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Review

Roles and significance of chelating agents for potentially toxic elements (PTEs) phytoremediation in soil: A review

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ABSTRACT

Phytoremediation is a biological remediation technique known for low-cost technology and environmentally friendly approach, which employs plants to extract, stabilise, and transform various compounds, such as potentially toxic elements (PTEs), in the soil or water. Recent developments in utilising chelating agents soil remediation have led to a renewed interest in chelate-induced phytoremediation. This review article summarises the roles of various chelating agents and the mechanisms of chelate-induced phytoremediation. This paper also discusses the recent findings on the impacts of chelating agents on PTEs uptake and plant growth and development in phytoremediation. It was found that the chelating agents have increased the rate of metal absorption and translocation up to 45% from roots to the aboveground plant parts during PTEs phytoremediation. Besides, it was also explored that the plants may experience some phytotoxicity after adding chelating agents to the soil. However, due to the leaching potential of synthetic chelating agents, the use of organic chelants have been explored to be used in PTEs phytoremediation. Finally, this paper also presents comprehensive insights on the significance of using chelating agents through SWOT analysis to discuss the advantages and limitations of chelate-induced phytoremediation.

1. Introduction

Global advancements with expanding industrial and human activities have dramatically resulted in several pollution in the soil. Soil pollution is a critical worldwide environmental problem due to its pronounced effects on the ecosystem and human health (Cristaldi et al., 2017; Ghazaryan et al., 2021). Due to the presence of various mutagenic and carcinogenic elements in the soil, soil contamination has brought several negative impacts to the ecosystem, such as deterioration of plant growth and PTEs accumulation in animals and consequently led to the food chain contamination at higher trophic levels (Shah and Daverey, 2020; Shen et al., 2022; Yadav et al., 2018). Soil contaminants may include the organic (halogenated volatiles, non-halogenated volatiles, polychlorinated-biphenyls, pesticides and cyanides) and inorganic (volatile metals, non-volatile metals and radioactive components) compounds (Cristaldi et al., 2017; Ng et al., 2020; Sharma et al., 2015). Among various soil contaminants, PTEs at a higher concentration have raised profound concerns about the environment and human health. PTEs can be described as naturally occurring trace elements that have a high atomic number (>20) and elemental density of greater than 5 g/cm^3 (Gul et al., 2021; Ng et al., 2020).

Multiple types of PTEs were reported to be present in the soil, such as arsenic (As), cadmium (Cd), chromium (Cr), cobalt (Co), copper (Cu), lead (Pb), iron (Fe), mercury (Hg), nickel (Ni), manganese (Mn) and zinc (Zn). Several types of PTEs, including Co, Cu, Cr, Fe, Mn and Zn, act as micronutrients for plants whereby the plants require it in a minute quantity for several biochemical and enzymatic processes (Gul et al., 2021; Shahid et al., 2014). On the other hand, other elements such as As, Cd, Pb and Hg serve no purpose in plant growth and the development of natural ecosystems (Ali et al., 2013; Gul et al., 2021). Besides, these elements are predominantly hazardous and have been regarded as the most dangerous elements by the Agency for Toxic Substances and Diseases Registry (ATSDR) and the United States Environmental Protection Agency (USEPA) (Ashraf et al., 2019; Shahid et al., 2014). The presence of these elements in the environment may cause severe impacts on the health of animals, plants and humans through the food chain (Ali et al., 2013; Ashraf et al., 2019). Furthermore, PTEs are non-biodegradable and persistent in the environmental bodies for a long time, consequently expanding the negative effect on the environment and ecosystem (Ali et al., 2013; Cristaldi et al., 2017; Shen et al., 2022).

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Therefore, remediation of the PTEs-contaminated areas is required to avoid any hazardous risk to ecosystems (Ali et al., 2013; Amir et al., 2020; Kurade et al., 2021).

As a result, many physical, chemical and biological remediation approaches have been developed to remediate the PTEs-contaminated sites. The chemical and physical remediation approaches include soil washing, soil flushing, chemical leaching, thermal treatment, vitrification, filtration, reverse osmosis and excavation (Ali et al., 2013; Saxena et al., 2016; Abou-Shanab et al., 2020; Yan et al., 2021). However, there are some limitations in physical and chemical remediation techniques due to cost and labour requirements, environmental risk and safety hazards which make its unsustainable to be applied (Gavrilescu, 2022; Ponce-Hernández et al., 2022). Hence, available bioremediation technologies such as bacterial remediation, mycoremediation, phycoremediation and phytoremediation seen as possible remediation techniques for remediating pollution with less environmental risk (Cristaldi et al., 2017). However, only phytoremediation, the plant-based bioremediation technology, has successfully gained public acceptance as one of the effective techniques due to its simple procedure, low-cost method and environmentally friendly approach (Gunarathne et al., 2019; Kafle et al., 2022; Ng et al., 2020; Sarwar et al., 2017). Correspondingly, many phytoremediation activities have been taken to remediate and recover assorted metals (As, Pb, Cu, Cr, Cd and Hg) with various types of plants and hyperaccumulators (Constantinescu et al., 2019; Ng et al., 2016; Patra et al., 2019; Radziemska, 2018).

Despite this remediation technology's numerous advantages and significant outcomes, some advancements in the current procedure are still required to improve the traditional phytoremediation. As a result, new research is still emerging in phytoremediation to maximise the removal efficiency of the metals, overcome current limitations and harness other benefits through this technique (Gavrilescu, 2022; Gunarathne et al., 2019). The capacity for phytoremediation is often limited by low bioavailability of the targeted metal in soil or contaminated sites due to residual insoluble forms of the metal and its strong binding to biotic and non-biotic soil ligands (Sinhal et al., 2010). Recently, there has been an increasing interest in using chelating agents in phytoremediation as they can enhance the bioavailability of metals in the soil and improve metal recovery by plants. Many previous studies have reported that chelate-assisted phytoremediation could boost the metals removal efficiency, high phytotoxicity tolerance and increase translocation and accumulation of metals in the aboveground parts of the plants (Aghelan et al., 2021; Hu et al., 2019; Jiang et al., 2019; Shahid et al., 2014; Tahmasbian and Safari Sinegani, 2016). Thus, this paper discusses the roles of the chelating agents in phytoremediation and summarises the impacts of chelating agents on PTEs uptake and plant growth and biomass. Moreover, the paper also provides an overview of traditional phytoremediation and the mechanisms of chelate-induced phytoremediation. Besides, the advantages and disadvantages of chelate-assisted phytoremediation are also described in this study using the SWOT analysis method. Lastly, several suggestions for future prospects of PTEs phytoremediation are also discussed at the end of the paper.

2. Available bioremediation strategies

Bioremediation involves plants and microorganisms (bacteria, algae and fungi) to remediate the contaminated soil and has no adverse effects on the environment (Gomes et al., 2016; Haleyur et al., 2019; Abtahi et al., 2020; Li et al., 2020). Biological remediation is eco-friendly and more economical compared to explored physical and chemical remediation approaches (Ayangbenro and Babalola, 2017; Mahbub et al., 2017; Foong et al., 2020; Sravya and Sangeetha, 2022). Besides that, the microorganisms involved also developed biological and metabolic activities that make them well-adapted to the PTEs in the soil (Lorenzo and Loza-Tavera, 2011; Oyetibo et al., 2016; Cruz et al., 2017; Chen et al., 2019a). It is also noteworthy that PTEs cannot be further degraded and can only be transferred into the less or non-toxic forms (Juwarkar and Yadav, 2010).

2.1. Bacterial remediation

Bacteria often remediate PTEs present in the contaminated sites through cell surface adsorption, metabolic uptake, oxidation and reduction (Kenney and Fein, 2011; Tourney and Ngwenya, 2014). Many scholars have studied the utilisation of various types of bacteria such as *Paenibacillus*, *Pseudomonas*, *Haemophilus*, *Micrococcus* and *Bacillus* to remediate contaminated sites with PTEs (Table 1). Chen et al. (2021) reported that *Bacillus pasteurii* reduces the content of Pb in soil up to 76.34%. *Pantoea agglomerans*, the gram-negative bacteria used to remediate soil contaminated with PTEs due to anthropogenic activities (Audu et al., 2020). The results showed that Pb and Cu absorption from contaminated soils showed the highest biosorption potential of 97% at the optimum pH and temperature (Anusha & Natarajan, 2020). Gram-positive bacterium, *Cellulosimicrobium* spp. is also capable of reducing Cr at higher concentrations (68–81%) (Naeem et al., 2013; Rehman and Faisal, 2015). Bharagava and Mishra (2018) highlighted that *Cellulosimicrobium* sp. able to reduce up to 97% of Cr present in tannery wastewater. Notably, the naturally occurring bacteria at the PTEs contaminated sites have developed resistance towards these elements. For instance, bacterial species like *Pseudomonas aeruginosa*, *Klebsiella pneumoniae* and *Staphylococcus* sp. found in PTEs-rich wastewater have developed resistance to certain PTEs such as As, Pb, Cr and Cu (Alboghobeish et al., 2014; Karakagh et al., 2012).

Many studies also showed that the bioremediation rate of PTEs depends on the cell density or biomass, which is observed in *Rhodotorula* sp. and *Halomonas* sp. (Cao et al., 2020; Manasi et al., 2014). It is also good to note that bacterial biofilms are the essential media for PTEs transportation and transformation (Fan et al., 2021; Rezania et al., 2016; Yang et al., 2021d). Besides that, indoleacetic acid and exopolysaccharides aid in producing plant growth promoting bacteria such as

Table 1
PTEs remediation capabilities among various types of bacteria species.

