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Review paper

Utilizing nano-biochar and biochar for sustainable heavy metal remediation and enhanced crop tolerance: Innovative approaches in nano-biosensing and environmental health

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ABSTRACT

Carbon-rich nano-biochar and conventional biochar have emerged as cost-effective, sustainable materials for heavy metal (HM) remediation and soil improvement. Derived from anaerobic thermal processes, these materials immobilize HMs through adsorption, complexation, and precipitation while enhancing soil fertility and climate resilience. The capacity of agricultural systems to withstand and recover from climate-induced stresses such as drought, extreme temperatures, and erratic rainfall should be increased. Nano-biochar, in particular, strengthens resilience by improving drought tolerance (via enhanced water retention and root-zone moisture regulation), nutrient retention (reducing leaching under heavy rainfall), and microbial stability (supporting beneficial soil microbiota under abiotic stresses). However, scalability challenges such as high energy inputs and feedstock variability remain key barriers to widespread nano-biochar adoption. Nevertheless, nano-biochar offers superior performance due to its nanoscale size & properties by employing advanced mechanisms like electrostatic attraction, redox reactions, and cation-π interactions that reduce HM uptake in crops by 30–95 % while increasing yields up to 59 % as demonstrated by aggregated research data discussed in this review. Compared to conventional biochar, nano-biochar demonstrates enhanced nutrient availability, water retention, and microbial activity, along with unique capabilities in nano-biosensing and carbon sequestration (up to 30 %). This review systematically compares their remediation efficiencies, production methods, and agricultural benefits while highlighting critical research gaps needing field validation, including long-term ecological impacts and scalable production economics. The results suggest targeting nano-biochar application in high-value situations-HM-contaminated soils, drylands, precision agriculture systems-coupled with policy incentives (subsidiies, carbon credits) and farmer training schemes to upscale take-up might be valuable. By integrating nano-biochar into climate-smart agro-ecosystem strategies, a collection of stakeholders can address simultaneously food security, environmental renewal, and climate change adaptation goals.

1. Introduction

Globally, heavy metal (HM) contamination affects approximately 24 million hectares of farmland, reducing crop yields by up to 40 % in polluted regions and posing severe health risks to over 16 million people through food chain exposure (Angon et al., 2024; Wen et al., 2024; UNEP, 2013; WHO, 2021). Soil contamination by HMs inflicts detrimental effects on human health and ecosystem stability (Emamverdian et al., 2022,2023a,2025). Excessive HM concentrations threaten soil quality and biological productivity and weaken food safety standards

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while creating health risks due to the accumulation of environmental pollutants in food chains (Perez et al., 2010; Awad et al., 2021; Emamverdian et al., 2017; Yazdanpanah-Ravari et al., 2022;). Anthropogenic activities that encompass waste disposal, wastewater irrigation and mining, smelting, vehicle emissions together with sludge application are primary sources that lead to environmental HM accumulation (Bolan et al., 2014; He et al., 2019; Fazeli-Nasab et al., 2022). Hence, plant uptake of HMs and food safety through the food chain is a top research priority (Mehmood et al., 2021). While various remediation methods exist for HM contamination, such as soil washing, chemical immobilization, and phytoremediation, they often suffer from limitations, including high operational costs, secondary pollution risks, and poor scalability for large-scale applications. Additionally, some techniques (e.g., excavation and disposal) are environmentally disruptive, while others (e.g., electrokinetic remediation) require significant energy inputs. These constraints underscore the need for sustainable, cost-effective alternatives like biochar and nano-biochar, which offer scalable solutions with minimal ecological impact while enhancing soil fertility and crop resilience. Some key requirements for sustainable agricultural practices include identifying economical organic materials as safe solutions for toxic soil contamination while ensuring reliable food production. In this regard, biochar demonstrates excellent potential as an environmentally sustainable material for HM remediation because of its distinct physicochemical features (Chen et al., 2023; Ahmad et al., 2018). Hence, it is currently a versatile tool for sustainable agriculture, serving both as a soil amendment for pollutant cleanup and as a foundation for advanced agricultural technology. With its porous carbon medium and adjustable surface chemistry, it can be used in both environmental remediation and precision agriculture, particularly with nano-biosensors for real-time monitoring of soil health indicators such as pH, nutrient status, and metal bioavailability. The adequate binding capacity of metal ions in the soil is enabled by these material properties, which decreases their bioavailability and toxicity. In addition, these molecules create an alkaline environment in soil that reduces HM availability because of biochar's pH-modifying effect (Wei et al., 2021). The immobilization mechanisms of HMs through biochar treatment primarily operate via electrostatic interactions and employ various processes such as sorption precipitation (Lian and Xing, 2017) metal-ligand complexation, surface co-precipitation, and ion exchange (Ding et al., 2016a). The sorption mechanisms of HMs on biochar surfaces rely primarily on functional group interactions, even though the surface area and pore volume also make contributions (Yu et al., 2019). The ability of biochar to limit HM migration and plant accumulation depends on three fundamental soil characteristics and production processes alongside biochar features (Bolan et al., 2014; He et al., 2019). Studies by Xiao et al. (2018) investigate how medium-temperature biochar efficiently binds metal cations. Other works have demonstrated biochar's efficacy in mitigating HM toxicity in various plant-soil systems, including the reduction of Cd availability by maize straw biochar (Zeng et al., 2018), the immobilization of Cd in smelter-contaminated soil by apple tree and apricot shell biochar (Ali et al., 2018,2020), the immobilization of HMs in plants by sewage sludge biochar (Chagas et al., 2021), and the significant reduction of As and Cd uptake in rice by soybean straw and tea biochar (Li et al., 2018; Pehlivan and Wang, 2022; Pehlivan et al.,2023).

Biochar applications have resulted in substantial reductions in metal content across various rice plant tissues, including shoots, grains, roots, and husks (Zheng et al., 2015). The nano-scale modification of traditional biochar called nano-biochar functions as a novel environmental remediation substance through its unique physicochemical properties. The reduction of biochar to nano-scale through organic biomass pyrolysis creates materials with enhanced surface area and porosity and multiple functional group characteristics when compared to traditional biochar (Zhang et al., 2020). The enhanced adsorption properties together with the high reactivity of nano-biochar enable its effective deployment for treating polluted soils. Multiple simultaneous mechanisms, including adsorption, ion exchange, complexation, and precipitation allow HM removal from nano-biochar (Wang et al., 2021). Recent research confirms that nano-biochar entities have proven superior at extracting toxic metals from both soil and aquatic domains (Yuan et al., 2019). The improved mineral content, storage efficiency, and bacterial activity come from nano-biochar, also creates tolerance for plants against HM stress (Li et al., 2022a). Notwithstanding its great potential, the utilization of nano-biochar necessitates a careful evaluation of its environmental behavior and long-term effects. Compared to traditional biochar, nano-biochar exhibits superior remediation efficiency due to its nanoscale structure, which provides a larger surface area, enhanced porosity, and a higher density of reactive functional groups (Saraugi and Routray, 2024). These properties enable advanced mechanisms such as targeted electrostatic attraction, redox reactions, and cation-π interactions (non-covalent bonds where positively charged metal ions (cations) attracted to the electron-rich π-systems in aromatic carbon structures (e.g., graphene-like sheets in biochar)), allowing for stronger immobilization of HMs and improved plant stress tolerance. While conventional biochar relies primarily on ion exchange and adsorption, nano-biochar's nanoscale reactivity offers greater precision in binding contaminants, making it particularly effective for low-concentration or complex pollution scenarios. Biochar immobilizes HMs primarily through ion exchange (replacing cations on its surface), adsorption (surface binding via functional groups), and complexation (forming stable metal-organic complexes), while nano-biochar enhances these processes through its nanoscale porosity and redox-active sites. Biochar-based sensors represent a novel approach to the detection and monitoring of HMs in soil and water systems. Utilizing the unique physicochemical properties of biochar, these sensors can detect metal ions through electrochemical methods, taking advantage of the high surface area and functional groups that enhance their sensitivity and selectivity (Shishehbore et al., 2022). The integration of biochar with nanostructured materials further improves the sensor's performance, allowing for real-time monitoring of pollutants and facilitating timely intervention strategies in agricultural and environmental management (Zheng et al., 2025). This technology not only enhances the effectiveness of HM remediation efforts but also contributes to sustainable practices by providing essential data for assessing soil health and water quality. Indeed, recent advancements demonstrate that nano-engineered biochar can encapsulate hazardous metals while acting as an in situ sensing platform, integrating traditional remediation techniques with agricultural industry practices. This technological synergy positions biochar as a fundamental element for climate-smart agriculture, enhancing its carbon sequestration potential through digital monitoring capabilities to improve crop management in contaminated soils.

Here this review analyzes the operations that make nano-biochar effective in HM removal alongside plant resilience enhancement and explores present limitations with potential investigation directions. This study addresses critical gaps in understanding the comparative efficacy and mechanisms of nano-biochar versus conventional biochar for HM remediation, particularly their long-term field-scale applicability and plant tolerance enhancement, which remain underexplored in current research. Beyond remediation, biochar and nano-biochar support climate-smart agriculture by enhancing soil carbon sequestration, reducing fertilizer dependency, and improving crop resilience in contaminated lands, which are key benefits for food security under climate change. Biochar and nano-biochar offer cost-effective remediation, with production costs up to 50 % lower than conventional methods, while improving soil fertility and crop yields, delivering both economic and environmental returns. While biochar and nano-biochar show promise, their efficacy depends on feedstock variability, potential long-term soil impacts, and lack of standardized regulations, which are challenges requiring further research for widespread adoption. This review systematically evaluates the mechanisms by which biochar and nano-biochar remediate HM contamination, enhance plant stress tolerance, and identify key research gaps for sustainable environmental applications.

2. HMs in the environment and their inhibitory roles on plant growth and development

The literature demonstrates that soil functions as a primary accumulation zone for HMs and metalloids through the continuous rise of anthropogenic activities (Al-Robai et al., 2023; Emamverdian et al., 2024a). Contaminants present in soil continue to persist and spread beyond the ground into water networks, exposing both the environment and humans to broad dangers (Shishehbore and Safaei, 2022). Soil HM contamination creates direct ecological risks to the food chain in proximity to rivers and estuaries along with surface waters (Zhuang et al., 2018). The critical nature of this problem requires precise knowledge about how HM affects these processes (Shishehbore and Safaei, 2022). For instance, HMs in China directly impact grain production to the extent of 1.2×10^7 tonnes each year, checking approximately 2×10^7 hectares of agricultural land and leading to economic losses of about 3 billion dollars (Zhao et al., 2015; Emamverdian et al., 2024b).

The non biodegradable nature of HMs allows them to accumulate within living organisms, leading to severe health issues, including cancer development (Yadav et al., 2019). One of the most severe threats causing this is their capacity to bind functional groups found in biomolecules, which include amino- and carboxylic acid and sulfur groups. The interaction creates disruption that harms both enzyme activities and protein functionality, leading to cellular process interference. Also, interfering with phosphate biomolecule degradation and precipitation processes are HM-caused events in cells (Turan et al., 2022).

Plants with their rich root systems uptake metal ions from the soil and convey them through diverse carrier proteins via rhizosphere exudation and pH alterations (Tangahu et al., 2011). Metals may infiltrate plants via stomata, injuries, and lenticels, resulting in accumulation within leaves (Shahid et al., 2017). Upon entering plant cells, HMs provoke oxidative stress through reactive oxygen species (ROS), damaging membranes and altering membrane permeability (Emamverdian et al., 2018,2023b). HMs presence in plants alters water balance, restricts nutrient absorption, induces chlorosis and senescence, diminishes photosynthetic efficiency, and may ultimately result in plant mortality (Singh et al., 2016).

The binding of these metals to sulfhydryl groups in proteins impedes enzyme activities, reduces plant metabolic efficiency, compromises membrane integrity, and affects cofactor activity (Dickinson and Scott, 2010; Ali and Mahmoud, 2013). HMs have been documented to interact with DNA and nuclear proteins, resulting in oxidative damage to biological macromolecules in plants (Briffa et al., 2020). Consequently, they adversely affect crops and plant species by disrupting morphological, physiological, cellular, and molecular processes, significantly reducing crop quality and yield.

Besides these immediate biological impacts, HMs accumulate in agro-ecosystems primarily through anthropogenic pathways. Indeed, anthropogenic practices, including mining, wastewater discharge, and urbanization have tremendously increased environmental HM concentrations during the last centuries. The contamination aggravates soil health and adulterates plant commodities with possible cascading risks to food security and ecosystem stability.An additional primary concern of environmental pollution caused by HMs is their severe impact on soil quality and plant products. In recent centuries, with the acceleration of urbanization and anthropogenic activities, the presence of HMs in the environment has increased significantly due to industrial processes, such as mining, wastewater discharge, and other industrial activities. Fig. 1 illustrates the relative contributions of different anthropogenic and natural sources to HM pollution. Industrial activities (e.g., smelting, manufacturing) and agricultural practices (e.g., pesticide/fertilizer overuse) dominate HM emissions, while vehicular exhaust and natural

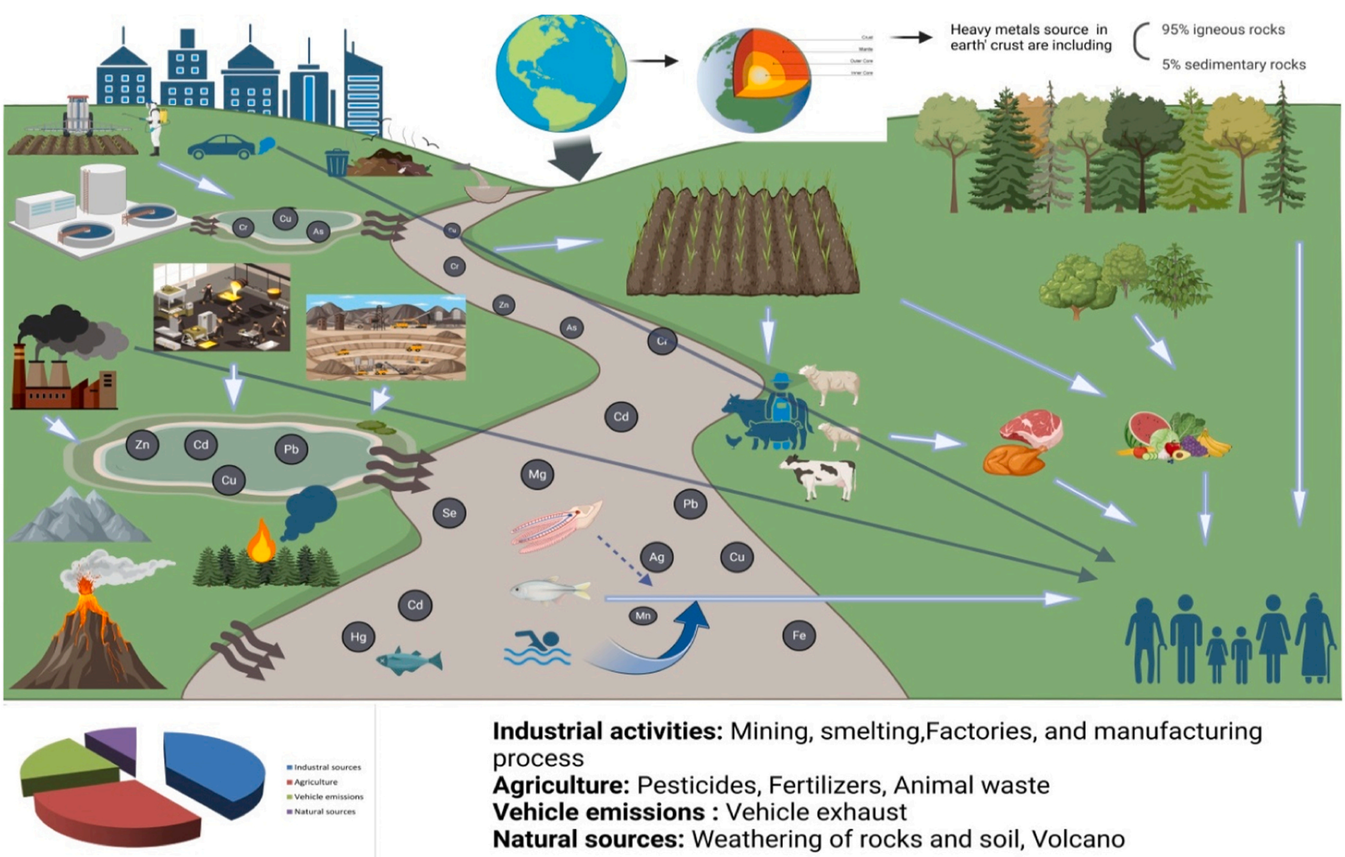

Fig. 1. Diagrammatic illustration of HMs sources in the soil.

weathering processes (e.g., volcanic activity, rock erosion) contribute minimally. The figure highlights the need for targeted mitigation strategies in high-impact sectors.

