

Review

# Realizing United Nations Sustainable Development Goals for Greener Remediation of Heavy Metals-Contaminated Soils by Biochar: Emerging Trends and Future Directions

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**Abstract:** The remediation of heavy metals (HMs) in soil is always an important topic, as environmental contamination by HMs is of serious concern. Numerous potential advantages, especially integrated with biochar produced from various biomass, might provide an ecologically beneficial tool for achieving the UN's sustainable development objectives for greener soil remediation. The aim of this study was to address how the soil-science professions may best successfully utilize biochar for greener remediation of HMs-contaminated soils. In this context, the biochar preparation method from different agricultural feedstock, and its use as a soil amendment for remediation of HMs-contaminated soil, were discussed. Furthermore, biochar-based nanocomposites containing functional materials have lately attracted much interest because of the unique properties emerging from their nanoscale size compartment, and present good promise in terms of reactivity and stability. The utility and potency of biochar-based nanocomposites, on the other hand, are determined by their ability to adapt to particular site circumstances and soil qualities. This overview summarized the current advances in the application for the remediation of HMs-polluted soils. Future views on the usage and possibilities for deploying biochar-based nanocomposites in polluted soils were discussed.

**Keywords:** biochar; nanomaterials; heavy metals; contaminated soil; soil remediation

## 1. Introduction

Soils provide for fundamental human requirements such as food, clean water, and clean air, and act as a primary carrier of biodiversity. Soil sustainability in the twenty-first century relies not only on the farmer, forester, and land-planner management practices but also on governmental decisions on laws and regulations, marketing, and subsidies [1,2]. Increasing anthropogenic impact on the natural environment has resulted in major global problems at the nexus of planetary and public health [2].

Heavy metals (HMs) pollution is a critical global environmental problem [3]. Heavy metals/metalloids have polluted five million locations throughout the globe, with current amounts above legal thresholds [4]. In the case of China, experts have estimated that more than 20 million hectares of farmland have been contaminated, accounting for 20% of the total landmass [5]. Heavy metals are present naturally in soils, but elevated levels may be derived from mining, industrial production, the usage of metal-containing chemicals, and anthropogenic activities [6]. In this context, the quantities of the HMs were reported to be Cu (20 mg/kg), Cd (0.06 mg/kg), Cr (20–200 mg/kg), Pb (10–150 mg/kg), Ni (40 mg/kg), and Zn (10–300 mg/kg). Heavy-metal concentrations in metal-rich soils, on the other hand,

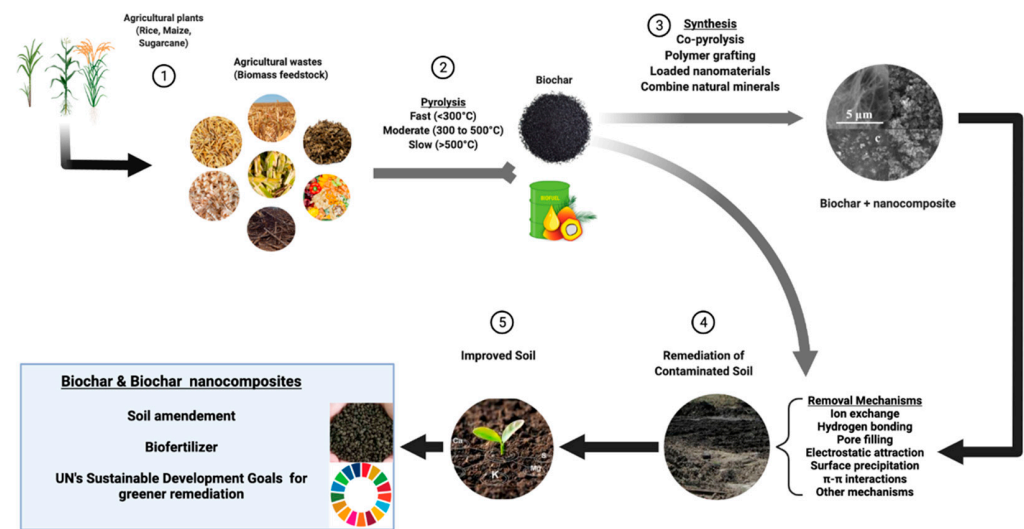
may reach 10–1000 times higher levels due to fundamental parent materials or pollution. According to statistics from a 2014 nationwide assessment of soil pollution in China, 16.1% of soil pollution sites (including agricultural and industrial) surpassed the second level of the Soil Environmental Quality Standard for the permissible limits, with 82.8% of them being HM- pollution sites. Cd, Hg, As, Cu, Pb, Cr, Zn, and Ni were found to be 7.0%, 1.6%, 2.7%, 2.1%, 1.5%, 1.1%, 0.9%, and 4.8% over national norms, respectively [7]. Hazardous metals ions, such as As, Cd, Cr, Cu, Hg, Pb, and Zn, are the most contaminating metals in soils. Their fundamental qualities include non-degradability, persistence, bioaccumulation, and biomagnification in a food chain [8]. To define and resolve pollution problems, HM contamination must be addressed to avoid several threats to the environment and people, all of which undermine food security and increase land tenure issues.

