Agrobiodiversity for Improving Nutrition, Health and Life Quality

Review Article OPEN ACCESS

Phytoremediation of Potentially Toxic Elements: Applications, Advantages, Limitations, and Future Perspectives

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Article Details:
Received: 2026-04-06
Accepted: 2026-05-10
Available online: 2026-05-31

DOI: https://doi.org/10.15414/ainhlq.2026.0011

Phytoremediation is an environmentally sustainable strategy for reducing the mobility, bioavailability, and ecological risk of potentially toxic elements (PTEs) in contaminated soils and aquatic environments. This review analyzes the main mechanisms and approaches of phytoremediation, including phytoextraction, phytostabilization, and phytovolatilization, emphasizing their advantages, limitations, and practical applications. This review provides a comparative overview of plant species and their capacity to remove or immobilize contaminants such as cadmium. Particular attention is also given to Jerusalem artichoke (*Helianthus tuberosus* L.) as a high-biomass C3 bioenergy crop with documented potential for phytoremediation of soils contaminated with Cd, Hg, and multiple metals, as well as for subsequent biomass conversion into biofuels and value-added bioproducts. Additionally, the review discusses the differences between C3 and C4 photosynthetic pathways and their influence on plant tolerance to stressful environmental conditions, highlighting how C4 species often exhibit higher stress resistance and remediation potential under drought and high-temperature conditions.

Keywords: potentially toxic elements; phytoremediation; phytomining; phytostabilization; contamination; bioenergy crops; contaminated biomass

Introduction

The rapid industrialization and intensification of agriculture over the past decades have led to the accumulation of potentially toxic elements (PTEs), including cadmium (Cd), lead (Pb), arsenic (As), and mercury (Hg) in soils and aquatic environments (Alloway, 2013). These contaminants pose serious risks to ecosystems, food safety, and human health due to their persistence, bioaccumulation, and toxicity even at low concentrations (Tchounwou et al., 2012). Conventional remediation methods, including excavation, soil washing, and chemical stabilization, are often expensive, energy-intensive, and may cause secondary pollution (Mulligan et al., 2001).

Phytoremediation has emerged as a promising, environmentally sustainable technology that leverages plants' natural ability to extract, stabilize, degrade, or volatilize pollutants from the environment (Ali et al., 2013). Its advantages include low cost, minimal disturbance to ecosystems, and the potential

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for aesthetic and ecological restoration of degraded landscapes. Phytoremediation efficiency depends on interactions among plants, soil properties, and rhizosphere microorganisms, as well as on physiological and biochemical traits that determine plant tolerance and metal accumulation capacity (Ma et al., 2011; Pilon-Smits, 2005).

In recent years, increasing attention has been paid to selecting efficient plant species and optimizing environmental and physiological conditions to enhance remediation efficiency (Yan et al., 2020). Moreover, differences between C3 and C4 plants provide valuable insights into plant adaptation to stress conditions such as drought, salinity, and PTE exposure (Sage et al., 2011).

This review aims to synthesize current evidence on plant-based strategies for reducing the mobility, bioavailability, and ecological risk of PTEs in contaminated soils and aquatic environments, with an emphasis on phytoextraction, phytostabilization, phytovolatilization, bioenergy crops, and post-harvest management of contaminated biomass.

Plant-based remediation strategies

Among the technologies used to restore metal-contaminated soils, phytoremediation is generally low-cost, causes limited site disturbance, and can support ecological restoration. Because of its relatively simple application at a large scale and low impact on local ecology, interest in this method has grown over the past 15 years. Additionally, by capturing PTEs, the method helps reduce soil erosion and metal leaching through contaminant stabilization, thereby reducing the risk of contaminant spread. Moreover, phytoremediation enhances soil fertility by increasing organic matter accumulation (Wuana, 2011; Jacob et al., 2018).

Plants can absorb dissolved ions from the soil solution, even at low concentrations. Plant roots can

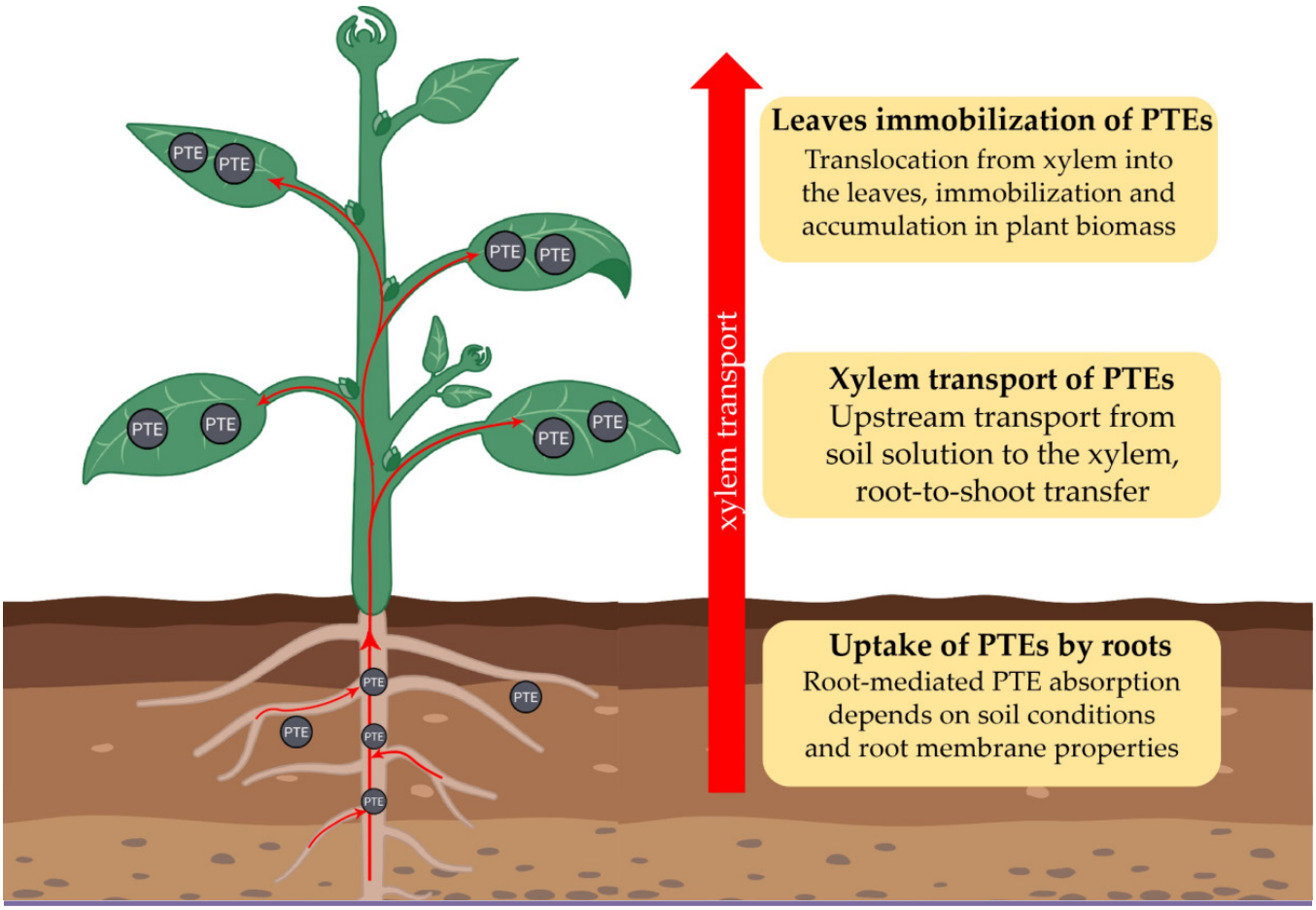

Figure 1 Schematic representation of PTE uptake and translocation in plants during phytoremediation. Potentially toxic elements are taken up by roots from the soil solution, depending on soil physicochemical conditions and root membrane properties. After root uptake, PTEs may be transported upward through the xylem from roots to shoots and leaves, where they can be immobilized, accumulated, or sequestered in plant biomass. This pathway illustrates the main physiological basis of phytoextraction and aboveground accumulation of contaminants

extend throughout large portions of the soil profile. Subsequently, rhizosphere interactions develop that can influence PTE accumulation and bioavailability, thereby contributing to soil restoration and improved soil stability (Jacob et al., 2018). Phytoremediation has been shown to remove or immobilize Cd, As, Ni, Pb, and Cu (Velasco-Arroyo et al., 2024). The general pathway of PTE uptake, xylem-mediated translocation, and subsequent immobilization in aboveground tissues is schematically presented in Figure 1.

Plants interact with contaminants through root uptake, translocation, transformation, sequestration, and volatilization. For organic pollutants, phytodegradation and rhizodegradation may contribute to contaminant breakdown. In contrast, inorganic PTEs are not degraded; instead, phytoremediation alters their speciation, mobility, bioavailability, compartmentalization, or transfer among environmental compartments.

Plant establishment on PTE-contaminated soils may be constrained by high metal bioavailability, low organic matter content, acidity, salinity, and nutrient imbalance. Biochar is used as a soil amendment in some phytoremediation systems because its alkaline functional groups, porous structure, and cation-exchange properties can decrease the soluble fraction of several cationic PTEs and improve conditions for plant growth (Beesley et al., 2011; Ahmad et al., 2014; Saletnik et al., 2019). However, the effects of biochar are not universal: immobilization efficiency depends on feedstock, pyrolysis temperature, ash content, pH, soil texture, contaminant speciation, and application rate. Therefore, biochar is better regarded as a site-specific amendment that may support phytostabilization or plant establishment, rather than as a standalone remediation method. In mine wastes and acidic substrates, alkaline biochars can reduce metal mobility and improve rhizosphere conditions, but field validation and monitoring of contaminant redistribution remain necessary (Drzewiecka et al., 2021).

Phytoremediation includes several major processes, among which phytostabilization and phytoextraction are particularly important for PTE-contaminated soils. Phytostabilization immobilizes contaminants in the rhizosphere or root tissues, whereas phytoextraction involves contaminant uptake from soil and subsequent translocation to harvestable aboveground biomass. For phytoextraction to be effective, plants must accumulate significant amounts of PTEs, tolerate contaminated soil, and produce large quantities of biomass under unfavorable conditions (McGrath et al., 2002). Until recently, phytoremediation research has focused on identifying species with high metal-accumulation capacity, although practical application is often limited by low biomass production (Luo et al., 2005; Shumaker and Begonia, 2005).

Phytostabilization of PTEs primarily reduces contaminant mobility and bioavailability through root retention, rhizosphere immobilization, adsorption to amendments, and erosion control. Rhizodegradation and phytohydraulics are more relevant to organic contaminants and are therefore best treated as complementary mechanisms (Hong et al., 2001; Bakshe, 2023).

In situations involving widespread contamination or soils with high concentrations of PTEs, phytostabilization is often the most economical and environmentally sustainable remediation approach. This is largely because phytoextraction is typically too slow for highly contaminated sites (Wong, 2003; Dary et al., 2010).