Phosphate fertilizer, meanwhile, is a significant source of Cd in agricultural soils, which can be extremely harmful to plants. Notwithstanding its toxicity, it is extensively utilized in numerous farm systems to enhance crop development (Wan et al., 2014). The Cd content in phosphate fertilizers varies by region, ranging from 5 to 41 mg/kg in the United States, 0.14–51 mg/kg in Brazil, and non-detectable to 27 mg/kg in China (Wan et al., 2014; Yu et al., 2018). Copper in agricultural soils predominantly originates from pesticide application. In France, the excessive application of pesticides, especially the Bordeaux mixture, in vineyards and orchards has resulted in approximately 30 % of the Cu present in agricultural soils originating from pesticide use (Belon et al., 2012). HMs from air deposition and fertilization are considerably more abundant in agricultural soils compared to those from sewage irrigation and sewage sludge applications (Angon et al., 2024; Peng et al., 2019). The contributions of various sources to soil HM concentrations are reported in Table 1. Table 1 compares the proportional inputs of various HM sources, demonstrating that atmospheric deposition and fertilization contribute substantially more to soil contamination than sewage irrigation or sludge application.

Plants clearly manage HM stress by using a variety of methods, including minimizing HM deposits, translocating metals to another organ, and increasing metal tolerance. These systems vary by plant species and soil type. Understanding the signaling pathways involved in HM stress responses will aid in plant survival and the development of efficient techniques to minimize HM pollution in soil and plants utilizing organic and cost-effective materials.

3. The potential of biochar as an amendment

3.1. Biochar: soil's black gold

Produced in high-temperature, oxygen-free conditions by pyrolyzing and carbonizing organic biomass such as agricultural waste and leaf matter, carbon-rich biochar (Chagas et al., 2022; Ji et al., 2022) benefits soil structure by enhancing moisture retention, cation exchange capacity (CEC), fertility, and remediation of xenobiotics. The primary mechanism by which biochar enhances soil quality is through microbial population increases, which impacts enzyme activity, nutrient cycling, soil carbon concentration, and mineralization (Palansooriya et al., 2019). While high lignin concentration may increase the number of some bacteria, its nutritional content, including magnesium (Mg), phosphorus (P), and potassium (K), enhances microbial enzyme activity, thereby improving soil health (Ji et al., 2022). In addition to soil advantages, biochar is a valuable tool in promoting the establishment of efficient electron-transfer pathways, therefore minimizing the harmful effects of ROS on plants (Yang et al., 2022).

3.2. Production methods and required materials for biochar production

3.2.1. Materials for preparation

The structural arrangement of the substrate, as well as the circumstances under which it is formed, have a significant impact on the characteristics of biochar. Sawdust, sludge, industrial solid waste, and agricultural wastes, such as fruit shells, straw, and animal manure, are among the raw materials/feedstocks used in biochar production. Each type of biomass exhibits a diverse range of elemental content (Wang et al., 2020b).

Biochar feedstock encompasses: (1) Agricultural residues, including corn stover, sugarcane bagasse, and rice husk, which serve as sustainable resources for biochar, alongside coconut husk, corncob, coconut shell, rice straw, and wheat straw. (2) Wood chips and branches, typically sourced from trees. (3) Sewage sludge, which contains organic matter as well as phosphorus, nitrogen, and potassium, albeit with contaminants. (4) Animal manure, including that from cows, chickens, and pigs, which serves as feedstock for managing soil fertility, along with woody biomass originating from trees and other woody sources. This includes classifications for sawmill residues, construction debris, forest residues (such as shed leaves and branches from mountainous areas), and various other organic wastes (Cheng et al., 2020).

3.2.2. Methods of production

Biochar can be created using numerous methods, such as hydrothermal pyrolysis, high-temperature pyrolysis, and flash pyrolysis (Du et al., 2017). Liu et al. (2021) and Du et al. (2017) state that feedstock type, carbonization technique, and pyrolysis temperature directly influence the chemical and physical features of the surface. The pyrolysis temperature is crucial since it influences the specific surface area and porosity of the biochar. When heated, biochar often expands surface area and creates more pores than the original material. Excessively high temperatures may alter the properties of biochar in many ways (e.g., reduced functional groups or excessive ash formation). Biochar produced by pyrolysis at elevated temperatures has a greater porosity compared to biochar generated via thermal carbonization.

Scanning electron microscopy (SEM) is often used to examine the pores and morphology of biochar generated via high-temperature pyrolysis. The majority of surface functional groups are eliminated, but a greater proportion is retained with alkaline carbonization. The minerals and nutrients in biochar ash are directly affected by the temperature and particular type of supply used in the pyrolysis procedure. As the

Table 1
The contribution of soil HM content in China and globe.

Region	City/Country	Cu (mg/kg)	Pb (mg/kg)	Cd (mg/kg)	Zn (mg/kg)	Cr (mg/kg)	Ni (mg/kg)	References
China	Nanjing	66.10	107.30	-	162.60	84.70	-	Lu et al., (2003)
	Qingdao	55.00	62.00	0.30	201.00	54.00	17.30	Yao et al., (2008)
	Shanghai	59.25	70.69	0.52	301.40	107.90	31.14	Shi et al., (2008)
	Beijing	23.70	28.60	0.15	65.60	35.60	27.80	Zheng et al., (2008)
	Changsha	51.40	89.40	6.90	276.00	121.00		Xi et al., (2008)
	Guangzhou	62.57	108.55	0.50	169.24		25.67	Lu et al., (2007)
Europe	Spain	57.01	1505.45	3.76	596.09	-	-	Rodrguez et al., (2009)
	France	20.06	43.14	0.53	43.14	42.08	14.47	Hernandez et al., (2003)
	UK	30.10	49.00	0.33	76.00	68.00	21.00	Rawlins et al., (2012)
	Netherlands	10.20	15.60	0.14	40.30	15.70	4.94	Brus et al., (2009)
	Finland	23.00	17.00	0.17	90.00	59.00	24.10	Salonen and Korkka-Niemi, (2007)
Middle East	Iran	60.15	46.59	1.53	94.09	63.79	35.53	Sayadi and Rezayi., (2014)
	Syria	34.00	17.00		103.00	57.00	39.00	Moller et al., (2005)
World	America	17.30		0.16		24.00	18.30	Burt et al., (2003)
	Africa	100.00	150.00	5.00	500.00	250.00	100.00	Yabe et al., (2010)
	India	79.05	41.80	-	178.50	147.05	52.46	Adimalla et al., (2019)

pyrolysis temperature increases, the mineral composition also tends to increase. Sludge and garbage serve as feedstocks due to their higher mineral content compared to plant matter. Temperature may also influence the crystallization of rocks in biochar. Chen et al. (2023) showed by X-ray diffraction that some minerals on the biochar surface exhibit increased solidity with higher pyrolysis temperatures.

Elevated temperatures (> 200 K min^{-1}) and brief reside durations ($<$ 2 s) induce rapid pyrolysis, resulting in minimal char production but yielding high-energy-density bio-oil. By contrast, slow-pyrolysis employs reduced heating rates (0.1–1 °C s^{-1}) and extended durations (ranging from several hours to days), prioritizing the production of biochar above liquid or gaseous by-products (Liu et al., 2015). Slow-pyrolysis produces biochar with a significantly higher surface area and porosity, which enhances its capacity for adsorbing HMs through various mechanisms such as adsorption, ion exchange, and complexation. This increased surface area allows for a greater number of active sites to interact with pollutants, making Slow-pyrolysis more effective in immobilizing metals in contaminated soils. In Slow-pyrolysis, increasing the peak temperature to 400–500 °C often results in biochar with a higher fixed carbon content (Shaheen et al., 2019). In contrast, gasification, while effective for producing syngas and other fuels, can pose risks associated with the release of volatile organic compounds (VOCs) and other toxins during the process. The byproducts of gasification may contain harmful substances that can contribute to environmental pollution. Additionally, the biochar produced through gasification often has lower stability and fewer advantageous properties for soil remediation compared to that produced by Slow-pyrolysis. Therefore, while both methods have their applications, the superior adsorptive qualities of slow-pyrolysis biochar, combined with its reduced environmental risks, make it the preferred choice for effective HM remediation in agricultural contexts. Intermediate pyrolysis on the other hand, occupies a position between rapid and slow-pyrolysis in terms of temperature and duration of the process. It produces biochar that is dense and has little reactive tar. This type of biochar is particularly suitable for fertilizer production and enhancing soil nutrients. Beyond pyrolysis, gasification offers an alternative approach but with distinct trade-offs (Table 2).

3.2.3. Gasification

The primary byproduct of gasification is a flammable gas composed of CO, H_2, and CH_4. Oxidation of dry biomass transpires by direct contact with air, oxygen, steam, carbon dioxide, nitrogen, or a mixture of these gases. Moreover, the biochar generated often harbors large concentrations of toxins, such as alkali polyaromatic hydrocarbons and alkaline metals, with diminished yields linked to high-temperature reactions that exacerbate the presence of these hazardous chemicals (Kambo and Dutta, 2015; Zhang et al., 2019,2019,2019).

3.2.4. Hydrothermal carbonization

The procedure occurs at temperatures between 180 and 300 °C and pressures ranging from 2 to 6 MPa. This approach employs feedstocks of dry biomass mixed with water or wet biomass, hence obviating the need for drying pre-treatment. Unlike gasification and pyrolysis methods, hydrothermal carbonization produces hydrochar, which demonstrates numerous benefits (Cheng and Li, 2018; Kambo and Dutta, 2015). Hydrochar is distinguished by a high concentration of surface functional groups, an increased degree of aromatization, enhanced purity, and superior yield. Furthermore, it demonstrates low metallic concentration and reduced alkalinity, leading to increased thermal conductivity and carbon content (Kambo and Dutta, 2015).

Table 2
Temperature ranges, yields, and primary applications in different pyrolysis methods (slow, fast, intermediate).

Pyrolysis Method	Temperature Range	Heating Rate	Primary Output	Best For
Slow pyrolysis	300–700°C	0.1–1 °C/s	Biochar	HM remediation, soil amendment
Fast pyrolysis	400–800°C	> 200 °C/min	Bio-oil	Energy production
Intermediate pyrolysis	250–400°C	Moderate	Biochar + gases	Fertilizer production

3.2.5. Torrefaction

Torrefaction, recognized as mild pyrolysis, results in a 30 % loss of biomass mass during the process. Furthermore, the materials produced through this method cannot be classified as ideal biochar. Because these materials typically consist of organic carbon compounds that exhibit a high specific energy density. Since this stage is merely an initial step to the pyrolysis process (Kambo and Dutta, 2015; Zhang et al., 2019), it does not qualify as a "real" biochar. While many methods demonstrate sufficient efficiency for biochar production, pyrolysis stands out as a suitable and effective process for preparing a regular biochar material. Fig. 2 illustrates the process of biochar preparation through pyrolysis in both industrial and laboratory settings. Biochar is typically produced via slow-pyrolysis (300–700°C for 1–4 h) using feedstocks like crop residues, wood chips, or manure, with temperature and duration dictating its physicochemical properties. Fig. 2 highlights key differences in feedstock handling, heating rates, and product collection methods between these operational scales.

3.3. Biochar and HMs

The primary byproduct of gasification is a flammable gas consisting of CO, H_2, and CH_4. The capacity of biochar to assimilate HMs can be influenced by its characteristics, which are contingent upon the pyrolysis conditions and source materials. When these characteristics interact with soil properties, they may influence the interaction between biochar and HMs, potentially affecting their mobility and phytotoxicity in the soil (He et al., 2019). Two main processes explain biochar's effect on HM behavior: HM mobility and abundance in polluted soils and soil condition modification, which affects HM dynamics. Table 3 outlines the mechanisms involved in HM removal and the substantial function of biochar in HM absorption. The table also highlights biochar's dual role in HM remediation—reducing mobility while improving soil health—making it a sustainable solution for contaminated sites.

3.3.1. Front-line mechanisms for addressing HMs

Biochar's key mechanisms for mitigating HMs include precipitation, electrostatic attraction, physical sorption, and complexation, which work synergistically to reduce the mobility and toxicity of HMs in contaminated soils (Khan et al., 2022) Biochar mitigates HMs through four synergistic mechanisms: (1) precipitation enhanced by alkaline minerals (e.g., $CaCO_3$, KCl), (2) electrostatic attraction, (3) physical sorption, and (4) complexation with functional groups. While conventional biochar relies on macroscale interactions, nano-biochar exhibits superior electrostatic attraction due to its enhanced surface charge density and nanoscale effects. Indeed, the types derived from plant materials under high temperatures tend to have an alkaline character, which promotes metal ion precipitation (Inyang et al., 2016). Other types contain various mineral phases, such as sylvite (KCl), quartz (SiO_2), and calcite ($CaCO_3$), which interact with metal ions to enhance precipitation (Cao and Harris, 2010). At high pyrolysis temperatures (300–600°C), these minerals have lower solubility, leading to slower sorption reactions and the formation of metal precipitates on the biochar surface (Inyang et al., 2016). The functional groups, including but not limited to hydroxyl and phosphate (CO_3^{2-}, PO_4^{3-}), contribute their share in the binding of metal ions, subsequently precipitating on the surface of biochar. Another important pathway for the HM immobilization process is through the electrostatic attraction between the biochar surface (negatively charged) and that of the metals. Specific metals with a charge of positive ions are electrostatically attracted toward the

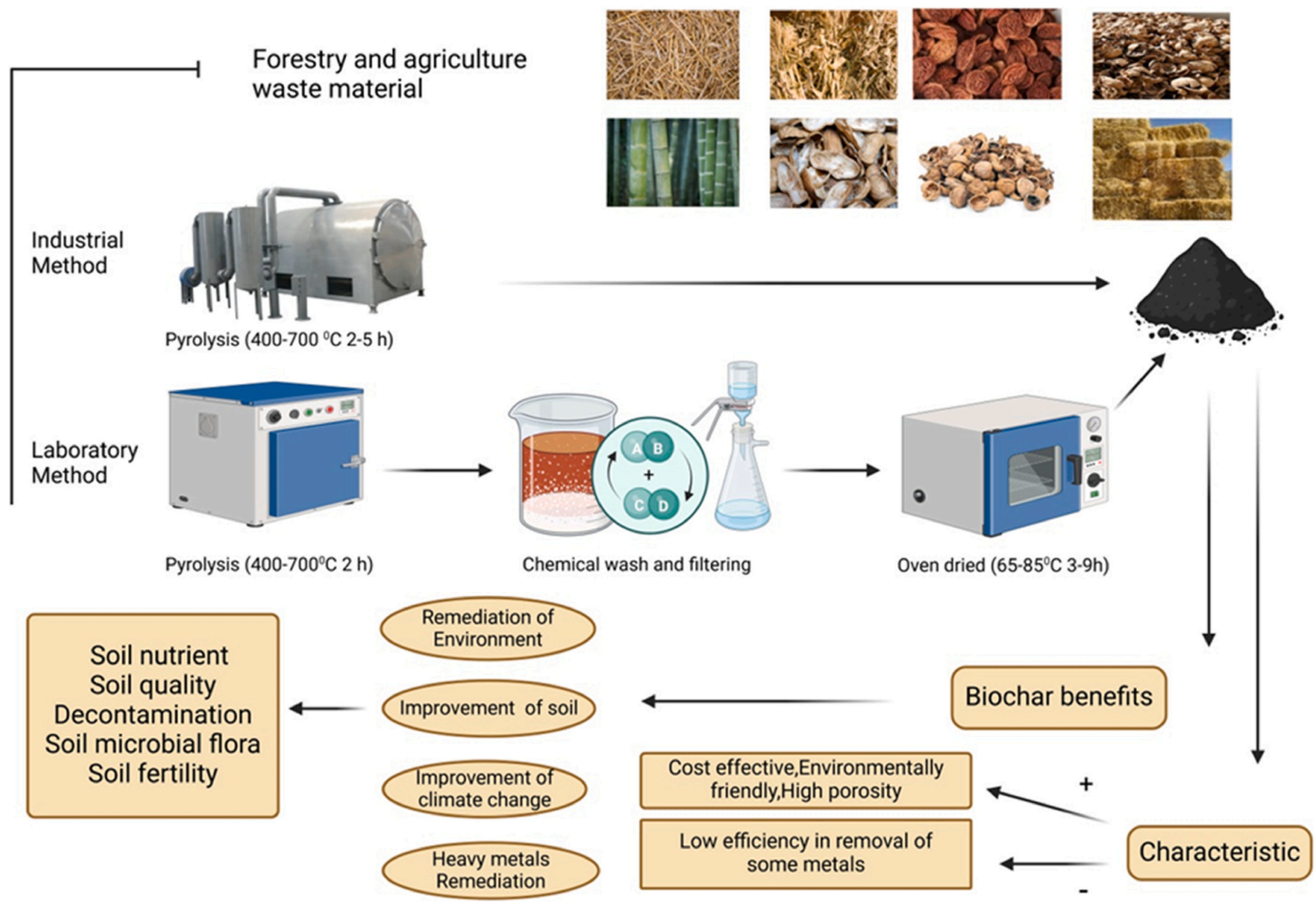

Fig. 2. Diagrammatic illustration of biochar preparation by pyrolysis at industrial and lab scale.