Nowadays, rapid growth in urbanization, population, and living standards play a crucial role in increasing agricultural waste generation, calling for a sustainable solution [9]. These challenges encourage the ecosystem science community to engage closely with policymakers to establish meaningful adaptation goals that benefit both people and the ecosystems they rely on [10,11]. This vision, aligned with the UN's Sustainable Development Goals, may create ever-growing pressure to have a cleaner environment with higher quality at a global level [12]. Different environmentally acceptable strategies have been discovered for sustainable soil management dealing with this problem [13]. Among various approaches, an adaption of a nature-based solution strategy [12] is preferable as the transformation of the waste feedstock into soil discipline would mitigate the effects of climate changes at the scale and pace needed [14].

The importance of the nature-based solution has been recognized by 141 adaptation components of the 167 Nationally Determined Contributions (NDCs) submitted to the United Nations Framework Convention on Climate Change by all parties to the Paris Agreement. In all, 103 countries include NbS in their NDC's adaptation component, 76 countries in their adaptation and mitigation components, and 27 countries exclusively include them in their mitigation plans. In other words, 130 countries, or 66% of all signatories to the Paris Agreement, have stated their commitment to engage with ecosystems in some way to address climate change's causes and effects [15,16].

As stated earlier, due to the importance of environmental and economic indicators, feedstock conversion into materials provides many opportunities for mitigating climate change effects. Several kinds of feedstock residues, such as coconut coir, sunflower husk, rice husk, sugarcane bagasse, ground nutshell, etc., can serve as source materials for making biochar (Figure 1). These various potential benefits, incorporated with the fact that biochar is produced from a wide range of biomass, can potentially be a cost-effective method to convert the trash into a usable and valuable material [17]. Moreover, the wastes transformed to biochar offer an environmentally beneficial tool for realizing the UN's Sustainable Development Goals for greener remediation of HMs-contaminated soils.

The use of biochar, a solid carbonaceous substance, is one cost-effective and environmentally beneficial remediation method for remediation of HMs from the soil. This study focuses on an overview of biochar's usage in the remediation of HM contaminants, the benefits of biochar, and the impact of process variables, including temperature, pressure, and heating rate [18,19]. Previous studies on biochar's application in the remediation of HM-contaminated environments have mostly focused on the use of pristine biochar. In order to address this gap, this overview intends to address this gap by discussing biochar-based nanocomposites. Briefly, we summarize recent progress in understanding (1) biochar production techniques from various agricultural feedstocks, (2) application of biochar for soil amendment, and (3) application of biochar for the remediation of HM-polluted soils. In addition, research gaps and future directions in understanding biochar-based nanocomposites in the remediation of HM-polluted soils are discussed.



