they stimulate plant growth and improve biochemical properties of these soils. However, the applicability and feasibility of this approach (including long-term) need further assessment (Paz-Ferreiro et al., 2014; Lu et al., 2015; Gascó et al., 2019). To obtain a clean residue, HMs accumulated in the bio-oil can be removed by cation exchange, solvent

extraction, sorption, etc. (Yang et al., 2010; Carrier et al., 2011; Wang et al., 2013). Moisture in the biomass also affects the yield and calorific value of the pyrolysis products, and ideally, it should be below 10% (Wang et al., 2021). The tar produced by pyrolysis can block the pipeline, the repair of which is expensive (Wang et al., 2021). All these drawbacks prevent pyrolysis from widespreading to industrial scales. As a result, pyrolysis-based phytoremediation is still in its infancy (Wang et al., 2021).

2.3. Gasification

During gasification, solids heated to a certain temperature convert into syngas, which is a mixture of CO, H_2, etc. Typically, reactive gasifying agents (e.g., steam, air, CO_2) are used during gasification to form char, ash, and tar (Situmorang et al., 2020; Cui et al., 2021; Fig. 1). The main difference between gasification and pyrolysis is their reaction conditions (Cui et al., 2021). Pyrolysis is conducted in an inert atmosphere, while the gasification is performed under oxidative conditions (Cui et al., 2021). Unlike the slow pyrolysis method, gasification is implemented at 550–900 °C. This temperature becomes even higher (1000–1600 °C) after gasifying agents are introduced (Arena, 2012). The reduction of phytoremediation biomass processed by gasification is higher than after the pyrolysis but lower than after the incineration (Wang et al., 2021). Typically, biomass gasification is divided into four stages (Cui et al., 2018), characterized by the following temperature ranges: (1) < 190 °C, 190–410 °C, 410–650 °C, and 650–900 °C (Cui et al., 2018). Boudouard reaction is thermodynamically favorable above 710 °C (Kwon et al., 2014). The high-temperature weight loss of phytoremediation residues could be described by the Boudouard reaction, the initiation of which starts with the tar cracking reaction (Cui et al., 2018). In practical applications, the gasification is performed above 750 °C (Shen et al., 2013).

The syngas produced during biomass gasification is often used to generate electricity using fuel cells or gas turbines (He et al., 2009). CO and H_2 from the syngas could also be used as initial materials to produce high value-added chemicals and fuels (Arena, 2012). CO_2 can be used to produce CO above 700 °C via the following gasification reaction: char + $CO_2 \rightarrow 2CO$. In some cases, the CO accounted for 77.8% of the total gases formed and released at 900 °C (Cui et al., 2018). The syngas formed during the gasification of *S. alfredii* at 500–800 °C was used for fuel production via the Fischer–Tropsch process because of its favorable H_2/CO ratio (in the 1.26–2.46 range, Cui et al., 2018). Gasification could also be self-sustaining because of its heat release but only if the optimum equivalence ratio (ER) is in the 0.3–0.4 range (Zhang et al., 2011). Such no-external-heat gasification was demonstrated using a fluidized pilot bed reactor by Arena and Di Gregorio (2014)). However, the major obstacle to the stable functioning of the gasifier is the feedstock heterogeneity, both in composition and size (Cui et al., 2021). Sun and Wu (2020) proposed a new autothermal CaO biomass gasification technology, which uses CO_2 as the gasification agent. The combination of CaO and high CO_2 levels releases significant heat needed for biomass gasification. As a result, the syngas output increases from 0.21 to 0.90 kg/h. Zhang et al. (2019) gasified (for 10 min at 500 °C and 23 MPa) cyanobacteria in supercritical water and obtained 2.92 mol/kg of H_2, which accounted for the 33.3% of the total gas.

