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REVIEW ARTICLE

# Environmental impacts of mine tailings and phytoremediation as a sustainable management strategy: A review

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**Abstract** Mining activities are often associated with significant environmental degradation, particularly due to the accumulation of mine tailings (MTs). These waste materials are frequently stored in dams or open ponds without adequate treatment, posing serious risk of heavy metals (HMs) contamination to surrounding ecosystems. Given these challenges, restoration of MTs to mitigate their negative impacts has become highly important. This study attempts to compile different types of MTs, their characteristics, and associated issues such as acid mine drainage (AMD) and HMs contamination, along with other environmental impacts. It also explores the fundamentals of phytoremediation, highlighting key processes, recent advancements, benefits, limitations, and strategies for post-harvest management. The findings indicate that MTs are a major source of HM pollution and contribute significantly to environmental deterioration. Phytoremediation has emerged as a promising, cost-effective, and eco-friendly solution for MT restoration. In addition to mitigating contamination, phytoremediation enhances soil quality, prevents erosion, reduces HM leaching into groundwater, and improves the visual appeal of degraded sites. Research suggests that revegetating MT-contaminated soils with specific plant species can effectively remediate these areas, reducing HM leaching risks while improving soil properties. This review serves as a valuable resource for researchers working on MT restoration, offering insights into the latest advancements in phytoremediation technology and its potential to address the environmental challenges posed by MTs.

**Keywords** Heavy metals · Mine-tailings · Acid mine drainage · Phytoremediation

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## 1 Introduction

Mining is an essential component of the global economy, providing the necessary resources for the progress and development of modern society. Nevertheless, the exponential increase in the population exacerbates consumption patterns, creating a significant strain on natural resources (Srivastava and Gupta 2020). Consequently, mining operations are carried out on a large scale to facilitate economic growth and fulfil the needs of the present generation (Yin et al. 2018). Typically, mineral processing results in two primary types of products: those with economic value and those without. The waste materials left after extracting valuable minerals from ore during mining operations, consisting of ground rock, process water, chemicals, and sometimes organic matter, are commonly known as tailings. The amount of tailings produced during mineral extraction can be almost equivalent to the quantity of raw material that is processed (Adiansyah et al. 2015). Each year, mine operations globally produce 10 billion tonnes of mine tailings (MTs) (Xie and Zyl 2020; Sibanda et al. 2019), and this significant amount presents a grave concern for the environment. The fine and dusty tailings tend to spread into the surrounding areas through wind or water erosion, and this dispersal can lead to contamination of the air and soil, which in turn can pose health risks to individuals (Sun et al. 2018). In addition to wind and surface water erosion, tailings can contaminate the environment through leachate formation, infiltration into groundwater, and runoff during heavy rainfall, which can lead to the mobilization of toxic substances, resulting in the degradation of soil quality and water contamination. These fine particles have a high surface area-to-volume ratio, which enhances their reactivity and ability to remain suspended in water for extended periods, and this prolonged suspension, known as turbidity, reduces water clarity, inhibits photosynthesis by limiting light penetration, and adversely affects aquatic ecosystems, including phytoplankton and fish populations. Contaminated water not only affects aquatic ecosystems and biodiversity but also poses serious health risks to nearby communities reliant on these sources for drinking and irrigation.

Further, the presence of significant metal content in the tailings is a cause for concern, as the interactions and combined effects of these components can worsen the environmental impact (Buch et al. 2021). Soil properties in terrestrial environments have the potential to impact the chemical speciation and fractionation of metals, which can in turn influence their mobility, bioavailability, leaching, and toxicity (Liu et al. 2020). The presence of heavy metals (HMs) like arsenic (As), cadmium (Cd), chromium (Cr), mercury (Hg), lead (Pb), and nickel (Ni) in mine waste poses a significant threat to living organisms due to their highly toxic and carcinogenic nature (Ali et al. 2019). HMs are distinguished by their high density, which is larger than 4–5 g/cm$^3$, and their atomic number, which is above 20, while their atomic weights range from 63.54 to 200.59. The metals include transition metals, metalloids, actinides, and lanthanides. The rise of HMs, sometimes known as 'trace elements,' in the environment is a cause of concern for two primary reasons. Firstly, both humans and animals can consume these toxic substances by eating contaminated food or feeding on them or inhaling them as dust particles. Their potential for harm to the environment and human health is significant due to their high toxicity, non-biodegradability, and ability to accumulate in the food chain (Ali et al. 2013).

Studies have shown that agricultural soils contaminated with metals can have negative impacts on the health of both humans and local wildlife. According to a report (USEPA 2016), it is estimated that around ten million individuals across the globe are affected by metal-contaminated soils. Secondly, elevated levels of HMs in soils have phytotoxic effects, hampering vegetation establishment and making soils susceptible to erosion, which leads to further dispersion of pollutants to new areas, posing health risks to larger populations. In addition, HMs have the potential to disrupt several important processes in plants, including photosynthesis, water relationships, and biochemical and enzymatic activities (Ahmad et al. 2015; Ghori et al. 2016). Further, HMs threaten soil microorganisms by accumulating in their cells and tissues, hindering their biological activities and increasing toxicity through the generation of reactive oxygen species, which can cause cell damage and DNA degradation (Ahmad et al. 2015). Furthermore, the failure of tailings storage facilities with time due to structural weaknesses or natural disasters like earthquakes, landslides, or extreme rainfall can lead to catastrophic spills, releasing vast quantities of toxic waste into surrounding water bodies and landscapes. Such events have long-term ecological consequences and social implications, including displacement and loss of livelihoods. Given the multitude of harmful effects caused by HMs on the environment, it is crucial to take immediate and sustainable measures to tackle this problem.

Phytoremediation is considered to be a promising alternative to traditional physical and chemical techniques for the treatment of HM-contaminated soil due to its efficiency, cost-effectiveness, environmental friendliness, sustainability, and social viability (Farooq et al. 2020). This method utilizes the inherent capacities of plants to remove, carry, stabilize, and break down contaminants, such as HMs and organic compounds, in the rhizosphere (Prasad 2021). The plant-centric nature of this provides visually appealing surroundings and makes it well-suited for extensive use in large areas. Phytoremediation offers multiple advantages when applied to polluted soils. Significantly, it improves the amount of organic matter in the soil and the activity of microorganisms, which is essential for restoring contaminated areas (Deepika and Haritash 2023a). Furthermore, it aids in diminishing soil erosion and the creation of dust, functioning as a barrier against direct sunlight exposure and preserving the moisture content of the soil. Phytoremediation offers a notable benefit in terms of its influence on the retention and movement of water. Plants utilize transpiration to absorb significant quantities of water through their roots, which limits the downward movement of water into the soil and prevents the migration of pollutants toward groundwater (Karaca et al. 2018). The application of this technique is particularly relevant during the post-mining period, when the tailings storage facilities or dams are no longer in active use. At this stage, preventing the leaching of hazardous substances and exposure to air becomes critical. Phytoremediation can serve as a natural protective barrier, helping to minimize the risk of environmental contamination from residual MTs. Integrating phytoremediation as part of post-mining land reclamation efforts can help mitigate these risks and promote long-term ecological stability. Previously, plant species such as *Pteris vittata* L. (Yang et al. 2020), *Pteris cretica* L. (Jeong et al. 2015), *S. alfredii* L. (Zhu et al. 2019), *Viola baoshanensis* L. (Wu et al. 2010), *Solanum viarum* L. Dunal (Afonso et al. 2019), *Miscanthus sinensis* L. (Ridošková et al. 2019), *Phragmites australis* L. (Srivastava et al. 2014), *Cynodon dactylon, Sorghastrum nutans, Acacia concinna* (Kumar et al. 2017), *Rumex bucephalophorus* L., *Chrysopogon zizanioides* L., *Silene colorata* L. (Chaabani et al. 2017), *Arabidopsis halleri* L. and *T. caerulescens* L. (Liang et al. 2009), *Rorippa* *globosa* L. (Sun et al. 2010), *Pongamia pinnata* L. (Yu et al. 2019), and *Phytolacca acinose* L. (Xue et al. 2010) have been reported for the phytoremediation of MTs.

The primary objective of this study is to provide a comprehensive review of HMs contamination in MTs, highlighting its impact on soil, water, and ecosystem health. The study aims to explore the potential of phytoremediation as a sustainable approach for removing, stabilizing, and reducing the mobility of HMs in MT-contaminated soils. In addition, the study is set to investigate the recent advances in phytoremediation research and to provide researchers and stakeholders with valuable insights into facilitating effective MT reclamation and promoting ecological restoration. Moreover, strategies for post-harvest management for treating contaminated biomass, ensuring the safe disposal or reuse of HM-laden plant material have been outlined.