negatively charged groups in the biochar, as evidenced by Liang et al. (2021). This process is affected by factors like solution pH and biochar's point of zero charge (PZC), which increases metal sorption (Mukherjee et al., 2011). Pyrolysis temperature also influences the development of graphene-like structures that enhance adsorption through these electrostatic interactions (Dong et al., 2024; Zhang et al., 2021). Biochar decreases HM toxicity through physical sorption, where the metal ions in motion diffuse into the pores of the biochar. The high surface area and numerous pores generated during the pyrolysis process facilitate this absorption (Liang et al., 2021). Functional groups, alkaline minerals, and negative charges on the biochar surface further enhance its sorption capacity. For example, biochar derived from pine wood and switchgrass at 300°C and 700°C has been shown to absorb Cu effectively (Dong et al., 2011). Therefore, functional group complexation is one of the most necessary HM toxicity mitigation pathways through reduced metal phytoavailability. These complexations are facilitated by various multiatom structures developed on the surface of the biochar via interactions in metal-ligand specificity (Inyang et al., 2016). Complexation quickly occurs with those metals that display d-orbitals because they show strong ligand affinity, as explained by Crabtree (2009). Biochar prepared at lower pyrolysis temperatures contains oxygen-functional groups that are efficient in forming complexes with HMs (Liu and Zhang, 2009). Over time, the oxidation of biochar increases its carboxyl groups and oxygen content, enhancing the complexation mechanism.Beyond direct HM interactions, biochar indirectly influences metal mobility through soil conditioning

3.3.2. s-line mechanisms for addressing HMs

Both the mechanisms at the front line (primary direct interactions) and second line of biochar characteristics (secondary soil-mediated effects) from mineral release, pH adjustment to cation exchange, and enhancement of organic matter depend on the correct preparation conditions of biochar, soil characteristics, and type of feedstock. For instance, several studies have reported the improvement of HM absorption in contaminated soil as a result of the release of minerals such as Mg, Ca, and K released by biochar to enhance HM absorption (Natasha et al. (2021), He et al. (2019), Mansoor et al. (2021). These minerals are released down the biochar surface, contributing to adsorption. Rees et al., 2014. Biochar addition increases the P content in the soil, which is helpful in precipitating phosphate minerals and enhancing the sorption of Pb (Yang et al., 2021). Ca, Si, and Mn reductions in a biochar product will result in enhanced surface sorption sites, contributing to HM sorption (Rees et al., 2014). Biochar's alkaline nature increases pH in acidic soils, improving soil conditions and facilitating this immobilization (Mansoor et al., 2021). This positive effect is attributed to proton (H^+) competition on the surface colloids and changes in the CEC of the soil (Puga et al., 2016). When the biochar presents high CEC, it facilitates the release of cations, such as Mg^{2+} and Ca^{2+}; this will contribute to the development of ionic exchange between material and metal ions (Li et al., 2015), increasing the soil pH and, therefore enhancing the CEC. With this process, there is a loss of protons from biochar's functional groups, and negative charges enhance HM sorption and CEC (Banik et al., 2018). These negative charges facilitate the electrostatic exchange of cations, reducing HM mobility and its availability in the soil (Singh et al., 2010). Typically, biochar produced at a lower temperature has a lower C: O ratio, which enhances CEC and HM sorption (Feng et al., 2018). The protons of nano-biochar's surface functional groups increase HM adsorption by ionic exchange. The

Table 3
HMs absorption with biochar via involved mechanisms.

Biomass material	Pyrolysis temperature (°C)	HMs	HMs adsorption capacity (mg/g)	References	HM removal mechanism	Contributor
rice straw biochar	500 °C 5 h	Cd, Pb	45.1 and 61.4 mg/g,	Fan et al., (2020)	Rice straw biochar reduces 40 % of Cd availability in soil and limits Pb immobilization of soil	China
H_3PO_4-modified chicken feather biochar	-	Cd and Pb	7.84 mg/g and 24.41 $mg^{1-(1/n)}.L^{1/n}.g^{-1}$	Chen et al., (2019)	Absorption of Cd^{2+}/Pb^{2+} determined by electrostatic interaction, O- and N-containing groups higher adsorption of Pb^{2+} than Cd^{2+}has determined by phosphate precipitation little impact of proline, glucose, and pH (4–6) on the absorbance of Cd and Pb	China
peanut shells	300 °C for 5 h	Pb	22.82 (mg/ g)	Xue et al., (2012)	Hydrochar removes aqueous HMs by modification of $H_2O_{2,}$ so $H_2O_{2,}$ with the enhancement of oxygen-containing functional groups on hydrochar, is affected by hydrochar. So, the obtained hydrochar introduced as effective absorption of HMs	China
pine chips	200, 350, 500, and 650 °C for 4 h.	Cu^{2+}	20.7(mg /g)	Peng et al., (2017)	The surface area of biochar significantly influences its ability to sorb Cu, with higher sorption levels observed in biochar produced at higher pyrolysis temperatures. The functional groups facilitate the binding of metals through mechanisms such as ion exchange, electrostatic attraction, and ligand coordination, thereby improving the biochar's ability to immobilize metals like Cu.	China
poplar chips	550 °C	Al^3	89.58(mg /g)	Yin et al., (2018)	There is a chemical absorption process between nutrients and Al-modified biochar, so Al-modified biochar is selected as a good option for the remediation of eutrophic water.	China
palm kernel cake residue	350 °C	$Cd^{2,}$ $Cr,^{3,}$ $Pb^{2,}$ Hg^{2+}	18.60(mg/ g), 19.92(mg/ g), 49.64(mg/ g), 13.69(mg/ g).	Maneechakr and Mongkollertlop, (2020)	After CP-Fe-Mn was easily provided from Palm kernel cake residue, removal of Cd^{2+}, Cr^{3+}, $Pb^{2+,}$ or Hg^{2+} was conducted. Adsorption behavior was found between endothermic procedures and physisorption.	Thailand
coconut shell	800 °C for 6 h	Cd^2	30.1	Liu et al., (2018)	Biochar influences soil biological activity and the availability of metals. Biochar improved microcosmic pore structure and surface functional groups (impact on immobilizing metals)	China
sugar beet tailing	300°C	Cr	123 (mg/g)	Dong et al. (2011)	The positively charged sites and functional groups on sugar beet biochar surface can remove Cr	USA
orange waste	300°C	Cu	4.9(mg/g)	Pellera et al. (2012)	The Cu removal was attributed to the surface functional groups and active sites on the surface of orange waste biochar improved sorption capacity	Greece
rice husk	175–180°C	Hg, and Zn	303.03, and 18.94	El-Shafey (2010)	Rice husk biochar has active sites for HMs binding and increases their removal with high CEC and high sorbent acidity	Oman

efficiency of this process depends on the amount of HMs and the distribution of the functional groups on the biochar surface (Inyang et al., 2016). Moreover, the biochar prepared from woody materials had lower CEC than biochar prepared from plant materials. Plant-based biochars tend to have more acidic functional groups and a higher oxygen content than woody biochars (Harvey et al., 2011). Biochar's high carbon content elevates SOC, enhancing the binding of HMs to biochar surfaces. The presence of oxygen-functional groups plays a crucial role in this binding process. An increase in SOC can reduce the mobility of Pb(II) and convert it into less mobile forms, limiting absorption/bonding by plants (Abdelhafez et al., 2014). Besides, high SOC content nurtures the soil microorganisms, which enhances the quality of the soil and decreases HM availability (Khan et al., 2022).

3.4. Soil HMs remediation by biochar

The nature of the biochar depends on the biomass source and pyrolysis conditions and, hence, its performance in HM interactions. Table 4 summarizes data from various research studies on the remediation of HM-polluted soils using different types of biochar under varying pyrolysis conditions across diverse soil types. These emphasize the roles, paying particular attention to the mechanisms of HM reduction. Various key factors contributing to the effectiveness of biochar in the binding of HMs include its mineral content, surface functional groups, π-electrons, alkaline metal ions, pore structure, especially micropores, and organic matter content. The material's actions in reducing bioavailability and mobility include cation exchange, complexation, reduction, precipitation, and electrostatic attraction. These mechanisms facilitate the binding of metals to the biochar surface, converting HMs from inorganic to organic forms (Lahori et al., 2017; Wang et al., 2017). In this respect, the interactions between biochar and HMs are of great importance for soil remediation. Among these, Cu, Cd, Pb, As, and Hg demonstrate specific toxicity in soils. The following section discusses, on an element-wise basis, how biochar decreases the toxicity of these HMs for effective soil management.

3.4.1. Copper (Cu)

The beneficial impact of biochar on Cu pollution is mainly due to its capacity to immobilize the metal, elevate soil pH, and create binding complexes with ions (Cheng et al., 2020). Chicken-manure-derived biochar has a significant affinity for Cu owing to its negative ζ-potential and functional groups, especially hydroxyl (-OH) groups, which interact with Cu ions. The liming impact of biochar elevates soil pH, and it adds to the biochar surface functional groups (C=O and phenolic-OH), hence enhancing its complexation ability with Cu(II) for the immobilization of the target metal. Meier et al., (2017). Biochar alkalinity is a significant attribute that may enhance soil pH (Gonzaga et al., 2018). Rechberger et al. (2018) demonstrated that hydroxides and carbonates in woodchip-derived biochar facilitated the chemical reaction of $CuCO_3$ and $Cu(OH)_2$, leading to Cu(II) immobilization. A similar formation method was noted by Zhang et al. (2018) in bamboo-derived biochar. Dissolved organic carbon (DOC) in soil influences Cu mobility. Biochar application can increase soil DOC levels, which may enhance the formation of soluble Cu(II)-organic complexes. For instance, Park

Table 4
The remediation of HMs in soil research emphasizes the feedstock, metals, soils, and pyrolysis conditions involved in the mechanism.

Biomass material	Condition of Biochar Production(pyrolysis conditions)	HM remediation	Soil condition	Mechanism and results	References
Spartina alterniflora-derived biochar	350, 450, 550, and 650 °C	Cd	saline-alkaline	Reduction of Cd available occurred in low pyrolysis temperatures (350 and 450 °C). Biochar enhanced the pH and organic matter in the soil, but it didn't impact soil salinity	Cai et al., (2020)
wheat straw	485 °C	Pb, Cd	tidal cinnamon soils (clay texture class, 38 % sand, 57 % clay, 5 % silt)	Fluctuations in soil redox and soil pH in drought and flood cycles play a role in the mobilization and immobilization of metals, and biochar has a lower impact on the reduction of HMs in the flooding cycle rather than in the drought cycle of soil.	Sui et al. (2018)
coconut shell, straw, and sludge-derived biochar	300–500 °C	Pb, Zn	mining area	Reduction of acid soluble fraction and leaching concentration of HMs Improvement of HMs remediation by using biochar with the extension of incubation time and enhancement of dosage	Xu et al., (2022)
lychee branches	500 °C	Pb, Cd, As, Zn	mining area	Increasing HM accumulation in the leaf - enhances the HM-extraction effect in the plant.	Jun et al. (2020)
modified coconut shell biochar (MCSB)	800 °C	Cd, Ni, and Zn	sandy soil	Reduction of acid soluble of Cd, Ni, and Zn in soil immobilizing of metals in soil Increasing soil physicochemical and biological properties	Liu et al., (2018)
bamboo wood	750 °C	(Pb, Cb)	paddy field (silty loam soil—71.6 % silt, 7.0 % sand, and 22.4 % clay)	Enhancement: soil alkalinity, electrical conductivity (EC), OC content, and soil PH. Reduction: Cd and Pb uptake by plant- Cd concentration in plant organs- N availability.	Xu et al. (2016)
pig manure and *Suaeda glauca* biochar	00 °C, 500 °C and 700 °C for 4 h	Cd and Pb	fluvo-aquic soil	Enhancement of extractable Pb and Cd in red soil Increasing the mobility and bioavailability of HMs by released ions from biochar	Liu et al., (2020)
orange bagasse, coconut husk, and sewage sludge	(500 °C)	Cu	ultisol	Reduction of Cu availability, enhancement of Cu associated with OM	Gonzaga et al. (2020)
wheat straw	300 and 700 °C	Cd	paddy soils	Biochar with the reduction of soil exchangeable fraction reduced 70 % of Cd concentration in soil	Gong et al., (2021)
sewage sludge-derived biochar, and wood biochar	500 °C	Cd, Pb, and Zn	mining soil	Application of biochar enhances soil and leachate pH, as well as decreases Zn, Cd, and Pb bioavailability	Penido et al., (2019)
(*Eucalyptus sp.*) and sewage sludge wood	(350 °C)and (500 °C)	(Pb, Cd, Zn)	mining soils	Enhancement of soil PH and leachate Reduction of bioavailability levels of Cd, Zn, and Pb	Penido et al. (2019)
olive tree pruning, pine woodchip	(450 °C)	Cu and As	sandy loam texture (39 % silt, 54 % sand, and 7 % clay)	Reduction of plant HM uptake, as well as HM availability and mobility in the soil	Brennan et al. (2014)

et al. (2011) observed this effect in their study using chicken manure-derived biochar. Similarly, Wagner and Kaupenjohnn (2015) reported elevated Cu concentrations in soil solutions after Miscanthus-biochar application, attributing this to Cu(II) desorption via organic complexation. Furthermore, Pehlivan and Wang (2022) evidenced that biochar obtained from pyrolyzed nutrient-rich black tea waste improved the adsorption of toxic Cu(II), whereas mitigated the detrimental effects promoting biomass growth and reducing phytotoxicity in maize plants.

3.4.2. Lead (Pb)

The Pb immobilization by biochar is a key mechanism for reducing Pb toxicity in soils. This happens primarily through precipitation and cation exchange reactions between Pb(II) and molecular groups in prepared biochar (e.g., CO_3^{2-}, OH^-, and alkaline metals (Ca^{2+}, Mg^{2+})), leading to the formation of $Pb_3(CO_3)_2(OH)_2$, which precipitates out of solution (Cheng et al., 2020). Biochar derived from soybean waste immobilized Pb in the soil through π-cation electron donor-acceptor interactions by the interaction between π-electron-deficient Pb(II) ions and the π-electron-rich graphene surface of biochar (Ahmad et al., 2016). On the other hand, biochars produced from sewage sludge (Ho et al., 2017) and poultry manure (Wang et al., 2015) have a high content of phosphate, which precipitates insoluble Pb-phosphate compounds like $Pb_5(PO_4)_3OH$, $Pb_5(PO_4)_3Cl$, and β-$Pb_9(PO_4)_6$, decreasing the mobility of Pb in soil (Huang et al., 2018; Ahmad et al., 2016). Moreover, functional groups present in biochar significantly contribute to the Pb immobilization process. For example, vegetable-waste biochar contains N-containing functional groups, particularly -NH_2, that form strong covalent bonds with Pb, drastically reducing extractable Pb concentrations in the soil as measured by NH_4OAc.

3.4.3. Cadmium (Cd)

Cadmium mobility in soil is strongly influenced by soil pH (Gao et al., 2019,2019). In alkaline conditions, biochar's binding capacity and Cd adsorption are enhanced through the presence of compounds like PO_4^{3-}, CO_3^{2-}, and OH^-, which promote the transformation of free Cd(II) into precipitates such as $Cd_3(PO_4)_2$, $Cd(OH)_2$, and $CdCO_3$ (Rechberger et al., 2018). The decrease in Cd bioavailability results from the high adsorption affinity of biochar is further augmented by cation exchange interactions with soil calcite ($CaCO_3$). Moreover, the extensive specific surface area and many functional groups of biochar are essential for the immobilization of cadmium (Zhang et al., 2019). Some studies have shown that biochar can reduce Cd concentration both in soil and water. For instance, rice straw biochar application significantly reduced Cd concentration in pore water within the soil rhizosphere (Yin et al., 2017), while biochar prepared from corn straw lowered Cd levels by 91 % in farmland soil (Gao et al., 2019). Applying 40 t ha^{-1} of wheat straw-derived biochar in paddy soil over three years led to a 24 % reduction in DTPA-extractable Cd and a 59 % reduction in $CaCl_2$-extractable Cd (Bian et al., 2014). Enhancing soil pH via biochar application is an essential factor in Cd immobilization. Furthermore, deprotonating acidic functional groups within the material can improve its capacity for Cd adsorption (Qian et al., 2019).

3.4.4. Arsenic (As)

Phosphorus has similar chemical properties to As, with particular importance in reducing As toxicity in contaminated soils, which was supported by works from Ahmad et al. (2016) and Ghorbani et al. (2024). This may be due to the significant correlation between the concentrations of AsV and phosphate in As-contaminated soils. A considerable amount of PO_4^{3-} induced the competition with As(V) for adsorption sites on biochar and, hence, promoted the desorption from the solid phase, significantly increasing As concentration in soil pore water (Ahmad et al., 2014). Besides P, another essential factor involving functions similar to the reductant of As is DOC; however, there are different underlying mechanisms for their actions. (Kim et al., 2018). Generally, increased DOC content enhances microbial reduction of As (V), hence stimulating the release of As(III) from paddy soil (Cheng et al., 2020). Biochar from oil palm fiber under anoxic conditions has demonstrated similar effects (Qiao et al., 2018). The use of rice straw-derived biochar in paddy soil under anaerobic scenarios raised the population of Fe-reducing bacteria, including Bacillus, Clostridium, and Caloramator, facilitating the reduction of As(V) adsorbed onto amorphous Fe/Al oxides (Wang et al., 2017). Biochar functions as an electron shuttle owing to its elevated alkalinity and aromaticity, akin to humus, facilitating the concurrent microbial reduction of As(V) and Fe(III) (Wang et al., 2017; Qiao et al., 2018). Also, biochar surfaces derived from chicken manure, with their rich functional groups, can induce the reduction of As(V) through the interaction with π-electrons (Choppala et al., 2016). Besides, Fe/Mn oxides on the surface of biochar could adsorb As, reducing its mobility and inhibiting the migration of As into soil solution. These findings highlight that there is diversity in the mechanism of biochar immobilizing As in contaminated soils (Yin et al., 2017; Yu et al., 2016).