To make gasification reliable for the phytoremediation efforts and to obtain clean syngas with desired composition and properties, the migration and transformation of HMs during gasification should be thoroughly studied (Cui et al., 2021). Duan et al. (2017) reported that the transformation behavior of As had the same trend with increasing the temperature during pyrolysis and gasification, and about 60% of As in *P. vittata* was released at 600 °C. Lin et al. (2019) investigated how the addition of silica sand, zeolite, calcium oxide, calcined coal and activated carbon affected the volatilization of HMs and H_2 production during the fluidized-bed gasification. The activated carbon captured the most HMs, and under its presence, 53.1% of the gas produced was H_2 (Lin et al., 2019). Jiang et al. (2016) applied entrained flow technology to gasify a variety of remediation plants. At gasification temperatures below 1000 °C, Cd, Pb, Zn and As were volatile, while Mn, Cu, Co and Ni were not (Jiang et al., 2016). The volatilization of HMs and their compounds are affected not only by the reaction temperature but also by the reaction gas velocity, bed material type, particle size and reaction pressure (Froment et al., 2013; Wu et al., 2014; Jiang et al., 2016). A recently demonstrated microwave-assisted chemical looping gasification (MACLG) method showed promise in syngas production from hyperaccumulator biomass (Zhang et al., 2019). Hematite presence with its high O content enhanced H_2 and CO yield during MACLG of *S. plumbizincicola* and *P. vittata* (Zhang et al., 2019). The residues still contained substantial amounts of HMs, but they were removed by the activated carbon (Zhang et al., 2019). Chen et al. (2020a) reported that calcium based ($CaSO_4$) oxygen carriers with SiO_2 and Al_2O_3 as inert carriers could reduce the volatilization content of HMs (Cd, Pb, Zn) in the gasification process. Too much tar dust in gasification gas can lead to energy waste, gasification efficiency reduction, gasification equipment corrosion, and gas pipeline blockage (Xiao et al., 2017).

2.4. Hydrothermal modification

Hydrothermal modification is a technology that uses supercritical or subcritical water to convert biomass into high calorific value fuel in an autoclave (Wang et al., 2021; Fu et al., 2021; Ahmad et al., 2021). Hydrothermal modification includes hydrothermal carbonization (HTC), hydrothermal liquefaction (HL), and hydrothermal gasification (HTG).

HTC involves biomass conversion into a solid carbonaceous substance often called hydrochar. HTC is performed at moderate temperatures (160–260 °C) and saturated pressures (2–10 MPa) (Cui et al., 2021; Table 1). The reduction of phytoremediation biomass processed by HTC is lower than after the incineration, pyrolysis and gasification (Li et al., 2018a; Wang et al., 2021). The hydrochar yield and properties strongly depend on the reaction temperature, pressure, and feedstock (Cui et al., 2020). The yield of hydrochars declines from 46.0−46.5 to 22.8−29.2% with increasing hydrothermal temperature from 200 °C to 260 °C, which is ascribed to the temperature-dependent degradation of cellulose and hemicellulose during the HTC process (Cui et al., 2020; Fig. 1). Cui et al. (2020) reported that *M. spicatum*-derived hydrochars had a lower carbon content and a higher ash content as compared with hydrochars derived from *H. verticillata* and *C. indica*.

Like biochar, hydrochar can also adsorb contaminants and assist in energy production (Lee and Park, 2021). The main predicament of hydrochar utilization is also the HM leaching risk, especially when the hydrochar contains an excess amount of HMs. An environmental safety assessment of the hydrochar should be implemented before it is used for the utilization. Hydrochar's pH and energy density are moderate and high, respectively (Kambo and Dutta, 2015). Unlike pyrolysis, HTC is conducted at lower temperatures and under the water presence. Thus, HTC can be successfully used for phytoremediation of treating hyperaccumulator plants with high water contents. Lee and Park (2021) reported that the concentration of Cd, Cu, Ni, Pb and Zn in hydrochars decreased compared to sunflower raw biomass after HTC treatment. In addition, as HMs were not decomposed but converted to unstable forms during HTC, the HMs that were separated from the sunflower biomass remained in the liquid phase (Lee and Park, 2021). However, a lot of research still needs to be undertaken to establish how HMs are distributed and in what chemical form in the hydrochar and the liquid phase. This data would be critical in determining whether the resulting hydrochar could be utilized. Mai et al. (2017) reported that Zn slowed down the degradation of the phenolic hydroxyl group during HTC of the corresponding biomass, which affected the redox functionality of the resulting hydrochar. HTC also exhibits stronger economic prospects since it requires less energy and no pre-drying steps. However, the resulting raw hydrochar-slurry still needs to be filtered and dried for further use. Another drawback is that HTC is conducted using batch and

semi-continuous setups, both of which need to be adapted for large-scale and consequent commercialization. The salt formed in the reaction process has low solubility in subcritical water or supercritical water and is easy to precipitate (Wang et al., 2021). When these precipitates combine with coke, the reactor can be blocked and the operation cost of the equipment can be increased (Wang et al., 2021).

HL is performed at 190–380 °C and 5–20 MPa in subcritical water conditions (Cui et al., 2021; Table 1). The resulting product is hydrochar and an organic liquid which is also called "bio-oil" (Qian et al., 2018; Cui et al., 2021; Fig. 1). The reduction of phytoremediation biomass processed by HTC is lower than after the incineration, pyrolysis and gasification (Li et al., 2018a; Wang et al., 2021).