### 1.1 Methodology

The methodology for this review paper involved a systematic and comprehensive literature search to compile relevant information on MTs, their environmental impacts, and phytoremediation as a restoration strategy. Peer-reviewed articles, books, and reports were sourced from databases such as PubMed, Scopus, Web of Science, and Google Scholar, using keywords including "mine tailings," "heavy metal contamination," "acid mine drainage," "phytoremediation," and "environmental restoration." The search on phytoremediation reports was limited to publications from the last ten years to ensure relevance, although foundational studies were included where necessary. Data were organized into categories covering MT characteristics, associated environmental issues (e.g., HM pollution and acid mine drainage), and phytoremediation processes, including mechanisms, plant species, advancements, benefits, and limitations. Studies were critically evaluated for methodological rigor and relevance to MT restoration. The review synthesized findings to highlight phytoremediation's efficacy, challenges, and post-harvest management strategies, providing a cohesive overview of current knowledge and research gaps.

## 2 Characteristics of HMs mine-tailings

MTs can contain metals, such as Fe, Cu, Ni, and Zn, in relatively high amounts (ranging from 0.5% to 3%), and occasionally valuable metals like gold (Au) and silver (Ag). As toxic metals, they can be found in amounts of up to 100 mg/kg (Ceniceros-Gómez et al. 2018). In addition, tailings lack essential nutrients such as nitrogen (N) and phosphorus (P), as well as organic matter, resulting in a low cation exchange capacity, which, in turn, leads to the leaching of inorganic nutrients (Ghosh and Maiti 2021). Although the physical and chemical properties of MTs differ depending on the type of ore and processing techniques used (as shown in Table 1), certain broad physicochemical features are typically found in most tailings (discussed below).

(1) Particle size: The particle size of tailings can vary significantly depending on the unique process requirements. In general, tailings are characterized as fine-grained particles, ranging in size from silt-sized to sand-sized particles, measuring between 1 and 600 µm (Sun et al. 2018; Wang et al. 2017). Because of their small particle size, tailings exhibit high reactivity and tend to disperse quickly in the surrounding areas by wind erosion during drought seasons, leading to pollution of the air, water, and soil.

(2) Bulk density: The bulk density of tailings is determined by the composition of the original rock. Usually, tailings have a bulk density that falls within the range of 1.8–1.9 t $m^{-3}$ (Kossoff et al. 2014). However, certain tailings, such as vanadium–titanium iron ore tailings, can have a significantly higher density, reaching 3.133 t/$m^3$ (Tian et al. 2024). Tailings are produced during the crushing and grinding phases of the beneficiation process, resulting in a decrease in bulk density and an increase in specific surface area. Storing fine tailings in unstable dams might lead to the potential dangers of collapse and landslides, especially during the rainy season (Liu and Huang 2017).

(3) Water content: MTs generally contain a substantial quantity of water, which might come from either the processing of ore or from rainfall and natural drainage. The moisture content can impact the stability and behavior of the tailings, especially their capacity for leaching and transportation (Hu et al. 2017).

(4) Chemical composition: The chemical composition of MTs varies significantly depending on the extracted minerals and processing methods. Major elements include Al, Ca, K, Mg, Mn, Na, P, Ti, Si, Fe, and S. Trace amounts of precious metals, such as Au, Ag, and Cu are also present. Additionally, chemical residues from beneficiation processes, such as cyanide from Au and Ag leaching or flotation reagents like potassium amyl xanthate and surfactants from sulfide ore processing, may persist in tailings. These residues can contribute to environmental toxicity, posing risks to ecosystems if not properly contained or treated (Li et al. 2021).

(5) Mineral composition: The mineral composition of MTs significantly influences their acid-producing and acid-neutralizing capacities. Aluminosilicate minerals, such as muscovite, chlorite, and plagioclase, offer limited acid-neutralizing potential due to their slow reaction rates, while carbonate minerals like calcite are far more effective at buffering acidity (Gitari et al. 2018). Conversely, sulphide minerals, including pyrite and pyrrhotite, drive acid production through oxidation, leading to low pH levels and the formation of AMD (Jamieson et al. 2012).

**Table 1** Characteristics of MTs and range of HMs concentration in contaminated soil

| Types of MTs | Physical properties | Geochemical properties | Mineralogical properties | References |
|---|---|---|---|---|
| Iron MTs | Particle size: fine (< 74 μm in diameter), coarse (up to 150 μm), specific gravity: 3.23 (coarse tailings), 3.08 (fine tailings) | Water content: ≤ 20%, bulk density- > 1.5 g $cm^{-3}$, less nutrients (C, N < 1%), salinity (~ 60 dS·$m^{-1}$), less organic matter (< 1%), extreme pH (2–11)<br>Range of HM concentration (mg·$kg^{-1}$): Fe- 2928–118,390, Mn- 2.9–2800, Cr- 43.7–150, Zn- 1.7–114, Ni- 1–43.73, Cu- 1.7–95, Pb- 0.5–44, Co- 0.1–1.9, Cd- 0–0.73 | Quartz ($SiO_2$), hematite ($Fe_2O_3$), magnetite ($Fe_3O_4$), goethite (α-FeOOH), kaolinite [$Si_2Al_2O_5(OH)_4$], bauxite ($Al_2O_3$), gibbsite (Al $(OH)_3$) | Carmignano et al. 2021; Fazekašová and Fazekaš 2020; Hu et al. 2017; Podgórska and Jóźwiak 2024; Sarathchandra et al. 2023 |
| Copper MTs | Texture: sandy silt, specific gravity: 2.77 (coarse tailings), 2.76 (fine tailings) | Extreme pH (due to the presence of pyrite and alumino-silicates), low organic carbon (0.22% ± 0.10%)<br>Range of HM concentration (mg·$kg^{-1}$): Cu- 231–864, Pb- 12–302, Zn- 180–1149, Cr- 15.3–54, Ni- 9–42, Mg- 287 | Quartz, azurite [$Cu_3(CO_3)_2(OH)_2$], tenorite (CuO), chalcopyrite ($CuFeS_2$), chalcocite ($Cu_2S$), covellite (CuS), cuprite ($Cu_2O^+$) bornite ($Cu_5FeS_4$), malachite ($Cu_2CO_3(OH)_2$), clinochlore [(Mg, $Fe_2{}^+$)$5Al_2Si_3O_{10}(OH)_8$], dolomite [$CaMg(CO_3)_2$], siderite, hematite, calcite ($CaCO_3$), pyrite ($FeS_2$), epidote ($Ca_2(Fe,Al)Al_2(SiO_4)(Si_2O_7)$ O(OH)), muscovite, and gypsum | Afonso et al. 2020; Gascó et al. 2019; Gitari et al. 2018; Hu et al. 2017; Jannesar Malakooti et al. 2014; Munir et al. 2020; Punia 2021; Shi et al. 2016 |
| Gold MTs | Specific gravity-2.6, particle size: 0.4 μm-355 μm, texture: silt loam | Organic carbon: < 0.6%, pH: generally acidic<br>Range of HM concentration (mg·$kg^{-1}$): As- 21–2936, Cr- 237, Ni- 35–150, Zn- 64–939, Cu- 2.43–73, Pb- 12–348 | Quartz, jarosite ($KFe_3{}^+(OH)_6(SO_4)_2$), pyrophyllite, sepiolite ($Mg_4Si_6O_{15}(OH)_2 \bullet 6(H_2O)$), albite, clinochlore, mica, chlorite, magnetite, amphibole, feldspar, calcite and pyrite | Gitari et al. 2018; Krisnayanti et al. 2012; Lemos et al. 2020; Ngole-Jeme and Fantke 2017; Petelka et al. 2019; Redwan and Bamousa 2019 |
| Lead/zinc MTs | Specific gravity: 2.91, bulk density: 1.49 g/$cm^3$ | Range of HM concentration (mg $kg^{-1}$): As- 13–11,092, Cd- 0.31–280, Hg- 1–597, Pb- 21–28,453, Zn- 116–23,031, Cu- 35–1405, Ni- 9–44.4 | Quartz, anglesite ($PbSO_4$), calcite, cerussite ($PbCO_3$), balena (PbS), bianchite ($ZnSO_4{\cdot}6H_2O$), goslarite ($ZnSO_4{\cdot}7H_2O$), hydrozincite ($Zn_5(OH)6(CO_3)_2$), pyrite, smithsonite ($ZnCO_3$), ankerite [Ca $(Fe,Mg,Mn)(CO_3)_2$], bernalite $Fe(OH)_3$, sphalerite (ZnS), greenockite (CdS), zincite (ZnO), grunerite [$Fe_7Si_8O_{22}(OH)_2$], dolomite, jarosite, hematite, and calcite | Behera et al. 2019; Fernández et al. 2017; Gutiérrez et al. 2016; Hasnaoui et al. 2020; Sun et al. 2018 |
| Coal MTs | Texture: sandy loam | pH: slightly acidic to slightly alkaline and negative neutralizing potential, Organic carbon (%): 0.69–2.72<br>Range of HM concentration (mg·$kg^{-1}$): Fe- 33–11,956, As- 7–151, Cd- 0.1–13.7, Cr- 3.8–40.8, Cu- 11–4630, Pb- 18–1075, Ni- 5.1–85.6, Zn- 18–742 | Quartz, bauxite, lime (CaO), hematite, magnesia (MgO), mullite ($3Al_2O_3.2SiO_2$), magnetite, mica, kaolinite, dolomite, feldspar, calcite | Desai et al. 2019; Mdumela and Sengani 2021; Yagüe et al. 2018; Yeheyis et al. 2009 |

## 3 Environmental impacts of MTs

The production of MTs poses substantial environmental challenges, with two critical and globally significant concerns being soil contamination by HMs and the occurrence of AMD. The following section provides a detailed analysis of both of these issues.