Plant selection plays a critical role in the effectiveness of metal phytostabilization strategies. Ideally, suitable plants should be able to develop a robust root system and produce substantial biomass, even in environments with high PTE concentrations. At the same time, suitable species should minimize metal translocation from roots to shoots. For a plant to function effectively in phytostabilization, it must absorb only limited amounts of PTEs, thereby reducing the risk of these contaminants entering the wildlife food chain (Alkorta et al., 2010).

To enhance the effectiveness of phytostabilization, most approaches incorporate soil amendments. In addition to organic materials, commonly used amendments include liming agents, phosphorus compounds, aluminosilicates, and metal oxyhydroxides. These substances improve soil conditions, thereby supporting plant establishment and growth. The combined use of stabilizing plants and soil amendments, such as compost, calcium oxide (CaO), and zeolites, for remediation is referred to as chemophytostabilization, assisted natural remediation, and aided phytostabilization (Alvarenga et al., 2008). In such strategies, the amendments act as immobilizing agents, reducing the bioavailable fraction of PTEs in the soil. Reduced metal bioavailability facilitates revegetation and promotes ecosystem recovery in contaminated areas. Additionally, vegetation helps limit the spread of pollutants by minimizing wind and water erosion. It may further aid in metal immobilization through biological activity and the production of organic matter (Bouwman et al., 2004; Ruttens et al., 2006).

In this context, Bidar and colleagues examined the potential of *Lolium perenne* and *Trifolium repens* to stabilize PTEs at a contaminated site near a decommissioned lead (Pb) smelter. Their findings showed that PTEs were predominantly retained in the roots rather than translocated to the shoots. The most accumulated metal was Cd, followed by Zn and Pb (Bidar et al., 2007). Lei and Duan investigated the phytostabilization potential of ten pioneer plant species for remediating lead-zinc (Pb-Zn) mine tailings. The species examined included *Artemisia roxburghiana*, *Clematis tangutica*, *Carex inanis*, *Cyperaceae hebecarpus*, *Plantago depressa*, *Cynoglossum lanceolatum*, *Potentilla saundesiana*, *Coriaria sinica*, *Oxyria sinensis*, and *Miscanthus nepalensis*. Their study revealed that PTEs were primarily accumulated in the root tissues of these plants. Among the ten species, *Artemisia roxburghiana* demonstrated the highest potential for phytostabilization. This was attributed not only to its metal retention capacity but also to its significant enhancement of rhizosphere soil fertility, including increased levels of available organic matter and macroelements (Lei and Duang, 2008).

An additional investigation assessed the effectiveness of pig manure and conventional fertilizer as soil amendments for remediating lead (Pb)-contaminated substrates. Within this framework, *Vetiveria zizanioides* and *Thysanolaena maxima* demonstrated considerable potential for chemophytostabilization in Pb-mining environments. The study reported minimal Pb translocation to aerial tissues, with shoot concentrations ranging from 8.3 to 179 mg·kg^{-1}. In contrast, root tissues accumulated substantially higher Pb levels, ranging from 107 to 911 mg·kg^{-1}, indicating a strong capacity for metal immobilization in roots (Rotkittikhun et al., 2007).

Plant uptake of PTEs is primarily governed by their bioavailability in the soil matrix (Glick, 2010). According to Lebeau et al. (2008), several soil characteristics influence metal bioavailability, including particle size, nutrient levels, pH, redox potential, organic

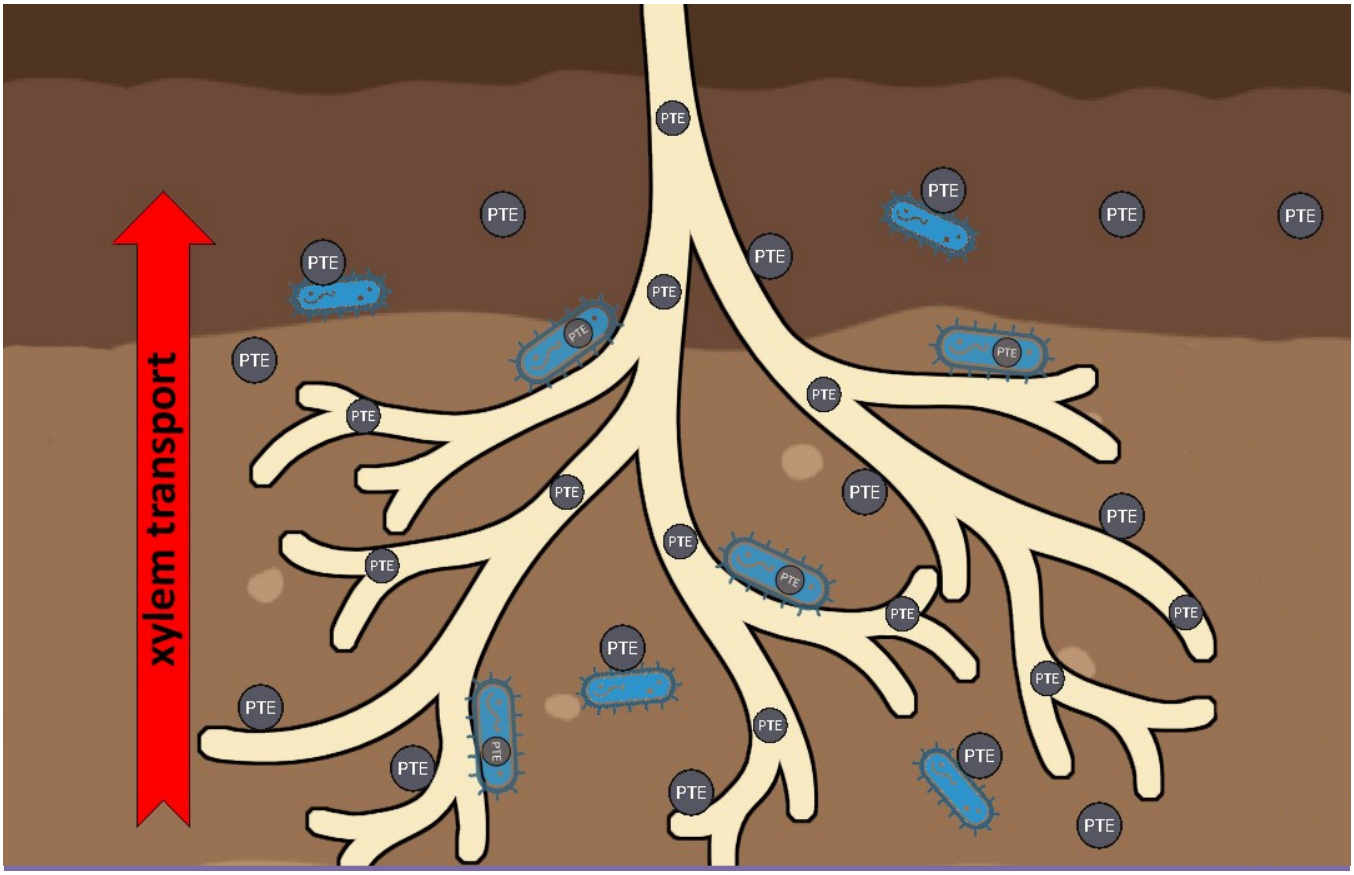

Figure 2 Rhizosphere-mediated uptake and root-to-shoot transport of potentially toxic elements (PTEs) in plants. PTEs present in the soil solution can interact with plant roots and rhizosphere microorganisms before entering root tissues. Microbial activity in the rhizosphere may modify PTE bioavailability by releasing organic acids, siderophores, biosurfactants, and other metabolites that influence metal mobility and uptake. After absorption by the root system, PTEs may be transported upward through the xylem, contributing to their accumulation, immobilization, or sequestration in plant tissues

matter content, and the presence of competing ions. Rhizosphere microorganisms can enhance the phytoavailability of PTEs by releasing chelating compounds (e.g., siderophores, biosurfactants, and organic acids), promoting soil acidification, engaging in redox reactions, and facilitating phosphate solubilization (Lebeau et al., 2008). The role of the root system and rhizosphere microorganisms in PTE mobilization, uptake, and subsequent xylem-mediated transport is schematically shown in Figure 2.

Phytovolatilization refers to the uptake of selected contaminants by plants, their biochemical conversion into volatile forms, and subsequent release from aerial tissues to the atmosphere. Among PTEs, this mechanism is most relevant for elements that can form volatile species, particularly Se and Hg, and, in some cases, arsenic-containing compounds (Terry et al., 2000; Yan et al., 2020; Bhat et al., 2022). Unlike phytoextraction, phytovolatilization does not concentrate contaminants in harvestable biomass; instead, it transfers contaminants from soil or water into the atmospheric compartment. For this reason, it should be evaluated together with atmospheric transport, redeposition, and risk-transfer considerations.

For selenium, plants can assimilate inorganic Se into organic selenoamino acids and subsequently transform part of the absorbed Se into volatile methylated compounds such as dimethylselenide or dimethyldiselenide. These forms are generally less toxic than many inorganic selenium species, but their release still represents transfer of the element from soil or water into the atmosphere (Terry et al., 2000). For mercury, engineered or naturally tolerant plant-microbe systems may reduce Hg^{2+} to elemental Hg^0, which is more volatile than ionic mercury and may be emitted from plant tissues (Bizily et al., 2000; Awa and Hadibarata, 2020). However, Hg^0 can be transported by the atmosphere and later redeposited; under suitable anaerobic conditions, it may contribute indirectly to methylmercury formation after redeposition (Chibuike and Obiora, 2014).

Accordingly, phytovolatilization is best described as a contaminant-transformation and risk-redistribution process rather than a complete remediation endpoint. It may be useful when volatilized products are demonstrably less toxic and when emissions are monitored, but it is less suitable where atmospheric release could create secondary exposure pathways. Any discussion of this mechanism should therefore include both its potential benefits and its limitations as a possible source of atmospheric contamination and redeposition (Vangronsveld et al., 2009).

Phytomining is a bio-based metal recovery strategy that employs metal-hyperaccumulating plant species to extract economically valuable elements from substrates such as low-grade ores, mine tailings, and contaminated soils. The harvested biomass, enriched in target PTEs, is subsequently processed into a "bio-ore" suitable for metallurgical recovery (Kikis et al., 2024). This approach capitalizes on the natural capacity of specific plant taxa to accumulate elevated concentrations of PTEs, including nickel, cobalt, zinc, rare-earth elements, and gold, in their aerial tissues (Sheoran et al., 2009; Dinh et al., 2022). Empirical evidence from both field and controlled-environment studies has demonstrated the potential of these species to extract measurable quantities of PTEs when cultivated on metal-enriched substrates (Li et al., 2020). Phytomining presents several advantages over conventional mining practices. It offers the potential to reduce environmental degradation, facilitate *t* remediation of contaminated sites, and enable resource recovery from substrates that are otherwise uneconomical to exploit through traditional methods.

Recent studies highlight the promise of nickel phytomining and the selective recovery of rare-earth elements, particularly in geochemically favorable environments such as serpentine soils and mine residues (van der Ent et al., 2015). Nonetheless, achieving commercial viability at scale necessitates further advancements in plant productivity, processing efficiency, and system integration. Current research focuses on improving agronomic performance through the application of soil amendments (e.g., biochar, fertilizers, microbial inoculants), the selection and breeding of superior hyperaccumulator genotypes, and the development of integrated systems that couple phytomining with bioenergy production to improve overall economic returns.

