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From Agricultural Waste to Safer Vegetables: A Critical Review of Biochar for Remediating Heavy Metals in Vegetable-Growing Soils and Plant Uptake

Long Ba Le · Binh Thanh Nguyen · Tri Dinh Mai

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Abstract Heavy metal (HM) contamination in agricultural soils poses a critical threat to food safety, particularly in vegetable production systems where direct human exposure occurs through consumption. Vegetables grown in contaminated soils often accumulate elevated levels of toxic metals such as lead, cadmium, chromium, and arsenic, as well as excessive zinc, increasing dietary and public health risks. Conventional remediation methods are frequently costly, inefficient, or environmentally unsustainable. Biochar has emerged as a promising, sustainable amendment for reducing HM bioavailability; however, its effectiveness and mechanisms in vegetable-growing soils remain insufficiently understood. This review critically evaluates biochar's remediation potential, associated risks, and practical challenges, and outlines future research and policy priorities for safe and sustainable vegetable production. While biochar's physicochemical traits—such as high surface area, alkalinity, and cation exchange capacity—facilitate metal immobilization through adsorption, ion exchange, and precipitation, remediation outcomes vary widely with production conditions and application rates. Potential drawbacks include nutrient immobilization or the remobilization of elements such as arsenic and chromium (VI). Broader adoption is further limited by variability in biochar quality, scarce long-term field data, high production costs, and weak policy support. Strengthening biochar governance through quality standards, financial incentives, and integration into soil health and carbon sequestration programs is essential to enhance feasibility and farmer adoption. By linking scientific evidence with policy and practice, biochar can serve as a scalable solution for safer vegetable production, improved soil resilience, and climate change mitigation.

Keywords Agricultural residues · Biochar · Heavy metal immobilization · Soil remediation · Sustainable agriculture · Environmental management

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L. B. Le · B. T. Nguyen (✉)
Institute of Environmental Science, Engineering and Management, Industrial University of Ho Chi Minh City, 12 Nguyen Van Bao, Hanh Thong Ward, Ho Chi Minh City, Vietnam
e-mail: nguyenbinh@iuh.edu.vn

T. D. Mai
Institute of Advanced Technology, Vietnam Academy of Science and Technology, 1B, TL29 Street, An Phu Dong, Ho Chi Minh City, Vietnam

1 Introduction

Heavy metal (HM) contamination in agricultural soils has emerged as a pressing global issue (Angon et al., 2024), particularly in vegetable production systems (Brahma et al., 2025)— "vegetable" referring to edible plant parts such as leaves, stems, roots, tubers, bulbs, and fruits grown for human consumption — posing direct risks to food safety and human health. Non-degradable metals such as lead (Pb), cadmium (Cd), arsenic (As), and chromium (Cr) can persist in soils for decades (Das et al., 2023), posing serious risks through bioaccumulation in edible plant tissues (Bibi et al., 2024). Leafy and root vegetables are especially vulnerable to HM uptake due to their shallow rooting systems and high rates of metal translocation, making them critical vectors for dietary exposure (Kaur et al., 2025; Zhou et al., 2016). Elevated metal concentrations in vegetables not only compromise food quality but also pose chronic health threats, including kidney dysfunction, neurological disorders, and carcinogenic effects (Blenis et al., 2023).

HM contamination in agricultural soils represents a critical entry point for toxic elements into the food chain, particularly through the uptake of these elements by vegetable crops (Rai et al., 2019). Contaminated soils, often resulting from industrial activities, mining, sewage sludge application, and the overuse of agrochemicals, serve as persistent reservoirs of non-degradable metals such as Pb, Cd, As, and Cr (Wuana & Okieimen, 2011). Moreover, the bioavailability of metals in soil is influenced by soil texture, organic matter content, cation exchange capacity, and pH, with acidic and low-buffered soils typically enhancing metal solubility and plant absorption (Olaniran et al., 2013). These metals can be readily taken up by plant roots and translocated to aboveground tissues, especially leaves, which are the primary edible parts of many vegetables. The presence of HMs in soil can disrupt key soil properties and functions—such as microbial activity, nutrient cycling, and pH buffering—thereby degrading soil quality and reducing productivity (Mohammad et al., 2025; Vasilachi et al., 2023). Soils contaminated with HMs often exhibit low microbial biomass, acidic pH, poor nutrient retention, and altered redox potential, all of which further exacerbate metal mobility and plant uptake (Angon et al., 2024; Zheng et al., 2024). This complex interplay between soil contamination and plant physiology highlights the urgent need for effective remediation strategies to reduce metal accumulation in edible plant parts and ensure food safety in vegetable production systems.

