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UNDERSTANDING THE FATE AND BEHAVIOR OF ARSENIC IN SOILS AND THEIR REMEDIATION METHODS: A REVIEW

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ABSTRACT

Arsenic contamination of soils is a critical global environmental issue due to its persistence, high toxicity, and potential for transfer into food chains and groundwater systems. Elevated arsenic levels arise from both natural sources, such as the weathering of arsenic-bearing minerals, and human activities including mining, industrial discharges, pesticide use, and excessive fertilizer application. The fate and behavior of arsenic in soils are controlled by complex physicochemical and biological interactions involving soil pH, redox conditions, mineralogy, organic matter, competing ions, and microbial processes. These factors determine arsenic speciation, mobility, bioavailability, and toxicity, which in turn influence environmental and health risks. Among arsenic forms, inorganic species, arsenite and arsenate are of greatest concern due to their high mobility and toxicity. Arsenic exposure negatively affects plant growth and metabolism, causes toxicity in animals, and poses serious health risks to humans, including carcinogenic and systemic effects from long-term exposure. Various remediation strategies, such as physical removal, chemical stabilization, and biological approaches using plants or microorganisms, are employed to reduce arsenic levels or limit its mobility. Choosing an appropriate remediation method depends on site-specific soil conditions, contamination severity, and long-term management objectives. Understanding the interactions between As mobility, toxicity, and remediation options is essential for protecting ecosystems and human health.

Key words: Arsenic, Soil contamination, Arsenic mobility, Soil remediation, Toxicity, Phytoremediation, Bioremediation

Introduction

Arsenic (As) is a highly toxic metalloid widely distributed in the environment and poses serious risks to human health, plants, and animals. The International Agency for Research on Cancer classifies arsenic and its compounds as Group I carcinogens, confirming their carcinogenicity to humans (Petursdottir *et al.*, 2015; Sarkar and Paul, 2021). Owing to its extreme toxicity and historical use in poisoning, arsenic is often termed the "king of poisons." The element was first described by Theophrastus around 300 B.C. (Caley and Richards, 1956), while its isolation from orpiment (arsenic sulfide, As_2S_3) was achieved much later, in the thirteenth century, by the alchemist Albertus Magnus.

Arsenic contamination of soils and water bodies has become a major global environmental concern due to both natural and anthropogenic inputs. Natural sources include the weathering of arsenic-rich minerals, geothermal processes, and volcanic emissions, whereas anthropogenic contributions arise from mining, smelting, industrial effluents, pesticide usage, and the extensive application of phosphate fertilizers (Mahimairaja *et al.*, 2005). In soil environments, arsenic occurs in both organic and inorganic forms. Among these, inorganic species such as arsenite (As(III)) and arsenate (As(V)) are of greatest concern because of their higher mobility and toxicity, which increase the risks of crop contamination and groundwater pollution.

The mobility, bioavailability, and toxicity of arsenic in soils are governed by several interacting physicochemical and biological factors, including soil pH, redox potential, clay and organic matter content, and the presence of competing ions such as phosphate (Lee *et al.*, 2015). Under aerobic conditions, As (V) is typically strongly adsorbed onto iron, aluminum, and manganese oxides, limiting its mobility. In contrast, As (III) becomes more soluble under reducing conditions, resulting in enhanced mobility and greater environmental risk. Microbial processes play a crucial role in arsenic cycling by mediating redox transformations and methylation reactions, which can either reduce toxicity or enhance volatilization (Dhuldhaj *et al.*, 2013).

To address arsenic-contaminated soils, various remediation strategies have been developed. Physical methods such as soil removal, capping, and electrokinetic remediation aim to isolate or extract contaminated materials. Chemical approaches, including soil washing and immobilization, focus on removing or stabilizing arsenic, while biological methods such as phytoremediation and microbial bioremediation offer more sustainable alternatives. However, challenges remain in applying these techniques at field scale, reducing costs, and ensuring long-term effectiveness without causing secondary environmental impacts.

Sources and occurrence

Arsenic (Z=33) is the 20th most prevalent element in the geosphere, with an average abundance of approximately 5 mg kg^{-1} in the Earth's crust (Garelick *et al.*, 2008). This element is characterized as colorless, odorless, and tasteless, possessing toxic properties (Katsoyiannis and Zouboulis, 2006). It occurs in nature primarily as ores, usually in combination with sulfur, such as realgar (AsS), orpiment (As_2S_3), arsenopyriterite (FeAsS) (Magalhaes, 2002).