Bacteria species	PTE tested	Observations	References
Bacillus pasteurii	Pb	Pb concentration in soil decreased up to 76.34%	Chen et al. (2021)
Bacillus cereus	Pb, Cu, Cr	At pH 7 and 35 °C, about 78–98% of PTEs was accumulated	Anusha & Natarajan (2020)
Bacillus circulans	Cr	Able to remove Cr effectively at 30 °C and pH 5.6	Chaturvedi (2011)
Cellulosimicrobium sp.	Cr	About 97% of Cr reduced from tannery wastewater with 300 mg L^{-1} of Cr	Bharagava and Mishra (2018)
Enterobacter asburiae	Cd, Ni	Accumulated about Cd (3000 µg/mL) and Ni (2000 µg/mL)	Bhagat et al. (2016)
***Exiguobacterium* spp.**	As	Highly resistance to As with the removal potential up to 99%	Javanbakht et al. (2014)
Halomonas sp.	Pb, Fe, Cd	Enhanced remediation rate with a shorter duration	Cao et al. (2020)
***Nostoc* spp.**	Pb, Cd, Cr	More than 90% efficiency in Pb removal whilst Cd and Cr through biosorption	Kumar et al. (2012), Roy et al. (2015)
Pantoea agglomerans	Pb, Cu	Highest biosorption potential up to 97%	Audu et al. (2020)
Pseudomonas spp.	Cr, Cu	The epiphytic biofilm had better positive effects on the Cr bioaccumulation than Cu	Geng et al. (2022)
Sporosarcina pasteurii	Zn	About 58–96% of Zn removal from the polluted sites	Li et al. (2013), Jalilvand et al. (2019), Nasrin et al. (2022)

Exiguobacterium spp. which have potential removal of As about 99% by reducing the toxicity (Javanbakht et al., 2014). *Sporosarcina pasteurii*, a gram-positive rod-shaped bacterium had been used in several studies to remediate sites polluted with Zn where the results indicated that up to 96% of Zn was removed from the soil (Jalilvand et al., 2019; Li et al., 2013). Together with extracellular polymeric substances, biofilms help in PTEs remediation by surface precipitation and ion exchange process (D'Acunto et al., 2016; Hu et al., 2019; Li and Yu, 2014). Geng et al. (2022) reported that *Pseudomonas* spp. had significantly higher Cr accumulation (8.3 mg kg^{-1}) than Cu (3.9 mg kg^{-1}). Cyanobacteria like *Nostoc*, *Synechococcus*, *Synechocystis* and *Phormidium* have also been identified as a potential PTEs bioremediation species (Hussein et al., 2020; Sen et al., 2018). *Nostoc* spp. was reported to have more than 90% of recovery efficiency of Pb, Cd and Cr (Kumar et al., 2012; Roy et al., 2015).

2.2. Mycoremediation

Mycoremediation is an effective method of removing environmental pollutants and toxic elements by employing fungi (Akhtar and Mannan, 2020). Fungi are considered as an essential agent for the bioremediation of PTEs due to their large biomass, robustness and simple nutritional requirement (Pietro-Souza et al., 2019; Urik et al., 2014). Besides that, fungi remediation is also a more lucrative option than bacterial remediation in remediating enzymes (Deshmukh et al., 2016; Shah, 2017). In addition, fungal cells can also live under high toxic metal concentrations with various ionisable sites and functional groups, allowing them to have the capability of broad absorption and specificity of PTEs (Khan et al., 2019; Xu et al., 2021). However, less bioremediation has been conducted using fungi as compared to other microorganisms due to longer duration required in the process (Vala et al., 2018; Zeng et al., 2015). Arbuscular mycorrhizal fungi (AMF) consist of a group of roots obligate biotrophs that exchange mutual benefits with plants (Berruti et al., 2016). AMF are often found in the PTEs polluted area where the mycelia, vesicles and spores act as the sink for the PTEs (Huang et al., 2018; Wang, 2017). *Funneliformis mosseae* is an AMF that was found to be an effective agent for Cu, Pb and Zn bioremediation with up to 68% of PTEs removed from contaminated soil (Adeyemi et al., 2021; Yang et al., 2015b; Thioub et al., 2019). Similar filamentous fungi species also have developed hydrophobin as part of resistance toward PTEs, which was also reported in *Trichoderma harizianum* for Hg (Puglisi et al., 2012).

Another filamentous fungi *Penicillium* spp. exhibited high volatilising capacity and bioremediation of Hg in polluted sites (Chang et al., 2020; Jiskra et al., 2015). It was also observed that both the intracellular and extracellular processes allow the *Penicillium* spp. to survive and bioremediate in the Hg-rich area. *Aspergillus fumigatus* has also shown exceptional performances in reducing the concentration of Cr, Cd, Cu and Ni below the permissible limits with about 37–98% of PTEs were absorbed from the contaminated sites (Dey et al., 2016, 2020). In another study by Hassan et al. (2020), *Aspergillus fumigatus* showed a maximum tolerance index (0.98) towards As, Cr, Cu and Mn as compared to other fungi (*Tremates versicolor* and *Daldinia starbaeckii*). Besides, it was also observed that the intracellular accumulation of toxic metals in fungi vacuole, played an essential role for both resistance and detoxification of PTEs (Hassan et al., 2020; Muneer et al., 2016). Furthermore, a native fungal isolate, *Aspergillus flavus* has been reported with 74% of Cr removal (Huang et al., 2022) while *Aspergillus penicillioides* has recorded to remove up to 73% of Pb from contaminated wastewater (Paria et al., 2022) (see Table 2) .

Table 2
PTEs remediation capabilities among various types of fungi species.

Fungi species	PTE tested	Observations	References
Aspergillus flavus	Cr	Up to 74% of Cr removal	Huang et al. (2022)
Aspergillus fumigatus	As, Cr, Cu, Mn	Efficient in the removal of As (77%), Mn (71%), Cr (60%), and Cu (52%)	Hassan et al. (2020)
Aspergillus fumigatus	Cr, Cd, Cu, Ni	About 37–98% of PTEs were accumulated	Dey et al. (2016, 2020)
Aspergillus penicillioides	Pb	At pH 8.85 and 32 °C, 73% of Pb was bio-absorbed from the contaminated wastewater	Paria et al. (2022)
Funneliformis mosseae	Cu, Pb, Zn	Removed up to 68% of PTEs from the soil	Yang et al. (2015b), Thioub et al. (2019), Adeyemi et al. (2021)
Penicillium spp.	Hg	Enhanced the Hg removal up to 26%	Cheng et al. (2020)

2.3. Phycoremediation

Phycoremediation refers to the ability of phototrophic algae to selectively accumulate PTEs from polluted sites (Chakdar et al., 2022). Phycoremediation techniques are generally efficient due to algae's robust structure, which is often small in size with higher surface area (Soni et al., 2019; Sharma and Shukla, 2021). Phycoremediation has several advantages over other microorganism bioremediation due to tolerance and high accumulation of toxic elements (Salama et al., 2019). The microalgae remediate PTEs in the polluted sites in two stages. First, the PTEs are passively absorbed by the cell surface (biosorption) and later, these toxic elements accumulate inside the cell by diffusion (bioaccumulation). Previous studies have been conducted on utilising microalgae as a possible alternative method to remediate PTEs-polluted sites (Carullo et al., 2018; Hoang et al., 2019; Rugninia et al., 2017; Stefan et al., 2019).

Moreover, Nugroho et al. (2017) reported that about 90% of Cu was adsorbed into *Scenedesmus subspicatus*, indicating algae's effectiveness for bioremediation. Besides that, Bauenova et al. (2021) found that Cd (0.3 mg L^{-1}) and Cr (30 mg L^{-1}) were accumulated in the cytoplasm of *Parachlorella kessleri*. Meanwhile, *Desmodesmus pleimorphus*, *Chlorella vulgaris* and *Scenedesmus obliquus* can remove Cd from the contaminated environment (Alam et al., 2015; Monteiro et al., 2010; Shanmugam et al., 2018). On the other hand, *Navicula submuscular* and *Chlorella vulgaris* were used to remediate Cr (Suray and Putri, 2015; Cherifi et al., 2016) whilst *Aphanothece* spp. was successfully utilised for Cd remediation (Satya et al., 2020). Nevertheless, *Chlorella sorokiniana* has also been reported to withstand a broad range of temperatures and adverse conditions with Cd accumulation up to 11232 mg kg^{-1} (Leon-Vaz et al., 2021; Liang et al., 2017; Raikova et al., 2019) (see Table 3).

2.4. Phytoremediation

Phytoremediation is a promising green in-situ remediation technique for growing plants on PTEs-contaminated sites to remove, degrade and detoxify the toxic metals (Antoniadis et al., 2017; Cristaldi et al., 2017; Sarwar et al., 2017; Sarma et al., 2021). Phytoremediation uses plants including hyperaccumulators to absorb PTEs from the soil (Bian et al., 2017; Li et al., 2018; Luo et al., 2018; Tian et al., 2022). Hyperaccumulators are plants that act as bioreactors that can withstand and accumulate high amounts of PTEs without any damage (Pandey et al., 2015; Redondo-Gomez, 2013; Van der Ent et al., 2013). The PTEs can be transferred into the harvestable aboveground aerial parts and/or stabilised in the rhizosphere area (Striker and Colmer, 2016; Yang et al., 2014).

Various plant species were studied to determine phytoremediation capability in remediating different types of PTEs (Table 4). *Amaranthus hypochondriacus* L. (Cui et al., 2021; Sun et al., 2020; Tai et al., 2018; Wang et al., 2019; Xie et al., 2020; Yu et al., 2020), *Amaranthus caudatus* (Cay, 2016), *Sedum alfredii* (Li et al., 2012; Wang et al., 2012; Tao et al., 2020) and *Lolium multiflorum* (Liu et al., 2017) were used to remediate soil contaminated with cadmium (Cd). On the other hand, *Helianthus*

Table 3
PTEs remediation capabilities among various types of algae species.

Algae species	PTE tested	Observations	References
***Aphanothece* spp.**	Cd	Recorded 60.24 mg L^{-1} removal of Cd	Satya et al. (2020)
Chlorella sorokiniana	Cd	Able to grow on various temperature, conditions and accumulation of Cd (11232 mg kg^{-1})	Liang et al. (2017), Raikova et al. (2019), Leon-Vaz et al. (2021)
Chlorella vulgaris	Cd, Cr	High tolerance to Cd and able to extract up to 21.50 mg g^{-1}, while 15–45 mg L^{-1} of Cr was extracted	Alam et al. (2015), Suray and Putri (2015)
Desmodesmus pleimorphus	Cd	The biomass reached 61.2 mg g^{-1} in one day after the removal of Cd	Monteiro et al. (2010)
Navicula subminuscula	Cr	Resistance to and removal of high concentration of Cr (98%)	Cherifi et al. (2016)
Parachlorella kessleri	Cd, Cr	Cd (0.3 mg L^{-1}) and Cr (30 mg L^{-1}) found accumulated in the cytoplasm	Bauenova et al. (2021)
Scenedesmus obliquus	Cd	High tolerance to Cd and enlarged vacuoles	Shanmugam et al. (2018)
Scenedesmus subspicatus	Cu	About 90% of Cu adsorbed in the cell surface	Nugroho et al. (2017)

Table 4
Various types of plant species used for PTEs phytoremedation.