3.4.5. Mercury (Hg)

Unlike As, mercury (Hg) exhibits unique toxicity pathways. Mercury is among the most toxic HMs found in soils, and its toxic effects are mitigated by biochar application in contaminated areas. Studies indicate that Hg(II) ions in soil may interact with the surface of biochar derived from hardwood to form complexes ($-COOHg^+$), subsequently limiting Hg mobility via precipitation. Additionally, sulfoxide and thiol functional groups in biochar can react with Hg(II) ions to create S(Hg) and [Hg $(OSR_2)_6^{2+}$] precipitates, respectively (Wang et al., 2019). This process mainly depends on the sulfate concentration in the biochar because it may increase the coordination reaction between Hg and sulfur to ensure the precipitation of the sulfide. For example, biochar derived from rice husk, with higher sulfate than biochar made from wheat straw, showed improved performance due to enhancing Hg and sulfur coordination and precipitation of sulfide (Wang et al., 2018a). Methylmercury (MeHg) is another highly toxic form of mercury that arises from the inorganic transformation of Hg in soil. It is highly toxic and bioaccumulative (Zhang et al., 2010). Rice root exudates in soils could decrease the pH of soils, which increases Hg methylation. Biochar application, due to its alkaline nature, could raise the soil pH and hence restrict this methylation of Hg (Cheng et al., 2020). Additionally, sewage sludge biochar, given its high organic matter content, may also increase activities and enhance the growth of heterotrophic microorganisms responsible for the formation of MeHg in acidic, Hg-polluted farmland soils. This is further not reflected in higher bioaccumulation within rice because the application of biochar has been discovered to reduce the level of MeHg within rice by limiting the Hg bioavailability and hindering its uptake (Zhang et al., 2018).

3.5. HM remediation in plants using biochar

3.5.1. Impact of biochar on plant HM uptake

The absorption capacity of HMs by the apical end, mainly the entire surface of the root, depends on the root's capacity, development, and the metal concerned (Haider et al., 2021; Begum et al., 2019). Root cortical cells first interact with HMs, metals as well as essential elements, taken up via competitive processes through both apoplastic (between cells), and symplastic routes (through cells) (Haider et al., 2021; Tangahu et al., 2011). It facilitates the contaminants' transfer to the root surface by mass flow mechanisms (Tran and Popova, 2013). The increase in negatively charged exchange sites on soil particles due to biochar has resulted in increasing the immobilization of cationic metals through increasing pH in acidic soils by converting them to more alkaline environments (Joseph et al., 2021; Palansooriya et al., 2020; Haider et al., 2022; Murtaza et al., 2022b). Surface properties such as oxygenated functional groups on biochar are crucial for HM immobilization through multiple mechanisms, including physisorption, reduction, anion attraction, electron shuttling, precipitation, cation attraction, and ion exchange (Xu et al., 2019a). For example, a study by Chen et al. (2018) indicates that applying biochar in soil reduces concentrations of HMs such as Cu, Zn, Cd, and Pb in plant tissues by 25 %, 17 %, 38 %, and 39 %, respectively. Other studies demonstrate that under higher application rates of 10 Mg ha^{-1} , similar reductions are observed. Biochars derived from manure with high Ca content were more effective in the immobilization of Cu^{2+} ions by ion exchange compared to plant-derived biochar. Besides, the biochars with high P content can precipitate Pb by forming insoluble compounds such as β-$Pb_9(PO_4)_6$; their alkaline property and high calcite content can favor the precipitation of $Pb_3(CO_3)_2(OH)_2$, which further precipitates Pb (Lei et al., 2019; Li et al., 2016). The C-coated minerals on biochar surfaces were also one of the most important means of reducing the bioavailability of HMs (Kumar et al., 2020). High-temperature pyrolyzed biochars of willow origin improved HM absorption due to heightened chemi-sorption and physio-sorption processes (Bogusz et al., 2019). The bioavailability of Ni, Pb, Co, As, Cr, and Cu is lowered, while that of Cd and Zn is raised in acidic soils by the biochar. Conversely, it may decrease the binding sites of As by changing the pH of the soil, thus increasing the mobility of anionic metalloids because of less positively charged sites (Vithanage et al., 2017). The efficacy of biochar is contingent upon several parameters, including the attributes of the biochar, the chemistry of metals, and soil functions (Li et al., 2022b; Nkoh et al., 2022). For instance, in maize, the incorporation of biochar decreased the bioavailability of Pb(II) by 71 % and exchangeable Pb(II) by 99 %, indicating a substantial reduction in Pb toxicity relative to untreated soils. In rice (Yue et al., 2019), biochar lowered the toxicity of Cd(II), Pb (II), and As in contaminated soils, thereby enhancing the growth of *Salix viminalis* (Lebrun et al., 2020). On the other hand, a %1 tea plant-derived biochar was reported to govern Ni, Mn, As, and Cr mobilities in the mining-affected soil matrix (Pehlivan et al., 2023). By contrast, the amendment of biochar in HM-polluted soils lowers the daily intake of HMs by 12.5 %, the risk of cancer by 30.6 %, and the hazard quotient by 30 % (Nkoh et al., 2022).

3.5.2. Impact of biochar on plant biochemistry under HM stress

Biochar, because of its unique chemical and physical structure in nature, can reduce the accumulation of HMs, which leads to a reduction of overgeneration of ROS. The porous surface and oxygenated functional groups of biochar assist in regulating the soil's water-holding capacity (WHC), enhancing mineral availability for plants. By improving plasma membrane functionality and thus reducing oxidative damage caused by substances such as malondialdehyde (MDA), hydrogen peroxide (H_2O_2), and electrolyte leakage (Khan et al., 2021b), by scavenging ROS or upregulating antioxidant gene expressions.

Various studies show biochar's ability to significantly reduce ROS generation—such as H_2O_2 and the superoxide anion (O_2^-) under HM stress. For instance, studies on Brassica chinensis exposed to Cd (Kamran et al., 2019) and Chenopodium quinoa exposed to Cd (Naeem et al., 2020) have all demonstrated that biochar reduces oxidative stress by limiting ROS generation in response to HM. The reduction in oxidative stress-mediated by biochar affects the integrity of plant cell membranes. When biochar is added to HM-contaminated soils, it alters the soil's physicochemical properties and reduces the phytoavailability of HMs through complexation, which helps prevent membrane degradation (Khan et al., 2022). Biochar, according to Kamran et al. (2019), may reduce Cd's negative impacts and avoid the destruction of cell membranes. Moreover, improved cell membrane quality by Ni sorption was the capability of biochar to delay the deterioration of membranes associated with the reduction in MDA level (Shahbaz et al., 2018). Besides, biochar has also been proven to protect photosystem II and decelerate chlorophyll degradation, hence increasing plant tolerance to metal stress (Khan et al., 2022). More importantly, the literature has identified that biochar protects plant health against metal-added stressors. For example, the amendment of biochar ameliorated Cr toxicity, which had an overall enhancement effect on chlorophyll levels and overall plant growth (Li et al., 2018). Besides improving the ultrastructure of stomata and leaves, along with their density, biochar increased photosynthesis rates in rice plants exposed to nitrate stress. These results have pointed out that biochar helps plants cope with hazardous metals and generally enhances their health (Khan et al., 2021a).

3.5.3. *Impact of biochar on plant defense mechanisms under heavy metal stress*

Stimulation of antioxidant activity, both enzymatic and non-enzymatic, is one of the major plant defense mechanisms against HM toxicity, which counteracts the adverse effects of ROS and balances cell turgor. The binding of radicals to enzyme sites can reduce their activities and alleviate toxic effects (Khan et al., 2022). By helping to balance water and nutrient availability for plants, biochar improves the effectiveness of antioxidant defenses and lowers HM levels (Khan et al., 2022.). In earlier studies, bamboo biochar significantly elevated antioxidant activity (SOD, POD, CAT, and PAL) in Brassica juncea grown in four HM-contaminated soils. Another investigation found that incorporating pistachio shell biochar increased antioxidant levels and enhanced Ni mobility in soil, thereby stimulating plant defenses against oxidative stress generated by HMs. Turan (2019) also showed in Cd-, As-, and Pb-contaminated soils that biochar derived from different feedstocks, including maize cob, rice husk, and peanut shell, enhanced antioxidant activity. However, only rice husk biochar showed a significant increase in antioxidant activity (Ibrahim et al., 2019). This implies that in the manufacturing of biochar, feedstock type is crucial for regulating the antioxidant defense mechanisms of plants. Still, the concentration of biochar might affect its antioxidant activity. Rehman et al. (2019) indicated, for example, that high rates of rice biochar application in Cu-contaminated soils lowered antioxidant levels, most likely owing to poorer Cu absorption by plants or because high biochar concentrations might hinder growth. Furthermore, in plants exposed to reduced Cd and ROS toxicity in biochar-amended soils, antioxidant-related genes like CAT, Mn-SOD, and GR were upregulated. Based on its content and the type of metals involved in the target process, the biochar supplement may reduce HM stress but has different effects (Kang et al., 2022).

3.5.4. *Impact of biochar on plant growth and development under heavy metal stress*

Heavy metal pollution in soil impedes nutrient absorption by plants, thereby restricting their growth and development. However, biochar can mitigate these effects by reducing HM phytoavailability and increasing nutrient availability, thus enhancing plant growth (Ye et al., 2020; Anae et al., 2021). The enhanced nutrient availability, synergistic effects of biochar on soil characteristics, and immobilization of HMs account for the improved performance of crops under HM stress. Biochar significantly improves crop growth and productivity by decreasing bioavailability and enhancing soil fertility. Khan et al. (2022) elucidated that several studies have shown the function of biochar in promoting plant development and its capacity to alleviate toxicity. For instance, studies have demonstrated that biochar improves root and shoot biomass in plants subjected to Cd toxicity (Abbas et al., 2017). Biochar promotes rice growth in As-contaminated soils (Lian et al., 2020a). Beyond its effects in contaminated soils, biochar enhances plant growth, crop yield, and development under both normal and stressed conditions, including salinity, drought, and HM exposure (Naeem et al., 2020; Shahbaz et al., 2018; Zhang et al., 2022b). Application of 2.5–5 % biochar was found to increase physiological parameters of photosynthetic efficiency, transpiration rate, CO_2 exchange rate, and stomatal conductance in plants under Cd stress, thus promoting the growth of *Brassica rapa* (Kamran et al., 2019). These materials affect the accumulation of osmolytes, soluble sugars, and proline. Accordingly, a recent study demonstrated that biochar mitigated osmotic stress in Brassica chinensis exposed to Cu, Cd, and Pb, increasing soluble sugar and proline levels (Xu et al., 2020). Table 5 indicates the impact of biochar on the reduction of HM toxicity and enhancement of plant tolerance in various plant species, key physiological improvements, and comparative effectiveness under different stress conditions, providing a comprehensive overview of its restorative potential in contaminated soils.

3.5.4.1. *Impact of biochar on plant molecular mechanisms under heavy metal stress.*

Beyond physiological effects, biochar modulates molecular responses to HM stress.The regulation of genes related to HM stress in plants can be affected by biochar. For example, one study reported that adding 3 % biochar derived from rice straw, modified with HNO_3, improved rice tolerance to 60 mg L^{-1} of vanadium (V) by stimulating antioxidant capacity. This enhancement occurred via upregulation of genes encoding antioxidant enzymes, including *OsCAT*, *OsPOD*, *OsSOD*, and *OsAPX*. These genes exhibited increased expression levels 5.57-, 5.04-, 4.97-, 5.25-, and 4.80-fold compared to control treatments, thereby improving rice's phytoremediation potential under V stress (Mehmood et al., 2021). For example, Hannan et al. (2021) found in another study that the high surface area of bamboo biochar reduces the damage in Brassica napus exposed to 100 mg kg^{-1} Ni through a reduction in the transcript levels of SOD genes. With the decrease in Ni toxicity thanks to biochar, genes involved in enzymatic antioxidants such as APX, CAT, and GR were upregulated. Biochar induced the expression of iron transport-related gene *BnIRT1* and down-regulated the expression of Ni transporters, including BnNi-T and BnNRAMP3. Thus, these results demonstrate that biochar decreases Ni transport and increases the expression of genes from the *Brassica* genus, which are related to stress tolerance. In agreement, silicon-rich rice husk biochar (RH-300 and RH-700) applied in a study under Cd stress reduced toxicity through the improvement of Cd immobilization in the soil and regulation of the expression of silicon transporter genes such as LSi1 and LSi3, taking part in rice roots in the regulation of Cd uptake and its translocation to plant grains. Si-rich biochar addition considerably reduced Cd uptake in plants by modifying the expressions of the target genes Wang et al. (2020a). However, all studies showed a consistent effect. Under Zn and Cd toxicity in Foxtail millet using corn straw-derived biochar, there were no significant differences observed in the expressions of CAT genes with biochar addition.This work also revealed that raising the pyrolysis temperature of biochar at 300 °C, 400 °C, and 500 °C reduced Mn-SOD and GR gene expression, implying that pyrolyzed conditions, especially temperature, considerably affect the antioxidant gene transcription in plants (Kang et al., 2022).

In some cases, the genetic background of the plants contributes more than biochar towards metal uptake reduction. According to Ma et al. (2021), sometimes the plant's genetics in regulating metal uptake decrease the potential ability of biochar to change metal bioavailability-e.g., of Cd. The role of biochar is important in the plants' molecular genetics in mitigating the stress imparted by the metals. For instance, the study of muskmelon plants under Cd stress revealed that wood-derived biochar down-regulated the expression of genes related to stress tolerance and phenylpropanoid pathways. Biochar also modulated the expression of WRKY transcription factors, the P450 protein family, and annexin genes, which all showed that it mitigates the toxic effects of

Table 5
Impact of different BCs from diverse sources to reduce HM toxicity and promote plant tolerance under different heavy metals in various plant species.

Biomass material	Soil type	Plant species	Heavy metals	Biochare rate	Mechanism	Contributor	References
Fe–Fe-modified biochar from rice husks	alkaline soil	maize (*Zea mays* L)	Cd	0.5 % Fe–modified biochar	Increasing soil organic carbon –Converting of acid soluble and reducible states of Cd into oxidizable and residual form Reduction of Cd accumulation in maize	China	Sun et al., (2021)
maize straw and hickory nutshell	Ferralic Acrisols	rice (*Oryza sativa*)	Cd	15 and 30 t ha^{-1}	Reduced Cd uptake by rice and immobilized Cd in soil Hickory nut shell-derived biochar had more potential to reduce Cd bioavailability than maize straw-derived biochar	China	Zhang et al., (2019)
wood, paper sludge, sewage sludge, and grapevine wood	alkaline soil.	Lolium perenne	Si, Al, Ca and Fe	0, 20 and 40 t ha (−1)	Showed a positive impact on germination rates and soil fertility	Spain	de la Rosa et al., (2014)
peanut shell	red soil (classified as Udic Ferralsols).	*Vetiveria zizanioides*	Cd, Cu, and Pb	Rate of 35 % Biochar	Increasment of contents of different pigments in plants. Reduced MDA and increased SOD, and POD.Reduction of Cu in leaves and roots	China	Ai et al., (2023)
P. orientalis biochar and pig biochar	Low(0.75 % C) and high(3.08 % C) organicarbon	*Brassica chinensis* L	Cd	2 % pig biochar	Increased soil pH, organic carbon content, and available phosphorus content. Reduced DTPA-extractable Cd. Reduced the plant uptake of Cd.	China	Chen et al., (2019)
tea waste	river sediments.	Ramie seedlings	Cd	100, 500, 1000 and 5000 mg kg−1 biochar	Increased accumulation and translocation of Cd in the plants, reduced Cd toxicity in plants and microbes. Increased plant phytoremediation	China	Gong et al. (2019)
rice straw	paddy soil	Rice	Cd, As, and Pb	3 %, w/w	Reduced Cd uptake (38 %) and remarkabl increase in plant growth	China	Wen et al., (2021)
bamboo biochar	Cd-polluted soil	Wheat	Cd	0 %, 0.1 %, 1 %, 5 %	Decreased uptake of Cd by 23.33 % in grains, 21.57 % in straws and 34.06 % in root	China	Ma et al. (2021)
rice straw	sandy clay loam,	wheat	Cd	0 %, 1.5 %, 3.0 % and 5 % w/w	Amilorated Cd toxicity in wheat, Reduced oxidative activity and increased plant antioxidant capacity. Also increases soil pH and reduced metal bioavalability in soil.	Pakistan	Abbas et al., (2017)
bamboo biochar	mining soil	Brassica juncea	Cu, Pb, Zn,and Cd	0 % 2.5 %, 5 %, 7.5 %, and 10 %	Increased soil and plant enzymes, soil physiochemical parameters, soil pH, and EC, plant photosynthesis and finally plant growth and development as well as phytoremediation	Iran	Emamverdiyan et al., (2024c)
concarpus biochar	two soil moisture levels	*maize plants (Zea mays L.).*	Fe, Mn, Zn, Cd, Cu and Pb	0.0, 1.0, 3.0, and 5.0 % w/w	Reduced metal immobilization, increased shoot dry biomass, decreased NH_4OAc- or AB-DTPA-extractable metal levels in the soils, reduced soil metal toxicity	Saudi Arabia	Al-Wabel et al., (2015)
sugarcane bagasse-derived biochar	-	mash bean	Cd, and Cr	15 g kg^{-1} (*w/w* basis)	Reduced Cr and Cd bioavailability by 85 % and 63 %, respectively. Increased soil microbial activities	Pakistan	Bashir et al., (2018)

Cd on the plants (Cheng et al., 2023). The impact of biochar on plant molecular mechanisms is demonstrated in Fig. 3. Which illustrates these molecular mechanisms, highlighting: (1) biochar-induced gene up/downregulation, (2) key pathways (antioxidant synthesis, metal transport), and (3) species-specific responses, providing a visual synthesis of biochar's genomic interactions in metal-stressed plants (Table 6).