A variety of As compounds, both organic and inorganic, are enter the environment through geological (geogenic) and anthropogenic (human activity) sources. Natural sources include weathering of rocks and minerals, geothermal activities, and volcanic eruptions (Smedley and Kinniburgh 2002). Biological sources play a minor role in As input, but plants and microorganisms influence As redistribution through bioaccumulation, biotransformation, and biovolatilization. Aquatic organisms are known to accumulate As, often reaching concentrations significantly higher than those in the surrounding water. However, major anthropogenic sources of As include industrial waste, sewage sludge, incineration, wood treatment, mining/smelting operations, and use of agricultural chemicals, which are a common practice throughout the world (Wang and Mulligan 2006).

The contamination of soil and aquatic systems with As is a significant environmental concern. Excessive As in soils is hazardous to organisms and threatens plant growth, as well as causing human diseases through the food chain. According to the World Health Organization (WHO, 2004), the permissible limit of As is 30 mg kg^{-1} in soil and 10 μg L^{-1} in water.

Transformation of arsenic in soil

Arsenic undergoes multiple transformation processes in soil, primarily governed by redox reactions, adsorption-desorption, microbial activity, and mineral precipitation-dissolution, which influence its mobility, bioavailability, and toxicity (Shrivastava *et al.*, 2015).

Arsenic exists in soil in both organic and inorganic forms, with inorganic forms being more prevalent than organic forms. Under anaerobic conditions, As(V) is reduced to the more mobile and toxic As(III), whereas aerobic conditions favor its oxidation back to As(V). As(III) can further react with sulfide (S^{2-}) to form stable minerals such as arsenopyrite (As_2S_3, As_3S_4, FeAsS), thereby reducing As mobility (Wang *et al.*, 2025). As(V) binds to iron (Fe), aluminum (Al), and manganese (Mn) oxides and hydroxides, restricting its movement in the soil (Ackermann *et al.*, 2008).

Microbial As methylation is a key process that transforms inorganic As into less toxic organic forms, such as monomethyl arsenic acid (MMA), dimethyl arsenic acid (DMA) and, dimethyl arsine, and ultimately volatile trimethylarsine. This process reduces As toxicity and facilitates its removal from the soil environment (Huang *et al.*, 2016).

Factors affecting arsenic availability and mobility in soil

The mobility, transformation, and transport of As in soil are controlled by several physicochemical properties, including pH, redox potential, temperature, clay and organic matter content, and the presence of competing ions and sesquioxides such as Fe and Al oxides (Lee *et al.*, 2015). These factors determine how strongly As binds to soil particles, how readily it moves through the soil solution, and how available it becomes to plants and groundwater.

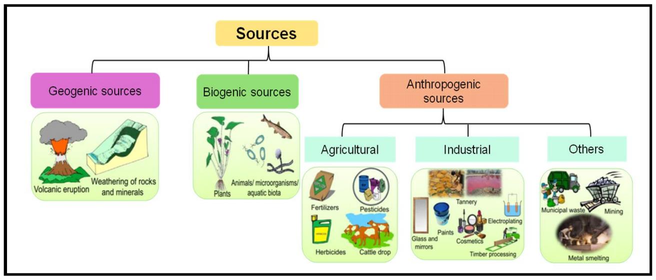

Fig. 1: Sources of arsenic in soil.

Soil pH plays a critical role in controlling arsenic speciation and mobility. Under aerobic and higher-pH conditions, As(V) predominates, whereas As(III), which is more mobile, is favored in anaerobic and lower-pH soils (Khalid *et al*., 2017). At low pH, As(V) shows strong adsorption, while As(III) is less reactive but can still associate with mineral surfaces (Pigna *et al*., 2015). At moderate pH, As(III) adsorption declines with increasing pH, following a parabolic trend with a maximum near pH 8.5 (Goldberg, 2002). At higher pH, increased OH{ induces negative surface charges, reducing adsorption and enhancing mobility of both species (Gustafsson & Bhattacharya, 2007). However, elevated pH may also promote stable mineral complex formation, potentially reducing arsenic risk in some soils (Martin *et al*., 2014; Ren *et al*., 2024).