Soil contamination arises from both natural and anthropogenic sources (Ali et al., 2019). In many countries, elevated levels of Cu, Pb, and Cd in agricultural soils are largely attributed to industrial activities, while other metals such as As, Hg, Cr, and Ni stem from a combination of natural mineral weathering and human inputs (Cao et al., 2024). Major anthropogenic contributors include mining, wastewater irrigation, sewage sludge, manure, and excessive use of chemical fertilizers and pesticides (Arunakumara et al., 2013). Among these, mining and wastewater irrigation are particularly influential, and the prolonged use of fertilizers and pesticides further exacerbates HM accumulation in cultivated soils (Zakaria et al., 2021).

Although a variety of remediation techniques have been developed to address HM contamination, most conventional methods present significant limitations. Soil washing and electrokinetic extraction are effective, but are cost-intensive and may lead to nutrient loss and secondary pollution (Blenis et al., 2023). Phytoremediation, though conceptually attractive and environmentally compatible, remains limited in large-scale application due to slow remediation rates, strong dependence on climatic and site-specific factors, and the low metal tolerance of many plant species (Cozma et al., 2025). Chemical immobilization using lime or phosphate offers temporary benefits, but lacks long-term stability in field conditions (Cheng et al., 2020). These shortcomings have driven the search for more sustainable and cost-effective alternatives.

Amid these challenges, biochar (BC) has attracted growing interest as a sustainable, low-cost soil amendment capable of mitigating HM contamination while simultaneously improving soil fertility and structure (Mohan et al., 2024). Its desirable properties—including high cation exchange capacity (CEC), large specific surface area, alkaline pH, and chemically stable structure—enable it to adsorb or co-precipitate HMs, reducing their mobility and bioavailability (Subramanian et al., 2024). BC applications have been shown to enhance soil organic carbon, buffer soil pH, increase moisture retention, and stimulate beneficial microbial populations (Blenis et al., 2023; Cheng et al., 2020). Additionally, BC can be modified (e.g., doped with minerals or functionalized chemically) to increase its sorption affinity for specific contaminants (Chen et al., 2022; Liang et al., 2021). Its production from agricultural residues aligns with circular economy principles, converting waste biomass into a value-added product for environmental remediation (Díaz et al., 2024).

Globally, an estimated 14–17% of croplands are contaminated with toxic metals, potentially exposing 0.9–1.4 billion people to heightened environmental and health risks (Hou et al., 2025). Vegetable crops are particularly susceptible, as they readily absorb and accumulate HMs from contaminated soil, water, and air, posing serious food safety concerns (Zhou et al., 2016). Conventional remediation methods are often costly and unsustainable, underscoring the need for alternative, nature-based solutions. Biochar has gained increasing attention as a low-cost and eco-friendly amendment capable of immobilizing HMs, improving soil properties, and enhancing crop safety (Sarraf et al., 2024). However, uncertainties persist regarding its long-term effectiveness, variability in performance across feedstocks and production conditions, and potential adverse impacts. This study is presented as a critical review, going beyond a simple summary of existing literature to evaluate, interpret, and integrate current knowledge on BC use for HM remediation in vegetable production. It provides a comprehensive assessment of how BC properties—such as feedstock type, production conditions, and physicochemical characteristics—influence metal immobilization in soils and accumulation in vegetable crops. The review also analyzes inconsistencies among studies, identifies key research gaps, and highlights environmental trade-offs and management challenges. In addition, it discusses adverse effects, microbial responses, and the technical and economic feasibility of large-scale applications. Specifically, this paper aims to: (1) provide an overview of HM contamination sources and impacts in agricultural soils; (2) examine the critical links between BC production, its physicochemical properties, and the mechanisms of metal immobilization; (3) critically evaluate the evidence for BC's effectiveness in reducing HM uptake in common vegetable crops; and (4) identify the key challenges, risks, and future research priorities necessary to translate this promising technology into a safe, effective, and scalable practice for sustainable agriculture.