**Figure 1.** Conversion of agricultural waste biomass to biochar to meet the sustainability goals.

## 2. Biochar Preparation Methods

Biochar is a solid organic residue produced from the pyrolysis of biomass. Biochar may be made on a small scale with a cooking burner or a big scale using a pyrolysis system. Pyrolysis is a thermochemical process that converts biomass into biochar at temperatures between 350 and 700 °C in the absence, or with a limited amount, of oxygen [20]. The solid carbon-rich product of such a process is defined as biochar or char, and the volatile portion of pyrolysis is partially condensed to a liquid fraction known as tar or bio-oil. Pyrolysis processes are divided into three categories based on the conditions: slow pyrolysis (slow heating rates for a long time, temperatures below 300 °C), moderate pyrolysis (temperatures between 300 and 500 °C), and fast pyrolysis (high heat-transfer rates in a short time, temperatures above 500 °C) [21]. According to the literature [22–24], primary cracking and the secondary breakdown and the production of oxygen functional groups begin at approximately 400–500 °C during biomass pyrolysis [22–24].

Various feedstocks resulted in different surface areas, pores, and functional groups in biochars, impacting biochar properties. Rice husk, wood bark, sugar beet tailing, fruit peels, pinewood, wood waste, and plant residues are the most common biochar feedstocks from the agriculture sector [22,24]. Understanding whether initial feedstock qualities affect end biochar characteristics is pivotal in the feedstock. Feedstocks have been demonstrated to have a significant influence on the development of biochars with unique chemical characteristics. Wood-based biochars have higher C and lower plant-available nutrients, while manure-based biochars have the reverse tendency. Grass-based biochars are usually somewhere in between woody and manure biochars. These characteristics, nevertheless, may be affected by the pyrolysis temperature and method often used to produce [25,26].

The pyrolysis settings and feedstock have a significant impact on the characteristics of biochar; additional variables include the rates at which heat is exchanged, temperatures, and residency period [20]. The pyrolysis temperature affects biochar's structural and physicochemical characteristics, such as surface area, pore architectures, surface functional groups, and elemental compositions. The release of volatiles at high temperatures may explain the impact of pyrolysis temperature on such characteristics. According to many researchers [27–29], higher pyrolysis temperatures resulted in a larger biochar surface area, higher pH, and higher percent C content but lower percent N content. As a result, choosing an appropriate pyrolysis temperature involves a tradeoff between the specified surface and chemical characteristics [27–29].

### 3. Biochar's Potential Role as a Soil Amendment

Heavy metals have long been recognized as serious environmental pollutants emitted by various sectors, including coal burning, battery manufacturing, leather manufacturing, and pesticide use [30]. Due to their high toxicity, carcinogenicity, and mobility, HMs represent significant dangers and health concerns to people [2]. Furthermore, different states of HMs ions may coexist in the environment, resulting in more complex and multi-toxic pollutions. These contaminants can build up in food systems, causing harm to plants, animals, and people (damage to the endocrine system, impact on immunity, neurological disorders, and cancer) [31].

Chemical decontamination procedures for HMs, such as excavation, precipitation, heat treatment, electro-remediation, and chemical leaching, are still expensive and are dependent on the pollutant and soil properties [7]. The primary difficulties and downsides of these procedures include changing soil parameters (particularly pH), the possibility of soil-fertility loss, small-scale application, and by-product formation [32]. On the other hand, chemical precipitation has a high cost and might cause secondary pollution. In contrast, phytoremediation has a very long working period, and the treatment of metal-loaded biomass wastes is still an environmental issue [33]. As a result, there is still a pressing need to create efficient, cost-effective, and “green” technologies capable of removing large amounts of HMs.

According to a market survey report by Grand View Research Inc. (2019), the global biochar market size is estimated to reach USD 3.1 billion by 2025 and is expected to grow with a CAGR of 13.2% [34]. Another independent agency, Fact.MR (2021), valued the global biochar market at USD 8 million in 2020 and estimated that the global sales of biochar would cross USD 23 million by 2031 with a CAGR of 11% [35]. In addition, Transparency Market Research in 2020 estimated that the global biochar market will boom with a CAGR of 15.35% during 2021–2031. However, stakeholders are yet to conduct a life-cycle cost analysis that can help to close the gap over the field application of biochar [36]. The variations in projections are primarily due to disruption of the industry–supplier–farmer axis in different countries affected due to the varying severity of the Covid pandemic.