Studies have demonstrated that particular plant species can accumulate measurable quantities of gold (Au) when grown in Au-enriched soils or treated substrates. For instance, conifer species growing naturally in gold ore regions of Canada were reported to accumulate up to 0.02 mg Au·kg^{-1} dry weight (DW), approximately 100 times higher than typical background concentrations in vegetation (Corzo Remigio et al., 2020). Similarly, when soils containing 0.6 mg Au·kg^{-1} were treated with sodium cyanide, *Brassica juncea* accumulated up to 63 mg Au·kg^{-1} DW, suggesting that a viable phytomining yield of 1 kg Au·ha^{-1} could be achieved if plants reached

a biomass of approximately 10,000 kg·ha^{-1} with tissue concentrations around 100 mg Au·kg^{-1} (Wilson-Corral et al., 2012).

Further experiments have demonstrated comparable effects in other plant species. *Daucus carota* (carrot) accumulated 0.779 mg Au·ha^{-1} when treated with ammonium thiocyanate and 1.45 mg Au·ha^{-1} when treated with thiosulfate under growth conditions containing 3.8 mg Au·kg^{-1} substrate (Msuya et al., 2000). More recently, *Helichrysum arenarium* was investigated for its gold uptake potential, showing translocation and bioaccumulation factors (TF = 2.04, BAF = 0.59), indicating efficient metal transport and storage within plant tissues. The study concluded that increasing gold bioavailability through soil amendments such as NaCN, SCN^-, and thiosulfates could enhance the efficiency of this species for phytomining applications (Vural and Safari, 2022).

Cadmium hyperaccumulation has been most consistently documented in metal-tolerant *Brassica*ceae, particularly species historically referred to as *Thlaspi* L. and now often treated within *Noccaea* Moench. *Noccaea caerulescens* (J. Presl & C. Presl) F.K. Mey. and related taxa can accumulate unusually high Cd concentrations in shoots, but reported values vary strongly with genotype, substrate chemistry, exposure duration, soil pH, competing ions, and whether experiments are conducted in soil or hydroponic systems (Küpper and Leitenmaier, 2012). For this reason, Cd accumulation data should be interpreted in relation to both biomass production and field applicability, rather than by tissue concentration alone.

Extreme Cd concentrations reported for hairy roots or hydroponic systems are useful for understanding uptake and sequestration mechanisms, but they should not be generalized directly to field-scale phytoextraction. For example, the high Cd accumulation reported for hairy roots of *Thlaspi caerulescens* under controlled nutrient-solution conditions (Nedelkoska and Doran, 2000) demonstrates physiological capacity under artificial exposure, whereas field remediation also depends on root access to bioavailable Cd, shoot biomass, harvest frequency, and safe disposal or processing of contaminated biomass. Therefore, Cd-hyperaccumulators are valuable mechanistic models and potential remediation tools, but their practical performance must be assessed under site-specific soil conditions.

Role of c3 and C4 photosynthetic pathways in phytoremediation potential

Three systems of carbon fixation exist in nature: C3, C4, and CAM. The C3 pathway is the most common mechanism of carbon fixation among terrestrial plants. Carbon dioxide enters leaf mesophyll cells and is directly fixed by ribulose-1,5-bisphosphate carboxylase/oxygenase (Rubisco). Rubisco catalyzes the addition of CO_2 to ribulose-1,5-bisphosphate (RuBP), producing two molecules of 3-phosphoglycerate. C4 species often show higher water-use efficiency, photosynthetic performance, and biomass production under high temperature, high light, and drought conditions. However, their phytoremediation efficiency remains species-, contaminant-, and site-specific. This advantage is partly due to their distinct leaf anatomy: C4 plants typically possess Kranz anatomy, in which photosynthetically active bundle sheath cells surround the vascular bundles. C3 plants also have bundle sheath cells, but these cells are generally less specialized for CO_2 concentration. The main anatomical and biochemical differences between C3 and C4 photosynthesis, which influence plant productivity and stress tolerance under phytoremediation conditions, are schematically shown in Figure 3.

In C4 plants, phosphoenolpyruvate carboxylase catalyzes the initial fixation of bicarbonate in mesophyll cells, producing four-carbon acids. Depending on the C4 subtype, decarboxylation in bundle sheath cells may involve NADP-malic enzyme, NAD-malic enzyme, or phosphoenolpyruvate carboxykinase. Nonetheless, several C3 plants have also been identified as effective candidates for metal remediation and biomass production.

C4 species were more prevalent and better adapted to the contaminated site, whereas C3 species were less represented, with only three C3 plants recorded. C4 photosynthesis reduces photorespiration by concentrating CO_2 around Rubisco in bundle sheath cells, thereby improving carbon fixation efficiency under warm, dry conditions. The study also demonstrated that the combined presence of four PTEs (Mn, Co, Cd, and Zn) negatively affected antioxidant stress responses in several biomass crops, even though these species are typically considered tolerant to PTEs (Sonowal et al., 2018).

The responses of C3 and C4 plants to environmental extremes should not be interpreted in isolation, because plant performance under both favorable and stressful conditions reflects the combined influence of the stressor and the prevailing environmental factors.

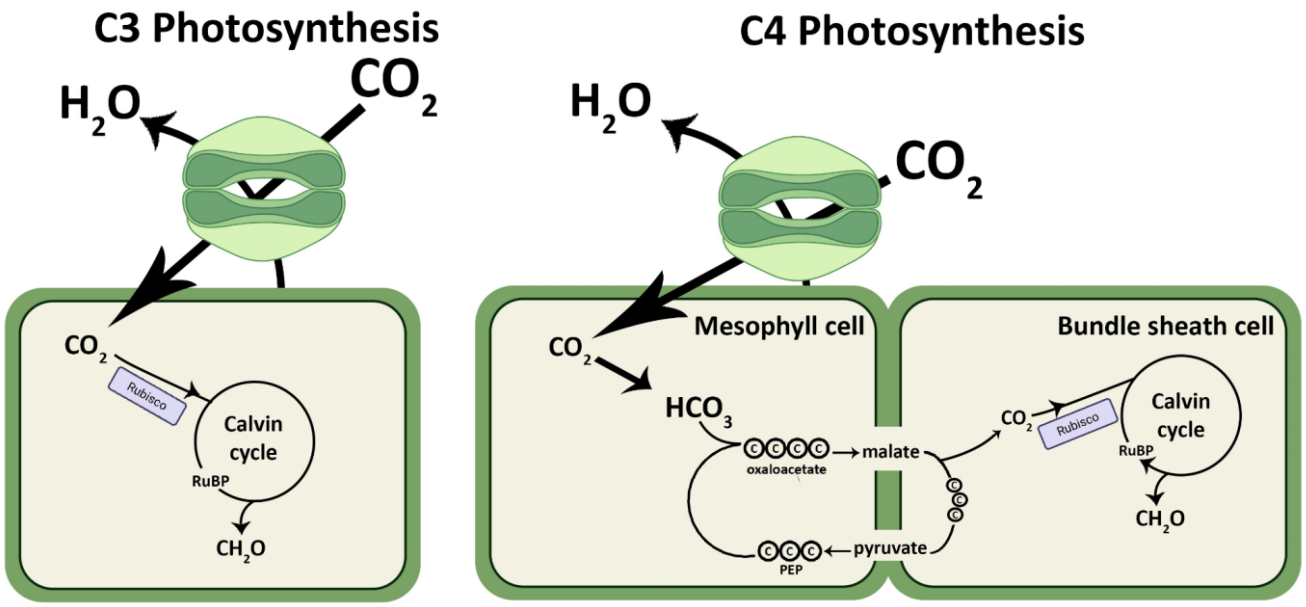

Figure 3 Schematic comparison of C3 and C4 photosynthetic pathways relevant to phytoremediation potential. In C3 plants, atmospheric CO_2 enters the mesophyll cells and is directly fixed by Rubisco in the Calvin cycle, resulting in carbohydrate formation. In C4 plants, CO_2 is initially converted to bicarbonate and fixed in mesophyll cells into four-carbon compounds, such as oxaloacetate and malate. These compounds are then transported to bundle sheath cells, where CO_2 is released and refixed by Rubisco in the Calvin cycle. This CO_2-concentrating mechanism improves photosynthetic efficiency, reduces photorespiration, and can enhance biomass production and stress tolerance under high temperature, drought, and contaminated-soil conditions

For example, C3 species generally perform poorly under warmer conditions, while C4 species respond more positively; the opposite may occur under cooler conditions. Such patterns are also supported by studies of *intermedia*te C3-C4 species like *Phragmites australis* and *Eleocharis vivipara* (Zheng et al., 2000; Ueno, 2001).

Environmental variability, including changes in weather, soil properties, water availability, and nutrient status, makes *in situ* experiments challenging. These variations complicate the interpretation of plant responses and make it difficult to identify the specific factor responsible for a given outcome. For instance, plants growing on mine tailings often show reduced growth, typically attributed to toxic metal exposure, yet the influence of surrounding environmental conditions is frequently overlooked. Overall, ecological stresses modify plant metabolic processes, including photosynthesis (Srivastava et al., 2012).

Switchgrass (*Panicum virgatum* L.) is a perennial C4 grass widely studied as a lignocellulosic bioenergy crop and as a candidate for phytoremediation on marginal or moderately contaminated land. Its relevance is linked to perennial growth, deep rooting, high seasonal biomass production, and relatively low input requirements compared with many annual crops (McLaughlin and Kszos, 2005; Wright and Turhollow, 2010). These traits may support contaminant stabilization, erosion control, and partial metal uptake, but they do not by themselves make switchgrass a universal phytoextractor.

Evidence for switchgrass phytoremediation is contaminant- and site-specific. Experimental work indicates that switchgrass can tolerate and accumulate selected metals, including Cd, Cr, Pb, and Zn, under controlled or amended conditions (Li et al., 2011; Chen et al., 2012; Balsamo et al., 2015; Johnson et al., 2015). However, metal uptake is strongly influenced by soil amendments, pH, organic matter, contaminant bioavailability, and biomass yield. In poorly amended or highly toxic substrates, plant growth may be reduced enough to limit total removal even when tissue concentrations are measurable (Shrestha et al., 2019). Thus, switchgrass should be discussed as a robust biomass crop with potential for phytostabilization and assisted phytoextraction, not as an intrinsically high-performing metal-removal species under all conditions.

Related *Panicum* species, such as *Panicum maximum*, have also been investigated for metal uptake, but these results should not be used interchangeably with *P. virgatum* unless species identity and experimental conditions are clearly stated. Data from *P. maximum* studies can be retained as evidence that the genus

includes metal-tolerant grasses, while conclusions about switchgrass should rely primarily on *P. virgatum*-specific studies.