PTEs	Plant used		References
	Scientific name	Common name	
As	*Pteris vittata*	Chinese brake fern	Fu et al. (2017), Abbas et al. (2018), Cai et al. (2019), Abou-Shanab et al. (2020)
Cd	*Amaranthus hypochondriacus* L.	Prince's-feather	Tai et al. (2018), Wang et al. (2019), Sun et al. (2020), Xie et al. (2020), Yu et al. (2020), Cui et al. (2021)
	Amaranthus caudatus	Love-Lies-Bleeding	Cay (2016)
	Sedum alfredii	NA	Li et al. (2012), Wang et al. (2012), Tao et al. (2020)
	Lolium multiflorum	Italian ryegrass	Liu et al. (2017)
Cr	*Chrysopogon zizanioides* *Vetiveria zizanioides*	Vetiver grass	Misinire et al. (2021) Kumar et al. (2013), Singh et al. (2015)
	Tagetes erecta	Aztec marigold	Coelho et al. (2017)
Cu	*Corchorus capsularis*	White jute	Parveen et al. (2020)
	Helianthus annuus L.	Sunflower	Hattab-Hambli et al. (2016)
	Cymbopogon citratus	Lemongrass	Gautam et al. (2017), Kumar et al. (2019)
	Salix spp.	Willow	Yang et al. (2015a), Cao et al. (2022), Yang et al. (2021c)
Pb	*Brassica juncea*	Mustard green	Kohli et al. (2018), Soares et al. (2020), Rathika et al. (2021)
	Brassica napus L.	Rapeseed	Bilal Shakoor et al. (2014)
	Spinacea oleracea L.	Spinach	Khan et al. (2016)
	Eucalyptus globulus	Blue gum	Luo et al. (2017)
	Sorghum bicolor	Great millet	Rathika et al. (2020)
Zn	*Sedum plumbizincicola*	NA	Li et al. (2014), Li et al. (2018)
	Brassica napus L.	Mustard green	Belouchrani et al. (2016)
	Brassica juncea	Rapeseed	Chaudhry et al. (2020)
	Tagetes erecta L.	Aztec marigold	Madanan et al. (2021)
	Pfaffia glomerata	NA	Huang et al. (2021)

Note: NA: Not available.

annuus L. (Hattab-Hambli et al., 2016), *Salix* spp. (Yang et al., 2015a, 2021; Cao et al., 2022), *Eucalyptus globulus* (Luo et al., 2017) *Tagetes erecta* L. (Aztec Marigold) (Madanan et al., 2021) and *Pfaffia glomerata* (Huang et al., 2021) were used to remediate Cu, Pb and Zn. Moreover, *Sedum plumbizincicola* (Li et al., 2014, 2018), *Brassica juncea* (Belouchrani et al., 2016) and *Brassica napus* (Chaudhry et al., 2020) were applied with Zn contamination. Other studies reported that *Pteris vittata* (Abbas et al., 2018; Abou-Shanab et al., 2020; Cai et al., 2019; Fu et al., 2017) had shown promising findings in remediating As contamination.

Besides that, *Chrysopogon zizanioides*, commonly known as vetiver grass and *Tagetes erecta* L. are used in the phytoremediation of Cr (Coelho et al., 2017; Kumar et al., 2013; Singh et al., 2015). *Brassica juncea* (mustard green), *Brassica napus* (rapeseed), *Spinacea oleracea* L. (spinach) and *Sorghum bicolor* (great millet) are among the hyperaccumulators that were used to remediate Pb polluted soils (Bilal Shakoor et al., 2014; Khan et al., 2016; Kohli et al., 2018; Rathika et al., 2020; Rathika et al., 2021; Soares et al., 2020; Li et al., 2018) whereas *Corchorus capsularis* (Parveen et al., 2020) and *Cymbopogon citratus* (lemongrass) (Gautam et al., 2017; Kumar et al., 2019) were used to remediate soil with Cu.

3. Overview of phytoremediation

The phytoremediation technique is suitable for lower concentration level of contamination that is carried out over a longer duration without causing disturbance to the site (Kumarathilaka et al., 2018; Missimer et al., 2018; Saha et al., 2017). The level of PTEs accumulation by plants varies due to a few factors such as type of plants (Tai et al., 2018; Yang et al., 2015a), plant biomass (Ali et al., 2020; Kindtler et al., 2019), type of soil (Rosenfeld et al., 2018; Wang et al., 2021) and environmental conditions (Laidlaw et al., 2015; Pilipovic et al., 2019; Salam et al., 2016). There are several mechanisms in phytoremediation: phytoextraction, phytostabilisation, phytovolatilisation, phytotransformation and rhizofilitration (Fig. 1).

3.1. Types of phytoremediation

3.1.1. Phytoextraction

Phytoextraction is the most common and economically feasible phytoremediation technique that is remarkable for removing PTEs in contaminated soil and water (Abbas and Abdelhafez, 2013; Patra et al., 2019; Yang et al., 2016). The extraction and accumulation of PTEs in shoots without affecting the soil properties is known as phytoextraction

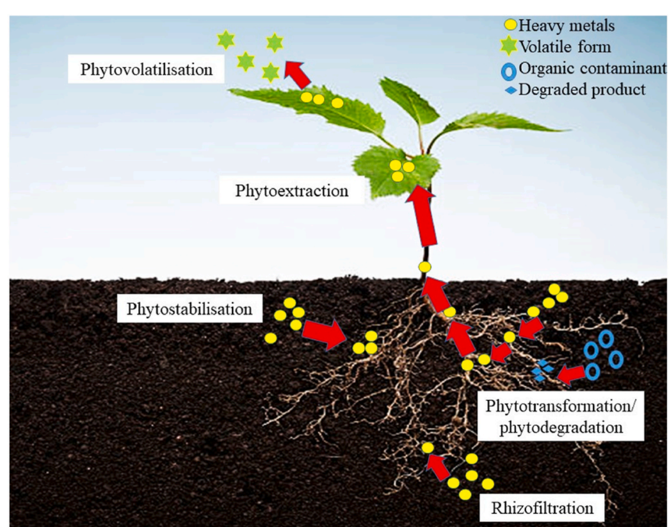

Fig. 1. Plants employ various mechanisms for PTEs remediation in the soil.

(Chandra et al., 2015; Chandra & Kumar, 2017; Zhu et al., 2020). The accumulated PTEs could be removed from the contaminated area by harvesting the plants (Ali et al., 2013; Liu et al., 2020). The plants extract the toxic elements and subsequently translocate them into assorted plant parts such as roots, stems and other edible sections (Haller et al., 2017; Liu et al., 2012; Wang et al., 2019).

Generally herbaceous plants for example, *Dysphania botrys*, *Lotus hispus*, *Plantago lanceolata* are preferred for the phytoextraction process as well as woody plants are also getting a wide acceptance recently (Li et al., 2012). Fast growth, high biomass and easy cultivation are some of the positive characteristics for phytoextraction plants (Patra et al., 2018a,b). Furthermore, plants selected for phytoextraction should also have high tolerance capacity towards PTEs (Ahmed, 2015; Kacalkova et al., 2015; Rezania et al., 2016). Plants such as *Chrysopogon zizanioides* are used as promising phytoextractors for As, Hg and Pb among the graminoids (Bernardino et al., 2016; Lomonte et al., 2014). In addition, *Chrysopogon zizanioides* was also proved as an excellent agent in phytoextraction of Fe, Mn, Zn and Cr (Banerjee et al., 2019; Melato et al., 2016). Meanwhile, willows (Haller and Jonsson, 2020) and poplars (Vollenweider et al., 2011) are among the well-known hardwood plant species that are capable for phytoextraction of PTEs such as Cd, Cr, Cu, Ni, Pb and Zn. *Helianthus annuus* L. showed about 71–95% of Pb and Cd (Alaboudi et al., 2018) and 60% of Cr (Bahadur et al., 2016, 2017) could be extracted from the contaminated soil. Two species of the genus Brassica such as *Brassica juncea* and *Brassica rapa* were also previously utilised to extract Pb, Zn and Cd (Ali et al., 2021; Benavides et al., 2021; Dhaliwal et al., 2022). Some previous studies also reported that soil bacteria could enhance the phytoextraction of Pb (Egamberdieva et al., 2016; Manzoor et al., 2019) but require a longer duration.

3.1.2. Phytostabilisation

Phytostabilisation is a process where the PTEs are immobilised within the plant's root (Girkina et al., 2018; Tangahu et al., 2011). PTEs can be switched from a toxic state to a non-toxic or low-toxic state through phytostabilisation (Li et al., 2019; Sharma and Pandey, 2014). Besides, the plant root exudates have stabilised and demobilised the toxic elements, thus preventing the movement of PTEs around the rhizosphere area of the plant (Lv et al., 2018; Van Oosten and Maggio, 2014). As the PTEs are absorbed and attached to the rhizosphere parts of the plants, it decreases the bioavailability of these elements (Ali et al., 2013).

An ideal plant for phytostabilisation should acquire a bio-accumulation value of lower than 1, highly tolerant to few environmental stresses (temperature, drought, salinity and soil nutrient) and with an extensive plant root system (Galal et al., 2017; Padmavathiamma and Li, 2007; Shackira and Puthur, 2019). In this context, Midhat et al. (2018) used *Medicago sativa* L. to remediate acidic mining sites polluted with different types of PTEs as it has increased the phytostabilisation capacity and acted as a good solution for soil neutralisation. Besides that, Zeng et al. (2018) also reported that ornamental plants such as *Osmanthus fragrans*, *Ligustrum vicaryi* L. and *Cinnamomum camphora* showed promising results to phytostabilise soil polluted with Cd. Similarly, Sricoth et al. (2018) found that *Eichhornia crassipes* and *Pistia stratiotes* are suitable for Cd and Zn phytostabilisation due to its bioconcentration and translocation factors results. Moreover, *Athyrium wardii* (Hook) is used to stabilise Pb (Zhao et al., 2016) whereby Constantinescu et al. (2019) revealed that *Agrostis capillaris* is suitable for phytostabilisation of Cd, Cu and Zn contaminated soils. Furthermore, rye plant (*Secale cereal* L.) are efficient to phytostabilise As and Cd due to its high resistance and tolerance to a wide range of climatic conditions, higher translocation of PTEs in the shoot and metal sequestration in the plant cells (Alvarez-Ayuso et al., 2016; Fresno et al., 2020). In addition, the black locust (*Robinia pseudoacacia 'Nyirsegi*), a pollutant-tolerant woody crop was also reported to have the ability to stabilise Cu and As (Wawra et al., 2018; Zhu et al., 2016).