### 3.1 AMD

AMD is a major global environmental problem characterized by its very acidic nature (pH less than 3), high amounts of sulfates, and the presence of toxic HMs and metalloids such as Cd, Cu, Fe, Mn, Pb, Zn, As, and Se (Park et al. 2019). Certain mine waste materials, namely those containing sulfide minerals such as pyrite (e.g., Cu and Pb/Zn tailings), can produce acidic rock drainage when they come into contact with air and water. The acidic runoff has the potential to release HMs and other pollutants into the nearby environment (Park et al. 2019; Xu et al. 2019). Its detrimental effects include increased suspended solids, HM mobilization, lowered pH in water bodies, groundwater contamination, and entry of toxic metals into the food chain. Exposure to toxic metals can damage cells and reduce cell viability in humans and animals (Dutta et al. 2020), while precipitated minerals from AMD can impact benthic organisms in receiving water bodies (Thomas et al. 2022). AMD not only negatively impacts aquatic ecosystems but also affects water quality intended for human consumption and irrigation (Zhu et al. 2020). Its entry into surface water bodies leads to direct toxicity to organisms, habitat alteration, visual staining of sediments, disruption of nutrient cycles, and renders water unsuitable for various uses (Evans et al. 2015; Skousen et al. 2019). Addressing AMD is challenging due to its severe environmental damage, high treatment costs, and poor accountability for remediation efforts (Carvalho 2017; Sheridan et al. 2018). For instance, in Johannesburg, South Africa, where approximately 250,000 cubic meters of AMD are discharged daily, the annual treatment cost through pH neutralization is estimated at 50 million USD (Sheridan et al. 2018).

#### *3.1.1 Chemistry and microbiology of AMD formation*

Pyrite ($FeS_2$) and arsenopyrite (FeAsS) are the main contributors to the formation of AMD. However, other minerals rich in sulfide, such as iron sulfides ($Fe_xS_x$), pentlandite [(Fe, Ni)$_9S_8$], chalcopyrite ($CuFeS_2$), villamaninite ($Cu_2S$), covellite (CuS), molybdenite ($MoS_2$), sphalerite [(Fe, Zn)S], millerite (NiS), and galena (PbS), also play a minor role in this process (Simate and Ndlovu 2014; Tabelin et al. 2017). Pyrite, which is frequently encountered in quartz veins of igneous and metamorphic rock, as well as in some sedimentary rocks such as coal beds and wetlands, plays a significant role in the creation of AMD (Thomas et al. 2022). Pyrite exhibits stability in both acidic and alkaline environments but undergoes oxidation upon exposure to molecular oxygen ($O_2$), water ($H_2O$), and microbes (Masindi et al. 2015). AMD development is related to four primary reactions. At first, the process of pyrite oxidation results in the production of ferrous iron and sulfate, which leads to the generation of 2 mol of acidity for every mole of pyrite (as shown in Eq. 1). Ferrous iron is then oxidized to ferric iron, consuming 1 mol of acidity (Eq. 2), this reaction can also take place when $Fe_2^+$ migrates to surface waters with higher pH, like rivers and dams, i.e., pH > 5 (Skousen et al. 2019). Ferric iron undergoes spontaneous hydrolysis to produce ferric hydroxide, which is observed as an orange-red precipitate in AMD (Eq. 3). However, this reaction is sluggish in acidic environments and only takes place at pH values more than 3.5 (Park et al. 2019). Additional metals present in MTs may experience analogous hydrolysis reactions under appropriate pH conditions. Equation 4 illustrates the fast process of pyrite undergoing oxidation by ferric iron, leading to the generation of 16 mol of acidity and ferrous iron until either ferric iron or pyrite is completely used up. This reaction explains the rapid generation of AMD when sulfide minerals are present in mining sites. Notably, pyrite oxidation occurs primarily with ferric iron, not oxygen. AMD development in surface layers of mine dumps is caused by the presence of atmospheric oxygen. In deeper sections of the dumps, ferric iron from the upper layer mixes with groundwater, which promotes additional oxidation of the MTs. Groundwater is crucial in the formation of AMD in underground materials. The lack of oxygen does not impede sulfide oxidation, posing difficulties in the rehabilitation of mine regions once AMD production begins (Karaca et al. 2018). Excessive $Fe^{3+}$ acts as a secondary agent that reduces pyrite, leading to the release of acidity into water, which is why AMD is characterized as acidic.

$$FeS_2 + \frac{7}{2}O_2 + H_2\mathrm{O} \rightarrow Fe^{2+} + 2SO_4^{2-} + 2H^+ \tag{1}$$

$$Fe^{2+} + \frac{1}{4}O_2 + H^+ \rightarrow Fe^{3+} + \frac{1}{4}H_2\mathrm{O} \tag{2}$$

$$Fe^{3+} + 3H_2\mathrm{O} \rightarrow Fe(OH)_3 + 3H^+ \tag{3}$$

$$FeS_2 + 14Fe^{3+} + 8H_2\mathrm{O} \rightarrow 15Fe^{2+} + 2SO_4^{2-} + 16H^+ \tag{4}$$

These reactions usually take place naturally and are facilitated by microorganisms that obtain energy from oxidation reactions. Acidophilic bacteria, such as *Acidithiobacillus* and *Thiobacillus* (at low pH) and sulfur-based bacteria like *Sulfolobus* and *Desulfovibrio* (at low to neutral pH), are important catalysts for these processes (Masindi et al. 2022). The breakdown of pyrite would be significantly slower without the presence of acidophilic bacteria. *Acidithiobacillus ferrooxidans*, also referred to as *Thiobacillus ferrooxidans*, is one of the extensively studied acidophiles. These bacteria live on sulfide-mineral surfaces and obtain energy by oxidizing iron. Nevertheless, when oxygen is not present, they can also undergo oxidation of reduced sulfur (and occasionally hydrogen). Several acidophilic microorganisms that oxidize iron also have the ability to oxidize sulfur. *A. ferrooxidans* is capable of thriving in surroundings with different pH values. Its inner cytoplasm maintains a pH of 7, while its outer membrane is specifically adapted to a pH of 2 (Jacobs et al. 2016). This modification enables them to undergo oxidation of iron or sulfur under neutral pH conditions, resulting in subsequent acidification, even in non-acidic surroundings. Acidophilic bacteria, particularly *T. ferrooxidans,* can greatly expedite Eqs. (2) and (4) in naturally acidic conditions (Moodley et al. 2018).

The formation of AMD is chemically simple, but the ultimate result is influenced by several factors such as the geological features of the mining area, the presence of microbes, temperature, and the availability of water and oxygen. The parameters mentioned exhibit substantial variation across different regions, emphasizing the importance of meticulous and precise evaluation when predicting, preventing, containing, and treating AMD (Simate and Ndlovu 2014).

## 3.2 HMs contamination in the environment

MTs are significant sources of HM contamination, impacting air, water, soil, ecosystems, and human health. The following subsections outline the exposure pathways, environmental pollution, ecological effects, and human health impacts caused by HMs from MTs.

### *3.2.1 Exposure pathways*

High-intensity winds mechanically erode tailings, generating coarse (2.5–10 µm), fine (2.5 µm), and ultrafine (< 2.5 µm) particles that can be transported over long distances due to their aerodynamic properties (Csavina et al. 2011; Ceniceros-Gómez et al. 2018). This effect is particularly pronounced in arid and semi-arid regions, where limited precipitation and high wind velocities enhance dust dispersion. Additionally, semi-arid conditions promote oxidation, forming highly soluble hydrated sulfates on tailings surfaces, which can carry HMs such as As, Pb, Sb, and Zn, increasing their mobility and environmental spread (Corona Sánchez et al. 2021; González-Sánchez et al. 2023). Water erosion also contributes to HM migration, with mine residue material detected in agricultural soil and road dust near abandoned tailings (Del Rio-Salas et al. 2019).

### *3.2.2 Environmental pollution*

Air Pollution: Tailings emit particulate matter containing metals and toxic aerosols, degrading air quality beyond the immediate vicinity of mining operations (Noble et al. 2017). Fine and ultrafine particles, due to their small size, can travel considerable distances, impacting regional air quality (Corona Sánchez et al. 2021).

Soil Contamination: Aerial deposition and water-mediated transport of HM-laden dust contaminate soils, including agricultural lands. For instance, soils near the Kombat tailings dam in Namibia showed high levels of Cu (up to 150 mg/kg) and Pb (up to 164 mg/kg) due to prolonged wind and water erosion (Mileusnić et al. 2014). Similarly, soil samples near a former Au-mine in Johannesburg, South Africa, contained significant As concentrations (13.46–234.6 mg/kg) (Olobatoke and Mathuthu 2016). Polluted soils can further act as secondary sources of HM contamination, exacerbating environmental degradation (Gholizadeh et al. 2015).