Recent research efforts focused on HM separation from hyperaccumulators using HL (Carrier et al., 2011; Deng et al., 2014; Qian et al., 2018). These efforts were accompanied by HMs, bio-oil, and hydrochar reclamation (Carrier et al., 2011; Deng et al., 2014; Qian et al., 2018). When the particle size, the liquid-solid ratio, the pressure, the temperature, the reaction time, and the catalyst are 75 μm, 13.3: 1, 23 Mpa, 373 °C, 30 min, and 0.1 mol/L K_2CO_3, more than 97% of Mn and other harmful metals in stems and leaves of *Phytolacca americana* L. can be separated into aqueous solution, and 86.24% of biomass can be converted into crude bio-oil (Deng et al., 2014). Qian et al. (2018) reported that about 90% of Zn was released from the *S. plumbizincicola* biomass during HL at an optimized temperature of 220 °C, and the release risk was mitigated via HL reaction for hydrochar production. However, more research still needs to be undertaken to establish how HMs are distributed and in what chemical form in the hydrochar and the liquid phase. HL-derived liquid fuel possesses higher heating values as well as lower oxygen and higher carbon contents than similar fuels obtained from pyrolyzed biomass (Huber et al., 2006). Bio-oils can not be used directly as a transportation fuel. However, they can be blended or upgraded to then be used to fabricate diesel-based fuels. Upgradation to generate liquid fuel from bio-oils could be performed catalytically, which is reviewed and summarized elsewhere (Tekin et al., 2014; Kumar et al., 2018). HM (e.g., Zn, Pb, and Ni) presence in the hyperaccumulators preceding these bio-oils was also reported to catalytically promote hydrogenation reaction and organic acids production from these bio-oils as initial materials (Yu et al., 2014; Wang et al., 2013). The HL of Zn/Pb/Ni-hyperaccumulator resulted in the in-situ up-gradation of the bio-oil, especially if the released HM compounds could be isolated and/or recovered. Similar to HTC, HL has a problem of blocking the reactor and increasing the operation cost of the equipment (Wang et al., 2021).

HTG decomposes the biomass under the supercritical water presence typically at pressures and temperatures above 22.1 MPa and 374 °C, respectively, and is compatible with the wet biomass (Cui et al., 2021; Table 1). Compared with reaction temperature, the reaction time is less effective, while the pressure is least effective on the conversion efficiency and product yields (Li et al., 2018b). The main gas products are H_2 and CH_4 (Cui et al., 2021; Fig. 1). The reduction of phytoremediation biomass processed by HTC is lower than after the incineration, pyrolysis and gasification (Li et al., 2018a; Wang et al., 2021).

Supercritical water acts as both a medium and a reactant during HTG. HTG can treat fresh hyperaccumulator biomass directly without drying (Cui et al., 2021), and is very efficient in alleviating environmental risks of HM leaking from the hyperaccumulator as well as achieving the energy recovery (Carrier et al., 2011; Li et al., 2018b). Relative to the subcritical water conditions during HTC, the solubility of HMs in supercritical water during HTG is lower (Sue et al., 1999), which is beneficial for the hyperaccumulator-derived liquid product treatment. HTG is classified into three sub-categories (Correa and Kruse, 2018). The first category includes processes, in which the aqueous phase reforming occurs at 215–265 °C with H_2 and CO_2 as final products (de Vlieger et al., 2012). The second category contains near-critical catalyzed gasification reactions occurring at 350–400 °C and producing CH_4 (Matsumura et al., 2005). The third category describes processes during which supercritical water gasification occurs at above 500 °C, and H_2 and CO_2 are the final major products (Kruse et al., 2003). HTG produces more H_2 from biomass no matter what its moisture content is, unlike general gasification routes. Additionally, the formation of high-pressure H_2 further reduces the gas compression cost, typically acquired during storage of the released H_2 (Demirbas, 2009). Unlike general gasification, HTG prevents HM volatilization because of its low temperatures (Huang and Yuan, 2016). For example, almost no HMs were detected in the gas formed during the HTG of *Pteris vittata* L. (Carrier et al., 2011; Li et al., 2018b). Although Pb and Zn tended to accumulate in solid residues with a maximum increment of about 50% in the total content, they were mostly converted to more stable oxidizable and residual fractions, and thus the ecotoxicity and bioavailability were greatly mitigated with no obvious increase in direct toxicity fractions (Li et al., 2018b). Still, relatively extreme reaction conditions of HTG demand advanced setups, which increases the fabrication and operation costs. Other drawbacks preventing HTG commercialization are salt deposition and reactor corrosion.