2 Methodology

This review employed a structured narrative synthesis approach to comprehensively capture and analyze existing research on BC for HM remediation in vegetable-growing soils. The literature search targeted peer-reviewed studies and authoritative sources published between 1993 and 2025, utilizing academic databases such as Scopus, Google Scholar, and ResearchGate. Keyword combinations such as biochar, heavy metals, soil remediation, vegetable cultivation, and specific vegetable species (*Lactuca sativa, Brassica juncea, Amaranthus*) were used in conjunction with Boolean operators (AND, OR) to maximize the relevance and breadth of results. In addition to academic sources, selected grey literature and institutional databases were used to supplement empirical findings with contextual data. This included statistics on agricultural residue generation and biomass availability from the FAOSTAT (the FAO's global agricultural statistics database, 2003–2023), the International BC Initiative (IBI), and UNIDO. These sources provided valuable perspectives on practical applications, ongoing BC projects, and policy developments relevant to low- and middle-income countries.

Rather than conducting a statistical meta-analysis, a qualitative narrative synthesis was undertaken due to the diverse methodologies and data types across the studies. The reviewed literature was categorized thematically to draw meaningful comparisons and synthesize trends. Key themes included: HM contamination in agricultural soils, BC as a tool for HM remediation in vegetable-growing soils, effects of BC on HM uptake in vegetables, scientific and technical challenges in BC applications, and Economic and environmental challenges and future perspectives for global adoption. A total of 199 references were selected for inclusion in the review (Table 1). The vast majority (183 of 199) were published in journals indexed in the Web of Science, reflecting the predominance of high-quality, peer-reviewed literature in this field. The number of relevant publications has grown rapidly, particularly after 2010, with over 98 studies published between 2021 and 2025 alone. This growth indicates an increasing global interest in sustainable, bio-based solutions for soil remediation, particularly in relation to food safety and vegetable systems.

3 Heavy Metal Contamination in Agricultural Soils

HM contamination of agricultural soils remains a persistent environmental issue due to the

Table 1 Number of reviewed articles by indexing category across publication periods

Publication period	ISI-indexed (WoS)	Scopus-indexed only	Other (not indexed)	Total
< 2010	17	0	1	18
2010–2015	22	1	2	25
2016–2020	46	1	1	48
2021–2025	98	9	1	108
Total	183	11	5	199

The "ISI-Indexed" category includes all articles published in journals currently indexed in the Web of Science Core Collection (including SCI/SCIE/SSCI). "Scopus-Indexed Only" covers articles in journals indexed by Scopus but not by Web of Science. "Other" refers to references not indexed in either database (e.g., book chapters, Reports or regional journals

non-degradable nature and high toxicity of elements such as Pb, Cd, As, Cr, Ni, Zn, Cu, and Hg. Unlike organic pollutants, these metals cannot be broken down biologically or chemically, allowing them to accumulate in soils over time. This persistence not only threatens soil fertility and microbial function but also facilitates metal transfer to crops, thereby posing substantial risks to food safety and human health (Chauhan et al., 2018; Srivastava et al., 2017).