Using biochar as a soil supplement substantially impacted soil fertility by changing the soil's chemical, biological, and physical properties [37]. Its use as a soil amendment improves soil quality and plant development, resulting in higher agricultural yields. Biochar resource, manufacturing method, soil type, and condition, as well as the kind of crop to be planted, may all have an impact on its efficacy [38].

### 4. Applications of Biochar in Remediation of Heavy Metals in Soil

Heavy metals that originate in soil, such as Cu, Zn, As, Cr, Co, Ni, Sb, Hg, Th, Pb, Se, Si, and Cd, may be exceedingly detrimental to human and plant life if soil and water are contaminated [39–41]. As HMs do not biodegrade, they may remain in contaminated soils for prolonged periods, emphasizing the necessity of using the most practical method, such as biochar, as an environmentally friendly procedure [42]. Biochar has been used to solve the issue of heavy-metal pollution and increase soil fertility for a long time. As shown in Table 1, various forms of biochar are utilized for different types of HMs depending on the soil type.

The use of biochar as a soil supplement in agriculture has been the emphasis of these early uses. Still, other applications in environmental-remediation engineering may be equally as essential as the soil practices [43]. Biochars may have a wide range of physical and chemical characteristics depending on the feedstock and thermochemical conversion (pyrolysis) processes. Consequently, the performance of biochar in diverse field applications is greatly influenced by both the manufacturing processes and the source material composition [44,45]. The link between biochar characteristics, manufacturing circumstances, and feedstock composition must be defined to understand better the current variety in available biochars and the implications for its usage as an engineered material [46].

Towards this purpose, a summary of our current understanding of the impact of source material and pyrolysis procedures on biochar properties is provided in Table 1.

High Cd(II) exposure risk in soils will enhance the metal's ability to transfer and hyper-accumulate in plants and crops, and its ability to leach into surface and groundwater, and cause detrimental ecosystem consequences [47]. To address this problem, Chen et al. (2022) studied the effect of biochar pyrolysis temperature on Cd transportation in water-saturated soil [48]. They revealed that biochar made at 500 °C dramatically inhibited Cd(II) transport at high ionic strength [48]. In another investigation, pine sawdust biomass biochar produced at 550 °C was a better amendment for Pb immobilization than the biochar produced at 300 °C (Table 1). Furthermore, biochar generated at higher temperatures has the potential to be more stable, making it suited for the rehabilitation of Pb-contaminated soils that are regularly inundated [49]. When released into the environment, HMs such as As pose a serious threat to animal and human health. The biochar derived from corn straw showed acceptable sorption affinities towards As(III). Moreover, a pH increase after utilization of biochar can neutralize acid soil, therefore potentially preventing red-soil acidification [50].

**Table 1.** Research studies reported the application of biochar in heavy metals-contaminated soils.

Focus of Study	Polluted-Soil Type and Conditions	Type of Biochar	Pyrolysis Conditions	Biochar's Contribution	Important Results	Reference
The effect of pyrolysis temperature on the transportation of Cd(II) in water-saturated soils	Upper-layer silty loam red soil contained Fe (47.0 g/kg) and Al (16.7 g/kg)	Wheat straw	350 and 500 °C for 2 h under N <sub>2</sub> atmosphere	High-temperature biochar showed a higher affinity towards Cd(II).	Biochar made at 500 °C biochar dramatically inhibited Cd(II) transport at high ionic strength.	[48]
The effect of biochar as an amendment for the As contaminated soil	Upper-layer red soil (pH:5.42, CEC:5.90 cmol/kg, OM:14.90 g/kg)	Corn straw	600 °C for 2 h under N <sub>2</sub> atmosphere	The increase in soil pH due to biochar by 0.4 units would potentially reduce the acidification of red soil.	The bonds (Mn-O/As and Fe-O/As) improved surface sorption capacity for As removal. Oxygenated functional groups, such as O-H, C=O, Si-O, and especially Mn-O, facilitated the oxidation of As(III) to As(V) in the contaminated soils. Variations of dynamic redox conditions were limited. Pb immobilization due to, potentially, alteration in redox chemistries due to resorption of Pb dissolved in soil.	[50]
Immobilization and speciation of Pb under redox conditions for soil amendment	Upper-layer sandy loam agricultural land soil (close to a gold mine) contained (As:2047 mg/kg and Pb:1680 mg/kg)	Pine sawdust	300 and 550 °C (residence time not available)	The biochars produced at higher temperatures were found to be more suitable for Pb immobilization under dynamic redox conditions.	Precipitation and complexation of available functional groups could be influential on Pb immobilization.	[49]