3. Extraction treatment

Extraction treatment involves HMs removal from the phytoremediation residues using an extraction agent. The reduction of phytoremediation biomass processed by extraction is lower than after heat treatment (Li et al., 2018a). The residues can be pressed to remove the squeezed liquid or heat-treated prior to the extraction to concentrate HMs in the squeezed liquid or biomass (Wang et al., 2017b, 2021). HMs in the squeezed liquid need to be treated if the phytoremediation residues are pressed prior to the extraction. Wang et al. (2017b) reported that using flocculant can reduce the concentration of Cd in the squeezed liquid from 1.98 mg/L to 0.066 mg/L. The commonly used extractants are sulfuric acid, ammonium acetate, ammonium oxalate, ammonium Nitrate, EDTA, and pure water (Table 2). The HMs in the extraction solution treated by extraction method can be recycled.

An et al. (2012) used distilled water to extract Cr, Cu, Zn and Pb in *Buxus megistophylla* Levl., and found that the vacuum degree, soaking time and decompression time have obvious effects on the removal of Cr, Cu, Zn and Pb. The maximum removal rate of Cr, Cu, Zn, and Pb in *Buxus megistophylla* Levl. is 78.8%, 93.3%, 65.9%, and 82.9% (An et al., 2012). Houzelot et al. (2018) studied the leachability of Ni and Mn from birch ash and found that the kinetics of acid leaching was not limited by external membrane diffusion or stirring speed but depended on the chemical elements present in the phytoremediation residues. However, the leachability and leaching rates could be significantly increased by elevating the temperature or by using concentrated acids (Houzelot et al., 2018). Núñez-López et al. (2008) performed leached Pb from *Robinia pseudoacacia* and established the following extraction intensity: EDTA > ammonium oxalate > water ~ ammonium nitrate > ammonium acetate. The maximum removal rate of Pb in *Robinia pseudoacacia* is 99% by EDTA (Núñez-López et al., 2008). The favored complex in EDTA system was the soluble species $PbEDTA^{2-}$, whereas for the oxalate system the favored species was the precipitate of lead oxalate $Pb(COO)_{2S}$ although a small fraction of lead oxalate can remain in solution as $Pb[(COO)_2]_2^{2-}$ (Núñez-López et al., 2008). Ge et al. (2020) advised that the hydrochloric acid with low concentration could be used as extractant of HMs in peanut meal. The removal rate of Pb, Zn, Cu, and Cd in peanut meals can reach 74.43%, 93.13%, 96.36%, and 91.39% by extractant (Ge et al., 2020). In order to further improve the efficiency and reduce the cost, it is necessary to study the deep mechanism of extraction agent and HMs.

The effective active components in hyperaccumulators may be recycled and used. The effective active components include essential oils, saponins, lignin, etc., as well as compounds with medicinal value. Traditional extraction technology is generally solvent extraction, but its safety and extraction rate is low. The new extraction technology has the advantages of high efficiency, energy saving and environmental

Table 2
Technical parameters of the extraction treatment method.

Device	Temperature (°C)	Extraction time (min)	Medium	Stirring speed (rpm)	References
Device with stirring function	20–90	30–960	Potassium citrate, potassium tartrate, hydrochloric acid, sulfuric acid, ammonium acetate, ammonium oxalate, ammoniumnitrate, EDTA, pure water	100–700	Núñez-López et al. (2008), Barbaroux et al. (2011), Hazotte et al. (2017), Houzelot et al. (2018), Ge et al. (2020)
Device without stirring function	16–60	60–120	Distilled water	/	An et al. (2012)

protection, including ultrasonic extraction, microwave extraction, supercritical fluid extraction, semi bionic extraction and so on (Wang et al., 2021). Xue (2016) used *Folium Perillaes*, a Cd hyperaccumulator, as the raw material to extract essential oil from leaves by various methods, which provided a more efficient way to extract essential oil from *Folium Perillaes*. Xing et al. (2012) used *Sedum alfredii* Hance, a hyperaccumulator of Zn and Cd, as the raw material, obtained a salidroside-type metabolite by ultrasonic assisted ethanol extraction and macroporous resin column (AB-8 type) separation technology, and contents of Zn and Cd in these metabolites were lower than national standards. Some hyperaccumulated plants are also medicinal plants, and the effective components extracted from them can be used as medicinal materials. However, before using these active substances, the HM content must be analyzed and evaluated.

4. Microbial treatment

Microbial treatment method is a technology to treat biomass through microbial stabilization and humification of organic matter (Li et al., 2018a). Microbial treatment can be carried out under aerobic (compost) or anaerobic (fermentation) conditions (Table 3; Fig. 2). Aerobic or anaerobic microorganisms can decompose plant organic matter and transform it into microbial cell substances, organic acid, alcohol, CO_2, H_2O, NH_3, H_2S, CH_4, PO_4^{2-}, SO_4^{2-} releasing energy (Fig. 2). The reduction of phytoremediation biomass processed by microbial treatment is lower than after heat treatment (Li et al., 2018a; Wang et al., 2021).