3.1.3. Phytovolatilisation

Phytovolatisation is usually used to convert and release low toxic elements such as arsenic, mercury and selenium into the atmosphere through volatilisation (Guarino et al., 2020; Sharma et al., 2015). The PTEs in the phytovolatilisation process are absorbed by the root system, translocated to the shoot system, and finally released into the atmosphere through the diffusion (San Miguel et al., 2013; Wang et al., 2012). The PTEs are released into the atmosphere in less toxic and volatile forms through the stomatal leaves (Gavrilescu, 2022; Van Oosten and Maggio, 2014). Phytovolatilisation also involved assimilating PTEs into organic compounds through the biomethylation process and as a result, the biomolecules could eventually reach to the atmosphere (Guarino et al., 2020). PTEs can be subject to biological transformation by alternating their chemical forms or changing their oxidation state biotically or abiotically (Ruppert et al., 2013). This process can remarkably alter its bioavailability and toxicity in the soil. However, phytovolatilisation is a highly debatable remediation process as it does not entirely remove the pollutant but only transfers it to another phase (Prabha et al., 2021). Plants like *Arabidopsis thaliana*, *Liriodendron tulipifera*, *Arachis hypogaea*, *Populus deltoids* and *Oryza sativa* showed the capability to absorb Hg(II) from the contaminated sites and converted it into much less toxic Hg(0) (Pant and Singh, 2014). Besides, *Pteris vittata* (Sakakibara et al., 2010) and *Polypogonmon speliensis* (Ruppert et al., 2013) were reported to phytovolatise As contaminated soil in the forms of dimethylchloroarsine [$AsCl(CH_3)_2$] and pentamethylarsine [$As(CH_3)_5$] and subsequently released methane gas into the atmosphere.

3.1.4. Rhizofiltration

Rhizofiltration is generally used in the water remediation technique to reduce the contamination level in estuary and wetland areas through adsorption and absorption (Zhang et al., 2010). Rhizofiltration is highly effective in remediating soil and water contaminated with pollutants rich in nitrogen and phosphorus (Mithembu, 2012). The plants used in rhizofiltration require the ability to take a large amount of water from the soil for PTEs precipitation on the surface of the roots (Awa and Hadibarata, 2020; Benavides et al., 2018). Besides, the PTEs around the rhizosphere were also absorbed by the roots, and fixation takes place inside the roots as an insoluble form (Oustriere et al., 2017; Sampaio et al., 2019). Both hydrophytes and mesophytes are used as hyperaccumulator plants through the rhizofiltration technique (Erakhrumen and Agbontalor, 2007; Prasad, 2007). Galal et al. (2018) examined the phytoremediation capability of a macrophyte, *Vossia cuspidata* to absorb and bioaccumulate PTEs pollutants in wetlands. Besides, sunflower and mustard plants are used to remediate Pb-contaminated water through precipitation method in the root (Kafle et al., 2022). Plants such as *Azolla caroliniana* and *Callitriche lusitanica* used to uptake As while, *Callitriche stagnalis*, *Fontinalis antipyretica* and *Lemna minor* are used for remediation of Uranium (U) (Favas et al., 2012; Pratas et al., 2012). In addition, the removal of PTEs through the rhizofiltration mechanism was also done through microbial activities (Sikhosana et al., 2020). Dimitroula et al. (2015) reported that specific strains of *Pseudomonas* and *Ochrobactrum* on the plant root surface had reduced Cr(VI) to Cr(III) whereby the reduced Cr quickly precipitated in the root generating a Cr-free site.

3.2. Challenges in current phytoremediation practices

In the phytoremediation approach, several abiotic and biotic factors such as soil properties, plant clones, growth conditions, atmospheric carbon dioxide and plant-pathogen have influenced phytoremediation efficiency and eventually render some challenges to the procedures (Ekta and Modi, 2018; Ramachandra et al., 2020; Shackira et al., 2021) (Table 5). The low remediation rate has been identified as a critical limitation for phytoremediation from being applied into commercial and large scales (Hrynkiewicz et al., 2018; Kang et al., 2022; Samsuri

Table 5
Current challenges and limitations of phytoremediation treatment.

Challenges in current phytoremediation procedure				
Low remediation rate (Hrynkiewicz et al., 2018; Samsuri et al., 2019; Kang et al., 2022)	Restriction on plant's growth (Huang et al., 2004, 2005; Akhtar et al., 2017)	Duration of the treatment (Chloe and Black, 2014; Siyar et al., 2020)	Different level of contamination level (Favas et al., 2014; Akhtar et al., 2017)	Low bioavailability of metal (Li et al., 2018; Triphati et al., 2020; Liu et al., 2022)
Plant's root system (Thakur et al., 2016; Shen et al., 2022; Tiodar et al., 2021)	pH of the soil (Zeng et al., 2017; Ashraf et al., 2019; Javed et al., 2019)	Interaction and diverse remediation pathways (Agnello et al., 2016; Wang et al., 2021; Cao et al., 2022)	Climate factors and anoxic environment (Yang et al., 2017; Cao et al., 2022)	Heterogeneity of targeted soil (Gawronski et al., 2017; Agarwal et al., 2019)

et al., 2019). The low remediation rate was attributed to restricted plant survival and growth in contaminated areas that limit the catabolism of PTEs (Akhtar et al., 2017; Huang et al., 2004, 2005). Following this, the total duration taken for phytoremediation treatment is another strong constraint (Chloe and Black, 2014; Couto et al., 2015; Siyar et al., 2020). Past studies also indicated that the effectiveness of phytoremediation depends largely on the targeted sites of contamination level. Studies conducted by Evangelou et al. (2007), Saraswat and Rai (2011), Favas et al. (2014) and Akhtar et al. (2017) showed that highly contaminated sites usually have moderate phytoremediation effectiveness.

Besides, the low bioavailability of the PTEs in the targeted area has also limited the full potential and capability of phytoremediation (Guo et al., 2021; Khalid et al., 2017; Lu et al., 2020; Wang et al., 2019; Zhu et al., 2020). The PTEs must be available within the range of the plant's root system and phytoavailable forms for the absorption process to take place (Schwitzguebel, 2017; Tripathi et al., 2020). Moreover, the depth of treatment to the root zone and shorter root system were also reported as a limitation for the traditional phytoremediation method as the shorter root system often affects the effectiveness of the process (Capuana, 2011; Thakur et al., 2016; Tiodar et al., 2021; Shen et al., 2022). Following that, the pH of the soil was also reported to be a significant factor in the dissolution of pH to increase the bioavailability of PTEs (Ashraf et al., 2019; Javed et al., 2019; Zeng et al., 2017). Nevertheless, the interaction between various contaminants and diverse remediation pathways increases the complexity of the phytoremediation process (Agnello et al., 2016; Cao et al., 2022; Wang et al., 2021).

Apart from that, past studies also demonstrated that climatic factors (Dhanwal et al., 2017; Karami et al., 2010; Mahar et al., 2016) and an anoxic environment (Cao et al., 2022; Yang et al., 2017) such as flood could affect the variation pattern of metal accumulation among different plants. Furthermore, the variation of environmental conditions between on-site (in-situ) and off-site (ex-situ) phytoremediation treatment of the same target samples could also constrain its application commercially (Agarwal et al., 2019; Corwin et al., 2006; Gawronski et al., 2017; Van Dillewijn et al., 2007). As a result, numerous new studies in the phytoremediation field are still being conducted periodically to improve its performance and overcome the limitations that hinder its practical application.

4. Roles of chelating agents in phytoremediation

4.1. Chelating agents and their uses

Chelation refers to the attaching a specific organic molecule with mineral or metal ions to form a metal complex (Kaushik, 2015; Martell, 1965; Tsang et al., 2012). In general, a chelating agent is a chemical reagent used to enhance the bioavailability of various PTEs in the soil such as Cu, Pb, Cd and others for metal translocation into a plant (Rena et al., 2022; Yang et al., 2021b). The limitations in the standard phytoremediation procedure, such as low decontamination rate, low bioavailability of targeted metals and reduction of plant growth due to metal toxicity, have led to many new scientific studies to enhance phytoremediation for PTEs clean-up in the soil (Ghazaryan et al., 2021; Gul et al., 2021). The chelating agents act as a chemical bond to form metal complex(es) which help to enhance the bioavailability of PTEs in the soil and facilitate the phytoremediation process for metal translocation into the roots and aboveground part of the plants (Huang et al., 2019; Rena et al., 2022).

The use of chelating agents causes metal solubilisation in soils, which enhances the dissolution of metals due to the formation of metal complexes (Shahid et al., 2014). This metal-chelate complex formation has increased the capacity to mobilise the metals in soil and improved the translocation of metals to the roots and aboveground plant parts. According to Shahid et al. (2014), adding ethylene-diamine-tetraacetic acid (EDTA) may increase up to 600 times metal solubilisation, depending on the type of metal and soil conditions. In the process of chelant-assisted phytoremediation, two primary mechanisms were involved in increasing the mobilisation of metals by chelating agents. Firstly, the chelates are adsorbed onto the surfaces of metals when they come in contact with the metals in the soil, which destabilises the weak bond between metal and oxygen in mineral structures (Leštan et al., 2008; Shahid et al., 2014). This adsorption process is vital for forming a stable ternary surface complex between the chelating agent and metals (Saifullah et al., 2015; Shahid et al., 2014; Tsang et al., 2012).

On the other hand, the second mechanism involves the dissolution of metals ion from minerals attributed to ligand exchange reactions where the surface metal-oxygen bonds are broken after the adsorption of chelating agents on the mineral surface. Hence, an intermediate complex such as Fe-EDTA and Pb-EDDS formed in chelate-assisted phytoremediation (Sarwar et al., 2017; Shahid et al., 2014; Tsang et al., 2012). Therefore, with the presence of this metal complex(es), it will generally increase in the bioavailability of metals and subsequently speed up the translocation process in plant's rhizosphere for uptake by plant roots (Leštan et al., 2008; Shahid et al., 2014; Wenger et al., 2008). Thus, the enhancement of phytoextraction efficiency through the addition of chelating agents such as ethylene-diamine-tetraacetic acid (EDTA), N, N-dicarboxymethyl glutamic acid tetrasodium salt (GLDA) and ethylene-diamine-N,N′-disuccinic acid (EDDS) were explored (Gul et al., 2020; Yang et al., 2021b). The addition of EDTA enhanced the phytoextraction of Cd and Pb by *Pelargonium hortorum* (Gul et al., 2020). Wang et al. (2019) and Yang et al. (2021a) suggested that the addition of GLDA could enhance Cd and As's bioavailability effectively. EDDS was found to improve the phytoextraction of Ni by *Coronopus Didymus* L. from polluted soils (Sidhu et al., 2018). In addition, Hussain et al. (2019) utilised citric acid for phytoextraction of metals from multi-metal contaminated soil by soybean plant aided by *Kocuria rhizophila* (Glycine max L.).