3.1 Sources and Pathways of Contamination in Vegetable-Growing Soils

HMs in agricultural soils originate from both natural (geogenic) and anthropogenic sources. While weathering of parent materials can release trace metals into the environment, elevated concentrations in intensively cultivated soils are primarily attributed to human activities. Key contributors include industrial emissions, mining, and wastewater irrigation, which introduce metal particulates and residues into farming systems. Additionally, the use of agrochemicals—especially phosphate fertilizers containing cadmium and certain pesticides—further exacerbates metal accumulation (Khan et al., 2009; Mishra & Kumari, 2021; Zahra et al., 2017). In vegetable production systems, where crops are directly consumed and often grown in proximity to urban centers, the risks are particularly acute. Repeated application of untreated wastewater or organic amendments such as sewage sludge and manure can lead to elevated levels of Pb, Zn, and Cd in surface soils. The accumulation of these metals not only impairs plant growth but also creates a pathway for their entry into the food chain, with direct impacts for public health and ecological stability (Oves et al., 2012; Zhyrgalova et al., 2024).

3.2 Regional Contamination Trends in Asia and Vietnam

HM contamination in agricultural soils varies significantly across Asian countries, reflecting differences in emission sources, agricultural intensity, and regulatory enforcement (Table 2). For instance, soils in Malaysia have reported As and Cd levels exceeding 50 mg kg^{-1}, far above natural background concentrations (Zarcinas et al., 2004a). In industrialized agricultural zones in China, Zn levels often exceed several hundred mg kg^{-1}, while Cr levels approach 300–360 mg kg^{-1} (Zhou et al., 2014). One of the most severe historical cases occurred in Toyama, Japan, where Cd levels of 400–600 mg kg^{-1} in rice fields led to the "itai-itai" disease outbreak (Herawati et al., 2000). In contrast, countries such as South Korea generally report much lower levels of Pb, As, and Cd in agricultural soils (Jo & Koh, 2004), which could be due to several reasons such as with stringent emission controls, advanced industrial waste management, and limited local emission sources. These lower concentrations are not solely due to reduced atmospheric deposition but also reflect effective regulatory enforcement and the spatial attenuation of deposition from point sources with increasing distance. Long-range atmospheric transport may still contribute to low background levels of some metals across East Asia, depending on prevailing wind directions and regional emission patterns. It is worth noting, however, that in some regions of Asia, naturally metal-rich parent materials can also result in elevated baseline concentrations, independent of anthropogenic influence.

In Vietnam, contamination patterns are highly heterogeneous (Supplementary Table S1). While some areas remain relatively unaffected, others—particularly those near industrial, urban, or mining zones—have experienced extreme accumulation of HM. For example, peri-urban vegetable-growing areas near Hanoi have reported Pb concentrations of ~ 132.5 mg kg^{-1} and Cd ~ 4.1 mg kg^{-1}, while rice paddies in the Red River Delta show As concentrations of around 20 mg kg^{-1} (Hoang et al., 2021;

Table 2 Concentration ranges of heavy metals in soils across selected Asian countries (mg kg^{-1})