The efficiency of biochar implementation for the reduction in Cu mobility in soil has been shown in another investigation [51]. It was shown that the use of biochar in polluted soil resulted in the change of fraction-group composition due to the reduction in weakly bound forms, and increase in the part of residual and metal fractions strongly bound with organic matter [51]. The other researchers also showed that the introduction of biochar into soil contaminated with Cu and Zn, along with a decrease in loosely bound compounds of these metals in the soil, reduced the content of Cu and Zn in spring barley (*Hordeum sativum distichum*), and reduced the accumulation coefficient [52].

Increasing the pyrolysis temperature increased the sludge-based biochar's adsorption capability. This might be attributable to a larger concentration of oxygen-containing functional groups on the surface of biochar samples produced at higher temperatures. The quantity of HMs in the biochar was lowered when the pyrolysis temperature was raised from 700 to 900 °C. The explanation for this was that at temperatures over 700 °C, the

rate of deoxygenation reaction was faster. As a result, the quantity of oxygen-containing functions dropped, and the amount of HMs absorbed by the biochar decreased [52,53].

Biochars made from various biomasses will undoubtedly have varying impacts on the efficiency with which HMs are removed from the soil. This might be owing to the structural differences between biochars produced from various biomass sources. Two biochar samples were made by Wang et al. (2017) from the pyrolysis of maize straw and pig dung at 350 °C [54]. The findings showed that adding biochar samples to the soil (20 g/kg) significantly decreased the level of HMs such as Cd, Cr(VI), Hg, and Pb in the soil. Using maize straw and pig manure biochar samples, for example, the content of Hg was lowered from 0.79 to 0.59 and 0.34 mg/kg, respectively. Due to its larger surface area (BET surface area for corn straw and pig manure biochar samples was 10.7 and 26.8 m<sup>2</sup>/g, respectively), pig manure biochar had a greater rate of HMs removal. Furthermore, the heavy-metal concentration of the pig manure biochar was greater than that of the corn straw biochar. Calcium levels in pig manure and maize straw biochar samples, for example, were 7.29 and 78.10 g/kg, respectively. The greater calcium concentration in the pig manure biochar improved the ion-exchange mechanism's ability to remove HMs [54].

Furthermore, the adsorption effect of different biochars on the coexistence of HMs (Pb<sup>2+</sup>, Cu<sup>2+</sup>, Zn<sup>2+</sup>, Cd<sup>2+</sup>) was reported [55,56]. The researchers found the simultaneous presence of Pb<sup>2+</sup>, Cu<sup>2+</sup>, Zn<sup>2+</sup>, and Cd<sup>2+</sup> in the soil led to the competitive adsorption of HM ions [55,56]. The adsorption impact of biochar on complex HMs is greatly decreased compared to single HM adsorption due to a limited number of adsorption sites on the surface of biochar. The degree of competitive adsorption increased as the initial concentration rose [57].

Biochar may attenuate the concentrations of various HMs in varied quantities, according to the literature. The physicochemical characteristics of biochar may be greatly altered by pyrolysis-process conditions and feedstock origin. In general, increasing the pyrolysis temperature increases the surface area and amorphous structure of biochar, which improves the biochar's HM-adsorption performance. The metals that were initially present in biochar may speed up the ion-exchange process in soil remediation. Biochar treatment might be a way to immobilize HMs in the soil and prevent them from entering the food chain. However, it is worth noting that the quantity of HMs removed may be increased by modifying biochar. For example, by incorporating the nanomaterials on the surface of the biochar or through synthesis procedure, the intensity of oxygen-containing functions and metal content in the biochar might be raised. As a result, several techniques of biochar-composite manufacturing, and their impact on HMs remediation in the soil, are explored in-depth in the next section.