3.6. The challenges and risks of using biochar in practical applications

Despite these benefits, practical challenges remain. The application of biochar in the remediation of soil has not only covered efficiency in immobilizing or mobilizing HMs but also needed long-term stability and ecological risk assessment in the soils. Such a factor relates much to its performance, characteristics, and type (Cheng et al., 2020). Biochar has the potential to contain volatile organic compounds (VOCs), as identified by Buss and Mašek (2016), along with a range of toxic metals noted by Xu et al. (2016), polycyclic aromatic hydrocarbons (PAHs) reported by Zielińska and Oleszczuk. (2016); Lyu et al. (2016),and Fadel et al., (2022) dioxins (PCDD/Fs), and other compounds posing environmental hazards. The utilization of biochar for HM remediation typically requires substantial quantities, varying from 1.5 to 72 t ha^{-1} or greater (Cheng et al., 2020). This application may elevate the potential for the release of toxic compounds into the soil, air, or water, thereby posing a risk of secondary pollution and ecological harm. This observation highlights the dual nature of biochar, as it has the potential to both introduce new contaminants into the environment and remediate existing ones. Despite the fact that biochar has been demonstrated to effectively sequester carbon and reduce greenhouse gas emissions (Zhou et al., 2019), there are conflicting reports regarding its effects. Under specific conditions, biochar may increase emissions of CO_2 (Yang et al., 2017), CH_4 (Ribas et al., 2019), and N_2O (Liu et al., 2014), yet these effects may vary depending on the type of biochar used, with some biochars potentially limiting CH_4 emissions but increasing N_2O emissions, or vice versa. Therefore, interaction with greenhouse gas emissions by biochar should be considered when applying biochar to the field since, in summary, it may not result in positive effects in general on greenhouse gases. Biochar could bind with some pesticides, thereby preventing such pesticides from degradation. Conversely, it may stimulate the growth of some beneficial bacteria like Anaeromyxobacter,

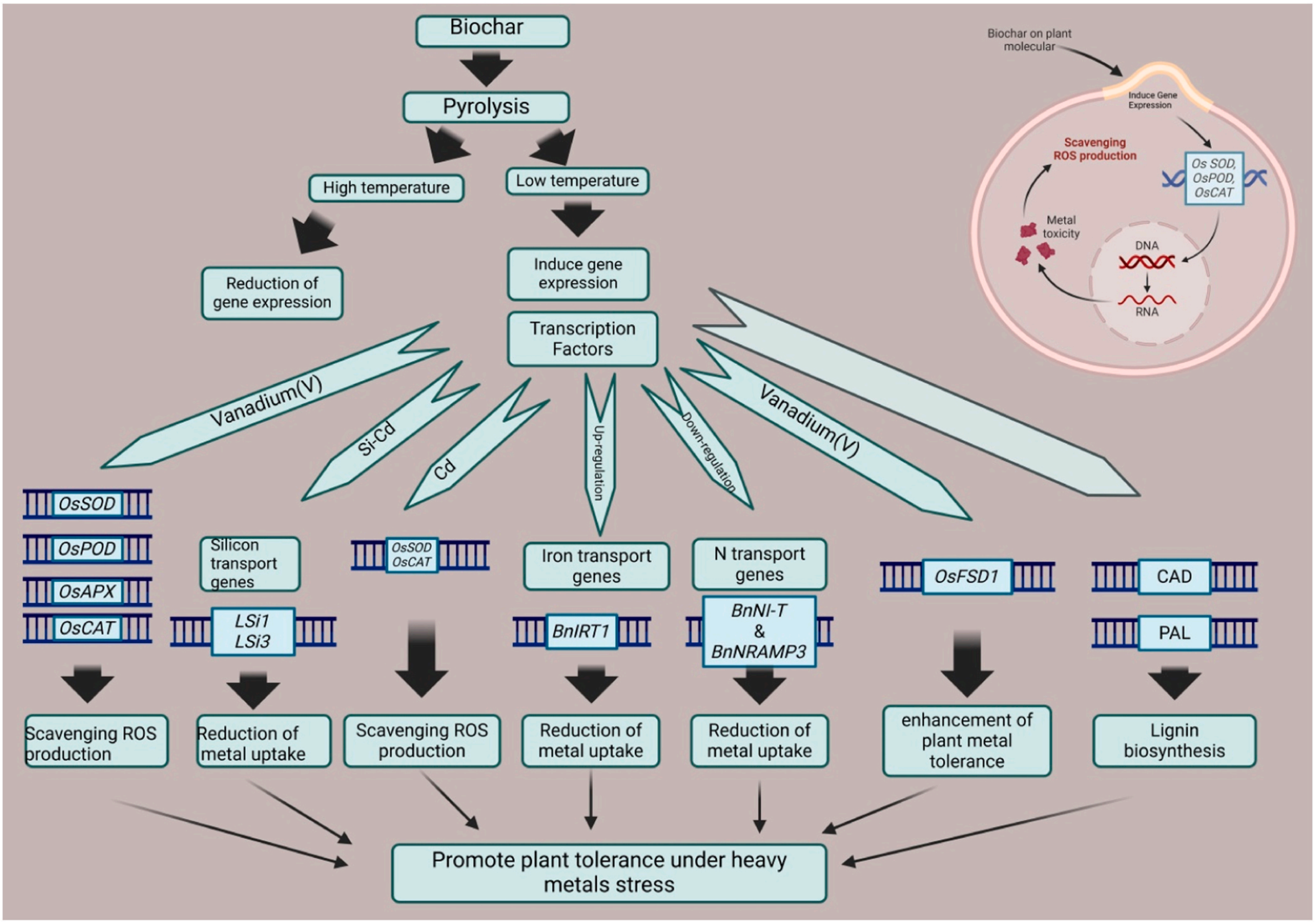

Fig. 3. Diagrammatic illustration of biochar impact on plant molecular mechanisms under HM stress.

Table 6
Gene expression changes (e.g., OsCAT, BnIRT1) under biochar treatment vs. control conditions.

Gene	Biochar-Induced Change	Functional Role	Plant Species	Reference
OsCAT	5.25-fold upregulation	Antioxidant enzyme (H_2O_2 detox)	Rice (*Oryza sativa*)	Mehmood et al. (2021)
BnIRT1	Upregulated	Iron transport (reduces Ni uptake)	*Brassica napus*	Hannan et al. (2021)
LSi1/ LSi3	Downregulated	Silicon/Cd transport regulation	Rice (*Oryza sativa*)	Wang et al. (2020a)

Geobacter, and Clostridium by Wang et al. (2017) and Qiao et al. (2018), which might alter microbial community structure. This could be followed by a decrease in fungal growth, survival, and diversity, as well as other organisms inhabiting the soil, such as acidophilic earthworms (Chen et al., 2013; Anyanwu et al., 2018). On the other hand, biochar may affect pesticide biodegradation in soil. Therefore, while choosing the type of biochar to be used for remediation, along with effectiveness and dosage, the particular conditions of the soil, kind of soil, and plant species become considerable factors. Fig. 4 depicts the mechanisms of HM mitigation by biochar in soil-vegetation systems. So that it illustrates these complex interactions, highlighting biochar's effects on HM immobilization, greenhouse gas emissions, and soil microbial communities in vegetation systems. Despite its advantages, the large-scale application of biochar faces practical challenges, including high production costs, inconsistent feedstock availability, and energy-intensive processing. Additionally, the lack of standardized guidelines for optimal application rates, long-term ecological impacts, and farmer awareness hinders widespread adoption. Addressing these barriers-through policy incentives, technological innovations, and cost-effective supply chains-is critical to unlocking biochar's full potential in sustainable agriculture and environmental remediation. "In conclusion, while biochar demonstrates clear agronomic and environmental benefits, future efforts should prioritize (1) cost-effective production methods using localized waste feedstocks, (2) standardized protocols for soil-specific applications, and (3) long-term ecotoxicological monitoring frameworks. Researchers should focus on developing predictive models for biochar-soil interactions, while policymakers must establish certification systems and farmer education programs to bridge the gap between laboratory potential and field-scale implementation."

4. Nano-biochar

Nano-biochar refers to a modified version of biochar composed of particles sized below micrometers, producing nanoscale dimensions (Jiang et al., 2023). Nano-scale synthesis produced nano-particles with enlarged surface areas which boosted the reactivity level and enhanced nutrient absorption capabilities (Elsawy et al., 2022). This structural refinement potentially mitigates HM stress by leveraging the unique benefits of the nano-scale properties (Elsawy et al., 2022), which could affect HM stress.

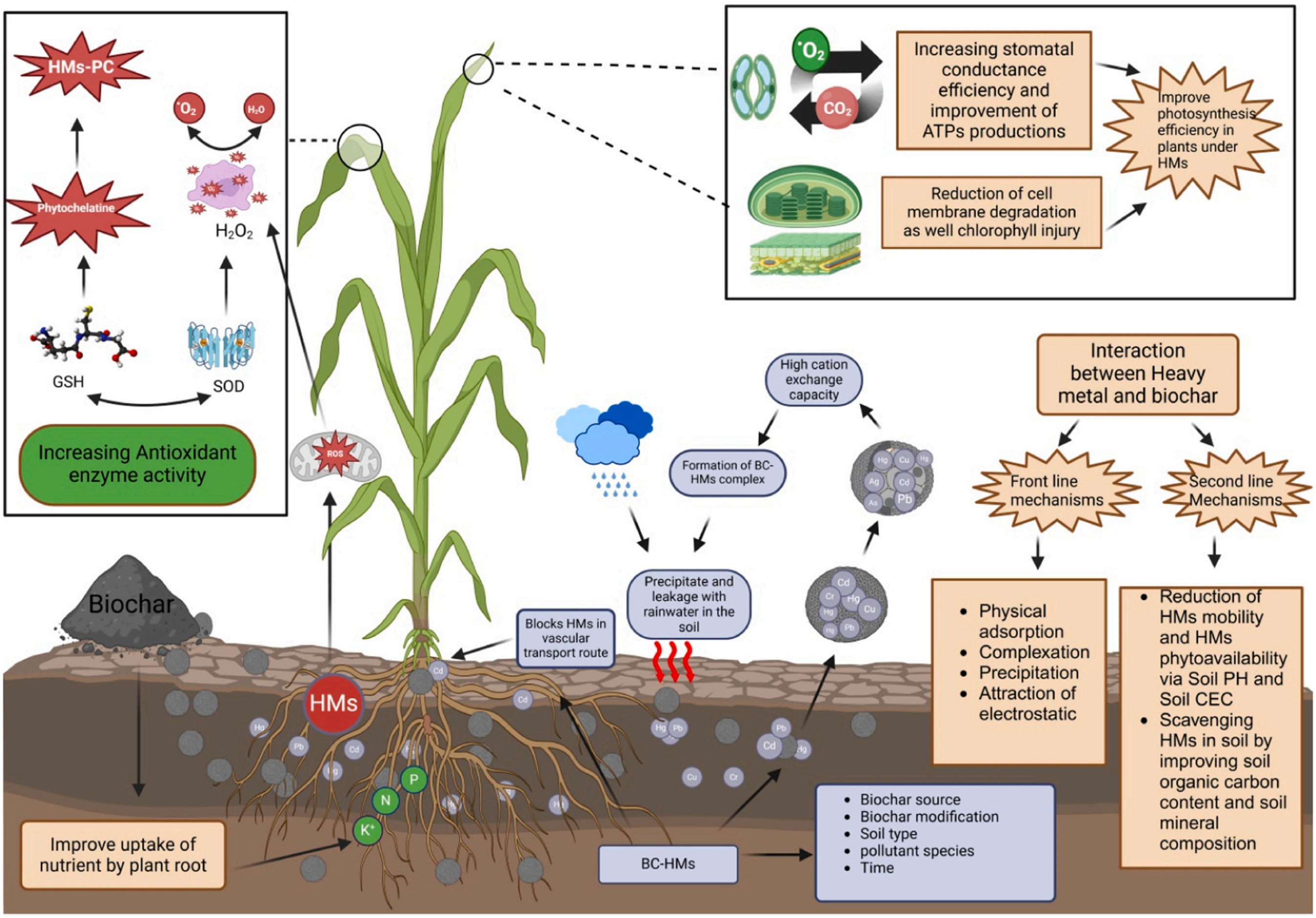

Fig. 4. Diagrammatic illustration of biochar impact on mitigates HMs in the soil and increasing plant tolerance under HM stress).

4.1. Production of nano-biochar

Raw material quality is one of the essential organic factors in the production of nano-biochar, affecting its physiochemical properties as the type of raw material choice is critical. For instance, raw materials containing high lignin led to the derivation of easily aggregated nano-biochars (Föhr et al., 2017). While biomass and raw materials with high hemicellulose produce nano-biochars with high oxygen content and low carbon content (Weber and Quicker, 2018). Oleszczuk et al. (2016) reported that nano-biochars derived from wheat straw and wicker had primary small particles with relatively open-structured while nano-biochars derived from miscanthus had large spherical particles in comparison with wheat straw and wicker, which exhibited weaker aggregation effects. On the other hand, a more direct relationship between the ash content of nano-biochar coming from organic waste and bulk biochar was reported, which had not existed in the generation process of nano-biochar coming from municipal waste (Song et al., 2019). The production of nano-biochar has usually been conducted in the same methods used in producing bulk biochar; however, there are a few additional steps in producing nano-biochar in comparison to traditional biochar (Chausali et al., 2021) (Fig. 5 demonstrates nano-biochar production process). The approach to nano-biochar production can start with two processes: top-down and bottom-up. The first process (top--down) includes the minimization of the macro size of biochar to nano-scale, while in the bottom-up process, the nanomaterial accumulates at the atomic level. The top-down process, which produces nano-biochar at a cost-effective level, uses cutting, grinding, centrifugation, and etching as its steps (Bhandari et al., 2023). The grinder serves as a size reduction method for macro-biochar, according to Dong et al. (2018), Lonappan et al. (2016), and Lian et al. (2020b). Different forms of nano-biochar can be produced through the bottom-up approach after subjecting biochar to disintegration and ball milling and sonication and carbonization processes, which generate unique characteristics of improved bioavailability, smaller dimensions, and enhanced mobilization (Rajput et al., 2022).

The process of ball milling represents a significant technique for nano-biochar design by mechanically grinding biochar to create nanoparticles, as identified by Chausali et al. (2021) and Lyu et al. (2018). During the milling process, biochar forms small, well-defined pores while its particles flatten, which results in increased specific surface area (SSA) (specific surface area) (Ramezanzadeh et al., 2021). Wet milling and dry milling constitute the two main modes of ball milling. Wet milling for twelve hours yields superior, refined biochar particle dimensions compared to dry milling. Additionally, both milling techniques generally produce biochars with larger surface areas than those created through manual grinding (Yuan et al., 2020). Ball milling represents a sustainable, low-cost method that offers reproducibility (Kumar et al., 2020; Naghdi et al., 2017). Vibrating disc milling acts as an alternative technique for manufacturing biochar particles in nanoparticle dimensions, yielding better particle uniformity than ball milling (Huang et al., 2022). In contrast, the double-disc milling technique tends to produce polygonal-shaped particles ranging from 50–150 nm (Ramanayaka et al., 2020a).