Over the 28-day study, soil concentrations of Pb, Cr, and Cd generally declined, while their levels increased in different tissues of *P. maximum*. The roots accumulated the highest amounts, with tissue concentrations decreasing in the order Cr > Pb > Cd. In control plants (grown in unspiked soils), metal concentrations ($\mu g \cdot g^{-1}$) measured at 7, 14, 21, and 28 days were 0.28–0.24 in soils, 0.08–0.06 in roots, 0.06–0.04 in stems, and 0.02–0.01 in foliage. Soil Cr levels are known to depend on factors such as pH, electrical conductivity, dissolved organic carbon, and amendments (Hattab et al., 2013). In Pb-treated soils, Pb concentrations decreased from 1.40 to 1.05 $\mu g \cdot g^{-1}$ (20 ppm treatment) and from 1.57 to 1.30 $\mu g \cdot g^{-1}$ (120 ppm treatment) over four weeks. Meanwhile, Pb uptake by *P. maximum* increased, reaching 0.38 $\mu g \cdot g^{-1}$ in roots, 0.30 $\mu g \cdot g^{-1}$ in stems, and 0.18 $\mu g \cdot g^{-1}$ in foliage. Bioaccumulation factors (BAF) ranged from 0.21–0.45 in roots, 0.17–0.35 in stems, and 0.08–0.21 in foliage at 20 ppm Pb, corresponding to 13–45% of soil concentrations. At 120 ppm Pb, tissue concentrations rose further (0.64–0.74 $\mu g \cdot g^{-1}$ in roots, 0.53–0.63 $\mu g \cdot g^{-1}$ in stems, and 0.21–0.34 $\mu g \cdot g^{-1}$ in foliage), with BAF values of 0.50–2.26 in roots, 0.41–1.94 in stems, and 0.03–1.00 in foliage (Olatunji et al., 2014).

Paulownia Siebold. & Zucc. is a genus of woody plants comprising approximately 17 species. It originates from East Asia and is widespread in China and Europe. It blooms in early spring with bell-shaped flowers. *Paulownia* trees can reach approximately 30 m in height and have large heart-shaped leaves 10–20 cm wide and 15–30 cm long. The edge of the leaf can be serrated, lobed, or entire. Available experimental evidence suggests that *Paulownia* extracts exhibit low acute toxicity under the tested conditions. Chen and colleagues conducted a study in which they administered a high concentration of 50 $mg \cdot mL^{-1}$ of paulownia flower extract three times a day, with a maximum dose of 0.4 $mL \cdot 10\ g^{-1}$ of body weight. Based on the observations, the animals were in a satisfactory mental state, moved freely, ate regularly, did not show abnormal salivation or sweating, and had respiratory rates within normal limits. *Paulownia* is characterized by exceptionally fast growth among woody plants: it can reach commercial maturity within approximately 10 years. The wood is characterized as strong and light; its physical and mechanical properties make it promising for planting and cultivation in Central Europe (Huber et al., 2023). Currently, the best conditions for growing paulownia for timber production are in the Middle East and Southern Europe (Jakubowski, 2022).

New plant hybrids are being developed through selective breeding. One example is the cross between *Paulownia fortunei* (Seem.) Hemsl. and *Paulownia tomentosa* (Thunb.) Steud., which resulted in the creation of *Paulownia Shan Tong*. This hybrid is notable for its rapid growth, making it suitable for industrial-scale cultivation. Among the most widely used hybrids are *Paulownia Clone in Vitro 112*, *Cotevisa 2*, and *Paulownia Shan Tong*. These varieties are typically produced through microclonal propagation techniques. *Paulownia* cultivation may alter soil nutrient composition and microbial communities. A possible solution to this problem is the use of fertilizers derived from woody biomass, such as pine bark and ash. Such amendments may help maintain microbial activity (Jakubowski, 2022).

Paulownia is a C3 plant species that fixes carbon dioxide through the Calvin cycle using ribulose-1,5-bisphosphate carboxylase/oxygenase (Rubisco). In C3 plants, Rubisco can bind both CO_2 and O_2, which may increase photorespiration under warm and dry conditions. In contrast, C4 photosynthesis concentrates CO_2 around Rubisco in bundle sheath cells, thereby reducing photorespiration and improving carbon fixation efficiency, water-use efficiency, and stress tolerance under high temperature and drought conditions (Icka et al., 2016).

Like many cultivated woody species, *Paulownia* can be affected by diseases. One of the most widely reported diseases is witches' broom. The disease is caused by a phytoplasma infection that spreads among paulownia trees. Disease development is associated with changes in plant gene expression in the presence of phytoplasma (Cao et al., 2021). Genetic resistance to phytoplasma infection has been suggested in *P. tomentosa* × *P. fortunei* hybrids. In addition, the plant may also be affected by *Phytophthora* spp. (Aloi et al., 2021).

The phytochemical profile of *Paulownia tomentosa* includes compounds with antioxidant, antibacterial, and cytotoxic activities (Schneiderová et al., 2014). The leaves of *Paulownia Clone in Vitro 112* are a valuable source of compounds with anticoagulant properties that inhibit platelet activation at various levels (Adach et al., 2021); triterpenoids have the highest activity among all other compounds present (Stochmal et al., 2022).

Soil contamination with PTEs is a persistent environmental problem because many metal(loid)s are

not degraded like organic pollutants and can remain in soils for long periods. Their risk depends not only on total concentration but also on chemical form, soil pH, organic matter, clay minerals, redox conditions, and interactions with plant roots and microorganisms (Doumett et al., 2008; Antoniadis et al., 2017). Some trace elements are essential micronutrients at low concentrations, whereas others, including Cd, Pb, As, and Hg, have no known beneficial function in plants and may impair physiological and biochemical processes when bioavailable concentrations increase (Clemens, 2006). These elements can enter food webs through plant uptake, dust, water movement, or soil ingestion; therefore, phytoremediation studies should evaluate both contaminant reduction and the risk of transfer into edible or ecological receptors.

In the context of phytoremediation, *Paulownia* has considerable potential due to its ability to tolerate high soil metal concentrations and accumulate them in its biomass. Thus, this fast-growing plant may contribute to the restoration of metal-contaminated soils under suitable conditions. Varieties such as *P. tomentosa* × *P. fortunei*-TF01 and *P. elongata* × *P. fortunei*-EF02 are used because they produce large amounts of plant biomass. Although the plant can be cultivated in soils containing PTEs, metal accumulation reduces growth and photosynthetic efficiency. The effect of Pb on the development of *P. fortunei* was stimulatory at low concentrations and inhibitory at high concentrations. The plant can tolerate lead concentrations up to 400 mg·L^{-1}, but concentrations above that cause toxicity. *Paulownia fortunei* seedlings, through physiological responses, contribute to environmental restoration under metal pollution. Fungal residues can enhance the phytoremediation potential of *Paulownia fortunei* (Han, 2020).

Jerusalem artichoke (*Helianthus tuberosus* L.), also known as topinambur, is a perennial high-biomass crop belonging to the Asteraceae family. Although it uses the C3 photosynthetic pathway, it is characterized by strong adaptability to marginal soils, tolerance to abiotic stress, and the ability to produce substantial aboveground and belowground biomass. These traits make it relevant to phytoremediation strategies that combine contaminant removal or stabilization with biomass production and post-harvest utilization.

Experimental studies have demonstrated that *H. tuberosus* can tolerate and accumulate several potentially toxic elements. In experiments conducted under different soil pH values and PTE concentrations, Willscher et al. (2017) reported that plant growth and remediation efficiency were strongly controlled by soil pH and metal load. The authors observed high accumulation of Fe, Mn, and Zn in roots, while shoots accumulated substantial amounts of Ni, Mn, and Zn, suggesting that *H. tuberosus* can be considered suitable for the phytoextraction of Fe, Mn, Zn, Cd, and Ni under appropriate site conditions (Willscher et al., 2017). Similar findings were reported for cadmium, where two Jerusalem artichoke cultivars exposed to Cd stress showed relatively high tolerance and Cd accumulation capacity, with roots accumulating more Cd than stems and leaves. The cultivar N5 showed stronger tolerance and higher accumulation than N2, indicating genotype-dependent differences in phytoremediation efficiency (Chen et al., 2011).

The remediation potential of *H. tuberosus* is not limited to Cd-contaminated soils. Lv et al. assessed its tolerance and phytoremediation capacity in Hg-contaminated soil and described it as a non-food energy crop suitable for the management of contaminated land (Lv et al., 2018). This is relevant because the use of non-food crops reduces the risk of introducing accumulated PTEs into the food chain. Therefore, Jerusalem artichoke may serve as a dual-purpose species: it can reduce environmental risks associated with contaminated soils while producing biomass that can be directed toward controlled industrial applications or bioenergy production pathways rather than food or feed use.

Biofuel from phytoremediation biomass

Biofuels are energy sources obtained from biological materials. Organic waste, plants, and algae are major feedstocks for biofuel production. Typical examples of biofuels include bioethanol, biodiesel, and biogas. A major advantage of biofuels is their potential to partially or fully replace fossil fuels in transportation, heating, or electricity generation (Demirbas, 2007). In addition, the ability of biofuel feedstocks to regenerate and the resulting lower net greenhouse gas emissions compared to conventional petroleum-based fuels are considered beneficial (Nigam and Singh, 2011). The widespread availability of biomass supports biofuel production, is compatible with existing fuel systems, and reduces reliance on non-renewable energy sources (Naik et al., 2010).

The combination of different techniques for creating a sustainable production pipeline, including phytoremediation and biofuel production, has emerged. Many plant species characterized by hyperaccumulation and fast growth, which are used to remove PTEs, often retain substantial lignocellulosic or lipid content suitable for producing biodiesel or bioethanol after harvesting. Using contaminated-

site biomass for energy prevents these plants from entering the food chain and creates a circular system where land restoration and energy production occur simultaneously (Pandey et al., 2016). Although challenges remain due to metal contamination in residues, integrating phytoremediation with biofuel production is considered a promising sustainable strategy (Vangronsveld et al., 2009). The possible post-harvest utilization pathways of phytoremediation biomass, including thermochemical conversion, composting, and bioethanol production, are summarized in Figure 4.

Bioethanol is a renewable transport fuel produced by fermenting carbohydrate-rich biomass such as sugarcane, corn, and lignocellulosic residues through microbial fermentation of simple sugars into ethanol (Hahn-Hägerdal et al., 2006). The process generally involves pretreatment and enzymatic hydrolysis to release fermentable monosaccharides from cellulose and hemicellulose, followed by fermentation with organisms such as *Saccharomyces cerevisiae* (Kumar et al., 2009). Lignocellulosic feedstocks, while harder to process, are sustainable because they are abundant and do not compete directly with food crops (Sun and Cheng, 2002). Advances in enzyme engineering, consolidated bioprocessing, and improved pretreatment methods have further enhanced ethanol yields and reduced production costs, positioning bioethanol as a key component of low-carbon energy systems (Balat et al., 2008).

Switchgrass can link phytoremediation with bioenergy production by directing harvested biomass to controlled lignocellulosic conversion pathways rather than to food or feed use. In the United States, *P. virgatum* L. has been widely evaluated as a model perennial energy crop because it can produce substantial biomass on some marginal lands and has established agronomic information for planting, harvesting, storage, and transport (McLaughlin and Kszos, 2005; Wright, 2007; Wright and Turhollow, 2010). These bioenergy traits are useful in remediation planning because economic value from biomass can partly offset the long time frames typical of plant-based remediation.