4.2. Types of chelating agents

Several chelating agents have been explored to examine their potential to enhance PTEs phytoremediation in soil (Aghelan et al., 2021; Kaushik, 2015; Mousavi et al., 2021). Chelating agents can be divided into two groups: natural and synthetic. Several synthetic agents such as ethylene-diamine-N,N′-succinic acid (EDDS) and diethylene-triamine pentaacetate (DTPA) have been employed to enhance bioavailability, uptake and root-shoot translocation (Barrutia et al., 2010; Gul et al., 2021; Tahmasbian and Safari Sinegani, 2016). Among all chelating agents, it has been found that EDTA is an efficient and most widely used chelator that increases plant potential in refining PTEs-contaminated areas by boosting up ion mobility, solubility and bioavailability at soil solution and root uptake stages (Gul et al., 2021; Mousavi et al., 2021). However, synthetic chelators may pose some environmental risks, such as leaching due to high mobility and durability in the soil (Mousavi et al., 2021). Hence, recently, there has been an increased interest in employing natural and/or organic chelants for phytoremediation technology (Chen et al., 2020; Huang et al., 2019; Singh et al., 2016). The natural and/or organic chelants, which consists of organic acids such as gluconic acid, citric acid (CA), 2,3-dihydroxy benzoic acid and homo citric acid are highly biodegradable in the soil which makes them suitable to be used on a large scale (Kaushik, 2015; Mousavi et al., 2021). Besides, metallothioneins (MTs) and phytochelatins (PCs) are also a part of natural chelators that have the capability to increase the mobility of PTEs in the soil (Gul et al., 2021).

Table 6 lists chelating agents used in phytoremediation for different metal types and plants. Based on previous studies, the application of chelating agents in phytoremediation technology has proven to improve the phytoremediation potential for diverse types of metals and plants in terms of PTEs accumulation (Huang et al., 2019; Rena et al., 2022; Tahmasbian and Safari Sinegani, 2016), plant growth (Aghelan et al., 2021; Ghazaryan et al., 2021) and microbial community in soil (Gul et al., 2021)..

4.3. Influence of chelating agent on phytoremediation

4.3.1. PTEs uptake in contaminated soil

Several previous studies have reported enhanced phytoremediation with the application of various chelating agents (Liang et al., 2021; Mousavi et al., 2021; Yang et al., 2021b). It has conclusively been shown that adding chelating agents for phytoremediation of PTEs could increase the toxic metal uptake and translocation from the root to the aboveground biomass (Jiang et al., 2019; Ng et al., 2019; Saifullah et al., 2015). The increase of PTEs uptake by the plants might have resulted from the capability of chelators to enhance the bioavailability of metal and subsequently increase the metal contents in the soil (Aghelan et al., 2021; Hseu et al., 2013). Table 7 shows that the application of various chelating agents in PTEs phytoremediation has increased the PTEs concentration in the roots and stems of various plants. Huang et al. (2019) found that the addition of DTPA in Pb phytoextraction using *Zea mays* L. led to a significant increase in metal uptake, bioconcentration factor (BF) and translocation factor (TF) by 26%, 26% and 45%, respectively as compared to the control plant (without a chelating agent). Similarly, Aghelan et al. (2021) also found that the application of chelators such as EDTA, CA and salicylic acid (SA) resulted in a significant increase in Pb content in the root of *Amaranthus caudatus* L. as compared to the control treatment.

Most of the studies indicated a high potential of using EDTA to increase the metal uptake or enhance the translocation of metal to plant parts (Aghelan et al., 2021; Jiang et al., 2019; Saifullah et al., 2015; Shahid et al., 2014). This may be related to the potential of EDTA to enhance root metal flux through apoplast in the root and considerably increase the metal uptake from root and shoot (Aghelan et al., 2021; Tahmasbian and Safari Sinegani, 2016; Zhang et al., 2016). EDTA possesses negatively charged hydroxyl or carboxyl groups, forming stable chelating compounds with positively charged metal, facilitating metal uptake and accumulation by plants (Jiang et al., 2019; Yang et al., 2021b). Zhang et al. (2016) compared the removal efficiency of PTEs in *Ricinus communis* L. using three different chelates such as CA, EDDS and EDTA; whereby EDTA was found to be effectively accumulated higher concentration of metals in plants, which increased by 4.4- and 8.9-fold as compared to the control treatments for Cd and Pb, respectively. Mousavi et al. (2021) demonstrated that the highest concentration of Cd in both shoots (0.70 mg kg^{-1}) and root (1.10 mg kg^{-1}) was observed after adding 1 mM EDTA as compared to MA treatment. Due to the efficiency of EDTA in enhancing the phytoextraction performance for different metal types and plant species, it was recognised as the most effective chelate to increase PTEs uptake by plants (Aghelan et al., 2021; Leštan et al., 2008; Shahid et al., 2014). However, recent developments in the phytoremediation studies have heightened the need to search for an alternative to EDTA due to its low biodegradability, which may affect the environment after the post-treatment of phytoremediation process (Ali et al., 2013; Leštan et al., 2008; Sarwar et al., 2017).

Table 6
Common types of chelating agents used for PTEs phytoremediation (Kaushik et al., 2015; Yang et al., 2021b).

Types of chelating agents	
Organic chelating agents	**Synthetic chelating agents**
Citric acid (CA)	Aspartate dibutyric acid ether (AES)
Maleic acid (MA)	Diethylene-triamine pentaacetate (DTPA)
Nitrilotriacetic acid (NTA)	Ethylene glycol tetraeacitic acid (AGTA)
Oxalic acid (OA)	Ethylene-diamine tetraacetic acid (EDTA)
Salicylic acid (SA)	Ethylene-diamine-N,N′-succinic acid (EDDS)
	Imino-disuccinic acid (IDSA)
	Tetrasodium-N,N-diacetate (GLDA)

The EDTA-metal complexes can be toxic to soil microorganisms and plants, allowing them to be persistent in the environment for a long period of time (Aghelan et al., 2021; Leštan et al., 2008; Mousavi et al., 2021). Therefore, the use of EDTA as a chelating agent in phytoremediation is least preferred, especially for commercial field applications (Aghelan et al., 2021; Zhang et al., 2016).

In recent years, the application and development of biodegradable chelating agents such as nitrilotriacetic acid (NTA), CA and others are getting more popular (Leštan et al., 2008; Zhang et al., 2016). In contrast to the synthetic chelating agents, natural chelating agents such as organic acid are currently preferred owing to their advantages, such as having low biological toxicity, less environmental impact and high biodegradability in soils which makes them suitable to be applied for the large-scale of phytoremediation (Mousavi et al., 2021; Yang et al., 2021b). Studies have revealed that various organic acids such as NTA, maleic acid (MA), oxalic acid (OA) and CA can successfully form metal complexes with various types of PTEs such as Cu (Ghazaryan et al., 2021), Cd (Hseu et al., 2013; Yang et al., 2021b), Cr (Qureshi et al., 2020), Pb (Amir et al., 2020; Huang et al., 2019; Jiang et al., 2019), U (Hu et al., 2019) and Hg (Amir et al., 2020). Additionally, higher metal uptake performances were reported with the use of organic chelating agents as compared to EDTA (Chen et al., 2020; Yang et al., 2021b; Zhang et al., 2016). This might be due to the selectivity of different chelating agents and metal, as various chelating agents may have affected the metals removal efficiency and translocation from roots to shoots (Chen et al., 2020; Leštan et al., 2008). Chen et al. (2020) indicated that both EDDS and CA have a good influence on the removal efficiency of U and Cd in the soil, whereby the translocation factor of Cd was found to reach a maximum (0.65) with the application of EDDS. On the other hand, CA showed a higher translocation factor (0.033) of U uptake compared to EDDS and OA treatment (Chen et al., 2020). Moreover, Yang et al. (2021b) found that the degradable chelating agents such as GLDA and NTA have significantly impacted the Cd uptake from the soil as compared to EDTA soil treatment. The concentration of Cd in the roots with 3 mmol kg^{-1} of GLDA, NTA and EDTA treatments were observed at 9.20 mg kg^{-1}, 10.50 mg kg^{-1} and 7.80 mg kg^{-1}, respectively. GLDA and NTA treatments have resulted in higher metal uptake in the plant, implying that these organic acids could perform better than EDTA. Similarly, Zhang et al. (2016) confirmed that EDDS could be a good alternative for EDTA to enhance the phytoremediation of Cd pollution in soil. The exciting results demonstrated by several past studies confirmed the potential and feasibility of using organic chelating agents as an alternative to the most widely used chelating agent, EDTA (Chen et al., 2020; Yang et al., 2021b; Zhang et al., 2016). However, the optimisation of the phytoextraction process in terms of desirable plant species, growth conditions and targeted metals should be conducted in the near future as the PTEs uptake is largely influenced by different types

Table 7
Comparison of various chelating agents on PTEs accumulation in the root and shoot among different plant species.

Chelating agents	Chelator concentration (mg/kg)	PTEs	PTEs concentration (mg/kg)	Plant species	Root concentration (mg/kg)	Shoot concentration (mg/kg)	References
CA	955	Cu	NA	*Artemisia absinthium* L.	367.92	109.44	Ghazaryan et al. (2021)
	960	Cr	100.00	*Ricinus communis* L.	500.00	250.00	Qureshi et al. (2020)
	1921	U	18.00	*Macleaya cordata*	NA	NA	Hu et al. (2019)
	960	Pb	NA	*Typha latifolia* L.	3569.33	2744.33	Amir et al. (2020)
		Hg			3809.33	4111.00	
DTPA	983	Pb	552.30	*Zea Mays* L.	504.87	243.92	Huang et al. (2019)
EDDS	1461	Pb	274.16	*Ricinus communis* L.	113.00	9.72	Zhang et al. (2016)
	730	Cd	2.50	*Ipomoea aquatic* Forsk	0.18	1.50	Hseu et al. (2013)
			5.00		0.20	1.03	
			10.00		0.20	1.29	
			20.00		0.09	0.82	
			30.00		0.08	0.66	
	1461	Cd	15.00	*Helianthus annuus* L.	108.27	70.38	Chen et al. (2020)
EDTA	730	Pb	200.00	*Amaranthus caudatus* L.	0.50	0.02	Aghelan et al. (2021)
			400.00		0.74	0.03	
	500	Pb	500.00	*Arundinaria argenteostriata*	229.30	94.83	Jiang et al. (2019)
			1000.00		769.83	119.56	
			1500/00		1117.70	245.27	
	2000	Pb	1220.72	*Helianthus anuss*	246.00	145.75	Tahmasbian and Safari Sinegani (2016)
	1461	Pb	274.16	*Ricinus communis* L.	284.00	121.00	Zhang et al. (2016)
		Cd	3.53		15.80	1.82	
	146	Cd	100	*Abelmoschus esculentus* L.	0.60	1.00	Mousavi et al. (2021)
	292				0.70	1.10	
	876	Cd	6.53	*Zea mays* L.	7.60	6.40	Yang et al. (2021b)
	1752				5.00	3.10	
	2628				3.90	2.30	
	1461	Zn	NA	*Brassica juncea*, Coss	610.00	400.00	Guo et al. (2019)
NTA	573	Cd	6.53	*Zea mays* L.	10.90	6.80	Yang et al. (2021b)
	1146				6.10	4.60	
	1720				5.00	3.10	
	477	Cd	2.50	*Ipomoea aquatic Forsk*	0.17	1.56	Hseu et al. (2013)
			5.00		0.18	1.38	
			10.00		0.15	1.02	
			20.00		0.12	0.96	
			30.00		0.12	0.81	
OA	225	Cd	10.71	*Sedum alfredii*	365.00	625.00	Liang et al. (2021)

Note: NA: Not available.

of soil, plant and metal as well as soil pH and cation exchange capacity (CEC) in the soil (Leštan et al., 2008; Shahid et al., 2014).