Temperature also influences As behavior because As adsorption is generally endothermic (Chen *et al*., 2017). Higher temperatures increase ion mobility and diffusion rates, enhancing As transport to adsorption sites and increasing the overall adsorption capacity of soils. Adsorption rates for As are faster at elevated temperatures, with reactions completing more quickly in soils like those from Beijing and Hainan (Feng *et al*., 2013).

Competing ions such as phosphate strongly influence arsenic mobility by competing for adsorption sites on soil particles (Pigna *et al*., 2015). Competitive sorption studies show that increasing phosphate concentrations significantly reduce both the extent and rate of As(V) adsorption (Zhang and Selim, 2008). Because As(V) and phosphate share similar tetrahedral oxyanion structures and pK_a values, they compete for the same binding sites on minerals such as iron (oxy) hydroxides and calcite (Liu *et al*., 2025). The strength of this competition depends on soil mineralogy and binding site density (Zeng *et al*., 2012). This interaction also influences plant uptake, as As(V) enters plants via phosphate transport systems.

Iron (Fe) and aluminum (Al) oxides, particularly in amorphous form, play a crucial role in As adsorption, with Fe oxides being particularly effective in immobilizing As in soils (Lee *et al*., 2015). These oxides possess high surface areas and positive charges that effectively bind As(V) through ligand exchange reactions (Goldberg, 2002). Fe oxides, such as ferrihydrite and goethite, exhibit a higher surface charge density, which facilitates stronger electrostatic interactions with As species (Gustafsson & Bhattacharya, 2007). As a result, soils rich in amorphous Fe oxides typically exhibit strong As-binding capacity and reduced As mobility.

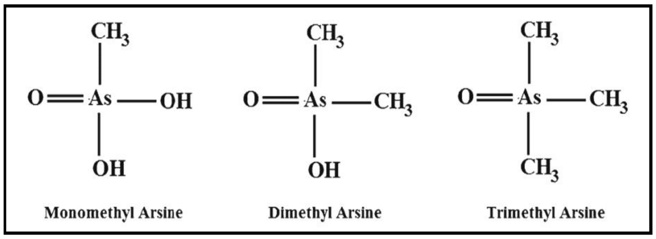

Fig. 2: Major Organoarsenic compounds.

The clay content and mineralogical makeup of soil impact As retention, with higher clay content generally enhancing adsorption and reducing mobility (Das *et al*., 2018). Clay content significantly influences As adsorption due to the high specific surface area and numerous reactive sites associated with clay minerals (Yuan *et al*., 2013). Arsenic is initially adsorbed onto clay surfaces and can eventually become incorporated into clay structures, reducing its mobility. Soils with higher clay content and elevated Fe oxide levels, such as some Californian soils, demonstrate enhanced As retention and lower risk of groundwater contamination (Manning and Goldberg, 1997).

Organic matter (OM) plays a key role in controlling arsenic (As) mobility and bioavailability through complexation processes, depending on its type and concentration (Anawar *et al*., 2013). Solid-phase OM can reduce As mobility by forming stable humic–clay complexes that enhance As retention. For instance, at pH 7, humic acid–coated kaolinite showed a higher partition coefficient for As(V) than uncoated kaolinite, indicating stronger sorption (Saada *et al*., 2003). Conversely, dissolved organic carbon may increase As mobility by competing for sorption sites or forming soluble As–organic complexes (Aftabtalab *et al*., 2022). Organic amendments such as vermicompost and bagasse can enhance As sorption and immobilization under suitable conditions (Raza *et al*., 2024).

Redox potential influences the oxidation state of As, with reduced conditions favoring the solubility and mobility of As due to the dissolution of Fe and Mn oxides/ hydroxides (Anawar *et al*., 2013). Under oxidizing conditions (500–200/ mV), As solubility is low and 65–98% of dissolved As is present as As(V). Under strongly

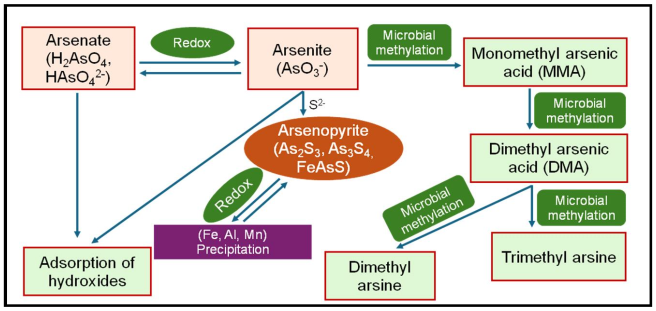

Fig. 3: Arsenic transformation in soil.