Cao et al. (2010) used compost to treat *Pteris vittata* L. containing As, and found that the content of total As and water-soluble As in the composted residue decreased by 25% and 32%, respectively. Most of the lost As was transferred to the leachate, which needed high-cost technology for subsequent treatment to prevent secondary pollution (Cao et al., 2010). In order to improve the composting method and reduce the harm of HMs after composting, some researchers found that adding lime, fly ash from biomass incineration, biochar, red mud and other substances for mixed composting could increase the residual content of HMs in the residue, thus reducing the leaching of water-soluble HMs and reducing the environmental risk and subsequent treatment costs (Soares et al., 2015; Singh and Kalamdhad, 2016; Asquer et al., 2019). Wei et al. (2020) studied effects of humus from different compost sources and HM tolerant bacteria produced by compost on the absorption of HMs, and found that HM tolerant bacteria had better binding ability of HMs than humus, and humus as an activator could improve the diversity and biomass of HM tolerant bacteria, thus promoting the adsorption of HM ions. The synergistic effect of humus and HM tolerant bacteria could reduce the leaching of HMs by 60−80% (Wei et al., 2020). Yang et al. (2019) reduced the content of HMs in the leachate by using functional bacteria with bed materials (sponge, zeolite, cotton). The combination of sponge and cotton with functional bacteria reduced the leaching of Cr by 19.1−26.4% (Yang et al., 2019). Tang et al. (2006) composted *Elsholtzia splendens* Nakai with Cu, and applied it as basal fertilizer in Cu deficient soil. The results showed that it could significantly increase the plant height, biomass and grain weight of the wheat, which was an effective way of resource utilization (Tang et al., 2006). Cao et al. (2015) found that compared to normal plants with low Cu content, the plants used in remediation with increased Cu levels (100 mg/kg) not only promoted anaerobic digestion and required a shorter anaerobic digestion time, but also increased the methane content in biogas. However, the duration of microbial treatment process is long (Table 3), and the reaction equipment is expensive. In addition, the residue and leachate after microbial treatment still contains high concentrations of HMs, which need to be further treated. In general, it is difficult to separate HMs from the plant body using microbial treatment. This method may be considered when the content of HMs in plants is low. When the content of HMs in plants is high, other disposal and utilization methods can be used to reduce the content of HMs in plants before microbial treatment.

Table 3
Technical parameters of the microbial treatment method.

Device	Temperature (°C)	Processing time (d)	Inoculum	References
Aerobic device	25–50	14–120	Mature biosolid	Tang et al. (2006), Cao et al. (2010), Singh and Kalamdhad (2016), Yang et al. (2019)
Anaerobic device	37	50	The biogas residue of a water hyacinth anaerobic digestion reactor at the Jiangsu Provincial Academy of Agricultural Science, China	Cao et al. (2015)

5. Compression landfill

Compression landfill is the most common solid waste disposal method, which is simple and easy to operate (Wang et al., 2021). The compression landfill system is mainly divided into waste storage system, leachate collection system, and landfill gas collection system (Fig. 3). The complex of high concentration HMs and chelating agent can be produced when the phytoremediation plants are compressed (Wang et al., 2021). The reduction of phytoremediation biomass processed by microbial treatment is lower than after heat treatment (Li et al., 2018a; Wang et al., 2021).

HMs mainly existed as soluble forms or bioavailable forms in the leachate produced by compression landfill of phytoremediation plants (Li et al., 2018a; Wang et al., 2021). Compared with microbial treatment method, the disposal time of compression landfill is shorter, but the leachate containing HMs produced by compression landfill has higher risk of secondary pollution (Wang et al., 2021). It is also a new problem whether the HM content of compressed phytoremediation plants can meet the standard of safe landfill (Wang et al., 2021). Before the large-scale application of the compression landfill method, it is necessary to conduct a deeper research on the treatment of the leachate and reduce the operation cost of the compression landfill method (Li et al.,