Another technique for producing nano-biochar is ultrasonication, wherein a probe generates shockwaves that break apart biochar particles. Sonication treatment causes structural exfoliation of biochar, generating new pores that expand microporosity (Oleszczuk et al., 2016; Sajjadi et al., 2020). The duration of sonication directly affects the

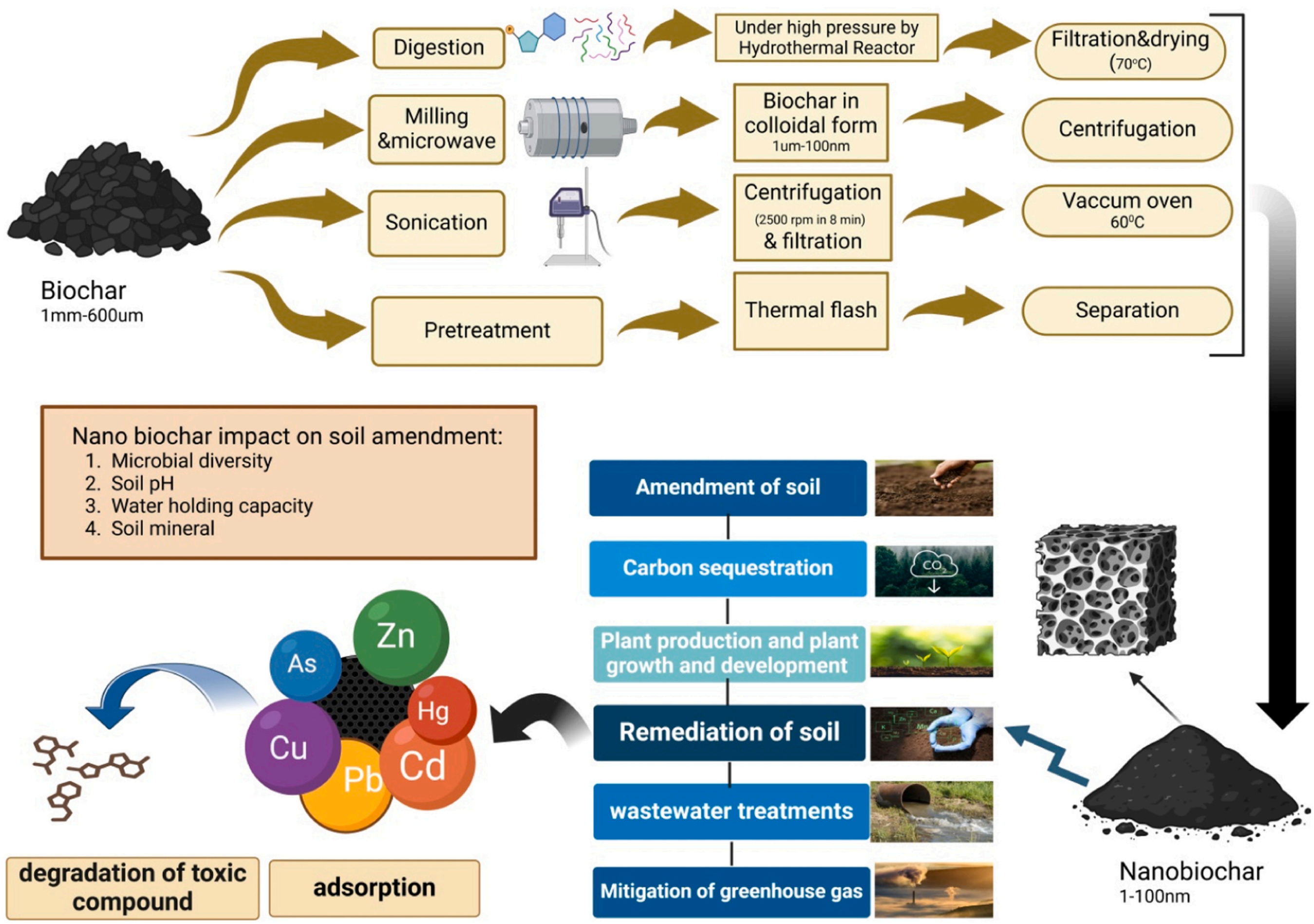

Fig. 5. Diagrammatic illustration of nano-biochar preparation in different methods.

formation of nano-biochar and the size of particles produced. Ultrasonication also assists in releasing fine particles trapped in the pores or on the biochar surface (Liu et al., 2018). For example, Song et al. (2019) achieved particle sizes under 100 nm using ultrasonic methods after treating biochar for 30 min, while Dong et al. (2018) produced nano-biochar particles measuring 42 ± 6 nm via ultrasonic treatment. Centrifugation serves as a direct method for isolating nanoparticles through systematic procedures (Lian et al., 2020b; Xu and Cölfen, 2021). Microwave pyrolysis reactors also represent a single-step economical system for generating nano-biochar (Wallace et al., 2019). Additionally, treating bulk biochar with acids like nitric and sulfuric acid has been reported as a method for producing nano-biochar (Guo et al., 2020). Over recent years, numerous approaches have emerged for modifying nano- and bulk biochar to enhance their structural properties. The transformative techniques for biochar treatment include ultrasonication (Li et al., 2017b), and thiol or acid or base modification (Lyu et al., 2020; Mahmoud et al., 2020a), and metal-assisted modification (Nath et al., 2019; Jenie et al., 2020). The research by Xu et al. (2019b) investigated the effect of ammonium hydroxide treatment during ball milling on biochar products, including pristine biochar along with nitrogen-doped ball-milled biochar and discovered that ammonium hydroxide addition significantly enhanced the SSA and pore volume results. The data indicates that ammonium hydroxide application during ball milling produces supplemental pore structures. A tendency towards surface agglomeration of nitrogen-doped biochar nano-biochar emerged as scientists attributed the cause to hydrophilic groups existing on the particle surfaces. A modified nano-biochar derived from rice husk with iron oxides exhibits more potential for pollutant removal when used together with non-modified nano-biochar, attributable to the enhancement of the C–O and Fe–O functional groups in the nano-biochar, which contribute to its adsorption capacity (Vishnu et al., 2022; Nath et al., 2019; Wang and Wang 2018b). Nano-biochar is synthesized through ball-milling or ultrasonication of bulk biochar (often post-pyrolysis at 400–600°C), reducing particles to < 100 nm while enhancing surface reactivity and porosity. Fig. 5 illustrates the nano-biochar production process, comparing top-down (size reduction via grinding/centrifugation) and bottom-up (atomic-level assembly) approaches while highlighting key steps like ball milling, ultrasonication, and chemical modifications that enhance surface area and functionality for improved environmental applications (the pyrolysis temperature range is, e.g., 300–700°C for biochar production).

4.2. Nano-Biochar Interaction with HMetals

4.2.1. Involved Mechanisms for Nano-Biochar to Cope with Heavy Metal Stress

Nano-biochar's nanoscale structure, tunable surface groups, and redox activity enable sensitive, selective HM biosensing through electrochemical, optical, and field-effect mechanisms. Thus, three distinct interactions occur between nano-biochars and HMs: electrostatic interactions combined with attractive and repulsive mechanisms (Aziz et al., 2023). According to Partheniades (2009), the electrostatic attraction mechanisms form an ionic bond by uniting opposing electrically charged elements. In contrast, nano-biochar is negatively charged; hence, two elements tend to form solid electrostatic bonds, which create a surface complex (Mohapatra et al., 2023; Tan et al., 2020). So, the oxygen-containing functional groups as electron donors in the interaction between the negatively charged surface of nano-biochar and positively charged metals are the major mechanisms of absorption of HMs by nano-biochar (Fan et al., 2020). The presence of these groups (nano-biochar surfaces are rich in oxygen-containing functional groups such as hydroxyl (-OH), carboxyl (-COOH), and carbonyl (C═O) essential for pre-concentration of target analytes and increasing detection sensitivity) contributes to the electrocatalytic properties of nano-biochar, enabling efficient electron transfer during sensing. This has been shown in the positively charged -OH groups with negatively charged Cr (VI) in solutions as an initial mechanism of Cr adsorption (Wang et al., 2020c). This surface complex facility removes HMs from the soil and water (Strawn and Sparks, 2022). On the other hand, because of the presence of –COOH functional groups on the surface of nano-biochar, it can interact with ions like Ni (II), Cu (II), and Cd (II), which can create surface complexes that improve nutrient capacity in the soil, and be limit nutrient leaching (Ding et al., 2016b; Zhang et al., 2022b). A lower pH at the zero-charge point, combined with a higher specific surface area, are crucial characteristics that enhance the adsorption capacity of nano-biochar (Wang et al., 2020d). Various functional groups, including hydroxyl, carboxyl, and phenolic groups, influence adsorption processes in nano-biochar. These functional groups enable electrostatic bonding and hydrogen bonding between nano-biochar functional sites and HM molecules (Bhandari et al., 2023; Strawn and Sparks, 2022). The attraction or repulsion effects, together with the electrostatic force intensity between solution components, depend on pH values. The best adsorption range for nano-biochar occurs at a pH between 1.8 and 9.8, enabling the absorption of methyl blue (Lyu et al., 2018). The incorporation of additional functional groups amplifies the electrostatic effect on adsorption by diminishing the surface potential of nano-biochar when the environment is acidic (Lyu et al., 2018). The potential of agglomeration in nano-biochar is attributed to the composition of the initial amount used as biomass (Ramanayaka et al., 2020b). The impact of low pH on nano-biochar induces a reduction of pore-filling processes as well as the limitation of adsorption capacity. Additionally, in acidic soils, nano-biochar particles with repulsive forces between nano-biochar particles may begin to agglomerate, which can be a result of concomitant reduction of surface charge and the protons of nano-biochar's surface functional groups (Ma et al., 2019). In this context, hetero-aggregation in the soil system has been created by agglomeration between soil minerals and nano-biochar. Because of using a small amount of nano-biochar in the remediation of the soil, heteroaggregation is usually dominant. When metal precipitation occurs at a higher pH, metals, as a form of heteroaggregates, adsorb to nano-biochar and adhere. Heteroaggregation inhibits the properties and migration of nano-biochar. In this context, it is different from soil physicochemical properties and nano-biochar concentrations (Zhang et al., 2022b). Lyu et al. (2018) reported that nano-biochar, because of having a high specific surface area, has superior adsorption performance in comparison with traditional biochar. Redox potential and cation-π interactions are two other adsorption factors that influence the mobility of HMs in the soil (Goswami et al., 2022; Jiang et al., 2023). Nano-biochar helps cation-π interactions form by creating an electric bond between aromatic rings and the positively charged cations that are structural parts of nano-biochar (Jiang et al., 2023; Infield et al., 2021). Aliakbar Tehrani, Kim (2016) explain that this interaction is caused by electrostatic forces between the electrons in aromatic rings and the positive charges of cations. It has been reported that compounds of aromatic molecules, including polycyclic aromatic hydrocarbons (PAHs), are created during the process of carbonization and pyrolysis (Abed Hussein et al., 2022) so that this aromatic ring consists of a delocalized π electron system, which leads to producing a massive electron cloud from above and below the ring plane (Wiedemeier et al., 2015). The attachment relationship between nano-biochar cations operates through π-electrons, which simultaneously draw positively charged cations above and beneath the surface. Even though cation-π interactions normally exhibit less strength than ionic bonds and covalent bonds, they remain effective at anchoring positive ions to biochar surfaces (Bhandari et al., 2023). Ball milling generates graphene structures on surfaces while expanding both external and internal surface areas—previous research concludes that ball milling produces stronger cation-π interactions, resulting in higher Ni absorption (Lyu et al., 2018). The surface of nano-biochar facilitates redox reactions and is instantiated as geoconductors and geobatteries (Sani et al., 2023; Sun et al., 2017). Key redox-active organic carbon components found on the surface include free radicals, phenols, quinones, and N- and S-functional groups, along with graphitic structures (Xu and Tsang, 2022). Interactions between soil redox conditions and organic carbon induce changes in toxic elements such as Mn, Fe, and O_2. A significant mechanism that supports the redox-induced transformation of toxic elements like oxidizing As (III), Cr (III), and Tl (I), and reducing Cr (VI)—involves the production of reactive oxygen species (ROS), as well as transitions between Mn (III)/Mn (II) and Fe (II) (Xu and Tsang, 2022). This reduction transforms toxic Cr (VI) into the less toxic Cr (III) and improves metal mobility in the soil (Xu and Tsang, 2022). The redox capability of organic carbon can be activated by utilizing biochar as a catalyst to promote electron transfer between toxic elements and organic carbon (Fei et al., 2022).

4.2.2. Amelioration of Heavy Metal Toxicity in Soil and Aqueous Systems Using Nano-Biochar

Nano-biochar has a strong potential to immobilize HMs in soil and the environment. For example, much research has reported that nano-biochar can absorb HMs such as Cu, Pb, and Cd from soil and aqueous systems. This removal potential is dependent on the rate of removal, which means that with increasing nano-biochar concentration, the removal rate of HMs gradually increases. On the other hand, other factors, such as adsorbent exchangeable sites, can be involved in the enhancement of the rate of removal (Vishnu et al., 2022; Li et al., 2017b; Yue et al., 2019). It's important to note, however, that overly concentrated biochar can adversely impact HM immobilization capability, particularly due to agglomeration effects, especially under acidic conditions (Rasaki et al., 2019; Mahmoud et al., 2019) While biochar effectively immobilizes HMs, studies indicate optimal dosage ranges of 2–5 % (w/w) for Cd and 3–8 % (w/w) for Pb in soils. The excessive concentrations (>10 %) may reduce efficacy by altering soil pH or pore structure. For instance, Lyu et al. (2018) observed that the adsorption rate of Ni decreased with higher rates of nano-biochar, while lower rates enhanced adsorption in soil. Furthermore, hetero-aggregation between nano-biochar and minerals may occur in soil systems. Excessive nano-biochar concentration promotes electrostatic interactions between positively charged soil minerals and negatively charged nano-biochar surfaces, enhancing heteroaggregation in soil systems (Zhang et al., 2022b). Therefore, it is vital to explore solutions for mitigating nano-biochar agglomeration and restricting hetero-aggregation in soil and the environment to enhance its absorptive capacity. Modified nano-biochar has been shown to exhibit higher adsorption of Cr (VI) than pristine nano-biochar (Wang et al., 2020d). Utilizing modified nano-biochar instead of pristine variants could enhance nano-biochar's efficiency in absorbing HMs in soil, which has also been observed in aqueous solutions. In a study conducted by Shabelskaya et al. (2022), a modified nano-biochar was synthesized using nanocomposite materials based on $CoFe_2O_4$, facilitating the removal of Cr (VI) compounds from aqueous solutions. Additionally, the removal of Cr (VI) and Cd (II) from aqueous mediums by employing graphitic nano-biochar derived from *Gliricidia sepium* was reported by Ramanayaka et al. (2020a). This material, with 28 m^2 g^{-1} SSA and surface chemistry akin to $-CH_3$, –OH, C═O, and –NH, demonstrated an adsorption capacity of 22 mg g^{-1} for Cd (II) at pH 9 and 7.46 mg g^{-1} for Cr (VI) at pH 4. However, employing modified nano-biochar in the environment may pose risks as hazardous substances, including HMs, could leach from its surface, penetrate soil, and cause environmental damage. Thus, the toxicity impact of using modified nano-biochar needs further investigation. Table 7 outlines the effects of nano-biochar on HM removal in soil.

Numerous studies have proven the effects of nano-biochar on

Table 7
Effect of different nano-biochars to reduce HM with emphasis on involved mechanisms.

NBC Resource	Pyrolysis condition	HMs	Soil structure	Rate of NBC	Mechanism	References
wood	650 °C	Cd	sandy soil	2 % (w/w) nano-biochar	The sorption of Cd was 400 % more than the control without nano-biochar. That was appromixaly1062.4 mg·kg^{-1}	Ramezanzadeh et al. (2021)
Ramie straw	700 °C for 2 h	Cd	clay mineral-coated quartz sand	-	Nano-biochar absorbs 40.65 mg·g−1. Cd^{2+} in the soil	Zhou et al. (2022)
corncob	-	Pb	shallot cropping	Biochar+compost (1:4)	Reduction of Pb bioavailability in soil	Purbalisa et al. (2021)
Bark chips	600 °C	Cu, Pb and, Zn	Acid Soil	10 % w/w	Reduction of metal in soil leachate. The adsorption capacity of the nano-biochar for Cu, Pb, and Zn was 121.5, 336, and 134.6 mg g−1, respectively	Arabyarmohammadi et al. (2018)
Wheat straw	600 °C	Pb(II)	-	0.2 g/L, 0.4 g/L, 0.6 g/L, 0.8 g/L, and 1.0 g/L	The absorption rate of Pb was 134.68 mg.g^{-1} The involved mechanisms include precipitation and ion-exchange	Cao et al., (2019)
Corn straw	700 °C, ball milling for 12 h	Cd, Pb	alkaline soil	2 % Nano-biochar	Adsorption rate for Cd and Pb was 8.7126 mg.g^{-1}, and 126 mg.g^{-1}	Zhang et al. (2022a)
Rice-hull	400, and 600 °C	Cd	-	200 mL suspension of nano-biochar	Nano-biochar has a high adsorption affinity for Cd, which significantly reduces the Cd uptake and phytotoxicity	Yue et al. (2019)
Wheat straw	350–550 °C	Cd	mixed with Cd (NO3)2	0 %, 0.2 %, 0.5 % and 1 % nano-biochar	The application of wheat straw nano-biochar reduced the available Cd in the soil	Liu et al. (2020)
Corn straw	700 °C,	Cd, Pb	alkaline soil	5 % red P (BPBC700)	The adsorption capacity of nano-biochar was 18.7 mg g^{-1} for Cd and 126.0 mg g^{-1} for Pb	Zhang et al. (2022a)

solutions. A study demonstrated the elimination of As, a significant concern in drinking water, from aqueous solutions with the use of nano biochar extracted from rice husk and covered with iron oxide, produced via chemical pyrolysis (Nath et al., 2019). The reduction leads to converting toxic Cr (VI) into less toxic Cr (III) and improves the chance of metal mobility in the soil (Xu and Tsang, 2022). The redox ability of organic carbon can be activated through the use of biochar as a catalyst that advances electron transfer between toxic elements and organic carbon (Fei et al., 2022). Hg^{2+} adsorption may arise by mechanisms including the creation of Hg-Cπ bonds, surface complexion, and electrostatic attractions. A study by Wang et al. (2020d) indicated that a complex between charged Cr^{6+} and charged –OH groups in nano-biochar facilitated Cr removal from aquatic environments. These show that nano-biochars can effectively eliminate HMs from soil and aquatic environments, and the SSA of nano-biochar and the availability of adsorbent exchangeable sites are crucial for HM adsorption. Table 7 summarizes nano-biochar's efficiency in HM removal from soil and aqueous systems, comparing adsorption capacities for various metals (Cr, Cd, As, Hg), highlighting key factors like surface chemistry (CH3, –OH, C═O), pH-dependent performance, and the superior effectiveness of modified nano-biochars (e.g., Fe-oxide coated) despite potential leaching risks. Surface functional groups like hydroxyl (–OH) and carbonyl (C═O) enhance HM adsorption via electrostatic attraction, complexation, or ion exchange, while methyl (CH3) groups may influence hydrophobicity and contaminant affinity. Modifying these groups optimizes nano-biochar's selectivity and capacity for target metals.