## 5. Applications of Biochar Nanocomposites in Remediation of Heavy Metals in Soil

The synthesis of biochar-based composites has opened up many new opportunities for both biochar and nanomaterials [58]. The functional groups, pore characteristics, surface-activity sites, catalytic-degradation capability, and ease of separation of the resultant composites are often drastically enhanced [59,60]. It is revealed that biochar modification by nanomaterials enhances its potential capacity for immobilization of HMs, rendering the bionanocomposite into an efficient heavy-metal sorbent in soils (Table 2). In this regard, a magnetic-based porous biochar sphere was synthesized to improve biochar's immobilization efficacy. The biochar composite showed excellent flotation and magnetism performance which eventually, by the addition of water into the soil, induced the spheres to leave the soil and float on the surface of the soil/water mixture. Electrostatic interactions between biochar spheres and HMs were responsible for the immobilization process [61].

After pyrolysis of residual bark chips at 600 °C, impregnation of chitosan/nanoclay onto biochar was performed to produce a homogenous composite to simultaneously immobilize Cu, Pb, and Zn metal ions within the contaminated soil. The nanocomposite hindered the leaching of Cu<sup>2+</sup>, Pb<sup>2+</sup>, and Zn<sup>2+</sup> by 100, 52.29, and 100%, respectively, much higher than pristine biochar [62]. According to the research by Mandal et al. (2020), the

attachment of nZVI nanoparticles and graphene oxide to biochar avoided aggregation and quick oxidation while preserving active nZVI reactivity and providing stability [63]. The biochar's complete acidic functional groups produce efficient pH and  $pH_{zpc}$ , which play an important role in Cu immobilization in the soil. Uniform nZVI particles with quasi-spherical centers surrounded by a thin layer of graphene oxide support develop on the charcoal surface. After 14 days of treatment, biochar/graphene oxide-nZVI substantially reduced the available Cu content in the soil and lowered Cu bioavailability [63].

Most biochars reported have no reusability upon aging and offer the risk of releasing immobilized components after short-term immobilization. To address the problem as mentioned earlier, the nano zero-valent iron (nZVI)/green-tea biochar produced at temperature 450 °C was investigated as a potential material for Pb immobilization in the soil. The composite showed a surface area of 38.08 m<sup>2</sup>/g, an average particle size of nZVI ~0.12 µm, and a saturation magnetization of 0.24 emu/mg, resulting in the production of superparamagnetic composite. The sequential extraction studies suggested the conversion of Pb species into oxides after 30 days of the experiment, suggesting a promising opportunity for heavy-metal immobilization in the soil [64]. Similarly, Fan et al. (2020) used one-pot pyrolysis of sawdust and Fe<sub>2</sub>O<sub>3</sub> mixture to enhance As immobilization to embed nZVI on biochar [65]. The mobility of As in soil was reduced as compared to the pristine biochar treatment. This phenomenon might be caused by As adsorption and co-precipitation on the surface of biochar caused by nZVI corrosion formation (amorphous FeOOH). Furthermore, following sorption by nZVI-BC, the majority of As(V) was reduced to As(III) [65]. Yu et al. (2015) produced a nano-MnO<sub>2</sub>-modified biochar composite, an excellent adsorbent of As(III) in red soil [50]. The specific surface area of the composite was dramatically reduced when nano-MnO<sub>2</sub> particles were deposited in the pores of biochar. As(III) reacted with oxygen-containing functional groups, creating Mn-O/As and Fe-O/As bonds, as shown by Fourier transform infrared spectroscopy (FTIR) and XPS analyses. MnO<sub>2</sub> and Fe-Mn oxide partly oxidized adsorbate As(III) to As(V) [50]. In another study, porous biochar-supported nanoscale zero-valent iron (BC-nZVI) was applied to immobilize Cd and Pb in clayey soil. With biochar-nZVI, simultaneous immobilization of Cd and Pb was accomplished, and both Cd and Pb availability were dramatically reduced. Moreover, stable Cd species such as Cd(OH)<sub>2</sub>, CdCO<sub>3</sub>, and CdO were created, while stable Pb species such as PbCO<sub>3</sub>, PbO, and Pb(OH)<sub>2</sub> were obtained, suggesting simultaneous immobilization of Cd and Pb in soil [66].