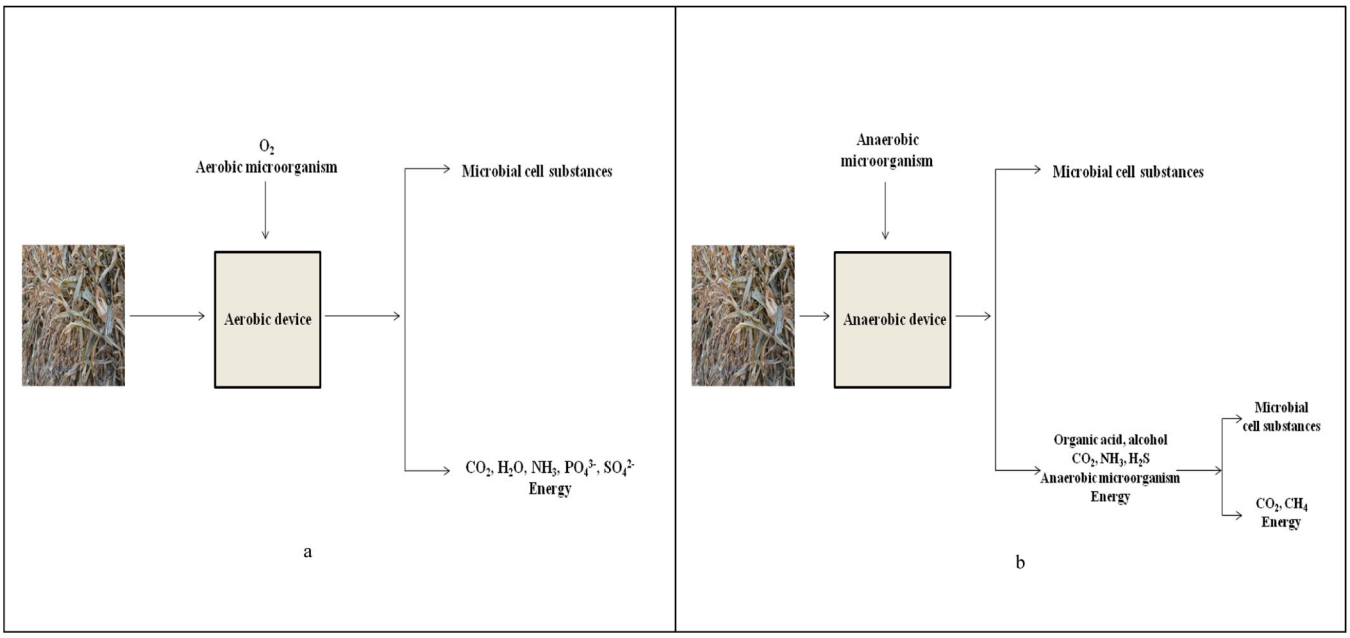

Fig. 2. Principle of microbial treatment (a: compost; b: fermentation).

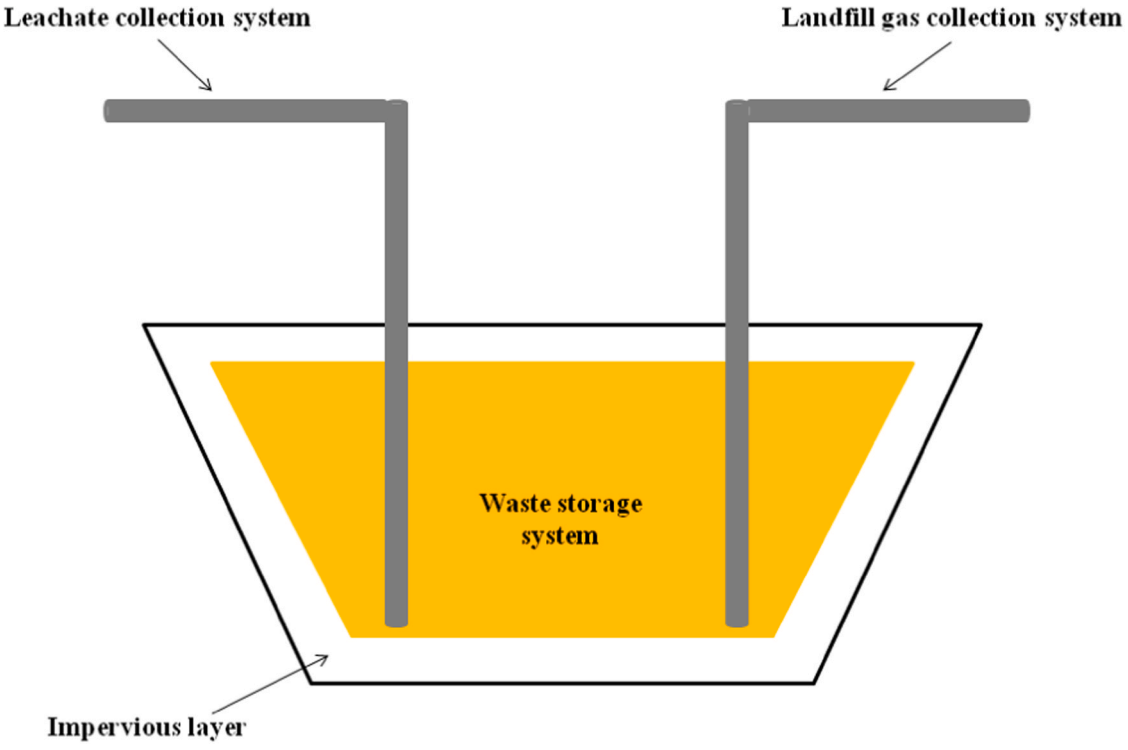

Fig. 3. Schematic diagram of landfill site.