4.3.2. Plant growth and biomass

Some of the ideal plant characteristics that are commonly used for phytoremediation include greater ability to accumulate high amounts of PTEs, produce high biomass, able to withstand high concentration of toxic elements and with fibrous root system (Ali et al., 2013; Sarwar et al., 2017). Besides, high tolerance to chemical or organic chelating agents is crucial for chelate-induced phytoremediation. Plants may experience some phytotoxicity as a result of the high concentration of added chelating agents and the formed metal-complex in the soil, which consequently affects the performance of the plants to uptake toxic metals and contaminants in the soil (Chen et al., 2020; Hseu et al., 2013; Shahid et al., 2014). Henceforth, several plant properties such as plant height, root and shoot biomass, total dry weight, leaf number, leaf width and diameter have been examined to explore the effects on plant growth and biomass of various plant types for the enhanced phytoremediation using chelating agents (Ghazaryan et al., 2021; Hseu et al., 2013; Jiang et al., 2019). Moreover, plant biomass has always been a critical parameter in examining plants' capability as metal phytoextractors (Rathika et al., 2021).

However, some inconsistent effects on plant growth and biomass were reported in the previous studies after adding chelating agents to the plant (Aghelan et al., 2021; Houben et al., 2013; Rathika et al., 2021; Sinhal et al., 2010). Most of the previous studies found that the higher concentration of chelating agents that were used in enhanced phytoremediation has reduced plant biomass and impacted other plant physiological properties (Table 8). Hseu et al. (2013) showed that the plant's leaves had necrotic spots after adding 5.0 mmol kg^{-1} of EDDS whereby the shoot height was also significantly lower than the other soil treatments. Besides, the root and shoot biomass were reduced in both EDDS and NTA treatments. Similarly, Chen et al. (2020) also found that chelating agents such as EDDS may cause toxic effect to the plants decreasing shoot biomass by 3.28–30.31% while the root biomass by 0.73–23.81% as compared to those of the control treatment. Moreover, Yang et al. (2021b) identified that the using various types of chelant such as EDTA, NTA and IDSA (Iminodisuccininc acid) has significantly decreased the aboveground biomass of *Zea mays* L. with the increasing concentrations of the chelating agents.

The reduction in the plant biomass production may be attributed to the role of chelating agents, which increase the bioavailability of PTEs in the soils and consequently promote the uptake of toxic metals and led to the phytotoxicity of the plant (Ghazaryan et al., 2021; Sarwar et al., 2017; Yang et al., 2021b). Furthermore, the reduction in plant biomass might be related to the competition between PTEs and micronutrients such as Mg, Zn and Fe in the soil (Chen et al., 2020). The higher bioavailability of PTEs after the addition of chelating agents has increased the concentration of the targeted metal and affected the absorption of vital nutrients into the plant (Chen et al., 2020; Hseu et al., 2013; Yang et al., 2021b). In addition, applying chelating agents can also reduce the chlorophyll content and plant biomass. The metal-chelate complex formed after adding of chelating agents to the soil can minimise the amount of chlorophyll in the plant due to the

Table 8
Dry biomass of root and shoot of different types of plants in response to various chelate treatments.

Plant Species	Root Biomass (g pot^{-1})		Shoot Biomass (g pot^{-1})		Chelating agent		References
	Control	Treated	Control	Treated	Types	Concentration (mmol kg^{-1})	
***Amarathus caudatus* L.**	0.05	0.03	0.03	0.02	EDTA	2	Aghelan et al. (2021)
Artemisia absinthium	14.10	10.27	27.57	18.60	EDTA	2	Ghazaryan et al. (2021)
		11.30		22.66	CA	5	
		13.24		22.30	MA		
***Helianthus annuus* L.**	2.73	2.81	27.78	26.00	CA	5	Chen et al. (2020)
		2.35		26.90	OA		
		2.40		23.30	EDDS		
***Ipomoea aquatic* Forsk**	0.17	0.18	1.36	1.25	EDDS	5	Hseu et al. (2013)
		0.15		1.78	NTA		
***Ricinus communis* L.**	0.32	0.29	1.71	1.47	CA	5	Zhang et al. (2016)
		0.34		2.90	EDDS		
		0.32		1.52	EDTA		
***Tagetes patula* L.**	0.02	0.04	0.02	0.06	EDTA	2	Aghelan et al. (2021)
***Zea mays* L.**	0.17	0.28	0.58	0.77	EDTA	6	Yang et al. (2021b)
		0.29		0.92	AES		
		0.15		0.62	NTA		

suppression of chlorophyll synthesising enzyme (α-aminolevulinic acid dehydratase) activity (Chen et al., 2020). The inhibition of this enzyme activity might limit the water absorption surface and photosynthetic activity thus suppressing the overall plant growth (Ghazaryan et al., 2021). Moreover, the formation of a high concentration of metal-chelate complex can also impair pigment generation and increase oxidative damage of plant cells (Chen et al., 2020; Patra et al., 2020; Tipu et al., 2021).

Contrary to all that precedes, several studies found that the application of chelating agents has improved the plant biomass and growth along with the increase in accumulation and metal uptake into the plants (Aghelan et al., 2021; Houben et al., 2013; Rathika et al., 2021; Zhang et al., 2016). Aghelan et al. (2021) reported that the average root and stem length and fresh weight of *Amarathus caudatus* L. and *Tagetes patula* L. were increased in EDTA treatment as compared to control plants. Moreover, Zhang et al. (2016) identified that the dry biomass of two cultivars of *Ricinus communis* L. (Zibo-3 and Zibo-9) had significantly increased by 28.8% and 59.4% with the presence of EDDS as compared to control treatment. Also, Rathika et al. (2021) demonstrated that adding EDTA as chelating agents improved the biomass of *Brassica juncea*. The mixed findings on plant growth and several plant properties obtained by previous work on using chelating agents to enhance phytoremediation have been attributed to the various chelating agents (Yang et al., 2021b). Besides, the plant biomass can also be influenced by numerous factors such as quantity and type of chelant, plant species, contamination level as well as time and rate of application of chelating agents (Huang et al., 2019; Kafle et al., 2022; Saifullah et al., 2015; Tahmasbian and Safari Sinegani, 2016).

Among different chelants, EDTA has been seen as a good type of chelant due to the capability to enhance the phytoremediation efficiency in various types of plants and contaminated sites (Kafle et al., 2022; Oladoye et al., 2022; Yadav et al., 2018). However, the earlier work has reported that the use of EDTA may cause phytotoxicity to the plants whereby the addition of EDTA has led to a more drastic reduction in plant biomass and growth as compared to the other types of chelating agents (Ghazaryan et al., 2021; Zhang et al., 2016). Zhang et al. (2016) demonstrated that the EDTA significantly inhibited the growth of *Ricinus communis* L. as compared to the citric acid application, which had almost no effects on plant growth. Ghazaryan et al. (2021) concluded that EDTA could affect plant growth, unlike citric acid and malic acid. Besides, the correlation analysis has confirmed that EDTA application at varying dosages may reduce the dry biomass of *Artemisia absinthium* (Ghazaryan et al., 2021). Tahmasbian and Safari Sinegani (2016) also examined that the application of EDTA had influenced the plant root dry weight to be significantly lower (2.41 ± 0.80 g) in EDTA treatments as compared to the cow manure (4.33 ± 1.02 g) and poultry manure extracts (4.29 ± 2.49 g) treatments. As a result, various researchers have been exploring several strategies and alternatives to use EDTA as chelant due to the phytotoxicity effects on plants and low biodegradability in soil (Oladoye et al., 2022; Saifullah et al., 2015; Sarwar et al., 2017). Combining chemical amendments with different types of organic chelant at the optimum dosages is compelling and emerging in the field of phytoremediation studies. Furthermore, past studies have shown convincing results that applying mixed organic and synthetic chelating agents could enhance the phytoremediation efficiency of the plant as well as to alleviate the toxic effects from specific chelating agents (Ghazaryan et al., 2021; Rathika et al., 2021).

5. Treatment and disposal of plant biomass after phytoremediation

One of the main challenges in phytoremediation procedure is the disposal of a large number of post-remediation plant biomass with PTEs (Han et al., 2018; He et al., 2019; Lin et al., 2016). The fate of every plant used for phytoremediation has become one of the biggest concerns for the method. However, there are only a handful of studies on the effective method for handling plant biomass collected from the phytoremediation process (Attinti et al., 2017; Gong et al., 2018a; Zhang et al., 2021). The post-phytoremediation treatment and disposal of plant biomass are seriously required to decrease the deposition of waste into landfills (Lin et al., 2014; Rizwan et al., 2018; Wei et al., 2021), in which could potentially be the secondary environmental pollutants (Du et al., 2019b; Gong et al., 2018a; Ozkan et al., 2016). Besides that, the treated biomass has the potential to be converted into other valuable bioresources such as biochar, biofuel and biogas (Pandey et al., 2016; Chang et al., 2019). Currently, several disposal and treatment methods are commonly based on the thermochemical (incineration, pyrolysis, hydrothermal carbonisation, gasification and liquefaction) and biochemical (anaerobic digestions and microbial degradation) (Delil et al., 2020; Kovacs and Szemmelveisz, 2017; Vocciante et al., 2019; Zhou et al., 2020).

5.1. Incineration

Incineration is one of the effective methods to minimise of post-phytoremediation plant biomass enriched with PTEs (Rizwan et al., 2018; Shen et al., 2022). The incineration method was observed to reduce up to 90% of the biomass, and the PTEs concentrated ash was produced at the end of the process (Huang et al., 2018; Vocciante et al., 2019). The final product (ash) can be easily handled without further contaminating the environment, whereby PTEs could be reclaimed from the generated ashes (Rizwan et al., 2018). Lu et al. (2012) and Zhang et al. (2021) observed that the incineration of *Sedum plumbizincicola* biomass yielded about 42% oxidisable state of Cd and the residual fraction of Zn. Temperature is one of the crucial parameters that could influence the incineration process. Zhu et al. (2019) highlighted that about 65% of Cd was recovered at 675 °C whilst Lei et al. (2019) and Zhou et al. (2020) reported 9.0–85.9% of As was produced in *Pteris vittata* L. biomass at 400 °C. However, there are also some arising environmental concerns about the incineration of post-phytoremediation biomass whereby possible secondary pollutants could emerge from the concentrated fly ash if it is not appropriately handled (Wang et al., 2021).