The remediation performance of switchgrass should nevertheless be presented cautiously. Models relating biomass yield to metal content have shown measurable Cd, Cr, and Zn uptake in switchgrass under experimental conditions, with uptake depending on both tissue concentration and dry biomass (Chen et al., 2012). Chromium-focused studies also indicate that roots may contribute substantially to Cr retention and uptake (Li et al., 2011). These findings support the use of switchgrass in selected phytostabilization or assisted phytoextraction systems, particularly on marginal land, but they do not justify describing the crop as universally effective for contaminated soils.

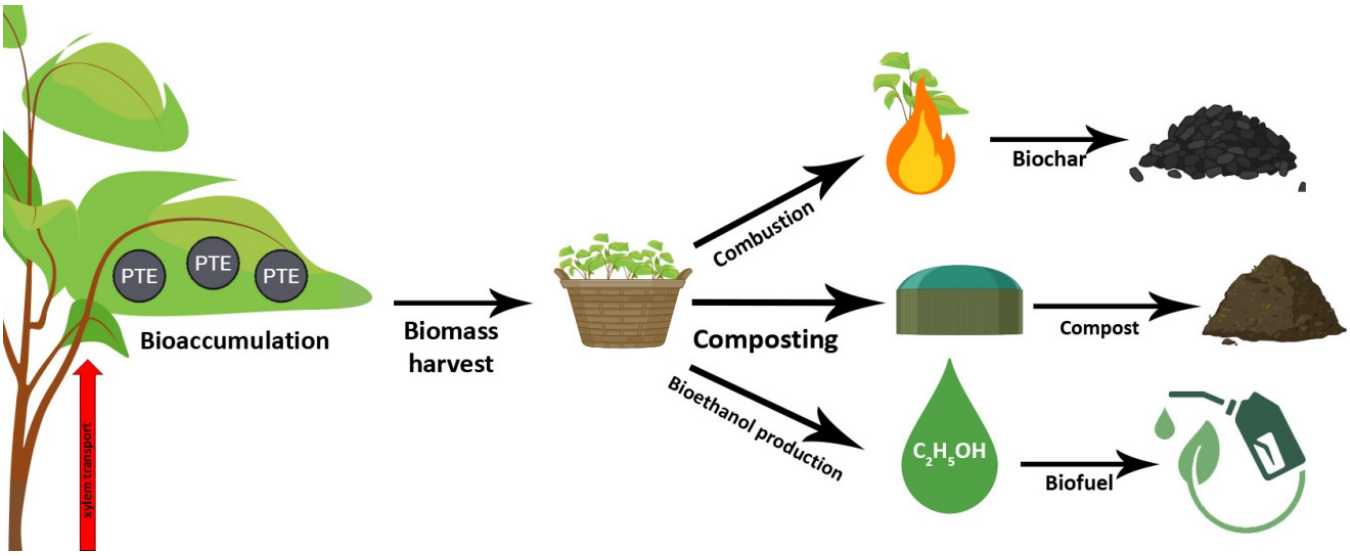

Figure 4 Integrated utilization pathways of biomass obtained after phytoremediation of PTE-contaminated soils. Plants used for phytoremediation can accumulate potentially toxic elements in aboveground biomass through root uptake and xylem-mediated translocation. After harvesting, this biomass may be directed to controlled post-harvest processing pathways, including thermochemical conversion for biochar production, composting under regulated conditions, or biochemical conversion into bioethanol and other biofuels. Such integrated systems can combine soil remediation with biomass valorization; however, the fate of accumulated PTEs in residues and final products must be carefully controlled to prevent secondary contamination

Site screening, contaminant speciation, amendment choice, and post-harvest biomass handling remain necessary before field implementation.

Biochar is produced by thermal conversion of biomass under oxygen-limited conditions and contains relatively stable aromatic carbon, ash, minerals, pores, and surface functional groups. These properties can increase pH, cation exchange capacity, water retention, and sorption of selected contaminants, although the magnitude and direction of these effects depend on feedstock type, pyrolysis conditions, soil properties, and contaminant chemistry (Beesley et al., 2011; Ahmad et al., 2014; Saletnik et al., 2019). In phytoremediation systems, biochar is therefore most appropriately described as an amendment that can support plant establishment and promote immobilization of some PTEs, rather than as a universal removal technology.

Evidence from amended-soil studies indicates that biochar can reduce the plant-available fraction of Cd, Pb, Zn, Cu, and other cationic elements in some contaminated soils, while improving biomass production under stressful substrate conditions (Beesley et al., 2010; Houben et al., 2013; Drzewiecka et al., 2021). However, immobilization may also reduce phytoextraction efficiency when the remediation goal is contaminant removal through harvestable biomass. The choice between biochar-assisted phytostabilization and phytoextraction should therefore be based on the target contaminant, the regulatory endpoint, the plant species, and post-harvest management options.

Compost is an organic amendment formed through aerobic decomposition of plant- and animal-derived residues. Mature compost supplies organic matter and nutrients, improves soil aggregation and water retention, and stimulates microbial activity (Insam and de Bertoldi, 2007; Bernal et al., 2009). In contaminated soils, compost can improve plant establishment and may reduce the bioavailability of some PTEs through complexation with organic matter, pH modification, and improved soil structure (Tejada et al., 2006). Because compost can also contain salts, nutrients, or trace contaminants depending on its source material, its use in phytoremediation should be preceded by quality testing and matched to specific remediation objectives.

Impact and remediation of potentially toxic elements

PTE-induced phytotoxicity results from a combination of direct metal interference with cellular targets and indirect disruption of nutrient balance, photosynthesis, water relations, and redox homeostasis. Excessive production of reactive oxygen species can damage lipids, proteins, nucleic acids, and photosynthetic structures, leading to chlorosis, necrosis, reduced biomass, inhibited root growth, and lower photosynthetic efficiency (Clemens, 2006; Pospíšil and Yamamoto, 2017). The severity of these responses depends on plant species, developmental stage, exposure duration, contaminant speciation, and soil properties. The main visible and cellular symptoms of PTE-induced phytotoxicity, including oxidative stress, chlorosis, necrotic spots, and leaf blight, are summarized schematically in Figure 5.

Arsenic is taken up primarily as arsenate As(V) and arsenite As(III). Arsenate may enter plant roots through phosphate transport pathways, whereas arsenite may be transported through aquaporin-related channels; once inside the plant, As may be reduced, complexed with thiol-rich compounds, sequestered in vacuoles, or transported to shoots depending on species and exposure conditions (Zhao et al., 2008). Arsenic toxicity commonly affects root growth, photosynthesis, nutrient homeostasis, respiration, and antioxidant metabolism (Finnegan and Chen, 2012).

Mercury is retained primarily in roots, with limited movement to shoots in many terrestrial plants. Its bioavailability is controlled by soil pH, organic matter, clay minerals, and binding to sulfur- and nitrogen-containing ligands (Greger et al., 2005). Hg exposure may inhibit root elongation, disrupt photosynthetic electron transport, and increase oxidative stress; plants respond by upregulating antioxidant enzymes, glutathione, phytochelatins, and compartmentalization mechanisms, although these defenses can be overwhelmed at high exposure levels (Sahu et al., 2011; Alcantara et al., 2013).

Cadmium is of particular concern because it is mobile in many agricultural soils and can enter crops even when present at relatively low concentrations. Anthropogenic inputs include mining and smelting activities, industrial emissions, sewage sludge, phosphate fertilizers, landfill leakage, batteries, and atmospheric deposition (Bigalke et al., 2017; Khan et al., 2017; Kubier et al., 2019). Cd can impair water relations, stomatal conductance, nutrient uptake, and chlorophyll formation; visible symptoms often include chlorosis, growth retardation, and necrotic injury under high exposure (Hermans et al., 2011). It can also disturb the uptake or distribution of mineral nutrients such as Fe, Zn, Ca, Mg, K, and Mn, thereby intensifying physiological stress (Xu et al., 2017).

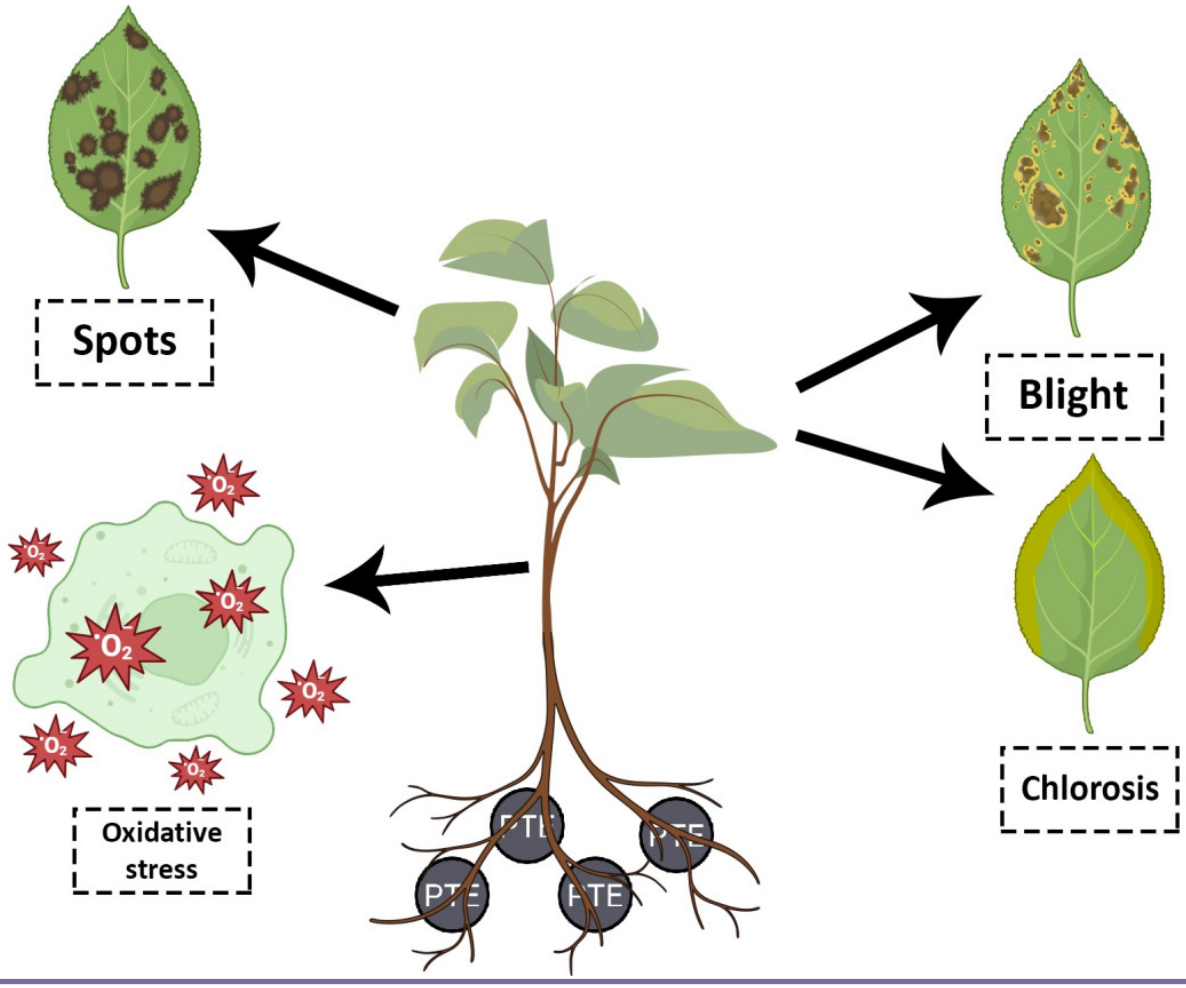

Figure 5 Major phytotoxic effects induced by potentially toxic elements in plants. Accumulation of PTEs in the root zone can disturb plant physiological and biochemical processes, leading to excessive production of reactive oxygen species (ROS) and oxidative stress. These stress responses may result in visible injury symptoms, including chlorosis, necrotic leaf spots, blight, reduced photosynthetic activity, and overall growth inhibition. The figure illustrates the connection between PTE accumulation and both cellular-level oxidative damage and aboveground morphological symptoms

Cadmium contamination may arise from localized sources, such as mines, smelters, and industrial waste deposits, or from diffuse sources, including fertilizer use, atmospheric deposition, wastewater reuse, and agro-industrial activities (Cloquet et al., 2006; Bigalke et al., 2017; Kubier et al., 2019). Because Cd behavior is strongly controlled by soil pH, organic matter, competing ions, and redox conditions, remediation strategies should focus on the bioavailable fraction and plant-transfer risk rather than total Cd concentration alone.