2018a; Wang et al., 2021).

6. Synthesis of nanomaterials

The particles with size ranging from 1 to 100 nm are called nanomaterials, and among them, metal nanoparticles and metal oxide nanoparticles are considered to be the most effective ones, because they increase the surface area to volume ratio (Ahmed et al., 2017; Chausali et al., 2021). Metal nanoparticles and metal oxide nanoparticles can be widely used in the fields of electrochemistry, catalysis, medical treatment, aviation industry and so on (Ahmed et al., 2017; Zammit et al., 2019; Priyadarshini et al., 2020; He et al., 2021; Liu et al., 2021; Yang et al., 2021; Pei et al., 2021). Compared with the traditional synthesis technology of metal nanoparticles, which is expensive and harmful to the environment, the biosynthesis of metal nanoparticles by phytoremediation plants has gradually become a new method (Ahmed et al., 2016). The phytosynthesis of nanoparticles and its mechanism is dependent on phytochemicals such as alkaloids, phenols, flavonoids, tannins, saponins, terpenoids, carbohydrates (Ahmed et al., 2017). The reduction rate of phytoremediation biomass processed by synthesis of

nanomaterials can be 100%, which is higher than other disposal methods (Li et al., 2018a; Wang et al., 2021).

Qu et al. (2014) synthesized metal carbon nanotubes with particle size of 110 nm loaded with Cu and ZnO using phytoremediation plants containing Zn and Cu. Wang et al. (2016) synthesized ZnO nanoparticles with photocatalytic activity for organic pollutants from *Sedum alfredii* Hance. Producing HM nanoparticles generally includes the concentrating of HMs and the combination of concentrated HMs and substances (such as chlorophyllin, Na_2S) (Fig. 4). Chen et al. (2020b) made CdS@C nanocomposites with hexagonal structure using *Sedum plumbizincicola* by pyrolysis and hydrothermal vulcanization process, which have good photocatalytic activity and can degrade pigment wastewater. At present, there is not much research of synthesis of nanomaterials using phytoremediation plants containing HMs. More studies about the preparation of nanomaterials from different plant species containing different kinds of HMs are needed. However, synthesis of nanomaterials for phytoremediation residues is complex and costly.

7. Future perspectives

The disposal and utilization methods (heat treatment, extraction treatment, microbial treatment, compression landfill, and synthesis of nanomaterials) have their own advantages and disadvantages. A comparison between various disposal and utilization methods for phytoremediation plants with HMs is listed in Table S3. The reduction rate of plant biomass with HMs is high using synthesis of nanomaterials, incineration, gasification, or pyrolysis (Table S3). The environmental risk of secondary pollution is low using extraction and synthesis of nanomaterials (Table S3). On the other hand, other methods (heat treatment, microbial treatment, and compression landfill) have relatively high risk of secondary pollution (Table S3). Synthesis of nanomaterials incineration, pyrolysis, and gasification are especially suitable for the disposal of high biomass plants (e.g. *Ricinus communis* L., *Pteris vittata* L., *Brassica juncea*) with HMs because of the strong capacity of plant biomass reduction. Hydrothermal modification, extraction, and microbial treatment are suitable for the disposal of low biomass plants (e.g. *Oxalis corniculata* L.) due to the low reduction rate of plant biomass. Incineration and gasification are especially suitable for the disposal of plants with high boiling point HMs (e.g. Ni, Cr, Cu), considering the secondary pollution caused by the transformation of HMs into gaseous state. Pyrolysis and hydrothermal modification is especially suitable for the disposal of plants with medium and high boiling point HMs (e.g. Pb, Cd, Zn, Ni, Cr, Cu). Extraction and synthesis of nanomaterials are especially suitable for the disposal of plants with low boiling point HMs (e.g. Hg, As) for the low risk of secondary pollution by extraction or synthesis of nanomaterials. Microbial treatment and compression landfill are suitable for the disposal of plants with low contents of HMs, as the disposal process can cause secondary pollution. All methods have by-products that can be recycled as resources. Each method has some shortcomings, which are the focus of future research.