4.2.3. Amelioration of Heavy Metal Toxicity in Plant Roots by Nano-Biochar

The efficiency of nano-biochar in remediation depends heavily on root exudates, which serve as key factors in explaining the mechanisms behind its HM reduction processes in plants. Plant roots exude various substances that produce leaks into plant growth soil spaces. These include higher molecular weight soluble organic secretions, like mucilage and exoenzymes, as well as low molecular weight secretions, such as organic acids and phenols (Bais et al., 2006). They significantly influence the physicochemical characteristics of the rhizosphere, hence impacting the bioavailability of soil HMs (Layet et al., 2017; Medyńska-Juraszek et al., 2020). Root exudates have been shown to induce nanoparticle-soil heteroaggregation (Ma et al., 2018), while organic compounds like malic, acetic, and citric acid exudates degrade the biochar matrix (Ma et al., 2018). Even so, the large SSA of nano-biochars plays a vital role in their adsorption capacity. When heteroaggregation or agglomeration occurs, it leads to reduced SSA for nano-biochar, subsequently diminishing its capacity to immobilize HMs. As root-released organic acids promote nano-biochar disaggregation, they also facilitate higher HM immobilization. The three process types that occur in nano-biochar are heteroaggregation, agglomeration, and adsorption, which mainly stem from the physical and chemical properties. Additional research focusing on root exudate mechanisms of HM adsorption must be conducted to advance decision-making and sound environmental management practices. The root exudates of Sedum alfredii create complexes with Cd, Zn, and Pb while releasing phytosiderophores, which substantially affect Fe bioavailability (Khan et al., 2016). Biochar serves as an exceptional Fe source that enhances phytosiderophores secretion by plant roots, according to research (Medyńska-Juraszek et al., 2020). Phytosiderophores play a key role in regulating plant Fe requirements and also influence the bioavailability of other HMs in the rhizosphere, thereby protecting plants from toxicity (Gupta and Singh, 2017). Consequently, investigating the role of exudates released from roots in metal speciation to mitigate toxicity in the soil using nano-biochar is crucial. For instance, researchers have proven that Pb ions form complexes with oxalic, acetic, or fumaric acid ligands from root exudates containing polysaccharides, which changes Pb mobility and/or bioavailability rates (Li et al., 2019b; Ghori et al., 2019). Cadmium, as one of the super toxic elements after entering the plant cell, leads to substituting elements such as Zn, Ca, and Fe, which impact plant growth, as well as Cd, which increases with the generation of free ions, leads to oxidative stress in plants with Fenton reactions (Amari et al., 2017; Loix et al., 2017). Nano-biochar reduces Cd solubility in soils to stabilize the chemical composition of Cd ions throughout the soil structure. When nano-biochar is added to soil, it increases the number of microorganisms and the types of microbial communities that live in it. It also helps beneficial bacteria like Actinobacteria and Bacteroidetes grow, which helps the soil rehabilitate. In a study by Kamran et al. (2019), *Brassica chinensis* grown, the results showed that nano-biochar remarkably reduced the levels of Cd^{2+} in the shoot and roots by 86.5 % and 95 %. Roots exhibited considerable variation in apoplastic pH during the absorption of minerals. Conversely, the absorption of NH_4^+ creates intense soil acidification, but NO3− absorption results in alkaline conditions. Active nitrates transport in roots enhances respiration rates that decrease redox potential, which promotes shoot-level oxidation processes, according to Husson et al. (2021). Tests have demonstrated that nano-biochar improves microbial abundance in soil, thus elevating soil nitrogen content and increasing biological activity. Its graphitic structures enable the material to serve as a conductor of electrons between organic acids and Cr (VI), which reduces Cr content. In the electron transfer process, minerals involved in redox reactions, as well as surface functionalities, undergo valence changes (Xu et al., 2021). These interactions enhance the uptake of essential plant nutrients, including PO_4^{3-}, NO_3-, and $K+$ (Masclaux-Daubresse et al., 2010; Elbehiry et al., 2022). Plants express particular genes from MATE (maize *ZmMATE1*) and ALMT (soybean *GmALMT1*) family genes when exposed to HM toxicity, which leads to the secretion of organic acids, including malic acid and citric acid. The compounds act to create chemical complexes with HMs while shaping how these HMs become accessible for the plants (Liang et al., 2013, 2011). Research shows that nano-biochar increases the release of root exudates from plants, according to Akhter et al. (2015) and Bais et al. (2006). Critical assessment of root exudate regulation by nano-biochar requires investigation into the genes responsible for this process. Fig. 6 illustrates nano-biochar's dual role in HM remediation - showing how root exudates (organic acids, phytosiderophores) interact with nanoparticles to alter metal bioavailability, while Tables 8 and 9 systematically compares the key mechanisms (adsorption, microbial stimulation, electron transfer) through which nano-biochar enhances plant HM tolerance across different species."

4.2.4. Challenges of Nano-Biochar Toxicity and Limitations in Nano-Biochar Production

Although nano-biochar can remediate the environment and soil, due to limitations and potential risks, more research can be used in large-scale field applications, the understanding of the long-term impact on the environment and soil, and water ecosystems as well as permissible limits of toxicity in using of nano-biochar is much important. For nano-biochar to be used sustainably and safely without unintended harm, proper instructions and rules must be in place. When nano-biochar is applied in agriculture through its strong adsorption properties, it may cause essential nutrients to bind with contaminants, thus leading to soil degradation and ecological compromise (Sani et. al, 2023). This can result in the leaching of nutrients into other soil layers, reducing nutrient retention and groundwater quality (Xiang et al., 2021). Furthermore, improper biochar combustion may introduce metal impurities, which can negatively affect soil nutrient balance and interfere with the cation exchange capacity (Sani et al., 2023). Research efforts concentrate on identifying the response of HMs after they attach to nano-biochar and then relocate from their original positions. Additional research needs to evaluate possible toxic effects on plant roots that occur when nano-biochar is placed in their vicinity. Regular soil and water system monitoring plays an essential role in assessing any possible harmful outcomes linked to nano-biochar usage. The high mobility nature of nano-biochar within the soil and water environment leads to potential site transport, which heightens the dangers of contaminant leakage and secondary environmental pollutants (Feng et al., 2022). The interaction of nano-biochar with soil and plants at its microscopic level presents unknown health risks that should be thoroughly studied to detect environmental safety concerns. Plants easily take in nano-biochar, which becomes accessible in water before entering dietary food chains at different trophic levels, which potentially produces harmful effects on organisms (Bhandari et al., 2023). Exposure to diets containing

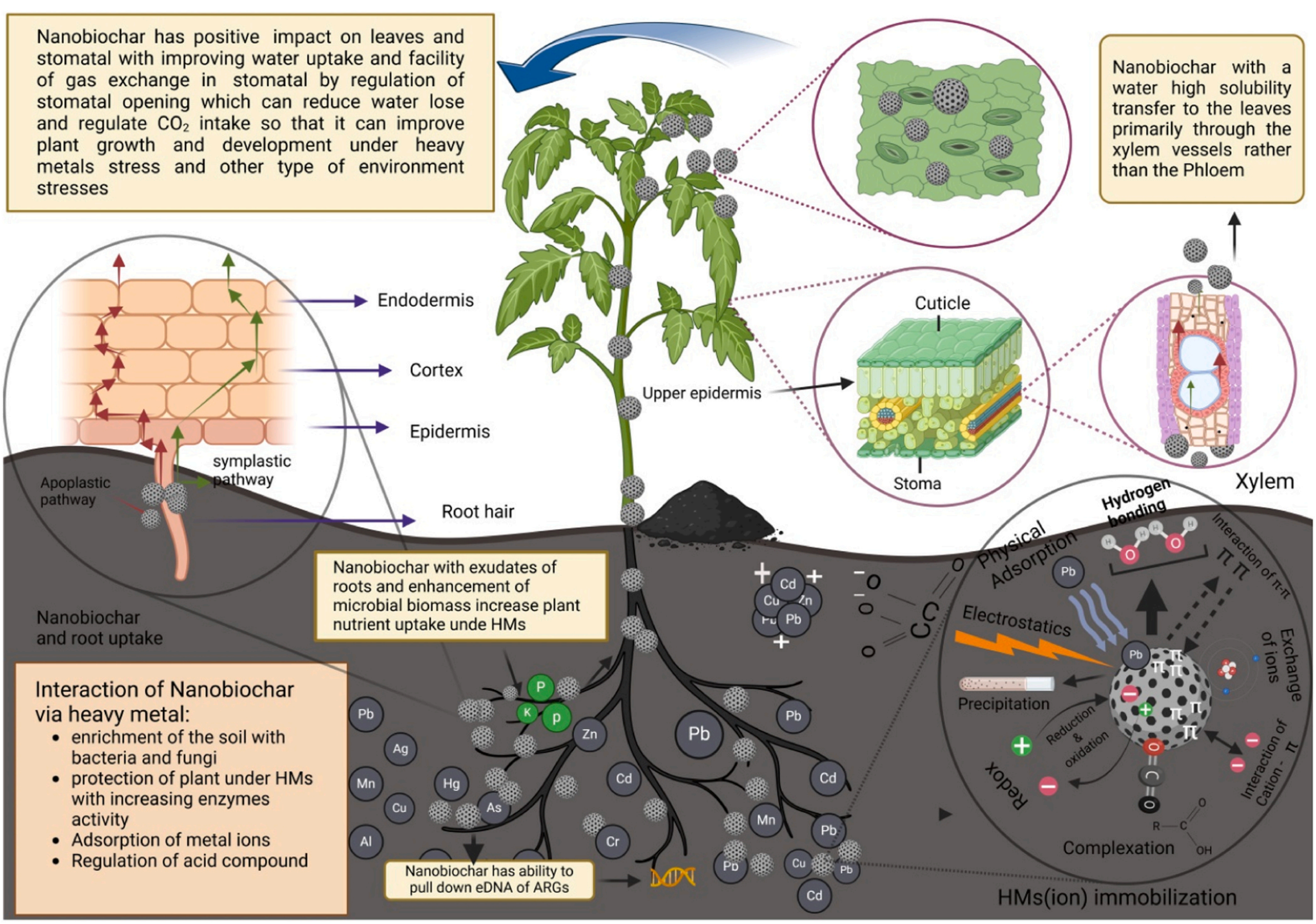

Fig. 6. Involved mechanisms of nano-biochar in ameliorated heavy metal stress in plants.

Table 8
Impact of different nano -biochar from diverse sources to reduce HM toxicity and promote plant tolerance under different heavy metals in various plant species.

Plant	Resource of NBC	Temperature condition	Rate of NBC	HMs	performance	References
Oryza sativa L.	Rice-hull BC	300–600 °C 10,000 rpm for 30 min	200 mL	Cd^{2+}	High temperature has more efficiency in amelioration of Cd toxicity in plants -Impact on the net flux of Cd^{2+} in various root tips zones -Induce oxidative stress	Yue et al., (2019)
Brassica chinensis L	wheat straw	350–550 °C	0, 50, 100, 200, 500 mg /L	Cd	Nano-biochar significantly increases microorganism abundance, microorganism diversity, and microbial biomass of Actinobacteria and Bacteroidetes in soil contaminated with Cd and decreases Proteobacteria diversity	Liu et al., (2020)
pak choi (*Brassica chinensis* L.)	wheat straw	300 °C for 2 h in an N2 atmosphere	100 mg/ L	Cd, As	Reduce the HMs concentrations in plants by an average of > 10 % Reduce HMs uptake of pak choi by changing soil pore environment.	Ouyang et al., (2021)
rice, tomato, and reed seedlings	rice straw, and wood sawdust	300 °C (low-temperature), 500 °C (mid-temperature) 700 °C (high-temperature).	1:50, w/w	-	Increasing the length of root/shoot. Induces rice growth and shows an inhibitory effect on reed growth	Zhang et al., (2020)
wheat	wheat straw-biochar	600 °C - 2 °C/min and a 100 mL/min nitrogen flow rate for 1 h	0.2 g/L	Pb (II)	Enhancement of Pb(II) adsorption on WS-CK Precipitation of Pb(II) Ion exchange of Pb(II) Competition between Precipitation and Ion exchange	Cao et al., (2019)

Table 9
Comparative Performance of Biochar vs. Nano-Biochar.

Property	Nano-Biochar	Biochar
Surface Area	300–1500 m^2/g (varies with production method)	Typically 50–300 m^2/g
Adsorption Capacity	High due to increased surface area and reactivity	Moderate (dependent on feedstock and process)
Stability in Soil	Potentially less stable due to nanoparticle mobility	Highly stable, long-lasting effects
Production Method	Ball milling, ultrasonication, chemical modification (more resource-intensive)	Pyrolysis (simple, low-tech)
Environmental Risks	Potential leaching of nanoparticles and unknown ecological impacts	Lower risk of releasing toxins
Effectiveness in HM Remediation	More effective for low-concentration and complex contaminants	Effective for moderate to high HM levels
Cost	Higher (potentially 10–50 USD/kg due to processing)	Generally lower (1.5–10 USD/kg)

nano-biochar supplements triggered liver damage when researchers analyzed the specimens. Mice exposed to eating nano-biochar developed both elevated oxidative stress markers together with abnormal tissue modifications as well as disturbances in microbiome metabolite production and species diversity, leading to adverse health results (Li et al., 2023). Excessive exposure to 50 % nano-biochar diluted the soil microbial community while decreasing its biomass carbon and nitrogen content and blocking the absorption of soil potassium in corn Rashid et al. (2023). Research needs to determine the potentially toxic effects of biochar so that its future usage can be studied adequately. In addition, variations in temperature, precipitation, and microbial activity may influence the properties of nano-biochar applied in the field, potentially causing transformations like erosion, fragmentation, and oxidation (Wang et al., 2020e). The widespread use of biochar to extract HMs could replace existing native soil remediation techniques, such as phytoremediation. It is important to note that the use of nano-biochar in biosensing entails significant regulatory hurdles. This is because standard protocols are required to assess the safety, efficacy, and environmental influence of nano-biochar, particularly on the potential toxicological consequences on human health and the environment. As regulators seek to address concerns about nanoparticle behavior in environmental contexts, establishing clear guidelines will be crucial for the wider adoption and application of nano-biochar-based sensing technologies. It is crucial to emphasize that the deployment of nano-biochar faces regulatory challenges primarily related to safety assessments and the establishment of acceptable use guidelines. The unique properties of nano-biochar need a thorough evaluation to determine its possible risks in terms of its environmental fate, human exposure, and long-term effects on water and soil ecosystems. Moreover, the lack of uniform testing standards holds back the regulatory framework, prolonging the approval process for its widespread adoption in biosensing applications and environmental remediation systems. Building robust regulatory guidelines that address these concerns will be essential to facilitating the safe and effective integration of nano-biochar into agriculture and environmental applications. Consequently, comprehensive research into the risks and environmental effects of nano-biochar is paramount before its adoption as a solution for soil, water, and ecosystem contamination in extensive agricultural practices. There is a pressing need for further investigation to ascertain the broader impact of nano-biochar on ecosystem dynamics, water quality, and soil health through prolonged application. Current evidence is insufficient to demonstrate how nano-biochar interacts with soil microbes or whether it can maintain its efficacy.