In the context of cadmium remediation, Jerusalem artichoke deserves particular attention because it combines Cd tolerance with high biomass production. Chen et al. showed that *H. tuberosus* cultivars exposed to increasing Cd concentrations maintained growth at relatively high Cd loads and accumulated more Cd in roots than in aboveground tissues. This root-dominant accumulation pattern suggests that Jerusalem artichoke may be useful not only for phytoextraction under selected conditions but also for phytostabilization-oriented strategies where limiting Cd transfer to harvestable or edible organs is important (Chen et al., 2011).

Conclusions

Phytoremediation is a sustainable plant-based strategy for reducing the mobility, bioavailability, and ecological risks of PTEs in contaminated environments. Its effectiveness depends on contaminant chemistry, soil properties, plant functional traits, rhizosphere interactions, and post-harvest biomass management. C4 species often exhibit advantages under drought, heat, and high-light conditions because of their high water-use efficiency and biomass productivity, whereas

selected C3 species, including *Helianthus tuberosus*, may be valuable for integrated phytoremediation–bioenergy systems. However, contaminated biomass management remains a critical limitation. Future research should prioritize field-scale validation, standardized phytoremediation performance indicators, contaminant-specific plant selection, and safe post-harvest conversion pathways that prevent secondary pollution.

Conflict of interest

The authors have no competing interests to declare.

Ethical statement

This article doesn't contain any studies that would require an ethical statement.

References

Adach, W., Żuchowski, J., Moniuszko-Szajwaj, B., Szumacher-Strabel, M., Stochmal, A., Olas, B., & Cieslak, A. (2021). *In vitro* antiplatelet activity of extract and its fractions of *Paulownia Clone in Vitro 112* leaves. *Biomedicine & Pharmacotherapy*, 137, 111301. https://doi.org/10.1016/j.biopha.2021.111301

Ahmad, M., Rajapaksha, A. U., Lim, J. E., Zhang, M., Bolan, N., Mohan, D., Vithanage, M., Lee, S. S., & Ok, Y. S. (2014). Biochar as a sorbent for contaminant management in soil and water: A review. *Chemosphere*, 99, 19–33. https://doi.org/10.1016/j.chemosphere.2013.10.071

Alcantara, H. J. P., Rivero, G. C., & Puzon, J. M. (2013). Tolerance mechanisms in mercury-exposed *Chromolaena odorata* (l.f.) R.M. King et H. Robinson, a potential phytoremediator. *Journal of Degraded and Mining Lands Management*, 1(1), 09–20. https://doi.org/10.15243/jdmlm.2013.011.009

Ali, H., Khan, E., & Sajad, M. A. (2013). Phytoremediation of heavy metals – Concepts and applications. *Chemosphere*, 91(7), 869–881. https://doi.org/10.1016/j.chemosphere.2013.01.075

Alkorta, I., Becerril, J., & Garbisu, C. (2010). Phytostabilization of metal contaminated soils. *Reviews on Environmental Health*, 25(2). https://doi.org/10.1515/reveh.2010.25.2.135

Alloway, B.J. (2013). Heavy metals in soils: Trace metals and metalloids in soils and their bioavailability. In *Heavy Metals in Soils*, 3rd ed., Springer: Dordrecht, Netherlands. https://doi.org/10.1007/978-94-007-4470-7

Aloi, F., Riolo, M., La Spada, F., Bentivenga, G., Moricca, S., Santilli, E., Pane, A., Faedda, R., & Cacciola, S. O. (2021). *Phytophthora* root and collar rot of *Paulownia*, a new disease for Europe. *Forests*, 12(12), 1664. https://doi.org/10.3390/f12121664

Alvarenga, P., Gonçalves, A., Fernandes, R., de Varennes, A., Vallini, G., Duarte, E., & Cunha-Queda, A. (2008). Evaluation of composts and liming materials in the phytostabilization of a mine soil using perennial ryegrass. *Science of The Total Environment*, 406(1–2), 43–56. https://doi.org/10.1016/j.scitotenv.2008.07.061

Antoniadis, V., Levizou, E., Shaheen, S. M., Ok, Y. S., Sebastian, A., Baum, C., Prasad, M. N., Wenzel, W. W., & Rinklebe, J. (2017). Trace elements in the soil-plant interface: Phytoavailability, translocation, and phytoremediation – A review. *Earth-Science Reviews*, 171, 621–645. https://doi.org/10.1016/j.earscirev.2017.06.005

Awa, S. H., & Hadibarata, T. (2020). Removal of heavy metals in contaminated soil by phytoremediation mechanism: A review. *Water, Air, & Soil Pollution*, 231(2), 47. https://doi.org/10.1007/s11270-020-4426-0

Bakshe, P., & Jugade, R. (2023). Phytostabilization and rhizofiltration of toxic heavy metals by heavy metal accumulator plants for sustainable management of contaminated industrial sites: A comprehensive review. *Journal of Hazardous Materials Advances*, 10, 100293. https://doi.org/10.1016/j.hazadv.2023.100293

Balat, M., Balat, H., & Öz, C. (2008). Progress in bioethanol processing. *Progress in Energy and Combustion Science*, 34(5), 551–573. https://doi.org/10.1016/j.pecs.2007.11.001

Balsamo, R. A., Kelly, W. J., Satrio, J. A., Ruiz-Felix, M. N., Fetterman, M., Wynn, R., & Hagel, K. (2015). Utilization of grasses for potential biofuel production and phytoremediation of heavy metal contaminated soils. *International Journal of Phytoremediation*, 17(5), 448–455. https://doi.org/10.1080/15226514.2014.922918

Beesley, L., Moreno-Jiménez, E., & Gomez-Eyles, J. L. (2010). Effects of biochar and greenwaste compost amendments on mobility, bioavailability and toxicity of inorganic and organic contaminants in a multi-element polluted soil. *Environmental Pollution*, 158(6), 2282–2287. https://doi.org/10.1016/j.envpol.2010.02.003

Beesley, L., Moreno-Jiménez, E., Gomez-Eyles, J. L., Harris, E., Robinson, B., & Sizmur, T. (2011). A review of biochars' potential role in the remediation, revegetation and restoration of contaminated soils. *Environmental Pollution*, 159(12), 3269–3282. https://doi.org/10.1016/j.envpol.2011.07.023

Bernal, M., Alburquerque, J., & Moral, R. (2009). Composting of animal manures and chemical criteria for compost maturity assessment. A review. *Bioresource Technology*, 100(22), 5444–5453. https://doi.org/10.1016/j.biortech.2008.11.027

Bhat, S. A., Bashir, O., Ul Haq, S. A., Amin, T., Rafiq, A., Ali, M., Américo-Pinheiro, J. H. P., & Sher, F. (2022). Phytoremediation of heavy metals in soil and water: An eco-friendly, sustainable and multidisciplinary approach. *Chemosphere*, 303, 134788. https://doi.org/10.1016/j.chemosphere.2022.134788

Bidar, G., Garçon, G., Pruvot, C., Dewaele, D., Cazier, F., Douay, F., & Shirali, P. (2007). Behavior of *Trifolium repens* and *Lolium perenne* growing in a heavy metal contaminated field: Plant metal concentration and phytotoxicity. *Environmental Pollution*, 147(3), 546–553. https://doi.org/10.1016/j.envpol.2006.10.013

Bigalke, M., Ulrich, A., Rehmus, A., & Keller, A. (2017). Accumulation of cadmium and uranium in arable soils in Switzerland. *Environmental Pollution*, 221, 85–93. https://doi.org/10.1016/j.envpol.2016.11.035

Bizily, S. P., Rugh, C. L., & Meagher, R. B. (2000). Phytodetoxification of hazardous organomercurials by genetically engineered plants. *Nature Biotechnology*, 18(2), 213–217. https://doi.org/10.1038/72678

Bouwman, L. A., & Vangronsveld, J. (2004). Rehabilitation of the nematode fauna in a phytostabilized, heavily zinc-contaminated, sandy soil. *Journal of Soils and Sediments*, 4(1), 17-23. https://doi.org/10.1007/bf02990824

Cao, Y., Sun, G., Zhai, X., Xu, P., Ma, L., Deng, M., Zhao, Z., Yang, H., Dong, Y., Shang, Z., Lv, Y., Yan, L., Liu, H., Cao, X., Li, B., Wang, Z., Zhao, X., Yu, H., Wang, F., Ma, W., Huang, J., & Fan, G. (2021). Genomic insights into the fast growth of paulownias and the formation of *Paulownia* witches' broom. *Molecular Plant*, 14(10), 1668–1682. https://doi.org/10.1016/j.molp.2021.06.021

Chen, B. C., Lai, H. Y., & Juang, K. W. (2012). Model evaluation of plant metal content and biomass yield for the phytoextraction of heavy metals by switchgrass. *Ecotoxicology and Environmental Safety*, 80, 393–400. https://doi.org/10.1016/j.ecoenv.2012.04.011

Chen, L., Long, X. H., Zhang, Z. H., Zheng, X. T., Rengel, Z., & Liu, Z. P. (2011). Cadmium Accumulation and Translocation in Two Jerusalem Artichoke (*Helianthus tuberosus* L.) Cultivars. *Pedosphere*, 21(5), 573–580. https://doi.org/10.1016/s1002-0160(11)60159-8

Chibuike, G. U., & Obiora, S. C. (2014). Heavy Metal Polluted Soils: Effect on Plants and Bioremediation Methods. *Applied and Environmental Soil Science*, 2014, 1–12. https://doi.org/10.1155/2014/752708

Clemens, S. (2006). Toxic metal accumulation, responses to exposure and mechanisms of tolerance in plants. *Biochimie*, 88(11), 1707–1719. https://doi.org/10.1016/j.biochi.2006.07.003