**Table 2.** Research studies reporting the application of biochar nanocomposites in heavy metals-contaminated soils.

Focus of Study	Polluted-Soil Type and Conditions	Type of Nanomaterials	Synthesis Method	Important Results	Reference
Immobilization of heavy metals in agricultural soil	Soil from agricultural land contaminated with Cd(II) 2.81 mg/kg and As(V) 60.23 mg/kg	Porous magnetic biochar sphere loaded with Fe <sub>3</sub> O <sub>4</sub> and FeCl <sub>2</sub> hydrates nanoparticles	One-step gelation and pyrolysis	The bioavailable fraction of Cd and As(V) was found to decrease from 1.55 to 0.32 mg/kg and 1.26 to 0.85 mg/kg, respectively. The nanocomposite adsorbed Cu <sup>2+</sup> , Pb <sup>2+</sup> , and Zn <sup>2+</sup> by 121.5, 336, and 134.6 mg/g, respectively, much higher than pristine biochar.	[61]
Immobilization of heavy metals in mine-impacted acidic waters and soils	Silty sand acid soil from a Cu mine contained (Cu, Pb, and Zn)	Chitosan and nanoclay	After pyrolysis, impregnation of chitosan/nanoclay suspension onto residual bark chips biochar was conducted to produce a homogenous composite	At 10% w/w, the nanocomposite reduced metal leaching from the soil by Cu <sup>2+</sup> : 100%, Zn <sup>2+</sup> : 100%, and Pb <sup>2+</sup> : 52.29%.	[62]

Table 2. Cont.

Focus of Study	Polluted-Soil Type and Conditions	Type of Nanomaterials	Synthesis Method	Important Results	Reference
Graphene oxide and nano zero-valent iron (nZVI) integration with biochar for Cu immobilization	Agricultural land soil spiked with 386 mg/kg and 488 mg/kg Cu	Graphene and nano zero-valent iron (nZVI)	After pyrolysis, impregnation of graphene and nZVI was performed through the post-pyrolysis and co-precipitation techniques, respectively.	The composite facilitated the conversion of accessible Cu to less easily accessible.	[63]
Synergistic effect of green-tea biochar and nZVI	Agricultural land soil spiked with 386 mg/kg Pb	Nano zero-valent iron (nZVI)	After pyrolysis, impregnation was performed through the co-precipitation technique.	Compared to green-tea biochar and pristine nZVI alone, the nanocomposite enhanced the immobilization efficiency of PB by 19.38% and 57.14%, respectively.	[64]
Magnetic biochars for the immobilization of heavy metals in a multi-contaminated soil	Paddy soil polluted with Cd, Cu, Zn, and Pb, with total contents of 1.4 mg/kg, 80 mg/kg, 1638 mg/kg, and 2463 mg/kg, respectively	Fe <sub>3</sub> O <sub>4</sub> /biochar	After pyrolysis, impregnation of biochar was performed through the post-pyrolysis and co-precipitation technique.	In soils amended with the magnetic composite, acid-soluble Cd was 8–10% lower than in control polluted soil.	[67]

Lu et al. (2018) investigated whether pyrolysis temperature, choice of feedstock, or magnetization played a predominant role in determining the sorptive biochar capacity [67]. The choice for the adequate temperature of pyrolysis and feedstock was more relevant than magnetization for preventing Cd, Pb, and Zn leaching [67].