Table l: Major hyperaccumulator plants used for arsenic phytoremediation.

Plants	Characteristics	Source
Pteris vittata L.	It is a fern exhibiting rapid uptake, xylem translocation, high tolerance, and exceptional frond sequestration, enabling effective phytoremediation of contaminated soils.	(Raj and Singh, 2015) (Xie *et al.*, 2009)
Pityrogrammacalomelanos	It is a fern that accumulates arsenic mostly in the fronds with ability to grow in highly contaminated areas. It can take up arsenic even from less contaminated areas.	(Luongo and Ma, 2005) (Francesconi *et al.*, 2002)
Brassica juncea	It is a herbaceous plant. It is a highly resistant plant with ability to grow on soils of different nature.	(Pickering *et al.*, 2000) (Vocciante *et al.*, 2019)
Pteris cretica	It is a fern with greater arsenate accumulation in roots.	(Raj and Singh, 2015) (Luongo and Ma, 2005)
Pteris umbrosa	It is a fern with greater arsenate accumulation in roots. They are fast growing plant and are also aesthetically pleasing.	(Luongo and Ma, 2005) (Koller *et al.*, 2007)

reducing conditions (e.g. –200/ mV), soluble As increases dramatically (13 fold compared to 500/ mV), indicating reduction of As(V) to As(III) and mobilization of As from soil solids (Masscheleyn *et al.*, 1991).

Overall, As mobility in soil results from the combined effects of pH, redox status, temperature, mineralogy, organic matter interactions, and competitive ions. These factors collectively determine the extent to which As is adsorbed, transformed, or transported, ultimately influencing environmental risk and the behavior of As in contaminated soils.

Arsenic toxicity in plants, animals and human beings

Arsenic toxicity poses significant risks to plants, animals, and humans, primarily due to its widespread environmental presence and accumulation in food chains. This metalloid, often found in groundwater and soil, can lead to severe health issues across various species. Understanding the mechanisms of As toxicity is crucial for developing mitigation strategies.

a. Arsenic toxicity in plants:

Arsenic is absorbed by plants, particularly through contaminated water and soil, leading to bioaccumulation. Inorganic forms, such as As(V) and As(III), are particularly harmful, affecting plant growth and function. Arsenic uptake by plant tissues causes severe damage to metabolic processes in mitochondria and important cellular components, such as lipids, proteins, DNA, and RNA (Dileep *et al.*, 2023). Increased As concentrations negatively affect germination rates and seedling height. For instance, indigenous rice varieties showed a marked decrease in germination percentage at As levels above 100 µM L^{-1}(Bag *et al.*, 2019). As exposure leads to reduced leaf area and dry matter production, with significant correlations observed between As concentration and growth parameter (Srivastava, 2019).

b. Arsenic toxicity in animals

The primary route of As exposure in animals is through contaminated food and water, which can lead to systemic absorption. Skin exposure can result in localized effects such as blisters, lesions, and infections (Sharma *et al.*, 2024). Common symptoms of As poisoning include abdominal pain, vomiting, diarrhoea, and salivation. Animals may experience weakness, paralysis, and eye abnormalities due to neurological damage (Martinez-Castillo *et al.*, 2021). Long-term exposure can lead to serious health issues, including cancers and cardiovascular diseases (Otles and Cagindi, 2010).

c. Arsenic toxicity in humans

Arsenic toxicity in humans occurs through either acute or chronic exposure. Acute toxicity results from short-term exposure to high arsenic levels, whereas chronic toxicity develops due to prolonged exposure. One of the earliest and most distinctive signs of chronic arsenic exposure is the development of skin lesions, particularly hyperpigmentation and keratosis. These hallmark symptoms of arsenicosis can worsen over time and may progress to skin cancer if left untreated (Sengupta *et al.*, 2008). Chronic arsenic exposure also affects multiple organ systems, including the lungs, liver, cardiovascular, and nervous systems. It has been associated with chronic bronchitis, non-cirrhotic portal fibrosis, hypertension, ischemic heart disease, polyneuropathy, and cerebrovascular disorders (Mazumder, 2015). Arsenic is a proven carcinogen linked to increased risks of skin, bladder, lung, and liver cancers, emphasizing the need for early detection and monitoring (Chikkanna *et al.*, 2019). Additional health impacts include diabetes mellitus, anemia, reproductive complications such as stillbirth and infant mortality, and epigenetic changes that promote gene mutations. At the molecular level, arsenic interferes with enzymatic functions and damages DNA, contributing to systemic toxicity, particularly within the nervous system (Kaur *et al.*, 2023).