Cloquet, C., Carignan, J., Libourel, G., Sterckeman, T., Perdrix, E. (2006). Tracing source pollution in soils using cadmium and lead isotopes. *Environmental Science & Technology*, 40, 2525–2530. https://doi.org/10.1021/es052232

Corzo Remigio, A., Chaney, R. L., Baker, A. J. M., Edraki, M., Erskine, P. D., Echevarria, G., & van der Ent, A. (2020). Phytoextraction of high value elements and contaminants from mining and mineral wastes: opportunities and limitations. *Plant and Soil*, 449(1–2), 11–37. https://doi.org/10.1007/s11104-020-04487-3

Dary, M., Chamber-Pérez, M., Palomares, A., & Pajuelo, E. (2010). "*In situ*" phytostabilisation of heavy metal polluted soils using Lupinus luteus inoculated with metal resistant plant-growth promoting rhizobacteria. *Journal of Hazardous Materials*, 177(1–3), 323–330. https://doi.org/10.1016/j.jhazmat.2009.12.035

Demirbas, A. (2007). Progress and recent trends in biofuels. *Progress in Energy and Combustion Science*, 33(1), 1–18. https://doi.org/10.1016/j.pecs.2006.06.001

Dinh, T., Dobo, Z., & Kovacs, H. (2022). Phytomining of rare earth elements – A review. *Chemosphere*, 297, 134259. https://doi.org/10.1016/j.chemosphere.2022.134259

Doumett, S., Lamperi, L., Checchini, L., Azzarello, E., Mugnai, S., Mancuso, S., Petruzzelli, G., & Del Bubba, M. (2008). Heavy metal distribution between contaminated soil and *Paulownia tomentosa*, in a pilot-scale assisted phytoremediation study: Influence of different complexing agents. *Chemosphere*, 72(10), 1481–1490. https://doi.org/10.1016/j.chemosphere.2008.04.083

Drzewiecka, K., Gąsecka, M., Magdziak, Z., Budzyńska, S., Szostek, M., Niedzielski, P., Budka, A., Roszyk, E., Doczekalska, B., Górska, M., & Mleczek, M. (2021). The possibility of using *Paulownia elongata* S. Y. Hu × *Paulownia fortunei* hybrid for phytoextraction of toxic elements from post-industrial wastes with biochar. *Plants*, 10(10), 2049. https://doi.org/10.3390/plants10102049

Finnegan, P. M., & Chen, W. (2012). Arsenic toxicity: the effects on plant metabolism. *Frontiers in Physiology*, 3. https://doi.org/10.3389/fphys.2012.00182

Glick, B. R. (2010). Using soil bacteria to facilitate phytoremediation. *Biotechnology Advances*, 28(3), 367–374.https://doi.org/10.1016/j.biotechadv.2010.02.001

Greger, M., Wang, Y., & Neuschütz, C. (2005). Absence of Hg transpiration by shoot after Hg uptake by roots of six terrestrial plant species. *Environmental Pollution*, 134(2), 201–208. https://doi.org/10.1016/j.envpol.2004.08.007

Hahn-Hägerdal, B., Galbe, M., Gorwa-Grauslund, M., Lidén, G., & Zacchi, G. (2006). Bio-ethanol – the fuel of tomorrow from the residues of today. *Trends in Biotechnology*, 24(12), 549–556. https://doi.org/10.1016/j.tibtech.2006.10.004

Han, L., Chen, Y., Chen, M., Wu, Y., Su, R., Du, L., & Liu, Z. (2020). Mushroom residue modification enhances phytoremediation potential of *Paulownia fortunei* to lead-zinc slag. *Chemosphere*, 253, 126774. https://doi.org/10.1016/j.chemosphere.2020.126774

Hattab, N., Hambli, R., Motelica-Heino, M., & Mench, M. (2013). Neural network and Monte Carlo simulation approach to investigate variability of copper concentration in phytoremediated contaminated soils. *Journal of Environmental Management*, 129, 134–142. https://doi.org/10.1016/j.jenvman.2013.07.003

Hermans, C., Chen, J., Coppens, F., Inzé, D., & Verbruggen, N. (2011). Low magnesium status in plants enhances tolerance to cadmium exposure. *New Phytologist*, 192(2), 428–436. https://doi.org/10.1111/j.1469-8137.2011.03814.x

Hong, M. S., Farmayan, W. F., Dortch, I. J., Chiang, C. Y., McMillan, S. K., & Schnoor, J. L. (2001). Phytoremediation of MTBE from a Groundwater Plume. *Environmental Science & Technology*, 35(6), 1231–1239. https://doi.org/10.1021/es001911b

Houben, D., Evrard, L., & Sonnet, P. (2013). Mobility, bioavailability and pH-dependent leaching of cadmium,

zinc and lead in a contaminated soil amended with biochar. *Chemosphere*, 92(11), 1450–1457. https://doi.org/10.1016/j.chemosphere.2013.03.055

Huber, C., Moog, D., Stingl, R., Pramreiter, M., Stadlmann, A., Baumann, G., Praxmarer, G., Gutmann, R., Eisler, H., & Müller, U. (2023). *Paulownia* (*Paulownia elongata* S.Y.Hu) – importance for forestry and a general screening of technological and material properties. *Wood Material Science & Engineering*, 18(5), 1663–1675. https://doi.org/10.1080/17480272.2023.2172690

Icka, P., Damo, R., & Icka, E. (2016). *Paulownia tomentosa*, a fast growing timber. *Annals "Valahia" University of Targoviste – Agriculture*, 10(1), 14–19. https://doi.org/10.1515/agr-2016-0003

Insam, H., & de Bertoldi, M. (2007). Chapter 3 Microbiology of the composting process. *Waste Management Series*, 25–48. https://doi.org/10.1016/s1478-7482(07)80006-6

Jacob, J. M., Karthik, C., Saratale, R. G., Kumar, S. S., Prabakar, D., Kadirvelu, K., & Pugazhendhi, A. (2018). Biological approaches to tackle heavy metal pollution: A survey of literature. *Journal of Environmental Management*, 217, 56–70. https://doi.org/10.1016/j.jenvman.2018.03.077

Jakubowski, M. (2022). Cultivation potential and uses of *Paulownia* wood: A Review. *Forests*, 13(5), 668. https://doi.org/10.3390/f13050668

Johnson, D. M., Deocampo, D. M., El-Mayas, H., & Greipsson, S. (2015). Induced phytoextraction of lead through chemical manipulation of switchgrass and corn; role of iron supplement. *International Journal of Phytoremediation*, 17(12), 1192–1203. https://doi.org/10.1080/15226514.2015.1045134

Khan, M. A., Khan, S., Khan, A., & Alam, M. (2017). Soil contamination with cadmium, consequences and remediation using organic amendments. *Science of The Total Environment*, 601–602, 1591-1605. https://doi.org/10.1016/j.scitotenv.2017.06.030

Kikis, C., Thalassinos, G., & Antoniadis, V. (2024). Soil phytomining: recent developments – A review. *Soil Systems*, 8(1), 8. https://doi.org/10.3390/soilsystems8010008

Kubier, A., Wilkin, R. T., & Pichler, T. (2019). Cadmium in soils and groundwater: A review. *Applied Geochemistry*, 108, 104388. https://doi.org/10.1016/j.apgeochem.2019.104388

Kumar, P., Barrett, D. M., Delwiche, M. J., & Stroeve, P. (2009). Methods for pretreatment of lignocellulosic biomass for efficient hydrolysis and biofuel production. *Industrial & Engineering Chemistry Research*, 48(8), 3713–3729. https://doi.org/10.1021/ie801542g

Küpper, H., & Leitenmaier, B. (2012). Cadmium-accumulating plants. *Metal Ions in Life Sciences*, 373–393. https://doi.org/10.1007/978-94-007-5179-8_12

Lebeau, T., Braud, A., & Jézéquel, K. (2008). Performance of bioaugmentation-assisted phytoextraction applied to metal contaminated soils: A review. *Environmental Pollution*, 153(3), 497–522. https://doi.org/10.1016/j.envpol.2007.09.015

Lei, D., & Duang, C. (2008). Restoration potential of pioneer plants growing on lead-zinc mine tailings in Lanping, southwest China. *Journal of Environmental Sciences*, 20(10), 1202–1209. https://doi.org/10.1016/s1001-0742(08)62210-x

Li, C., Ji, X., & Luo, X. (2020). Visualizing hotspots and future trends in phytomining research through scientometrics. *Sustainability*, 12(11), 4593. https://doi.org/10.3390/su12114593

Li, C., Wang, Q.-H., Xiao, B., & Li, Y.-F. (2011). Phytoremediation potential of switchgrass (*Panicum virgatum* L.) for Cr-polluted soil. *2011 International Symposium on Water Resource and Environmental Protection* (pp. 1731–1734). https://doi.org/10.1109/iswrep.2011.5893582

Luo, C., Shen, Z., & Li, X. (2005). Enhanced phytoextraction of Cu, Pb, Zn and Cd with EDTA and EDDS. *Chemosphere*, 59(1), 1–11. https://doi.org/10.1016/j.chemosphere.2004.09.100

Lv, S., Yang, B., Kou, Y., Zeng, J., Wang, R., Xiao, Y., Li, F., Lu, Y., Mu, Y., & Zhao, C. (2018). Assessing the difference of tolerance and phytoremediation potential in mercury contaminated soil of a non-food energy crop, *Helianthus tuberosus* L. (Jerusalem artichoke). *PeerJ*, 6, e4325. https://doi.org/10.7717/peerj.4325

Ma, Y., Prasad, M., Rajkumar, M., & Freitas, H. (2011). Plant growth promoting rhizobacteria and endophytes accelerate phytoremediation of metalliferous soils. *Biotechnology Advances*, 29(2), 248–258. https://doi.org/10.1016/j.biotechadv.2010.12.001

McGrath, S., Zhao, J., & Lombi, E. (2002). Phytoremediation of metals, metalloids, and radionuclides. *Advances in Agronomy*, 1–56. https://doi.org/10.1016/s0065-2113(02)75002-5

McLaughlin, S. B., & Adams Kszos, L. (2005). Development of switchgrass (*Panicum virgatum*) as a bioenergy feedstock in the United States. *Biomass and Bioenergy*, 28(6), 515–535. https://doi.org/10.1016/j.biombioe.2004.05.006

Msuya, F. A., Brooks, R. R., & Anderson, C. W. N. (2000). Chemically-induced uptake of gold by root crops: Its significance for phytomining. *Gold Bulletin*, 33(4), 134–137. https://doi.org/10.1007/bf03215491

Mulligan, C. N., Yong, R. N., & Gibbs, B. F. (2001). An evaluation of technologies for the heavy metal remediation of dredged sediments. *Journal of Hazardous Materials*, 85(1–2), 145–163. https://doi.org/10.1016/s0304-3894(01)00226-6

Naik, S., Goud, V. V., Rout, P. K., & Dalai, A. K. (2010). Production of first and second generation biofuels: A comprehensive review. *Renewable and Sustainable Energy Reviews*, 14(2), 578–597. https://doi.org/10.1016/j.rser.2009.10.003