## 6. Future Prospects and Challenges

Several restrictions and problems remain for the usage of biochar to fulfill the ever-increasing demand can be summarized as follows. (1) Biochar has been used as a multi-functional platform for the uptake of HMs. However, new unique nano-based materials should be synthesized to develop functional composites ways to reduce manufacturing costs and boost removal efficiency/capacity to improve economic viability. Furthermore, to account for the possible secondary pollutions formed during the fabrication of biochar nanocomposites, more “green” synthesis techniques must be developed to achieve a highly effective and long-lasting HMs-removal amendment. Further research into the formation processes might aid in improving the characteristics of biochar nanocomposites in order to attain better HMs-removal effectiveness. (2) In order to increase the removal efficiency/capacity of HMs, a comprehensive investigation of the precise removal processes of various HMs is necessary. Due to the complex components present, multiple HMs and other organic/micropollutants are often found in genuine polluted soil. To increase the potential and practicality of using the biochar nanocomposite to remove HMs from actual contaminated soil, the competing adsorption processes of numerous HMs on nanoscale-metal-aided biochar must be explored further. The synthesized biochar nanocomposites may display variable removal capabilities under various soil conditions due to the restricted active sites and interacting connection between HMs and dissolved organic matter. To address the requirement of practical applications, the sequential HMs extractions with the investigation of the competition/synergetic processes among the coexisting HMs under varied actual soils should be examined in the following phase. (3) Due to actually complicated soils, the practical application may confront some additional obstacles.

The most innovative and promising nanomaterials and related biochar-based composites are more likely to become a real alternative in the short term. Apart from metal oxide-based composites, the enormous potential of metal-organic frameworks (MOFs) has not gone unnoticed in the soil field. MOFs provide a large family of micro-mesoporous crystalline materials with highly tunable characteristics such as extremely large surface areas (>5000 m<sup>2</sup> g<sup>-1</sup>) [68]. These properties could play a significant role in abating HMs in polluted soil. The utilization of nanomaterials with greener methods is preferred since



generated waste within the synthesis procedure is minimized sustainably. The employment of the green approach for the synthesis might open up a new path to eliminate the risk of harsh reagents to the ecosystems. The green synthesis involving using abundant nature-based materials is highly recommended.

The large-scale preparation of the biochar-based composites is still costly and time-consuming. Hence, the “more is better” approach must be avoided due to an economic point of view. Furthermore, this approach can lead to the production of wasted resources and the depletion of reactants. There are only a few large-scale approaches for composite preparation, while most of the processes are cumbersome and require using expensive reagents. Unfortunately, challenges remain to implement efficient, cost-effective, and timely HMs removal in practical systems. In addition, the cost assessment on a practical scale should be extensively explored to emphasize the benefit of biochar. As a result, creating long-term, large-scale HMs-removal systems that use biochar nanocomposites as an amendment could boost economic potential. Even though various studies of biochar nanocomposites as HMs sorbents have been published in recent years, attempts to improve operation tactics and build scale-up HMs-removal systems remain missing. This obstructs the commercialization of biochar nanocomposite-based sorbents for HMs removal from soils and should be the focus of future research.

## 7. Conclusions

Environmental and natural resource concerns have prompted a quest for finding renewable energy sources as a long-term approach to encompass ecological restoration. This review examined how the soil-science profession may best successfully utilize biochar for greener remediation of HMs-contaminated soil. Biomass is a resource that can be replenished. The thermochemical pyrolysis of biomass waste can be a practical and sustainable option for turning waste into useful biochar products. The resultant biochar is porous, with a large specific surface area and a high concentration of hydrophilic groups, which may be used for soil improvement, ecological restoration, waste management, and soil remediation.

The knowledge of the approaches to implement biochars for HMs-polluted soils is still insufficient. Thus far, there are no applicable standards for regulatory biochar application. Therefore, establishing the regulatory framework is necessary to ensure the safety of the application in the long term.

Nanoscale metals deposited on biochar might boost the effectiveness of HMs elimination. Understanding how to make biochar nanocomposites and how HMs are removed is critical for their future use. Direct interactions (e.g., electrostatic adsorption, ion exchange, complexation, and precipitation) and indirect interactions (e.g., via altering soil parameters such as pH, CEC, mineral content, and organic carbon content) between biochar and HMs must be all considered. Moreover, soil conditions such as pH, type, and other constituents are influential factors for the removal of HMs and should be handled ideally to increase biochar and biochar-nanocomposites effectiveness. There are various laboratory-scale instances for biochar nanocomposites to remediate polluted soil. However, more work needs to be done on running full-scale systems in order to make the technology more practical and dependable.

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