Water Pollution: HM-laden dust and eroded tailings can contaminate water bodies, negatively impacting aquatic systems through the deposition of metals like Hg and Pb (Cleaver et al. 2021; Xiao et al. 2017).

### *3.2.3 Ecological effects*

HM contamination from MTs disrupts ecosystems by affecting soil, plants, and animals. Elevated HM concentrations in soils reduce microbial activity, population, and enzymatic functions, altering soil ecosystems (Ngole-Jeme and Fantke 2017). Plants uptake HMs, leading to reduced agricultural output due to inhibited physiological processes such as photosynthesis, chlorophyll formation, and reproduction (Singh and Kalamdhad 2011). For example, Cu impairs photosynthesis and reproduction, Pb reduces chlorophyll production, and As disrupts metabolic activities, all of which diminish plant growth (Abdul-Wahab and Marikar 2012). HM deposition on plant surfaces further affects photosynthesis and transpiration rates. Plants accumulating HMs in their tissues pose risks to herbivores, with examples including rice and leafy vegetables near a tailing pond in Guangdong, China, showing high HM levels (Liang et al. 2017a, b). Arthropods, small mammals, and large mammals are similarly affected by HM-contaminated soils, leading to broader ecological impacts (Gall et al. 2015).

### *3.2.4 Human health effects*

HMs from MTs pose significant risks to human health through direct and indirect exposure pathways. Direct exposure occurs via inhalation of HM-laden dust and dermal absorption, while indirect exposure happens through the food chain, particularly from consuming contaminated crops (Su et al. 2024; Kamanzi et al. 2023). Fine and ultrafine particles penetrate deeply into the respiratory system, increasing bioavailability and causing pneumoconiotic diseases (Su et al. 2024). Long-term HM exposure can lead to severe health complications, e.g., chronic exposure to Cd can cause Itai-Itai disease (Pan et al. 2010), while Zn can result in kidney dysfunction and anemia (Nriagu 2011). Gastrointestinal distress, liver or kidney damage can occur due to heavy metal exposure (Achparaki et al. 2012). Ni exposure can lead to cancer, heart disease, and skin irritation (Duda-Chodak and Blaszczyk 2008), while Pb exposure can cause cardiovascular problems and damage to the nervous system (Singh and Kalamdhad 2011). Lastly, Cr exposure can result in cancer and ulcers (Shekhawat et al. 2015). Additionally, high HM levels in food can reduce essential nutrients like vitamin C and iron, weakening the immune system and contributing to malnutrition-related disabilities (Liang et al. 2017a, b). Health risk assessments have shown elevated cancer risks (above $10^{-6}$ to $10^{-4}$) from HM-contaminated rice and vegetables near tailings (Liang et al. 2017a, b) (Fig. 1).

## 4 Phytoremediation: a sustainable strategy

The mineralogy of tailings and site characteristics like soil permeability, leachability of chemicals, and contamination depth are important factors to be considered while designing a management strategy for a mine tailing contamination (Lwin et al. 2018; Palansooriya et al. 2020). Depending upon the soil characteristics and the distribution of contaminants, various physicochemical techniques have been developed to stabilize, remove, or recover HMs from contaminated sites. These include, soil replacement (Song et al. 2022), immobilization (Vasarevicius et al. 2020), vitrification (Ballesteros et al. 2017), soil washing/flushing (Alaboudi et al. 2020), chemical leaching (Alghanmi et al. 2015), electrokinetic extraction (Sun et al. 2023), and ion-exchange (Hussain and Ali 2021). These processes help mitigate the risk of exposure, weathering, and dissolution of HMs into the environment by the action of rain, water runoff, and wind. Overall, the physicochemical remediation processes are fast, effective, and widely used for remediating metal-contaminated sites. Despite the benefits of these interventions, there are concerns about their sustainability and environmental impact. These methods are energy, chemicals, and labor-intensive and require technological intricacy, adding up to the cost of treatment (Deepika and Haritash 2023a; b; Karaca et al. 2018; Wang et al. 2017). The use of chemical agents may change the soil properties, damage soil structure, and produce secondary pollutants, which negatively affect the environment. These techniques employed in soil remediation frequently render the area unfavorable to plant growth by eradicating all biological processes, including beneficial microorganisms like nitrogen-fixing bacteria and mycorrhizal fungi, as well as fauna. Considering these

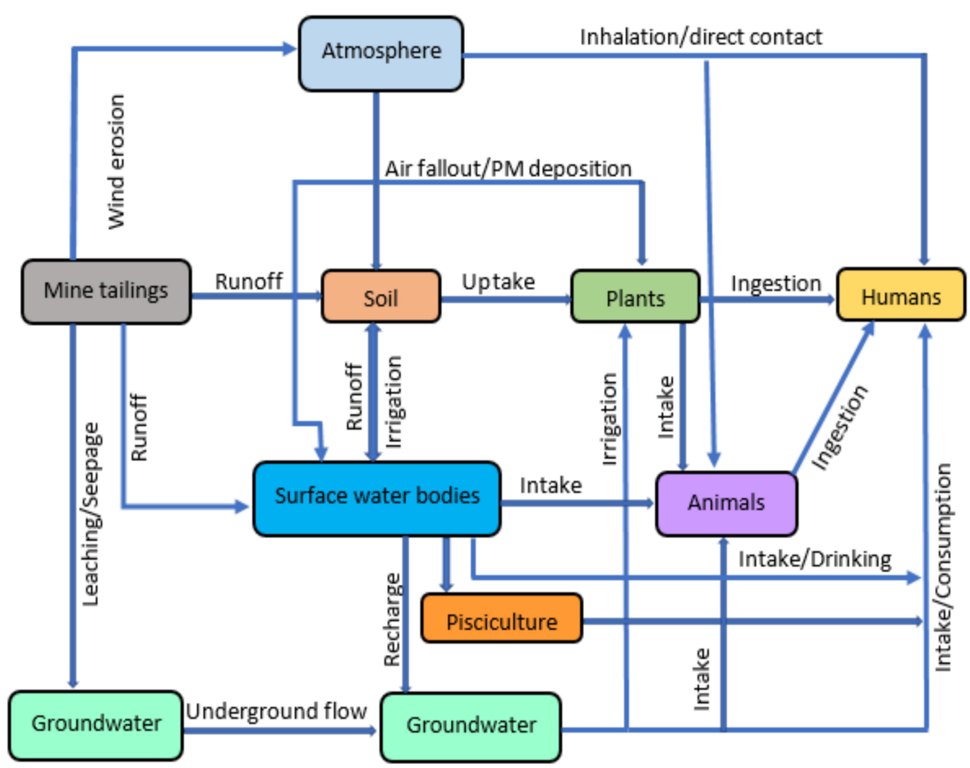



**Fig. 1** Fate pathways of HMs emissions from MTs in the environment

constraints, it is necessary to investigate more feasible alternatives. Unlike physical and chemical treatments, biological techniques like phytoremediation require minimum excavation of soil, thereby reducing erosion and preserving the landscape, while it requires minimum inputs and operational expenses as the method relies on natural plant growth and metabolism. Plants improve soil health and quality by adding organic matter and improving soil structure, thereby transforming contaminated areas into green spaces.

Plants, like all other living beings, require nutrients and minerals to grow and develop. Soil nutrient availability varies due to factors such as temperature, precipitation, soil type and pH, oxygen levels, and the presence or absence of various inorganic and organic components (DalCorso et al. 2019). Being immovable organisms, plants have evolved adaptable and versatile mechanisms for detecting and reacting to changes in the availability of nutrients to maximize their growth, development, and reproduction in various environmental situations (Krämer et al. 2007). HMs are naturally present elements in the earth's crust, and certain of these (namely Zn, Cu, Ni, Co, and Mo) are necessary for the growth and development of plants (Kim et al. 2015), while other HMs such as Cd, Hg, Ag, Pb, and Cr are biologically non-essential and show toxicity even at low concentrations (Ramírez 2013). The similarity of certain non-essential metals to essential ones allows the latter to enter plants replacing their essential homolog and interfering with biological functions (Broadley et al. 2012). Plants have developed a homeostatic network to regulate the intake, trafficking, storage, and detoxification of HMs in order to minimize their negative impacts while still allowing the absorption of vital elements (DalCorso et al. 2019). In addition, bacteria found in the rhizosphere of plants possess the capacity to eliminate various pollutants from the environment through a variety of enzymatic mechanisms. Due to their versatility, adaptability, and diversity in the environment, a variety of microbes and plants can be considered an excellent system for the remediation of environmental pollutants, including both organic and inorganic ones (Chatterjee et al. 2013). Given these characteristics, plants can be seen as a "natural, solar-powered pump and treat system" (Pilon-Smits 2005) that cleans contaminated environments, and the process is known as phytoremediation. It is a process of utilizing green plants, such as grasses and woody species, to eliminate, confine, or neutralize environmental pollutants such HMs, metalloids, trace elements, organic compounds, and radioactive substances found in soil or water. This concept encompasses all the biological, chemical, and physical processes controlled by plants that assist in the absorption, storage, breakdown, and transformation of pollutants. These processes can be carried out by plants themselves, soil microorganisms, or by interactions between plants and microorganisms (Sharma and Pandey 2014). This technique is economical, environment-friendly, visually appealing, and sustainable for managing contaminated environments. Furthermore, there are other benefits to this approach:

(1) Biomass utilization: Plants grown in a contaminated area can be harvested for biomass to be used as biofuel. Alternatively, these plants can continue to grow on-site, serving as pioneer species to help the ecosystem recover, increase local biodiversity, and contribute to the absorption of atmospheric $CO_2$ and the restoration of disturbed soils (Bell et al. 2014; Jiang et al. 2015).