5.2. Pyrolysis

Previous studies exhibited that pyrolysis is an effective, promising and sustainable approach for disposing of plant biomass waste and converting it into a stable form (Bert et al., 2017; Cui et al., 2021; ~~Giudocianni, 2017;~~ Li et al., 2018; Zhang et al., 2020). Besides, this process has also produced many valuable by-products such as biochar, condensable liquid (biofuel) and biogas (Bortoloti and Baron, 2022; Huang et al., 2018; Zhong et al., 2016; Wang et al., 2018) which are beneficial to generate energy and stimulate plant growth such as fertiliser (Debalina et al., 2017; Gasco et al., 2019). Compared to incineration, pyrolysis is an anaerobic process that performed under a closed container and at a moderate temperature (350–650 °C) (Mohan et al., 2014; Qian et al., 2016). Past studies have indicated that the slow charring process of pyrolysis reduces the number of pollutants in the biomass (Kan et al., 2016) while the mild heating temperature helps to retain the PTEs, thus providing cleaner by-products (Gong et al., 2018a; Gonsalvesh et al., 2016). Zhang et al. (2010) and Du et al. (2019a) have utilised *Silphium perfoliatum* L. as biomass feedstock to produce biochar through pyrolysis. Besides, pyrolysis method also demonstrated that PTEs could enhance the production of biogas and bio-oil from biomass as they can act as catalysts (Doroshenko et al., 2019; Zeng et al., 2019). However, rising concerns are reported that the produced biochar may cause environmental pollution similar to the incineration process (Brendova et al., 2016; Igalavithana et al., 2018; Lu et al., 2017). Li et al. (2021) mentioned that about 90% of PTEs such as Zn, Cu, Cr and Ni were remained in the biochar through pyrolysis. Meanwhile, incineration about 84% of PTEs are accumulated in biochar through incineration process (Vocciante et al., 2019). It is also notable that there is a significant reduction in the bioavailability of PTEs in the biochar through the pyrolysis process. Nevertheless, the pyrolysis biomass could be further treated to avoid secondary pollutants from the generated solid, liquid and gas (Rizwan et al., 2018).

5.3. Hydrothermal carbonisation, gasification and liquefaction

In hydrothermal treatment, water is utilised to convert post phytoremediation biomass fuel in an autoclave (Ahmad et al., 2021; Fu et al., 2021; Lee and Park, 2021). The hydrothermal treatment is divided into hydrothermal carbonisation, gasification and liquefaction. Hydrothermal carbonisation turns the carbonaceous biomass into hydrochar, as observed in the studies conducted by Lee et al. (2021) and Kim et al. (2017). This method is usually conducted under a controlled temperature and pressure. However, the reduction of biomass post-treatment is lower than the incineration and pyrolysis. Nonetheless, the hydrothermal treatment is somehow similar to incineration and pyrolysis treatment, as the final yield depends on the temperature of the process (Cui et al., 2020), whereby the hydrochar yield decreases from 22 to 47% when the temperature increases from 200 to 260 °C. On the other hand, hydrogen and methane gas were produced as the post-treatment products in the hydrothermal gasification process where the biomass in this method was treated without drying, thus decreasing the risk of having secondary pollutant (Correa and Kruse, 2018). Similarly, Carrier et al. (2011) also observed no PTE in the biogas post-treatment. Meanwhile, the hydrothermal liquefaction process produced both hydrochar and bio-oil at the end of the process (Qian et al., 2018). Deng et al. (2014) examined that *Phytolacca americana* L. biomass with Mn was converted into 87% of bio-oil under the controlled parameter such as temperature, pressure and the reaction time. Furthermore, Tekin et al. (2014) also reported that about 90% of Zn was retrieved from the *Solanum plumbizincicola* biomass. In addition, Yu et al. (2014) observed that PTEs in the *Brassica juncea* biomass act as catalysts, thus increasing bio-oil yield.

5.4. Anaerobic digestion and microbial degradation

Microbial treatment of plant biomass can be conducted under both aerobic and anaerobic conditions (Li*et al.*, 2018). The post-phytoremediation plant biomass with PTEs is treated and decomposed using microorganisms under controlled aerobic and anaerobic conditions, in which biogas is produced (Burak et al., 2012; Thewys et al., 2010; Yang et al., 2019; Wei et al., 2020). Nevertheless, the reduction of plant biomass through this method is lower than in the incineration, pyrolysis and hydrothermal treatments. Previously, post-phytoremediation biomass of *Elsholtzia splendens* (Xia et al., 2012), *Oenothera biennis* L. (Guo et al. 2013), *Zea mays* L., *Brassica napus* L. (Cao et al., 2015) and *Pteris vittata* L. (Cao et al., 2010) were treated using anaerobic digestion. Moreover, *Phanerochaete chrysosporium* (white-rot fungus) (Zhao et al., 2015), *Cadophora* sp. (Op De Beeck et al., 2015) and *Funneliformis mosseaewas* (Berthelot et al., 2018) plant biomass with Cd were also treated utilising biodegradation process. Bano et al. (2020) and Jing et al. (2020) observed that the degradation process of plant biomass is more effective with fungi than bacteria. Following that, Zhang et al. (2013) and Chen et al. (2016, 2018; 2019b) reported on the effective degradation of plant biomass by utilising fungi. However, the post-phytoremediation biomass treatment by microorganisms required a longer duration to complete and there is possibility of leaching throughout the process, which requires further attention (Hazotte et al., 2017; Liu et al., 2021).

6. SWOT analysis of chelate-induced phytoremediation

The main aim of SWOT analysis depends on the internal and external system characteristics which also support the operational decision at the same time. Generally, SWOT analysis consists of two main factors: (a) internal (strengths and weaknesses) and external (opportunities and threats) (Baycheva-Merger and Wolfslehner, 2016; D'Adomo and Rosa, 2016). Table 9 presents the general SWOT analysis of PTEs phytoremediation using chelating agents.

Table 9
SWOT analysis of chelate-induced phytoremediation for soil PTEs contamination.

Strengths (S)	Weaknesses (W)
• S1: Enhanced the phytoremediation efficiency for PTEs removal. • S2: Minimise time required for the phytoremediation process.	• W1: Interference with biochemical process in plant and soil. • W2: Vary performance with different chelating agent, plant species, type of contaminant and climatic condition.
Opportunities (O)	**Threats (T)**
• O1: Minor association with secondary pollutants in the post-phytoremediation process. • O2: Application of various biological chelator to repurpose the utilisation of waste. • O3: Ensure maximum metal recovery and provides economic benefits.	• T1: Environmental risk of non-biodegradable properties in certain chelating agents. • T2: Lack of understanding on the working cost for commercial practices. • T3: Controlled experiment may not produce a representative result in actual application.

6.1. Strengths

6.1.1. Enhanced the phytoremediation efficiency for PTEs removal

Including a chelating agent enhances the bioavailability of PTEs in soil, thus promoting the phytoremediation efficiency (Luo et al., 2018; Salehi et al., 2020). Previous studies have found that various chelators like EDTA, EDDS, DTPA, ethylene glycol-bis(β-aminoethyl acid)-N,N,N′, N′-tetraacetic acid (EGTA) and cyclohexane-diamine-tetraacetic acid (CDTA) are often used to enhance the solubility of PTEs in the soil (Najeeb et al., 2017; Shahid et al., 2014). Besides, it was also proved that the plant's tolerance toward PTEs was also increased in chelate-induced phytoremediation (Gong et al., 2010; Singh et al., 2016; Stingu et al., 2012). Following that, the addition of chelator also reduces various damages and injuries to plants by high-level phytotoxicity (Antoniadis et al., 2017; Mondaca et al., 2017). Moreover, the chelators could also increase the plant biomass production and soil microbial activities throughout the remediation phase (Chirakkara et al., 2016;); hence in return, improving the removal efficiency of pollutants.

6.1.2. Minimise time required for the phytoremediation process

The time required for the PTEs remediation process is one of the significant impediments to the phytoremediation approach from being applied at the field scale (Oladoye et al., 2022; Shahid et al., 2014). The higher time required for every phytoremediation process might be attributed to the slow metal uptake and translocation process that is resulted from the lower bioavailability of the metals in the soil (Kumar et al., 2022; Yang et al., 2022). However, various chelating agents such as EDTA, DTPA and NTA may assist the translocation and bio-accumulation process of metal uptake from the contaminated soil due to the dissolution and formation of metal-chelator complex that facilitate the movement of metals into the root (Chen et al., 2020; Oladoye et al., 2022; Sarwar et al., 2017). Consequently, adding chelating agents can minimise the time required by increasing the bioaccumulation and translocation indices of metal in plants (Shahid et al., 2014). Additionally, the chelate-induced phytoremediation may assist in harvesting higher amounts of PTEs in a shorter period (Sarwar et al., 2017; Tipu et al., 2021).

6.2. Weaknesses

6.2.1. Interference with biochemical process in plant and soil

Adopting chelating agents in phytoremediation may improve the metal uptake performance and facilitate the translocation of the metals in plants (Jiang et al., 2019; Mousavi et al., 2021; Oladoye et al., 2022). However, the application of chelating agents may significantly impact the natural biological components, thus disrupting the biochemical processes in plants (Chen et al., 2020; Ghazaryan et al., 2021; Hseu et al., 2013). For example, the presence of chelating agents might affect the soil microbial and enzymatic activities of the cultivated plants (Kafle et al., 2022; Shahid et al., 2014; Shen et al., 2022). Hence, the increase in metal concentration that is sequestrated in plants has increased plant toxicity and led to the failure of the plant system (Rostami and Azhdarpoor, 2019). Tipu et al. (2021) showed that the addition of EDTA has generally decreased plant growth and development as the photosynthetic rate dropped due to the declining number of photosynthetic pigments such as chlorophyll contents (chl-a and chl-b) and carotenoids. Besides, the uptake of nutrients enzymes such as phosphorus (P) and nitrate reductase activities in the cultivated plants was further reduced after the additional of EDTA as soil amendment (Patra et al., 2020; Tipu et al., 2021).