4.3. *Biochar and nano-biochar interaction in soil under heavy metal stress conditions*

The soil structure benefits from biochar just as it does from nano-biochar through aggregation enhancement while reducing soil compaction. Forms of biochar combined with nano-biochar make the soil more porous by expanding its spaces while they enhance stable aggregate formation (Hossain et al., 2020). Biochar along with nano-biochar improve both the structure of soils and root development within long-lasting soil aggregates according to Wu et al. (2023) and Ghassemi-Golezani and Farhangi-Abriz (2021). The drainage efficiency improves when biochar application reduces metal ion growth and root area waterlogging (Du et al., 2016; Wu et al., 2023). Furthermore, mixing biochar and nano-biochar enhances water transfer capacity, as demonstrated by mixing biochar and nano-biochar enhances water transfer capacity, as demonstrated by Mukherjee et al. (2022), due to increased soil permeability. Both biochar and nano-biochar exhibit superior pore structures that facilitate better soil water flow and prevent the formation of harmful substances within the earth (Hasnain et al., 2023). Consequently, these materials are crucial for absorbing retained ions within the soil, including Cl^-, Na^+, and other non-essential HMs (Xiao and Meng, 2020). Mehmood et al., (2023a); Mehmood et al., (2023b); Mehmood et al., (2023c) in their work showed that the porous structure, increased surface area, and negatively charged sites on the surface promote the attachment of positively charged ions in HMs, thereby hindering the uptake of these ions by plant roots (Hasnain as al., 2023). Consequently, the ion adsorption ability of biochar and nano-biochar minimizes non-essential ions in the soil and mitigates the deleterious effects of HMs on plants (Tao et al., 2023). Conversely, biochar and nano-biochar, based on their characteristics, result in the formation of acidic and alkaline soils. When these materials are derived from feedstock with elevated ash content, it raises soil pH, resulting in alkaline soil; conversely, biochar produced from acidic feedstock lowers soil pH, creating acidic soils (Mukherjee et al., 2022; Suleymanov et al., 2023). So that the pH of the soil can be balanced and buffered by biochar. This enhances the availability of plant nutrients and their absorption by plants, as well as soils contaminated with HMs; this feature can aid in immobilizing metal toxicity (Sultan et al., 2020). The primary function of biochar and nano-biochar in the remediation pertains to their capacity to adsorb ions, control metal leaching, modify pH levels, enhance soil structure, and improve electrical conductivity (Chaganti et al., 2015; Lateef et al., 2019). These functions illustrate that both biochar type can modify soil, hence promoting a more conducive environment for plant growth and lessening the detrimental effects on agricultural crop yields.

4.4. Tolerance response in heavy metal-stressed plants by biochar and nano-biochar

The capacity of nano-biochar and biochar to promote root development serves as a main mechanism for metal stress alleviation. Nano-biochar and biochar elicit better plant growth by expanding root surfaces which increases nutrient acquisition potential even in the presence of toxic substances (Ghassemi-Golezani et al., 2021; Wei et al., 2019). The higher cation exchange capacity (CEC) of nano-biochar along with biochar prevents essential nutrients such as N, P and K from leaching because they stay close to plant roots (Yuan et al., 2023). Both materials enhance soil nutrient accessibility by improving its chemical properties (Jeffery et al., 2017) while Soothar et al. (2021) demonstrated their nutrient stress reduction capacity. Another advantage is that biochar and nano-biochar also support the microbial communities that aid in nutrient cycling and availability (Jeffery et al., 2017; Lehmann and Joseph, 2015). Furthermore, biochar and nano-biochar play a critical role in maintaining osmotic balance and ion regulation within plants exposed to metal stress. They maintain the right levels of body ions by both blocking unwanted sodium absorption while enhancing potassium uptake that HMs typically disturb (Yan et al., 2021). The porous structure of biochar and nano-biochar allows them to store water in the root zone thereby decreasing osmotic stress in plants (Al Hinai et al., 2023; Tuyishimire et al., 2022). This water retention happens because of their porous nature which improves water-holding capacity (Fayez and Bazaid, 2014). The retained water in the system upholds turgor pressure for better cell functions while it minimizes osmotic stress that affects cells at a molecular level (Farouk and Al-Huqail, 2022). Macrodefensins gain strength through enzymatic activity stimulation, which strengthens plant defense systems. Plants exposed to HMs generate ROS, causing oxidative stress due to tissue metal accumulation (Ghorbani et al., 2024, 2024; Emamverdian et al., 2024 a,b,c,d). Research indicates that both types of biochar independently activate plant antioxidant defenses, enhancing enzyme activity of superoxide dismutase (SOD), peroxidase (POD), and catalase (CAT) to neutralize ROS and protect plant cells from damage (Emamverdian et al., 2024a,b,c; Sultan et al., 2024, 2025; Wu et al., 2023; El-Sharkawy et al., 2022). This could assist in maintaining an active antioxidant system and, by diminishing oxidative stress, enhance plant tolerance to HM stress. Conversely, enhancing antioxidant activity with biochar and nano-biochar defends plant membranes from higher lipoperoxidation induced by HMs (Gill and Tuteja, 2010; Kamran et al., 2020), hence conserving membrane integrity and plant viability. Photosynthesis efficiency enhancement serves as a vital method through which biochar and nano-biochar improve HM stress tolerance in plants. Research indicates that biochar applications as well as nano-biochar applications boost photosynthetic efficiency in plants. Biochar application enhances both the availability of nitrogen which serves as a fundamental factor for chlorophyll synthesis (Zhu et al., 2019). The specific interactions involving biochar and nano-biochar vary based on plant species; some plants exhibit superior growth in response to nano-biochar, while others respond more favorably to traditional biochar, rendering the optimal choice dependent on the target crop specifications. Ultimately, biochar and nano-biochar enhance nutrient uptake and biomass production during metal stress (Jabborova et al., 2022; Zeng et al., 2022), contributing to increased crop yield, quality, and productivity. Fig. 7 illustrates the beneficial effects on plants under HM stress conditions. As well Fig. 7 illustrates how nano-biochar and biochar enhance plant resilience to HM stress through four key mechanisms: (1) improved root development and nutrient uptake, (2) enhanced water retention and osmotic balance, (3) activation of antioxidant defenses (SOD, POD, CAT), and (4) protection of photosynthetic efficiency by preserving chlorophyll and stomatal function."

4.5. Nano-biochar and biochar: which one demonstrates greater efficiency in reducing heavy metal toxicity?

While both standard and nano-biochar reduce HM toxicity, their efficacy depends on cost, mobility, and environmental stability. The efficiency of biochar depends on its specialized composition which includes the selected HM type and environmental conditions along with application protocols. The article shows the performance differences between these two biochar types. One of the most critical considerations when employing environmental remediation materials is the cost-effective choice. Biochar is an economical and readily available technology that may be rapidly created via multiple biomass sources, like agricultural waste and residual materials. Conversely, nano-biochar is an emerging production technology that necessitates supplementary processing stages and incurs higher costs relative to typical biochar products. The choice of biochar, produced through environmentally friendly and basic low-tech methods such as pyrolysis, promotes environmental sustainability far more effectively than nano-biochar, which requires specialized equipment and extra energy. Biochar use may reduce potential ecological risks. Because nano-biochar's smaller particle size enhances mobility but may compromise long-term stability, and shows greater potential to degrade through time or endure single-compound movement, thus decreasing its longevity performance. Traditional biochar maintains better ecological stability, which enables long-term retention of HMs. Nano-biochar shows better efficacy than

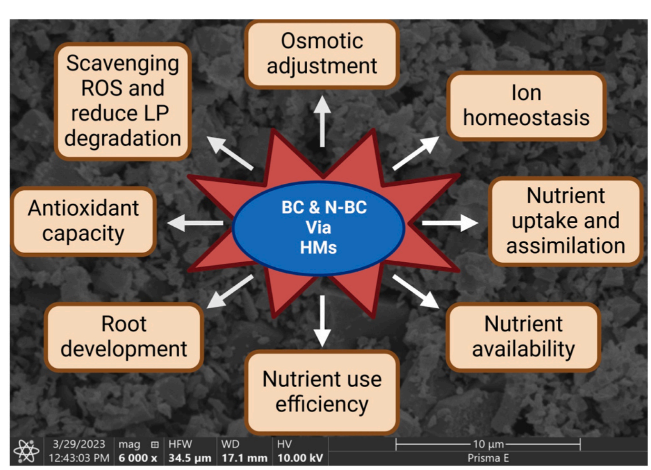

Fig. 7. The interaction between biochar and Nano-biochar with HMs in plants.

conventional biochar for hazardous metal removal because its small dimensions improve its interaction with soil ions through increased surface area and better-retained nutrient capacity. Thus, the particles contribute to ion adsorption and storage capability. In plants, this trait may improve nutrient availability by increasing nutrient retention and release. Because of nano-biochar's tiny size, it can spread more easily through soil or water and get into contaminated sites more easily where traditional biochar might not. This enhances its efficacy in varied contexts. The higher particle size of biochar can restrict its mobility and dispersion, hence compromising its effectiveness in certain situations. Under HM stress, plant growth is influenced differently by biochar and nano-biochar; certain plants exhibit superior yields with nano-biochar, while others respond more favorably to traditional biochar. Therefore, the optimal choice is dependent upon the specifications of the intended crop. Despite growing research, field-scale data remain limited, and there is an inadequate amount of field-scale trials and long-term studies, especially concerning nano-biochar. This gap underscores the need for further research for informed decision-making in specific crop management practices and for tailoring choices to local conditions. Table 8 shows the comparative performance of biochar versus nano-biochar; and encapsulates the comparative advantages and challenges associated with both biochar and nano-biochar, providing essential insights for researchers and practitioners seeking to choose the appropriate material for sustainable agriculture and effective environmental remediation.

5. Conclusions and future perspective

The presence of HMs in the environment signifies a considerable threat to agricultural production and ecological stability. Future research needs to emphasize cost-effective and scalable biochar production technologies appropriate for local feedstock sources in an effort to address these challenges since such methods need both to cut costs and be eco-friendly. Biochar is a good mobility suppressor of HMs across different soil conditions since it can easily be obtained from organic agricultural wastes and biomass. Multiple operational elements of biochar, including adsorption ion exchange electrostatic attraction pH modification, complexation precipitation, redox reactions, and effects on nutrient and water availability and soil microbial activity, enable straightforward operation in this way. The success of treatments depends on various additional variables, which include the feedstock choice and pyrolysis temperature conditions, as well as the way biochar absorbs materials, the amount of treatment and soil pH, and the particular HMs found in the target area. These materials reduce HM uptake in plants by scavenging ROS, controlling nutrient regulation, expressing associated genes, and enhancing microbial activity. Nano-biochar, on the other hand, serves a dual role in environmental management by providing effective HM remediation and enabling advanced biosensing capabilities. Its high surface area and reactivity make it ideal for real-time monitoring of contaminants in soils and water. The nanoscale structure, distinguished by a high surface area, porosity, and many functional groups, facilitates enhanced adsorption and immobilization of contaminants, including Cd, Pb, As, and Cr. The phytotoxicity effects of HMs decrease when nano-scale variants lower HMs' bioavailability through surface complexation and ion exchange and precipitation processes within the soil. The ability of nano-biochar to control oxidative stress while enhancing antioxidant plant enzymes makes it an excellent sustainable solution for treating HM-polluted sites. The pyrolysis of organic biomass (e.g., agricultural waste, wood, or manure) at controlled temperatures is the typical process for the production of nano-biochar. Subsequently, mechanical or chemical size reduction is employed to achieve nano-scale particulates. Advanced techniques, including ball milling, ultrasonication, and chemical activation, have been utilized to modify the physicochemical characteristics of nano-biochar, hence improving its efficacy in HM remediation and enhancing plant tolerance. Changes in soil structure and hydraulic conductivity, HMs leaching, ion adsorption, and the effects on soil pH and electrical conductivity are all consequences of using biochar and nano-biochar for soil remediation under HMs. The effects of biochar and nano-biochar on enhancing plant tolerance to stresses involving HMs can be wrapped under several principal mechanisms that include osmotic adjustment, ion homeostasis, root development, nutrient availability and efficiency, ROS scavenging, inhibition of lipid peroxidation, enhancement of photosynthetic efficiency, regulation of stomatal functions, and ultimately, their impact on yield, and quality optimization. The outstanding features of nano-biochar, which include superior surface area, reactivity, and mobility, enable it to lessen HM toxicity with superior performance. Both environmental dangers and higher costs should be carefully considered. The widespread production benefits from biochar due to its safer nature and ease of mass application, although its effectiveness may decrease under specific conditions. Subject matter experts need to choose between these solutions by evaluating the targeted application together with the type of contamination and the need for sustainable performance above instant results. The assessment of large-scale remediation activities and reduction of HM movement toward the food chain needs extensive field research. The most beneficial outcomes emerge from employing biochar alongside other approaches in various applications. Scientists must study both types of biochar before and after remediation activities to measure metal bioavailability and maintain environmental sustainability.

According to the literature, while nano-biochar demonstrated superior efficiency in HM remediation due to its enhanced surface functionality and tunable properties, conventional biochar remained a cost-effective alternative for large-scale applications. The economic viability of biochar, derived from low-cost feedstocks and simpler production processes, offers practical advantages where ultra-high remediation performance is not critical. Key findings from each section highlight: 1) the physicochemical properties of biochar (e.g., surface area, porosity, functional groups) are highly dependent on feedstock type and pyrolysis conditions, with nano-biochar offering superior reactivity but requiring stricter control over synthesis parameters. 2) Biochar improves nutrient retention, water-holding capacity, and crop yields, while nano-biochar shows promise in targeted pollutant removal due to its high adsorption capacity. However, long-term field studies are scarce, particularly in diverse soil types and climates. 3) Potential drawbacks include chemical leaching (e.g., HMs, PAHs) and unintended effects on soil microbiota, underscoring the need for standardized risk-assessment protocols.

While nano-biochar and biochar offer significant potential for soil improvement, carbon sequestration, and contaminant remediation, their environmental risks should not be overlooked. Potential ecological concerns include 1) chemical leaching: pyrolysis conditions and feedstock type may release HMs, polycyclic aromatic hydrocarbons (PAHs), or residual toxins, posing risks to soil and water systems. 2) nano-biochar toxicity: the small size and high reactivity of nano-biochar could disrupt microbial communities or accumulate in organisms, necessitating further ecotoxicological studies/ecotoxicological screening of nano-biochar before field applications. 3) soil ecosystem impacts: long-term effects on soil biota, nutrient cycles, and plant growth require careful monitoring, especially in diverse environmental conditions.

To bridge knowledge gaps, future studies should explain the nano-biochar-soil biota interactions at molecular scales to establish ecotoxicity, large-scale, long-term experiments over various agroclimatic conditions to evaluate real-world performance and unintended effects, sustainable sourcing of feedstocks (e.g., crop residues, invasive biomass) and lifecycle analysis to enhance environmental benefit, limits on nano-biochar application, safety thresholds, and monitoring regimes to ensure safe implementation (subsidies for biochar can motivate farmers to adopt such green measures, enhancing soil quality and yields). Tighter regulations regarding HMs in agriculture also need to be implemented to provide food security and public health. Research should also focus on developing field-deployable sensors that utilize nano-biochar for accurate and timely assessments of HM levels. Such sensors could enhance environmental monitoring and support sustainable agricultural practices by providing critical data for informed decision-making. An extensive investigation of the production techniques is needed to validate their practical application because of the need to assess both cost-efficiency, scalability, and environmental safety. Forthcoming studies should analyze nano-biochar's prolonged environmental behavior and its toxicological potential, along with its effect on soil microorganisms, because its advantages remain obvious. Simulated field examinations are required to evaluate the effectiveness of nano-biochar in actual environmental conditions because these tests validate proper usage methods. Hence, site-specific assessments to optimize biochar benefits (e.g., crop yield, pollution remediation) while minimizing unintended consequences (e.g., leaching, soil disruption) are important. Policymakers should also support large-scale trials, incentivize sustainable feedstock use, and establish safety thresholds to integrate biochar into climate-smart agriculture and soil restoration programs, ensuring scalable and environmentally responsible adoption. By bridging these gaps, researchers and practitioners can advance biochar technologies from experimental stages to scalable, environmentally responsible solutions. Therefore, the compiled knowledge of this review has direct implications for policy and practice in agriculture and environmental sciences, emphasizing the need for standardized biochar production guidelines, field-specific application protocols, and regulatory frameworks to mitigate ecological risks.

Author's contributions

All authors contributed to the study conception and design. The first draft of the manuscript was written by [Abolghassem Emamverdian], [Ahlam Khalofah], [Necla Pehlivan], and [Abazar Ghorbani] and all authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.

CRediT authorship contribution statement

Necla Pehlivan: Writing – review & editing, Writing – original draft, Visualization, Validation, Project administration, Methodology, Investigation, Conceptualization. **Ghorbani Abazar:** Writing – review & editing, Writing – original draft, Visualization, Validation, Investigation, Conceptualization. **Abolghassem Emamverdian:** Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Resources, Project administration, Methodology, Conceptualization. **Khalofah Ahlam:** Writing – review & editing, Writing – original draft, Supervision, Resources, Project administration, Investigation, Funding acquisition, Conceptualization.

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Declaration of Competing Interest

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

Data availability

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

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