(2) Metal recovery and waste reduction: Harvested biomass can enable the recovery and reuse of valuable metals through thermal, microbiological, or chemical methods to reduce size or mass (Patra and Mohanty 2013). In the case of lower metal concentration, the ash produced during biomass processing is significantly less in volume, approximately 20–30 t per 5,000 t of soil (Ghosh and Singh 2005).

(3) Ease of application: This can be performed either in situ or ex situ. In situ applications are often preferred due to their ability to minimize soil and environmental disturbance, as well as decrease the spread of contamination by air and waterborne waste (Moosavi and Seghatoleslami 2013).

(4) Soil carbon sequestration: This process captures $CO_2$ from the atmosphere and stores it in soil carbon pools, including humus, root exudates, and microbial biomass (Thomas et al. 2022).

(5) Soil quality improvement: This has a tendency to enhance the quality of soil by increasing the amount of organic matter, adjusting the pH level, promoting the growth of soil microorganisms and preventing soil erosion (Jacob et al. 2018).

(6) Environmental adaptability: This has the potential to be used in a wide range of temperatures and under extreme environmental circumstances. For instance, invasive plants exhibit significant efficacy in nations lacking native phytoremediation plants when cultivated in polluted areas (Prabakaran et al. 2019).

(7) Cost-effectiveness and scalability: This is cost-effective and suitable for larger areas and therefore does not necessitate expensive equipment or highly specialized personnel.

(8) Versatility: This has the capacity to remediate places contaminated with multiple types of pollutants (Hegedusova et al. 2009).

Phytoremediation utilizes many methods, including degradation (photodegradation), accumulation (phytoextraction, rhizofiltration), dissipation (phytovolatilization), and immobilization (phytostabilization), to break down, remove, or immobilize pollutants (Balint-Ponici et al. 2019) (Fig. 2). Plants utilize many techniques to reduce the concentrations of harmful substances in soil, which vary based on the types of contaminants found. Phytoextraction and phytostabilization are the main methods used to remediate HMs in soil.

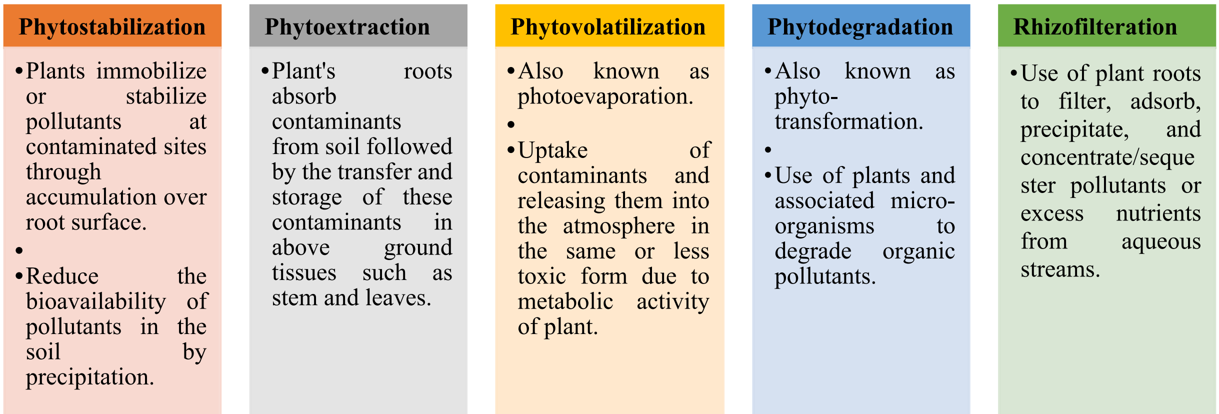



**Fig. 2** Removal mechanisms during phytoremediation

At first glance, phytoextraction and phytostabilization seem to have different goals, and they do indeed differ in terms of various practical aspects. Nevertheless, it can be argued that both approaches share the goal of reducing the amount of mobile/bioavailable HM constituents in the soil. Phytoextraction involves the removal of HM components, whereas phytostabilization reduces the mobility and bioavailability of HMs without actually eliminating them (Barbafieri et al. 2013; Liu et al. 2018). Phytostabilization is accomplished by the roots absorbing and accumulating metal ions, as well as the precipitation of these ions in the rhizosphere zone due to their interaction with organic molecules and changes in the metals' oxidative state. Positively charged metal ions establish robust interactions with pectins in the cell walls of plants and with the negatively charged plasma membranes (DalCorso et al. 2019). Phytoextraction is mainly concerned with extracting the easily accessible and movable portion of HMs using plants. Nevertheless, it is possible to improve the movement of HMs and to make them more accessible for plant absorption by implementing supplementary techniques such as the utilization of additives (Barbafieri et al. 2013). Plants that utilize the phytoextraction method to efficiently eliminate and store substantial amounts of contaminants in their aboveground biomass are known as hyperaccumulators (Ahmad et al. 2011), which are plant species that possess branches with concentrations over 100 mg Cd $kg^{-1}$, over 1000 mg Ni, Pb, and Cu $kg^{-1}$, or more than 10,000 mg Zn and Mn $kg^{-1}$ (dry weight) when grown in soils abundant in metals (Baker and Brooks 1989). Currently, 721 plant species have been identified as metal hyperaccumulators. These plants are classified into many families, including Brassicaceae, Fabaceae, Euphorbiaceae, Asteraceae, etc. (Wu et al. 2018; Huang et al. 2020).

The success of phytoremediation depends on several factors, such as the bioavailability of HMs, plant species, soil characteristics, climatic conditions, etc. Although many of these elements are independent and challenging to regulate, care must be taken while selecting suitable plant species. The efficacy of phytoremediation relies on the careful evaluation and choice of an optimal plant species. The most desirable vegetation characteristics include the ability to adapt to local climates, a deep root structure, the capacity to thrive in the specific type of soil, the ability to extract or break down contaminants into less toxic forms, a fast growth rate, high biomass production, a high capacity for accumulating substances, ease of planting and maintenance, and non-attractive to herbivores (Kafle et al. 2022). Moreover, the bioavailability of HMs in the soil plays a vital role in successful phytoremediation. This is because the entire concentrations of HMs in the soil are not easily accessible for bioaccumulation (Chang et al. 2014). Certain HM ions, such as Cd and Zn, have a higher degree of mobility and accessibility for absorption by plants compared to others, such as Pb, which are generally less mobile (Laghlimi et al. 2015). For metal ions to be accessible for absorption by plant roots, they must first be released into the soil solution. This release is primarily regulated by the pH of the soil (Elekes 2014). Additional parameters, including soil organic matter, root size, external metal concentrations, temperature, metal interactions, nutrient inputs, and salinity, have an impact on the movement of metal ions in soil to an extent (Deepika and Haritash 2023b). Plants employ various strategies to increase the bioavailability of metal ions, including the secretion of phytosiderophores, carboxylates, and acidification of the rhizosphere to facilitate the chelation and solubilization of soil-bound metals (Jabeen et al. 2009; Kushwaha et al. 2015). In addition, various other techniques can be employed to enhance the effectiveness of phytoremediation, such as the utilization of chemical chelating agents, the introduction of microorganisms, and the incorporation of organic matter into the soil (Khan et al. 2021).