6.2.2. Vary performance with different chelating agent, plant species, type of contaminant and climatic condition

Phytoremediation efficiency mainly depends on several factors such as plant species, climate conditions, types of metal and contamination levels (Kafle et al., 2022; Shahid et al., 2014). The phytoremediation performance can vary in various plants and metals as different plant species may have various biochemical routes and remediation techniques (Guo et al., 2019; Sharma et al., 2015; Zhang et al., 2016). Generally, a plant species may work differently in varied cultivated soil and climate (Ali et al., 2013; Ashraf et al., 2019; Saifullah et al., 2015). Besides, various plant species may also have different tendencies to absorb and accumulate specific types of contaminants such as PTEs (Shen et al., 2022). The site, climate and contaminant specificities in phytoremediation have limited the development of a good remediation technology (Kafle et al., 2022). In addition, various types of chelating agents involved in phytoremediation have made it complex to identify suitable chelating agent for specific plant species and PTEs (Hseu et al., 2013; Yang et al., 2021b). Therefore, aside from the nature of phytoremediation techniques, it is required to examine the plant's physiological properties and the uptake mechanisms employed for each plant species (Mahar et al., 2016; Ponce-Hernández et al., 2022). Besides, identifying a suitable plant species and chelating agents for each type of metal is needed to optimise the plant phytoremediation capability (Oladoye et al., 2022; Tipu et al., 2021).

6.3. Opportunities

6.3.1. Minor association with secondary pollutants in the post-phytoremediation process

A considerable portion of plant biomass was produced in a post-phytoremediation process which is rich in PTEs (Sheoran *et al.*, 2014). Post-phytoremediation treatment and disposal methods such as incineration, pyrolysis and hydrothermal mechanism are conducted to process these plant biomass (Deng et al., 2014). The chelate-enhanced phytoremediation biomass was reported to reduce the generation of secondary pollutants during the treatment and/or disposal process (Sharma et al., 2022). Since chelator enhances the PTEs absorption in the phytoremediation process, fewer PTEs might be leached into the environment (Fasani et al., 2018; Tang et al., 2019; Wan et al., 2016). At the same time, plant biomass concentrated with PTEs can be treated prior to the post-phytoremediation process. Consequently, Lei et al. (2022) explored that the post-phytoremediation process also produced various by-products like biochar, biofuel and biogas that can be used in other applications; such as biochar to enhance agriculture whilst biofuel and biogas as the alternative fuel (Kim et al., 2014; Jaruwat et al., 2019; You et al., 2016). Studies also showed that biochar produced through the post-phytoremediation process with PTEs could enriched plant biomass that can be further used to remove PTEs (Khan et al., 2019). Nonetheless, these by-products should undergo immobilisation treatment to detoxify the metals before using it for other applications (Krueger et al., 2013; Liu et al., 2021).

6.3.2. Application of various biological chelator to repurpose the utilisation of waste

The incorporation of waste as a chelator in phytoremediation embraces a new technique to repurpose various biological wastes. Cow and poultry manure, risk husk ash and fly ash are some examples of valuable wastes that previously studied as biological chelators in the phytoremediation of PTEs (Fergusson, 2015; Shen et al., 2017). The results exhibited that the biological wastes are potentially added as a chelator to enhance the uptake of PTEs and create an alternative for the disposal of these wastes (Hai et al., 2022; You et al., 2022).

6.3.3. Ensure maximum metal recovery and provides economic benefits

Chelators are excellent in enhancing the efficiency of metal extraction in plants and can potentially alleviate plant toxicity from the high metal concentration in the contaminated soil (Liang et al., 2019; Wang et al., 2019). Previous studies on chelator-enhanced phytoremediation found that various plants have developed increased tolerance towards PTEs (Stingu et al., 2012; Singh et al., 2016). The alleviation of phytotoxicity from high concentrations of PTEs and chelators is required to improve the phytoremediation procedures, whereby it also provides economic benefits and maximum metal extraction from the contaminated sites (Sarwar et al., 2017). Besides, the application of chelating agents in phytoremediation allows the use of low phytoaccumulator plants with high biomass production, which assists the plants in enhancing the metal uptake and ensuring maximum metal recovery in a short time (Sarwar et al., 2017; Tipu et al., 2021). As a result, innovative phytoremediation and a cost-effective approach will be employed to extract high metal concentrations, especially for the field-scale application (Gong et al., 2018b; Li et al., 2022). However, the optimum concentration of chelating agents must be standardised in order to explore the economic feasibility of the phytoremediation process for future prospects (Tipu et al., 2021).

6.4. Threats

6.4.1. Environmental risk of non-biodegradable properties in certain chelating agents

Despite the remarkable effects of chelating agents in improving phytoremediation performance, the presence of chelating agents has created several environmental risks due to their non-biodegradable properties (Chen et al., 2022; Yadav et al., 2018). Synthetic chelating agents such as EDTA and EDDS can persist in the soil for several weeks or months. For instance, the widely used chelating agent EDTA can be persisted in the soil for up to six months (Shahid et al., 2014). Besides, the aid of chelating agents to solubilise and mobilise several metals in soil has facilitated the leaching of metals in the soil (Oladoye et al., 2022; Tsang et al., 2012). On the other hand, cultivated plants only absorb a limited fraction of mobilised metals, while the other considerable amounts of metals are leached into the soil and potentially taint the groundwater system (Shahid et al., 2014). However, the extent of metal leaching in the presence of chelating agents can be reduced by careful and proper application, and thereby chemical amendments should be applied at an optimum concentration level to minimise adverse environmental effects (Kafle et al., 2022; Tipu et al., 2021). Moreover, field-scale application of EDTA to the plant at a specific growth stage, particularly when the plants have reached a sufficient biomass, can boost metal uptake from soil and reduce downward leaching (Shahid et al., 2014). Besides, the use of biological chelating agents could indirectly substitute synthetic chelating agents for phytoremediation to reduce the risk of PTEs leaching into the environment (Ponce-Hernández et al., 2022; Saifullah et al., 2015).

6.4.2. Lack of understanding on the working cost for commercial practices

Aside from the environmental and societal acceptance aspects, economic factors also play a crucial influence in the actual application of the phytoremediation technique. The low cost of phytoremediation has been regarded as a significant advantage as compared to the other remediation techniques, such as excavation and soil washing (Kafle et al., 2022; Prasad, 2021; Shahid et al., 2014). Moreover, phytoremediation may save up to about 60–80% to remediate PTEs-contaminated areas as compared to the other traditional remediation technologies (Ashraf et al., 2019; Gunarathne et al., 2019). However, the combined applications of physical and chemical agents, such as chelates to enhance the phytoremediation efficiency may no longer be regarded as a low-cost remediation technology. Utilising high charges of synthetic chelating agents may increase the overall cost associated with phytoremediation (Sarwar et al., 2017; Shahid et al., 2014). As a result, this limitation can affect the practical application of phytoextraction at a large scale as it may be over-priced, including the designing, installation, operation and maintenance expenses (Sarwar et al., 2017; Shahid et al., 2014). Besides, the actual operating cost of PTEs phytoremediation has yet to be extensively explored by many researchers at present. The variation of plant species, types of metal as well as soil and chelating agent may also influence the overall cost of phytoremediation (Kafle et al., 2022; Shahid et al., 2014).

6.4.3. Controlled experiment may not produce a representative result in actual application

Despite the extensive experimental studies in PTEs phytoremediation, the obtained findings may not yield an accurate representation of results for field application. Most of the past phytoremediation studies were conducted in a controlled environment within a specific short time frame (Hu et al., 2021; Kafle et al., 2022; Ng et al., 2019). Hence, the effects of chelating agents on the efficiency of phytoremediation may differ in the actual field application. Besides, biotic and abiotic factors of the contaminated sites need to be taken into account as these factors greatly influence the metal uptake performance and the overall phytoremediation efficiency of the plants (Patra et al., 2020; Yadav et al., 2018). Therefore, more studies need to be further carried out into a full-scale practical field application to assess the true potential of chelate-assisted phytoremediation within a longer time frame in the actual environment (Gunarathne et al., 2019; Kafle et al., 2022; Yang et al., 2022).

7. Future research and way forward

The exploration on the use of chelating agents is still emerging in the field of PTEs phytoremediation technologies. This paper has successfully reviewed the effects of various chelating agents on the phytoremediation performance in terms of metal uptake and translocation as well as plant growth and biomass. Due to the specificity effects of phytoremediation, the performance of chelating agents may be varied by different chelating agents, concentration levels of chelating agent, contaminant levels as well as the plant species (Kafle et al., 2022; Sarwar et al., 2017; Shahid et al., 2014). Therefore, further experimental and extensive optimisation studies on the efficiency of phytoremediation are required to evaluate the efficacy among different types of chelating agents with various PTEs. Furthermore, the study on plant physiology and biochemical processes in the chelate-induced phytoremediation are urgently required to assess the effects of chelating agents on the biochemical components such as essential nutrients, photosynthetic pigments as well as microbial and enzyme activities in the soil and plant sections as all of these factors are highly related to plant growth and development during the phytoremediation process (Sarwar et al., 2017; Tipu et al., 2021). Besides, a wide field scale of research work is ought to be developed soon to validate the performance of chelating agents and to ensure the achieved results under different environmental conditions can be directly replicable in the actual field applications (Guo et al., 2019; Hu et al., 2019). Lastly, the fate of PTEs in post-phytoremediation plants needs to be studied in near future to avoid the introduction of secondary pollutants and the leaching of concentrated PTEs into the environment.

8. Conclusions and recommendations

In recent years, the use of chelating agents in PTEs phytoremediation practices has been one of the leading research topics of interest to overcome the current limitations such as low bioavailability of metals, low recovery efficiency as well as low translocation and accumulation rate. A considerable amount of studies has been published on the utilisation of numerous organic and synthetic chelating agents to assess the impacts of chelating agents on the current phytoremediation practices. Hence, the influence of chelating agents in PTEs uptake and plant growth, benefits and limitations of chelate-induced phytoremediation as well as the direction for future development were thoroughly discussed in the present paper. As a result, the following conclusions can be drawn.

1) The addition of chelating agents for PTEs phytoremediation has enhanced the toxic elements uptake and translocation rate from roots to the aboveground plant parts. This can be attributed to the dissolution of metal to form a chelate-metal complex, which indirectly increases the metal mobility in soil and plants.

2) Conflicting findings of the application of chelating agents in phytoremediation affecting plant biomass being negatively and positively, can be attributed to the use of different types of chelating agents along with the diverse application time, contamination levels and plant species.
3) The growing concern about synthetic chelating agents' environmental risk and leaching potential has prompted many researchers to explore the use of organic chelating agents such as citric acid and oxalic acid, which are biodegradable and more environmentally friendly options.
4) Extensive research on the PTEs phytoremediation efficiency of numerous chelating agents and a full-scale experimental study must be undertaken to examine and provide a comprehensive understanding on the impacts of chelating agents in actual field phytoremediation practices.

Author contributions

Nur Hanis Zulkernain: Conceptualization, Investigation, Writing-Original draft preparation, Project Administration. **Turkeswari Uvarajan**: Visualisation, Investigation, Writing-Original draft preparation. **Chuck Chuan Ng**: Conceptualization, Project Administration, Funding acquisition, Supervision, Writing-Review & Editing.All authors have read and agreed to the submitted version of the manuscript.

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.

Data availability

No data was used for the research described in the article.

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