Remediation methods for arsenic contaminated soil

Remediation methods for As contaminated soil can be categorized into three main types physical, chemical and biological methods.

A. Physical remediation methods

a) Soil replacement and soil cover: Soil replacement and soil cover both require importing clean soil but are applied under different conditions (Wan *et al.*, 2020). Soil replacement involves excavating contaminated soil and substituting it with clean soil, making it suitable for heavily polluted sites. However, this method is often hindered by high costs, energy consumption, and the risk of secondary pollution during transport. In contrast, soil cover entails placing a 20–40 cm layer of clean soil over contaminated soil, effectively reducing heavy metal uptake by crops and minimizing direct exposure to contaminants (Lee *et al.*, 2017).

b) Turnover and attenuation: Mixing polluted topsoil with clean deep soil to lower the total concentration of contaminants in the soil is called turnover and attenuation. The turnover depth is typically set at 40, 60, or 80 cm. The depth of turnover is influenced by how contaminants are spread throughout the soil profile (Heinkele and Bullmann, 1995). For instance, deeper turnover may be necessary in areas with significant contamination at greater depths. Higher turnover depths can lead to increased costs due to the need for more clean soil and labor for mixing (Chen and Chiou, 2008).

c) Electrokinetic remediation: Electrokinetic remediation (EKR) includes the development of a direct current electric field by putting electrodes into polluted soil solutions. Along with the electric field the pollutants migrate from the treatment area to the electrode area and it can then be removed by electro deposition or ion exchange extraction (Wan *et al.*, 2020). It is a quick and effective method to control As contamination in soil. The As removal efficiency can be as high as 44.8% (Yuan and Chiang, 2008). EKR showed a high removal rate for top soil (removal rate of 59% at the depth of 0–0.5 m) but comparatively lower removal rates in deeper layers (Kim *et al.*, 2014).

Despite the high remediation efficiency of these physical technologies, they are often regarded as too costly for implementation in large-scale projects. Thus, the field experiences of physical technologies, especially for large-scale field remediation projects, are few compared with those of chemical and biological technologies.

B. Chemical remediation methods

a) Soil washing: Soil washing (leaching) removes contaminants by applying chemical agents that dissolve or mobilize pollutants, followed by recovery of the contaminant-rich leachate. It has been widely used for arsenic-contaminated soils with acids, bases, chelants, organic ligands, and more recently biosurfactants (Wei *et al.*, 2016). Acids such as HCl and H_3PO_4 and bases like NaOH remove As by dissolving Fe minerals or increasing soil pH (Im *et al.*, 2015). Although higher concentrations enhance As removal, they can damage soil structure and increase disposal costs. Phosphoric acid is especially effective, achieving up to 90% removal due to strong competition between phosphate and As(V). Chelants generally show limited As extraction efficiency (Qiu *et al.*, 2010; Wen & Marshall, 2011), and persistent agents such as EDTA may cause secondary pollution. To minimize environmental impacts, recent studies emphasize biosurfactants (e.g., tannic acid, saponin) and natural organic eluents, including DOC, humic substances, citrate, NTA, and oxalate (Gusiatin, 2014). DOC can remove up to 88% of As (Lin *et al.*, 2017), while combined or sequential eluents may achieve removal efficiencies approaching 98%. However, high costs and soil degradation limit large-scale application.

b) Soil immobilization: Soil immobilization uses chemical amendments to reduce the mobility and bioavailability of contaminants, offering a low-cost and convenient remediation approach. For As-contaminated soils, common agents include Fe oxides, waste materials, and biochar (Doherty *et al.*, 2017). Fe oxides immobilize As through surface complexation with As(V), oxidation of As(III), and Fe–As precipitation. Fe-based nanoparticles further enhance effectiveness due to their large surface area (Gil-Diaz *et al.*, 2017; Arenas-Lago *et al.*, 2019). Waste materials such as compost, mine drainage sludge, and steel slag can also immobilize As, though their own metal content must be evaluated (Ko *et al.*, 2015; Cui *et al.*, 2018). Biochar alone is less effective but performs well when combined with Fe, enabling co-immobilization of other metals like Cd (Alozie *et al.*, 2018; Zhu *et al.*, 2019). Field-scale studies show nanoscale zero-valent iron can immobilize As over long periods (Gil-Diaz *et al.*, 2019).