Previous studies have documented the ability of various plant species to remove HMs from soil. Examples include *Magnolia grandiflora* L. (Liang et al. 2017a, b), *Datura* *inoxia* L. (Waoo 2016), *Brassica oleracea* L. var. Acephala (Haghighi et al. 2016), *Solanum nigrum* L. (Khan et al. 2014), *Mirabilis jalapa* L. and *Tagetes patula* L. (Li et al. 2020a, b), *Virola surinamensis* L. (Andrade Júnior et al. 2019), *Acanthus ilicifolius* L. (Shackira and Puthur 2017), *Vossia cuspidate* L. (Galal et al. 2017), *Helianthus annuus* L. (Alaboudi et al. 2018), *Calendula officinalis* L. (Goswami and Das 2016), *Sedum alfredii* L. Hance. Planta (Zhang et al. 2016a, b), and *B. juncea* L. (Ali et al. 2013). Research has been conducted for over 60 years to demonstrate the effectiveness of using metal-tolerant plants and soil amendments to reestablish vegetation on metal-toxic soils contaminated by MTs, which are a major source of HM contamination in soil. Successfully revegetating mine soils involves using a combination of native plants that are well-suited to the specific stress factors at the target site, along with non-native species that are highly resistant to environmental stress. Additionally, soil management techniques, primarily in the form of amendments that enhance soil properties, are employed (Fernández et al. 2016). *Thlaspi caerulescens* L., *Astragalus racemosus* L., and *Sebertia acuminata* L. are capable of extracting several metals, including Cd, Zn, Co, Cu, Ni, and Se, from contaminated environments (Narayanan et al. 2020; Yang et al. 2017). *Zea mays* L.*, Brassica juncea* L.*, and H. annuus* L. have demonstrated an increased ability to tolerate and remove metals, as reported by Khan et al. (2020). Rahim et al. (2019) reported on the phytotranslocation characteristics of *Jatropha curcas* L. on soil polluted with metals from a bauxite mine. The afforestation of an open-cast coal site over a span of 14 years resulted in considerable reductions, of 52% for Cd, 48% for Cu, 47% for Zn, 44% for Pb, and 35% for Mn (Desai et al. 2019). These reductions were observed specifically under the presence of *Alnus glutinosa* (L. Gaertn) and *Betula pendula*. According to Narayanan et al. (2021), *J. curcas* L., *Ricinus communis* L., *Macrotyloma uniflorum* (Lam.), *Oryza sativa* L., *Vigna unguiculata* L., *Pennisetum glaucum* L., and *Gossypium hirsutum* L. have been identified as effective plants for removing metals (Cd, Pb, Zn, Mn) from magnesite MTs. A few more examples of research on the use of plants to clean up soil contaminated with HMs from mining waste are listed in Table 2. Further, the soils in the mine area are typically inhospitable for plant growth due to their fluctuating pH levels, which can become acidic in the presence of sulfide mineral wastes or alkaline in the presence of carbonate minerals. These soils also lack organic matter and nutrients, have high concentrations of metals, and are subject to physical limitations (Asensio et al. 2018; Chu et al. 2017). So, revegetation is not the only goal associated with the restoration of mining sites, as the improvement of soil quality is also a major concern. Based on the optimal growth conditions of plants, the amelioration of MTs has been widely attempted using organic and inorganic materials, including woodchips, composted green waste, and manure (Zhang et al. 2020a, b). Indeed, the addition of organic residues has been reported as a viable alternative for the restoration of mine-contaminated soils (Acosta et al. 2018; González Polo et al. 2015; Simiele et al. 2020).

Despite various studies performed, phytoremediation has not gained significant attention worldwide. To determine the current state of phytoremediation studies on soil contaminated with metals due to mining activities, the published articles from the last ten years were analysed (Table 3). Research revealed that most of the studies have been performed using soil contaminated by Pb/Zn MTs (36%) followed by Cu (19%) and Au (15%). In addition, one-quarter of the total studies were carried out in China, indicating how unpopular the method is elsewhere. Additionally, the study makes it evident that the majority of research has been restricted to pot trials or laboratory size, demonstrating the method's limited application on a larger scale.

### 4.1 Case studies

#### *4.1.1 Forestry-based phytoremediation at Varteg Opencast Coalmine, South Wales, UK (Desai et al. 2019)*

A long-term study at the former Varteg opencast coal mine in South Wales, UK, evaluated the potential of forestry-based phytoremediation to reduce soil metal contamination over 14 years. Researchers compared mixed woodland plantings with conventional grassland reclamation methods to assess reductions in soil concentrations of metals, including Zn, Cd, Mn, Pb, and Cu. A chronosequence approach was used to monitor soil metal loadings and tree metal uptake. Mixed woodland areas showed significant annual reductions in Cd, Cu, Zn, and Pb concentrations by 4.3, 2.1, 7.3, and 7.1 mg $kg^{-1}$, respectively, amounting to total decreases of 35%–52% over the study period. Among the tree species tested, *Betula pendula* (silver birch) demonstrated a higher capacity for absorbing Cd, Cu, Zn, and Mn, while *Alnus glutinosa* (common alder) was more effective at accumulating Pb. The study highlights the potential of mixed forest plantations as a sustainable strategy for mitigating HM contamination and restoring ecological function to post-mining lands.

#### *4.1.2 Vetiver grass for iron ore spoil rehabilitation at Joda East Iron Mine, Odisha, India (Banerjee et al. 2019)*

A pilot study at the Joda East Iron Mine in Odisha, India, investigated the efficacy of vetiver grass (*Chrysopogon zizanioides*) in rehabilitating degraded iron ore spoil dumpsites. Four genotypes, S2 (diploid), S4 (tetraploid derivative of S2), TH (originating from Thailand), and BL (broadleaf), were evaluated over 12 months for growth performance, soil stabilization, and HM tolerance. Vetiver was cultivated on

**Table 2** Plants used in ecological restoration of MTs (Greenhouse studies and field studies)

| Type of MTs | Plant species | Heavy metal(s) | Additives | Outcomes | References |
|---|---|---|---|---|---|
| Gold MTs | *Paspalum conjugatum* L | Hg | Hg-resistant bacteria (*Brevundimonas vesicularis* and *Nitrococcus mobilis)* + ammonium thiosulfate | Application of amendments and soil remediation with *P. conjugatum* reduced Hg content in soil by 18–20% | Ustiatik et al. 2020 |
| NR | *B. juncea* L | Al, Cu, Pb, Fe, Cr, Cd, Mn, and As | Plant growth-promoting bacteria (PGPRs) | PGPRs helped in the mobilization and accumulation of metals within plant, mainly Al and Pb | Mendoza-Hernández et al. 2019 |
| Copper MTs | *J. curcas* L | Cr, Fe, Ni, Cu, Zn, Cd, Hg, Pb, and As | No additive (NA) | The plant assimilated significant quantities of Fe (> 3000 mg $kg^{-1}$ plant) along with considerable levels of Pb, Zn, Cu, Cr, and Ni, as well as traces of As. Metals (Cd, Hg, and Sn) present in trace amounts were fully eliminated from the soil | Álvarez-Mateos et al. 2019 |
| Rare-earth MTs | *Paspalum conjugatum* L | Cd and Pb | NA | The utilization of *P. conjugatum* resulted in a substantial reduction in the content of Pb and Cd in the soil, with Cd levels approaching the permissible threshold of 0.209 mg $kg^{-1}$. The restored soil exhibited a greater level of microbial diversity | Zhang et al. 2020a, b |
| NR | *S. nigrum* L | Cu, Zn, Cd, Hg, Pb, and Mn | Biochar/attapulgite | The synergistic effect between the plant and amendments was effective in terms of metal phytostabilization | Li et al. 2019a, b |
| Non-ferrous mine and smelting slag | *Arundo donax* L | Cd and Pb | NA | The plant is capable of tolerating soil with a pH range of 2.9–9.59, as well as high levels of soil Cd and Pb concentration, reaching up to 525 and 57,194 mg/kg, respectively | Liu et al. 2019a, b |
| Lead/zinc MTs | *Cytisus scoparius* L | Cd, Pb, and Zn | NA | The plant can perform as a Zn accumulator, Pb stabilizer, and Cd excluder | Lago-Vila et al. 2019 |
| Lead MTs | *H. annuus* L., *Sorghum bicolor* L., and *B. chinensis* L | Pb | NA | *B. chinensis* exhibited more efficacy in Pb absorption compared to *H. annuus* and *S. bicolor* | Hamvumba et al. 2014 |

**Table 2** (continued)

| Type of MTs | Plant species | Heavy metal(s) | Additives | Outcomes | References |
|---|---|---|---|---|---|
| Lead/zinc and gold MTs | *B. juncea* L | Zn, Pb, Cd, and Cu | *Streptomyces pactum* (Act12) + medical stone compost | Results showed a significant accumulation of metals in the shoot and root of *B. juncea* | Ali et al. 2017 |
| Gold MTs | *Typha latifolia* L. and *C. zizanioides* L | Hg, As, Pb, Cu, and Zn | EDTA and aluminum sulfate | The use of chemical amendments facilitated the accumulation and movement of HMs within the plant species | Anning and Akoto 2018 |
| Not recorded (NR) | *Alnus firma* L | Pb, Zn, As, Cd, Cu, and Ni | *Bacillus thuringiensis* GDB-1 | *A. firma* seedlings accumulated significant quantities of HMs in the following order: Pb > Zn > As > Cd > Cu > Ni in both soil with bacteria amendment and soil without bacteria amendment | Babu et al. 2013 |
| NR | *S. bicolor* × *sudanense* L | As, Cu, Pb, and Zn | *Penicillium aculeatum* PDR-4 and*Trichoderma* sp. PDR-16 | The utilization of mycorrhizal-assisted phytoremediation has been discovered to be advantageous in eliminating HMs from soil affected by mining tailings | Babu et al. 2014 |
| Iron MTs | *C. zizaniodes* L | Fe, Zn, Mn, Cu, Cr, and Pb | NA | Grass has proven to be highly efficient at stabilizing soil at areas contaminated with significant amounts of HMs, specifically iron (Fe), manganese (Mn), zinc (Zn), and chromium (Cr) | Banerjee et al. 2019 |
| NR | *Glycine max* L., *Dianthus chinensis* L.*, and Bassia scoparia* L | Cu, Zn, Pb, Cd, and Mn | Amendments (organic fertilizer, rice husk, biochar, ceramsite) + microrganims (*Mortierella* sp., *Trichoderma asperellum*, and *Mucor circinelloides*) | The combination of *G. max*, *M. circinelloides*, and amendment (organic fertilizer: rice husk: biochar: ceramsite = 1:1:2:1) was found to be the most effective remediation of metal contaminated soil | Li et al. 2019a, b |
| Copper MTs | *Bidens Pilosa* L. and *Plantago lanceolata* L | Cu | NA | Both *B. pilosa* and P. *lanceolata* plants exhibited traits of being highly efficient in accumulating Cu | Andreazza et al. 2015 |
| Chromite-asbestos MTs | *Cymbopogon citratus* L. and *C. zizanioides* L | Cr and Ni | Chicken manure, farmyard manure, and garden soil | The study found that *C. citratus* and *C. zizanioides* can be utilized for phytostabilization of abandoned chromite-asbestos mining waste when combined with amendments | Kumar and Maiti 2015 |