Although the migration and transformation of HMs in hyperaccumulators during heat treatment processes have been studied in some research, analysis of the evolution pathway of different kinds of volatile HMs (such as Hg) should be further promoted. It is necessary to do more research about the mechanism of the migration and transformation of HMs in hyperaccumulators during heat treatment processes. In addition to conventional offline methods such as sequential chemical extraction and XRD analysis, some accurate and online detection technologies should be employed to deeply and accurately illuminate the behavior of HMs. The offline techniques are usually based on physicochemical analysis of biomass and solid/liquid products, whereas they can't describe the time-resolved release of HMs and HM species in the gaseous products. Yields and quality of products of plant biomass treated by heat need to be further studied. What's more, the content, and the risk of HMs in products of plant biomass treated by heat should also be further studied.

At present, the research on the mechanism of extraction method for phytoremediation residues with HMs is insufficient. Therefore, more research about the deep mechanism of extraction agent and HMs is needed. Moreover, the treatment of the extraction solution also needs to be considered and studied. Metals and some useful substances can be extracted and recycled, and research in this area is also interesting.

Microbial treatment and compression landfill have the problem that leaching solution or leachate contains a lot of HMs, which may cause secondary pollution. It is necessary to conduct a deeper research on the treatment of the leaching solution or leachate and reduce the operation cost of microbial treatment and compression landfill. When the content of HMs in plants is high, other disposal and utilization methods can be used to reduce the content of HMs in plants before microbial treatment and compression landfill.

At present, the research on the mechanism of synthesis of nanomaterials for phytoremediation residues is insufficient. More research about the deep mechanism of synthesis of nanomaterials for phytoremediation residues is needed. Moreover, whether more simplified methods of can be found needs more research.

As the disposal and utilization methods (heat treatment, extraction treatment, microbial treatment, compression landfill, and synthesis of nanomaterials) have their own advantages and disadvantages, the combination of some methods may bring their advantages into play and overcome their disadvantages. However, there are not many studies on this aspect. Thus, it is worth studying technical parameters and mechanism of the combination methods.

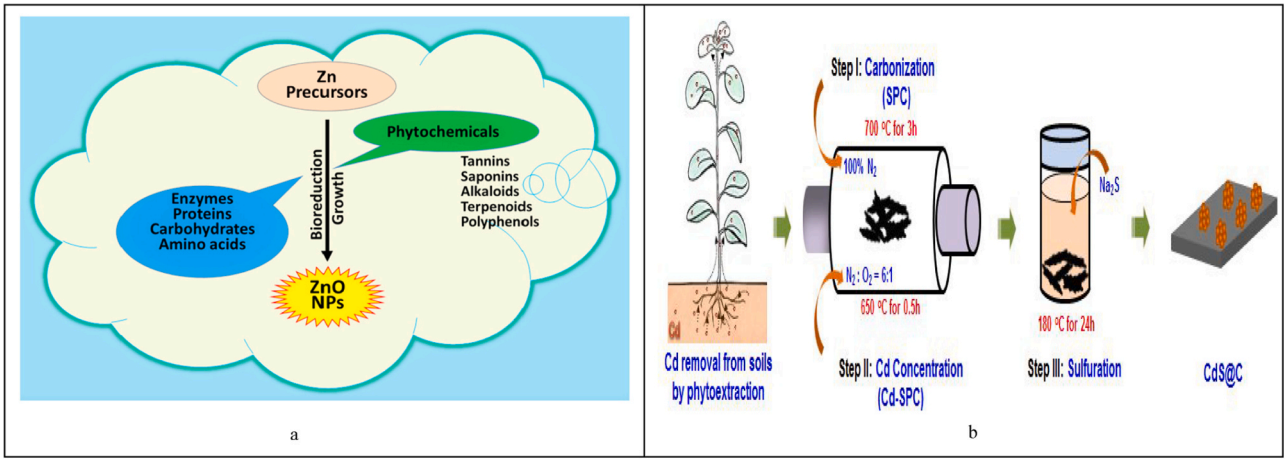

Fig. 4. Schematic illustration of the procedures for HM synthesis (a: ZnO nanoparticles; b: CdS@C nanocomposites) (Ahmed et al., 2017; Chen et al., 2020b).

8. Conclusions

Plant biomass derived from the phytoremediation of HM contaminated soils should be properly disposed. This review provides reference for technical parameters and characteristics of various disposal and utilization methods (heat treatment, extraction treatment, microbial treatment, compression landfill, and synthesis of nanomaterials) for phytoremediation plants with HMs. The reduction rate of plant biomass with HMs is high using synthesis of nanomaterials, incineration, gasification, or pyrolysis. The environmental risk of secondary pollution is low using extraction and synthesis of nanomaterials. Solid, liquid and gas products produced in the treatment process can be used as resources. The disposal and utilization methods have their own advantages and disadvantages. More research is needed to address the shortcomings of these methods. The combination of some methods may bring their advantages into play and overcome their disadvantages, which may be a feasible choice.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.ecoenv.2021.112821.

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