No	Country	Concentration of heavy metals in soil (mg kg^{-1})									Ref
		As	Cd	Co	Cr	Cu	Hg	Ni	Pb	Zn	
1	Bangladesh	5.04–23.14	1.67–6.90	-	20.47–59.09	20.50–74.99	-	20.09–69.13	31.81–67.65	47.46–128.06	Islam et al., 2016)
2	China (Huabei plain)	2.30–12.80	0.22–1.10	-	35.0–154.2	6.70–80.70	0.01–0.27	-	11.30–57.00	20.70–223.4	Huang & Jin, 2008)
3	China (Huanghuai plain)	3.07–22.30	0.06–0.52	-	47.70–361.00	-	0.01–0.67	9.74–230.00	14.30–48.60	34.10–619.00	Zhou et al., 2014)
4	China (Sanjiang plain)	-	0.10–0.36	4.87–19.40	38.60–61.30	13.20–27.80	-	13.90–30.10	13.30–22.90	21.40–62.00	Shan et al., 2013)
5	India	-	0.11–1.10	-	3.60–111.70	4.05–67.60	-	12.80–165.00	5.60–69.60	20.60–282.00	Chabukdhara et al., 2016)
6	Indonesia	-	36.60–148.40	-	-	9.60–10.80	-	-	-	52.40–78.20	Herawati et al., 2000)
7	Japan	-	403.40–601.80	-	-	17.80–25.30	-	-	-	83.20–112.03	Herawati et al., 2000)
8	Malaysia	0.28–56.07	0.28–56.70	0.05–6.70	1.10–60.90	0.37–47.30	0.37–47.30	0.40–41.30	0.85–65.00	2.90–92.00	Zarcinas et al., 2004a)
9	Nepal	-	0.21–0.34	-	-	23.76–29.32	-	-	20.96–31.81	75.42–101.75	Yan et al., 2012)
10	Pakistan	-	1.60–4.30	-	28.10–34.60	11.20–32.20	-	2.20–6.20	21.10–67.40	104.60–208.60	Jamali et al., 2007)
11	South Korea	0.00–5.62	0.00–1.28	-	0.00–4.98	0.00–78.24	0.01–0.54	0.00–60.24	0.00–66.44	0.00–252.00	Jo & Koh, 2004)
12	ThaiLand	0.08–29.00	0.01–0.17	0.10–21.1	0.14–79.40	0.16–43.60	0.01–0.10	0.10–43.90	0.10–54.60	0.10–71.00	Zarcinas et al., 2004b)
13	UAE	0.01–0.01	0.46–0.69	0.03–0.37	43.43–71.55	10.29–21.70	-	32.86–52.12	2.83–8.84	42.39 – 66.92	Al-Taani et al., 2021)

Huong et al., 2010), all of which exceed the national permissible limits for agricultural soils (70 mg kg^{-1} for Pb, 2 mg kg^{-1} for Cd, and 2 mg kg^{-1} for As) (Ministry of Science and Technology, 2002). Notably, areas affected by mining activities display exceedingly high metal concentrations. For example, soils near mines in Thai Nguyen Province have accumulated As levels around 3,468 mg kg^{-1} (Anh et al., 2011) and Pb concentrations of approximately 3,476 mg kg^{-1} (Nguyen et al., 2024). Similarly, in the lead–zinc mining regions of Bac Kan Province, Pb and Zn concentrations were as high as ~ 6,209 and ~ 14,030 mg kg^{-1}, respectively (Ha et al., 2011). These values drastically exceed the national agricultural soil quality standards, indicating strong anthropogenic contamination from mining operations. Conversely, in agricultural regions distant from heavy industries, HM concentrations typically align with natural geogenic backgrounds. For example, paddy soils in Ha Giang Province show negligible As content and only approximately 0.05 mg kg^{-1} of Cd, consistent with natural geochemical baselines (Tra & Egashira, 2001). Similarly, rice cultivation areas in the Mekong Delta generally have HM concentrations below hazardous thresholds and close to natural background values (Chu et al., 2010).

3.3 Toward Remediation: Need for Sustainable Solutions

Addressing HM contamination in vegetable production requires comprehensive and proactive strategies. Key measures include regular monitoring of soil and crop metal concentrations alongside the enforcement of stringent regulations to control industrial emissions and ensure the safe disposal of agro-industrial wastes. In instances of contamination, remediation techniques must be employed to immobilize or remove metals from the soil. Among these techniques, the application of organic soil amendments such as compost and BC has garnered attention for their potential to sequester metals and restore soil health (Ibrahim et al., 2023; Medyńska-Juraszek et al., 2022). BC, produced from agricultural waste via pyrolysis, offers a promising, low-cost, locally available solution aligned with circular economy principles. The next sections of this review will explore the production and characteristics of BC, its mechanisms for immobilizing metals in soil, and its effects on reducing HM uptake in vegetable crops.