**Table 2** (continued)

| Type of MTs | Plant species | Heavy metal(s) | Additives | Outcomes | References |
|---|---|---|---|---|---|
| NR | *J. curcas* L | Cu, Zn, Cd, and Pb | Corn biochar + *Acaulospora* sp. | The plant exhibited a great capacity to thrive in MTs and showed no signs of phytotoxicity | Gonzalez-Chavez et al. 2017 |
| Copper MTs | *Prosopis tamarugo* L., *Schinus mole* L., and *Atriplex nummularia* L | Cu, Fe, Zn, Mn, and Pb | $CaCO_3$ + compost + *Glomus intraradices* (AMF) | Each plant had high TF values and the ability to be employed in phytoextraction | Lam et al. 2017 |
| Lead/zinc MTs | *Centella asiatica* L | Cd | *Enterobacter* sp. FM-1 | The study determined that the introduction of *Enterobacter* sp. FM-1 in *C. asiatica* has the potential to be beneficial for the Cd removal | Li et al. 2018 |
| Rare earth MTs | *Dicranopteris dichotoma* L | Light and heavy rare-earth elements (REE) | NA | The growth of the plant increased during the entire experimental period. The total REE content in the shoots of the plant was > 1000 $mg\text{-}kg^{-1}$ and the BCF and TF were also > 1 | Zhiqiang and Zhibiao 2020 |
| Copper MTs | *B. juncea* L | Co, Cr, Cu, Ni, and Zn | Technosol and compost | The plant exhibited suitability for the accumulation of Cu and Zn with TF > 1. Thus, it is a suitable candidate for phytoextraction | Novo et al. 2013 |
| Chromite MTs | *Sesbania sesban* L. and *Brachiaria mutica* L | Cr | NA | *S. sesban* showed better results than *B. mutica* in terms of tolerance index, BCF, and transportation index | Patra et al. 2020 |
| Gold MTs | *Paspalum fasciculatum* Willd. ex Flugg | Cd and Pb | NA | The plant showed the phytostabilization ability for Cd, whereas it was suitable for the phytoextraction of Pb | Salas-Moreno and Marrugo-Negrete 2020 |
| Lead MTs | *Thysanolaena latifolia* L., *Mimosa pudica* L., and *Bidens pilosa* L | Pb | NA | From the results, *B. Pilosa* was proposed to be a Pb hyperaccumulator and *T. latifolia* and *M. pudica* were found to be excluders | Yongpisanphop et al. 2017 |
| Uranium MTs | *Rumex nepalensis* L. and *Polygonum viviparum* L | U and As | NA | *R. nepalensis* showed higher accumulation efficiency for U, whereas, high As accumulation was observed in *P. viviparum* | Li et al. 2019a, b |

**Table 2** (continued)

| Type of MTs | Plant species | Heavy metal(s) | Additives | Outcomes | References |
|---|---|---|---|---|---|
| Lead/zinc MTs | *Quercus virginiana* L. and *Salix integra* L | Cd, Pb, and Zn | NA | *Q. virginiana* was found to be a suitable option for phytostabilization of tailings and *S. integra* may be beneficial for phytoextraction | Shi et al. 2017 |
| Copper MTs | *R. communis* L | Cu | NA | The plant grew well in contaminated soil and showed high tolerance index but BCF and TF of the plant were observed to be < 1 for Cu | Palanivel et al. 2020 |

*NA* no additive, *NR* not reported

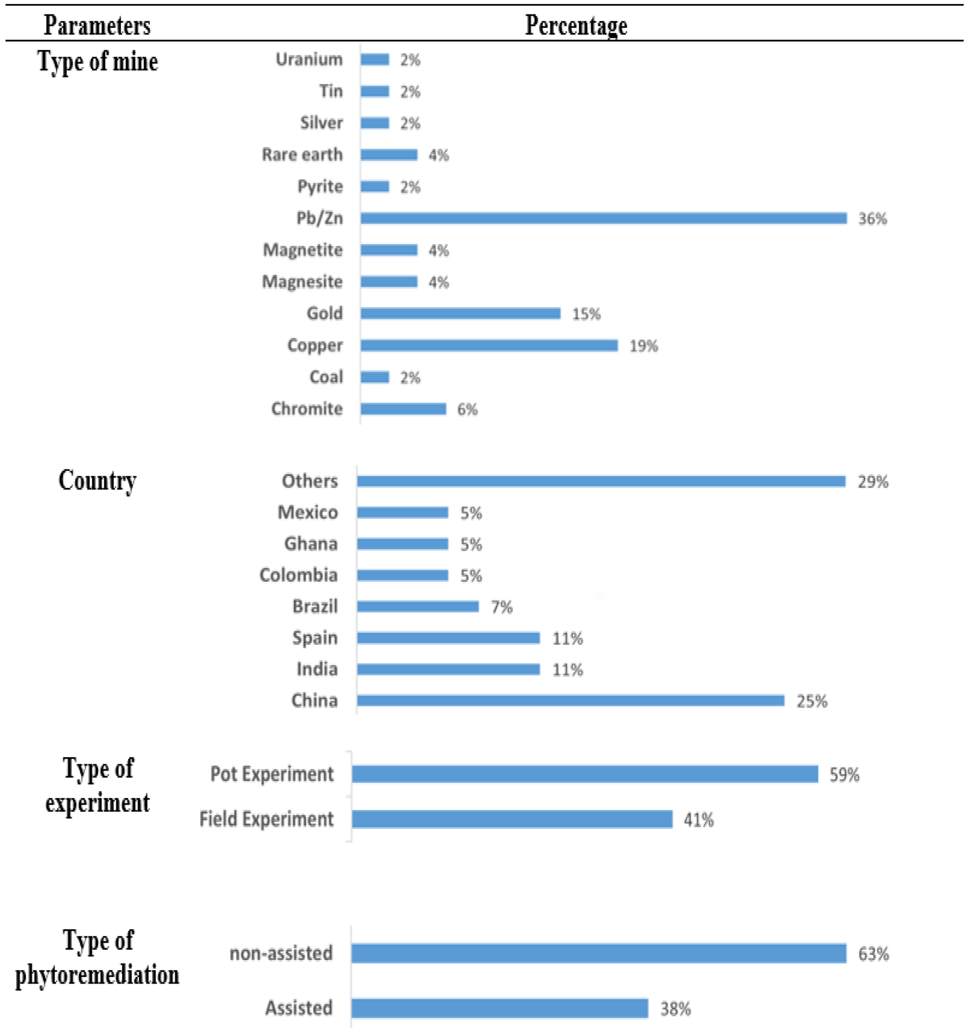
both iron mine soil and garden soil (control), with parameters such as shoot and root length, chlorophyll and carotenoid content, and biomass production being monitored. Scanning electron microscopy (SEM) and Perl's Prussian blue staining revealed iron (Fe) accumulation in roots and shoots, while zinc (Zn), manganese (Mn), chromium (Cr), and copper (Cu) were also detected in plant tissues. Although initial growth and biomass were lower in mine soil, the plants exhibited recovery and appeared healthy by the end of the study. Among the genotypes, the BL genotype demonstrated the highest tolerance and capacity for phytoremediation, followed by S4. This study underscores the potential of vetiver grass as a cost-effective and sustainable solution for restoring metal-contaminated spoil dumpsites.

## 5 Socio-economic impacts of phytoremediation

Phytoremediation, a sustainable and eco-friendly method for environmental remediation, offers substantial social and economic benefits while enhancing ecosystem functionality and ecological balance. The adoption and successful implementation of phytoremediation depend significantly on social acceptance, which is facilitated by its natural and aesthetically appealing characteristics. Phytoremediation contributes to several social benefits, primarily through improving the visual and ecological quality of contaminated sites (Erickson et al. 2021). The establishment of green spaces with ornamental vegetation not only provides aesthetic value but also creates better habitats for native wildlife. Transforming these sites into public parks or arboretums offers recreational opportunities and enhances community well-being (Capuana 2020). For instance, Teodorescu et al. (2011) suggested that plants like willows, which are widely used in phytoremediation, can be developed into urban hedges, enhancing the visual appeal and functionality of reclamation sites. Also, reclaimed urban brownfields through phytoremediation provide significant ecosystem services, including improved urban hydrology, heat mitigation, noise reduction, increased biodiversity, and $CO_2$ sequestration (Guidi Nissim et al. 2023). These contributions are associated with social cohesion, health, and overall quality of life. Moreover, phytoremediation projects generate employment opportunities for remediation activities, biomass harvesting, and the subsequent conversion of biomass into usable products. This not only uplifts the local economy but also fosters a sense of community involvement. Additionally, improvements in site safety and quality of life contribute to broader social acceptance of phytoremediation (Erickson et al. 2021).