C. Biological remediation methods

a). Phytoremediation: Phytoremediation employs soil–plant systems in which metal-accumulating plants are cultivated on contaminated sites, providing a low-cost and environmentally sustainable remediation option. Its effectiveness is site-specific and generally involves three main strategies: phytostabilization, rhizofiltration, and phytoextraction. Phytostabilization aims to immobilize contaminants in soil, thereby reducing their bioavailability and limiting their spread through erosion or leaching. Plants used in this approach enhance soil structure and restrict contaminant migration, making it suitable for sites with contamination levels too high for phytoextraction (Geeta & Choudhary, 2024). Rhizofiltration relies on plant root systems to absorb, concentrate, and precipitate heavy metals from contaminated water, and is particularly effective for treating wastewater and surface water (Geeta & Choudhary, 2024; Mir *et al.*, 2017). Phytoextraction involves plants that accumulate high metal concentrations in aboveground tissues, enabling repeated harvesting to reduce soil contamination. The arsenic-hyperaccumulating fern *Pteris vittata* has been widely used for remediating arsenic-contaminated soils, wastes, and waters, tolerating up to 1,500 ppm As. The cultivation of the arsenic hyperaccumulator Pteris vittata in five distinct contaminated paddy soils was conducted to assess arsenic uptake by rice through the process of phytoremediation (Ye *et al.*, 2011). In a related study, de Oliveira *et al.*, (2017) evaluated the efficacy of the arsenic hyperaccumulator *Pteris vittata* with organic amendments in mitigating arsenic absorption by lettuce from contaminated soil matrices.

b) Bioremediation: Bioremediation of As contaminated soils depends on microorganisms that transform As through three main mechanisms: bioaccumulation, microbial redox reactions, and methylation–volatilization (Dileep *et al.*, 2023). In bioaccumulation, microorganisms absorb As via biosorption on cell surfaces or metabolic uptake, even at low concentrations. This process is affected by soil pH, aeration, temperature, As speciation, and rhizosphere conditions (Anawar *et al.*, 2013). Several bacteria, fungi, and algae, such as *Fucus gardneri* and *Chlorella vulgaris*, are capable of As bioaccumulation (Patel *et al.*, 2023). Microbial redox reactions play a key role in As detoxification and immobilization. Many heterotrophic bacteria oxidize toxic As(III) to the less toxic and strongly adsorbing As(V); for example, *Pseudomonas* sp. HN-2 oxidizes up to 92% of As(III) under aerobic conditions (Zhang *et al.*, 2016), reducing toxicity and enhancing soil adsorption (Zhu, 2009). Under anoxic conditions, some microbes reduce As(V) to As(III), increasing mobility (Zhang *et al.*, 2015). Methylation converts inorganic As into volatile forms, such as dimethylarsenate and trimethylarsine, which escape to the atmosphere and lower soil As levels (Huang *et al.*, 2016; Zhang *et al.*, 2017; Di *et al.*, 2019).

Conclusion

Arsenic contamination in soil is a serious environmental concern due to its persistence, toxicity, and widespread occurrence through both natural and anthropogenic sources. Its complex behaviour in soil (governed by redox conditions, pH, mineral interactions, and microbial activity) determines its mobility and bioavailability, thereby influencing its risk to plants, animals, and humans.

Remediation of As contaminated soil requires a multifaceted approach, combining physical, chemical, and biological techniques. While chemical and physical remediation methods such as soil washing, immobilization, and electrokinetic remediation give effective results, they necessitate substantial financial investment and may pose significant ecological consequences. Conversely, biological remediation techniques, particularly phytoremediation and microbial bioremediation, offer more sustainable and environmental friendly alternatives, though they may be slower and site-specific while soil washing and electrokinetic methods offer more immediate but cost-intensive remediation strategies.

Continued research into the development of low-cost, efficient, and field-applicable technologies such as the use of biochar, nanomaterials, and integrated strategies is essential for long-term mitigation. Ultimately, addressing As contamination requires a combination of scientific innovation, policy support, and community awareness to ensure environmental safety and public health.

Conflict of interest: All authors declare that they have no conflict of interest**.**

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