Nedelkoska, T.V., & Doran, P.M. (2000). Hyperaccumulation of cadmium by hairy roots of *Thlaspi caerulescens*. *Biotechnol. Bioeng.*, 67, 607–615. https://doi.org/10.1002/(SICI)1097-0290(20000305)67:5%3C607::AID-BIT11%3E3.0.CO;2-3

Nigam, P. S., & Singh, A. (2011). Production of liquid biofuels from renewable resources. *Progress in Energy and Combustion Science*, 37(1), 52–68. https://doi.org/10.1016/j.pecs.2010.01.003

Olatunji, S. O., Ximba, J. B., Fatoki, S. O., & Opeolu, O. B. (2014). Assessment of the phytoremediation potential of *Panicum maximum* (guinea grass) for selected heavy metal removal from contaminated soils. *African Journal of Biotechnology*, 13(19), 1979–1984. https://doi.org/10.5897/ajb2014.13635

Pandey, V. C., Bajpai, O., & Singh, N. (2016). Energy crops in sustainable phytoremediation. *Renewable and Sustainable Energy Reviews*, 54, 58–73. https://doi.org/10.1016/j.rser.2015.09.078

Pilon-Smits, E. (2005). Phytoremediation. *Annual Review of Plant Biology*, 56(1), 15–39. https://doi.org/10.1146/annurev.arplant.56.032604.144214

Pospíšil, P., & Yamamoto, Y. (2017). Damage to photosystem II by lipid peroxidation products. *Biochimica et Biophysica Acta (BBA) – General Subjects,* 1861(2), 457–466. https://doi.org/10.1016/j.bbagen.2016.10.005

Rotkittikhun, P., Chaiyarat, R., Kruatrachue, M., Pokethitiyook, P., & Baker, A. (2007). Growth and lead accumulation by the grasses *Vetiveria zizanioides* and *Thysanolaena maxima* in lead-contaminated soil amended with pig manure and fertilizer: A glasshouse study. *Chemosphere*, 66(1), 45–53. https://doi.org/10.1016/j.chemosphere.2006.05.038

Ruttens, A., Mench, M., Colpaert, J., Boisson, J., Carleer, R., & Vangronsveld, J. (2006). Phytostabilization of a metal contaminated sandy soil. I: Influence of compost and/or inorganic metal immobilizing soil amendments on phytotoxicity and plant availability of metals. *Environmental Pollution*, 144(2), 524–532. https://doi.org/10.1016/j.envpol.2006.01.038

Sage, R. F., & Zhu, X. G. (2011). Exploiting the engine of C4 photosynthesis. *Journal of Experimental Botany*, 62(9), 2989–3000. https://doi.org/10.1093/jxb/err179

Sahu, G. K., Upadhyay, S., & Sahoo, B. B. (2011). Mercury induced phytotoxicity and oxidative stress in wheat (*Triticum aestivum* L.) plants. *Physiology and Molecular Biology of Plants*, 18(1), 21–31. https://doi.org/10.1007/s12298-011-0090-6

Saletnik, B., Zaguła, G., Bajcar, M., Tarapatskyy, M., Bobula, G., & Puchalski, C. (2019). Biochar as a Multifunctional Component of the Environment – A Review. *Applied Sciences*, 9(6), 1139. https://doi.org/10.3390/app9061139

Schneiderová, K., & Šmejkal, K. (2014). Phytochemical profile of *Paulownia tomentosa* (Thunb). Steud. *Phytochemistry Reviews*, 14(5), 799–833. https://doi.org/10.1007/s11101-014-9376-y

Sheoran, V., Sheoran, A., & Poonia, P. (2009). Phytomining: A review. *Minerals Engineering*, 22(12), 1007–1019. https://doi.org/10.1016/j.mineng.2009.04.001

Shrestha, P., Bellitürk, K., & Görres, J. H. (2019). Phytoremediation of Heavy Metal-Contaminated Soil by Switchgrass: A Comparative Study Utilizing Different Composts and Coir Fiber on Pollution Remediation, Plant Productivity, and Nutrient Leaching. *International Journal of Environmental Research and Public Health,* 16(7), 1261. https://doi.org/10.3390/ijerph16071261

Shumaker, K. L., & Begonia, G. (2005). Heavy Metal Uptake, Translocation, and Bioaccumulation Studies of *Triticum aestivum* Cultivated in Contaminated Dredged Materials. *International Journal of Environmental Research and Public Health,* 2(2), 293–298. https://doi.org/10.3390/ijerph2005020013

Sonowal, S., Prasad, M. N. V., & Sarma, H. (2018). C3 and C4 plants as potential phytoremediation and bioenergy crops for stabilization of crude oil and heavy metal co-contaminated soils-response of antioxidative enzymes. *Tropical Plant Research*, 5(3), 306–314. https://doi.org/10.22271/tpr.2018.v5.i3.039

Srivastava, J., Chandra, H., R., A., & S. Kalr, S. J. (2012). Response of C3 and C4 plant systems exposed to heavy metals for phytoextraction at elevated atmospheric CO_2 and at elevated temperature. In *Environmental Contamination*. https://doi.org/10.5772/31333

Stochmal, A., Moniuszko-Szajwaj, B., Zuchowski, J., Pecio, Kontek, B., Szumacher-Strabel, M., Olas, B., & Cieslak, A. (2022). Qualitative and quantitative analysis of secondary metabolites in morphological parts of *Paulownia* Clon *In Vitro* 112® and their anticoagulant properties in whole human blood. *Molecules,* 27(3), 980. https://doi.org/10.3390/molecules27030980

Sun, Y., & Cheng, J. (2002). Hydrolysis of lignocellulosic materials for ethanol production: a review. *Bioresource Technology*, 83(1), 1–11. https://doi.org/10.1016/s0960-8524(01)00212-7

Tchounwou, P. B., Yedjou, C. G., Patlolla, A. K., & Sutton, D. J. (2012). Heavy metal toxicity and the environment. *Experientia Supplementum*, 133–164. https://doi.org/10.1007/978-3-7643-8340-4_6

Tejada, M., Garcia, C., Gonzalez, J., & Hernandez, M. (2006). Use of organic amendment as a strategy for saline soil remediation: Influence on the physical, chemical and biological properties of soil. *Soil Biology and Biochemistry*, 38(6), 1413–1421. https://doi.org/10.1016/j.soilbio.2005.10.017

Terry, N., Zayed, A.M., De Souza, M.P., & Tarun, A.S. (2000). Selenium in higher plants. *Annual Review Plant Biology*, 51, 401–432. https://doi.org/10.1146/annurev.arplant.51.1.401

Ueno, O. (2001). Environmental Regulation of C3 and C4Differentiation in the Amphibious Sedge *Eleocharis vivipara*. *Plant Physiology,* 127(4), 1524–1532. https://doi.org/10.1104/pp.010704

van der Ent, A., Baker, A. J. M., Reeves, R. D., Chaney, R. L., Anderson, C. W. N., Meech, J. A., Erskine, P. D., Simonnot, M. O., Vaughan, J., Morel, J. L., Echevarria, G., Fogliani, B., Rongliang, Q., & Mulligan, D. R. (2015). Agromining: Farming for Metals in the Future? *Environmental Science & Technology*, 49(8), 4773–4780. https://doi.org/10.1021/es506031u

Vangronsveld, J., Herzig, R., Weyens, N., Boulet, J., Adriaensen, K., Ruttens, A., Thewys, T., Vassilev, A., Meers, E., Nehnevajova, E., van der Lelie, D., & Mench, M. (2009). Phytoremediation of contaminated soils and groundwater: lessons from the field. *Environmental Science and Pollution Research*, 16(7), 765–794. https://doi.org/10.1007/s11356-009-0213-6

Velasco-Arroyo, B., Curiel-Alegre, S., Khan, A. H. A., Rumbo, C., Pérez-Alonso, D., Rad, C., De Wilde, H., Pérez-de-Mora, A., & Barros, R. (2024). Phytostabilization of metal(loid) s by ten emergent macrophytes following a 90-day exposure to industrially contaminated groundwater. *New Biotechnology*, 79, 50–59. https://doi.org/10.1016/j.nbt.2023.12.003

Vural, A., & Safari, S. (2022). Phytoremediation ability of *Helichrysum arenarium* plant for Au and Ag: case study at Demirören village (Gümüşhane, Turkey). *Gold Bulletin*, 55(2), 129–136. https://doi.org/10.1007/s13404-022-00313-z

Willscher, S., Jablonski, L., Fona, Z., Rahmi, R., & Wittig, J. (2017). Phytoremediation experiments with *Helianthus tuberosus* under different pH and heavy metal soil concentrations. *Hydrometallurgy*, 168, 153–158. https://doi.org/10.1016/j.hydromet.2016.10.016

Wilson-Corral, V., Anderson, C. W., & Rodriguez-Lopez, M. (2012). Gold phytomining. A review of the relevance of this technology to mineral extraction in the 21st century. *Journal of Environmental Management*, 111, 249–257. https://doi.org/10.1016/j.jenvman.2012.07.037

Wong, M. (2003). Ecological restoration of mine degraded soils, with emphasis on metal contaminated soils. *Chemosphere*, 50(6), 775–780. https://doi.org/10.1016/s0045-6535(02)00232-1

Wright, L., & Turhollow, A. (2010). Switchgrass selection as a "model" bioenergy crop: A history of the process. *Biomass and Bioenergy*, 34(6), 851–868. https://doi.org/10.1016/j.biombioe.2010.01.030

Wright, L. L. (2007). H*istorical perspective on how and why switchgrass was selected as a "Model" high-potential energy crop*. https://doi.org/10.2172/929781

Wuana, R. A., & Okieimen, F. E. (2011). Heavy metals in contaminated soils: a review of sources, chemistry, risks and best available strategies for remediation. *ISRN Ecology*, 2011, 1–20. https://doi.org/10.5402/2011/402647

Xu, Q., Li, X., Ding, R., Wang, D., Liu, Y., Wang, Q., Zhao, J., Chen, F., Zeng, G., Yang, Q., & Li, H. (2017). Understanding and mitigating the toxicity of cadmium to the anaerobic fermentation of waste activated sludge. *Water Research*, 124, 269–279. https://doi.org/10.1016/j.watres.2017.07.067

Yan, A., Wang, Y., Tan, S. N., Mohd Yusof, M. L., Ghosh, S., & Chen, Z. (2020). Phytoremediation: A promising approach for revegetation of heavy metal-polluted land. *Frontiers in Plant Science*, 11, 359. https://doi.org/10.3389/fpls.2020.00359

Zhao, F. J., Ma, J. F., Meharg, A. A., & McGrath, S. P. (2008). Arsenic uptake and metabolism in plants. *New Phytologist*, 181(4), 777–794. https://doi.org/10.1111/j.1469-8137.2008.02716.x

Zheng, W. J., Zheng, X. P., & Zhang, C. L. (2000). A survey of photosynthetic carbon metabolism in 4 ecotypes of *Phragmites australis* in northwest China: Leaf anatomy, ultrastructure, and activities of ribulose 1,5-bisphosphate carboxylase, phosphoenolpyruvate carboxylase and glycollate oxidase. *Physiologia Plantarum*, 110(2), 201–208. https://doi.org/10.1034/j.1399-3054.2000.110209.x