From an economic perspective, phytoremediation involves specific costs related to site characterization, remediation plan development, and implementation. Annual expenditures include site maintenance, biomass harvesting,

**Table 3** Analysis of reported literature (%) on the phytoremediation of metal-contaminated soil by mining sector




and product sales. Wan et al. (2016) calculated the total cost of phytoremediation at approximately \$37.7 per cubic meter, with initial capital and operational costs constituting 46.02% and 53.98%, respectively. However, these costs are offset by multiple revenue streams and economic benefits. For instance, the sale of biomass for anaerobic digestion or renewable energy production can significantly enhance farmers' incomes. The recovery of valuable metals such as Se, Mn, Pb, Zn, Cd, U, Cu, Ni, Co, and Au through phytomining presents another lucrative opportunity. Many mining companies profit from metal recovery and biomass utilization for energy generation (Riaz et al. 2022).

Phytoremediation also increases the economic value of contaminated sites by improving soil quality. Previously unusable land gains marketability when vegetation is established, and biomass products like wood can be harvested and sold. Additionally, carbon sequestration contributes to economic benefits through the avoidance of social costs associated with greenhouse gas emissions. The estimated global value of reducing 1 metric ton of $CO_2$ is US\$42.00 (Mikhailova et al. 2019). Furthermore, phytoremediation reduces the costs of groundwater purification and environmental remediation by preventing contaminant migration. This protection of environmental resources not only lowers long-term remediation expenses but also ensures sustainable management of natural resources. Therefore, the integration of phytoremediation into environmental management offers dual benefits: improved ecosystem services and significant socioeconomic gains (Riaz et al. 2022). By enhancing site aesthetics, providing employment, reducing environmental remediation costs, and creating valuable by-products, phytoremediation represents a sustainable approach to addressing environmental contamination.

## 6 Limitations and challenges of phytoremediation

In recent years, phytoremediation has attracted much interest and acceptance. In contrast to other remediation approaches, phytoremediation has many benefits, but it also has some drawbacks that impact its efficacy and practical application (Muthusaravanan et al. 2019; Ramamurthy and Memarian 2012; Shah and Daverey 2020). As a slow process reliant on biological cycles, it often requires months or even years to show results, necessitating multiple crop cycles. The method is constrained by the root depth of remediation plants, which limits its effectiveness in addressing deeper soil contamination. Many hyperaccumulator plants also have low biomass and moderate growth rates, reducing their efficiency in extracting contaminants. Phytoremediation is suitable for application during the post-mining phase, specifically, decommissioned tailings storage facilities and inactive mining sites. It cannot be applied while the facility is still operational, as the ongoing deposition and physical instability of materials inhibit plant growth and interfere with remediation efforts. However, for such sites, the technique may be performed ex situ, where contaminated tailings or waste material are excavated, mixed with soil, and treated in controlled environments such as lined cells or biopiles. This batch-wise approach allows for more effective management of contaminant mobility, making it suitable for smaller-scale remediation efforts where in situ application is not feasible. Managing the resulting metal-laden biomass poses challenges, as handling, storing, and disposing of the material can be problematic (Farraji et al. 2016). While techniques like compaction and composting reduce transportation costs, they can also accelerate the leaching of metal–organic compounds. Environmental factors such as weather, climate, and susceptibility to diseases or insect attacks—exacerbated by climate change—can further hinder plant productivity and resource accumulation, particularly in tropical and subtropical regions. Additionally, the bioavailability of metals often limits phytoremediation's effectiveness, as many pollutants are strongly bound to soil particles. Synthetic chelating agents like EDTA and citric acid are used to improve metal mobilization, but are toxic, non-biodegradable, and may persist in the soil, creating long-term environmental risks (Khalid et al. 2017). Care must also be taken to avoid introducing invasive plant species as hyperaccumulators, which can threaten native biodiversity. Soil amendments and agronomic techniques, while intended to enhance remediation, can sometimes negatively affect pollutant mobility, further complicating efforts. Phytoremediation is most suitable for sites with low to moderate contamination, as heavily polluted soils inhibit sustainable plant growth. Bioaccumulation, where metals or pollutants concentrate in plants, poses additional risks as contaminants can enter the food chain or be consumed by animals unless the biomass is safely disposed of (Mahar et al. 2016). Despite its potential, comprehensive research on the time requirements and cost–benefit analysis of phytoremediation remains limited, hampering its practical implementation (Phang et al. 2024). There is also a significant challenge in managing stakeholder expectations. Phytoremediation is often misperceived as a universal solution for site decontamination, leading to disillusionment when results fall short. This skepticism can deter broader adoption of the technology, reducing its impact as a viable remediation strategy (Beans 2017).

## 7 Post-harvest management strategies

The successful implementation of these phyto-technologies, which are both environmentally friendly and sustainable, relies heavily on the effective post-phytoremediation procedures associated with the disposal of biomass contaminated with harmful metals (Khan et al. 2021). There is a lack of data among researchers about the outcome of polluted biomass harvested after phytoremediation (Song and Park 2017; Vigil et al. 2015). Although numerous published studies primarily concentrate on phytoremediation, there is a scarcity of studies and management techniques concerning the biomass-generated aftermath of remediation (Khan et al. 2023; Song et al. 2016; Song and Park 2017). Due to the substantial production of contaminated biomass during the phytoremediation process, it is essential to implement appropriate disposal management to prevent its entry into the food chain. Initially, a range of methods were employed to securely eliminate the polluted biomass, including composting and compaction, combustion and gasification, phytomining, and pyrolysis. Currently, the biomass produced by phytoremediation is transformed into important products and services through post-treatment. These include solid and composite wood products, carbon sequestration, biofuel production, and bio-fortification (Table 4).

## 8 Conclusion and future perspectives

MTs present a critical environmental challenge due to their low pH, poor organic content, and high concentrations of HMs such as Cu, Pb, Zn, Cd, Cr, and As. Their fine, dusty nature facilitates HM dispersion through wind and water erosion, leading to widespread contamination of air, soil, and water, with severe ecological and human health consequences (Mileusnić et al. 2014; Liang et al. 2017a, b). The persistent, non-biodegradable, and bio-accumulative properties of HMs exacerbate these impacts, degrading ecosystems and posing health risks through food chain contamination. Conventional physicochemical remediation methods are often impractical due to high costs, energy demands, and prolonged treatment times, necessitating sustainable alternatives.

Phytoremediation emerges as a viable, eco-friendly, and cost-effective solution for MT restoration. It stabilizes tailings, reduces HM leaching and AMD, improves

**Table 4** Post-harvest management strategies

**Post-harvest Management**

**Composting & comapction**

• Composting & compaction are integrated process where former results in the volume reduction of contaminated biomass by biological degradation and later functions by dehydrating the compost and collecting leachate.

• Both techniques reduces the transportation and handling costs of waste biomass.

**Thermochemical conversion**

• Incineration: Burning under controlled conditions and volume reduces to 2-5%. Ash formed can be used as bio-ore.

• Gasification: Incomplete combustion of biomass at 700-1200°C. Valuable products are syngas, hydrogen and biochar.

• Pyrolysis: Heating of biomass in complete absence of oxygen and results in the formation of biochar.

**Phytomining**

• It is based on phytoextraction and includes growing of hyperaccumulator plants, harvesting and combustion of plant biomass to obtain bio-ore.

• This process may result in the revenue generation by extracting saleable heavy metals.

**Other strategies**

• Contaminated biomass can be utilized for manufacturing solid/composite wood products for long term storage.

• Oil plants and algal species can be used for biofuel generation.

• Biofortification-coupled phytoremediation can be performed for production of trace metal enriched plants.

soil properties, and supports ecosystem recovery (Sun et al. 2018). Despite its potential, phytoremediation's slow pace, site-specific limitations, and high initial costs hinder widespread adoption. To address these challenges, integrated approaches combining phytostabilization, chemical fixation, soil amendments, and microbial inoculation are recommended to enhance efficiency. Research should prioritize fast-growing, high-biomass plant species tolerant of environmental stressors, alongside biotechnological innovations like genetically engineered plants and advanced rhizosphere management. Long-term field studies are essential to validate laboratory findings and ensure scalability. Furthermore, government-led financial incentives, subsidies, and regulatory frameworks are critical to overcoming economic barriers and encouraging stakeholder investment. By advancing these strategies, phytoremediation can transition from an emerging technology to a scalable, sustainable solution, effectively mitigating HM contamination from MTs and safeguarding ecosystems and human health for future generations.

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**Data availability** The datasets generated during and/or analyzed during the current study are available from the corresponding author on reasonable request.

**Declarations**

**Conflict of interest** The authors have no relevant financial or non-financial interests to disclose.

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