4 Biochar as a Tool for HM Remediation in Vegetable-Growing Soils

BC is a carbon-rich material produced through the thermal decomposition (pyrolysis) of organic biomass under oxygen-limited conditions. It has attracted considerable attention as a sustainable solution for remediating soils contaminated with HMs, particularly in regions where vegetable growth is a concern, such as in the food industry (Blenis et al., 2023; Lehmann & Joseph, 2015). The effectiveness of BC in immobilizing HMs is attributed to its advantageous physicochemical properties, including a high carbon content, extensive surface area, high cation exchange capacity (CEC), alkaline pH, and persistent aromatic structure (Sachdeva et al., 2023). These properties facilitate the sorption and retention of metal ions while simultaneously enhancing soil quality (Ahmad et al., 2014a; He et al., 2019). These properties are not static; they are significantly influenced by the choice of feedstock and the conditions under which BC is produced. By selecting suitable biomass sources and optimizing production parameters, BC can be tailored to enhance the immobilization of HMs to specific contaminants or soil conditions. Consequently, BC has proven to be a versatile tool for soil remediation, with applications ranging from industrialized nations to local farms in developing countries, such as Vietnam.

4.1 Biomass Availability and BC Production Potential

Supplementary Table S2, which summarizes global and regional agricultural productivity and biomass residue potential for major crops (2003–2023) based on FAOSTAT data, shows that global agriculture generates approximately 5.5 billion tonnes (Gt) of crop residues annually, with an estimated BC yield of ~ 21%, translating to a theoretical global BC potential of 1.16 Gt per year. Cereal crops, particularly rice and maize, contribute the bulk of this biomass. Southeast Asia alone produces 520 million tonnes (Mt) of agricultural residues each year—about 10% of the global total—offering a regional BC potential of 109 Mt. Rice farming dominates, accounting for 271 Mt of residues or one-quarter of global rice-derived biomass. Vietnam contributes 88–160 Mt, primarily from rice, followed by maize, sugarcane, and industrial crops such as coffee and cashew. However, current practices like open field burning result in significant feedstock losses and environmental harm (Bhattacharyya et al., 2021). Harnessing these residues for BC production represents a major opportunity for waste valorization, pollution reduction, and soil remediation within circular bioeconomy frameworks (Lentini et al., 2025).

The abundant availability of biomass residues all over the world presents considerable opportunities for the advancement of sustainable agriculture and environmental management. The transformation of agricultural waste into BC offers dual advantages: it valorizes waste, thereby mitigating open burning or landfilling, and it produces a stable soil amendment that enhances soil health. The diverse array of available feedstocks, each characterized by unique chemical compositions, plays a crucial role in determining the properties and remediation effectiveness of the resultant BC. Biomass with elevated cellulose and hemicellulose contents, such as rice straw and corn stalks, generally produces BC with a highly porous structure after pyrolysis. This porosity enhances the surface area and adsorption capacity for contaminants (Deng et al., 2016; Marrugo et al., 2016). For instance, BC derived from rice straw, which typically contains significant ash content and functional groups, has been demonstrated to effectively reduce the bioavailability of Cd, Pb, and Zn in contaminated paddy soils by increasing the soil pH and adsorbing metals onto its surface (Dang et al., 2018). In contrast, feedstocks rich in lignin, such as woody materials or coconut shells, yield BCs with greater aromatic carbon stability and carbon content but often exhibit fewer surface functional groups, potentially limiting their capacity for certain types of chemical adsorption (Volpe et al., 2018; Zhang et al., 2015). Feedstocks with a high ash content, such as rice husks, can produce BC with an elevated mineral content that facilitates the precipitation of metals as insoluble compounds (Wang et al., 2017). Recognizing these variations, researchers have explored modified BCs, such as those produced with chemical additives or post-treatment, to enhance their performance. Notably, BC produced from cotton stalks pre-treated with $MgCl_2$ demonstrated a phosphate removal capacity of approximately 130 mg g^{-1}, while pepper stem BC achieved approximately 131 mg g^{-1} Pb adsorption, both significantly higher than their unmodified counterparts (Park et al., 2016; Yu et al., 2016). These modifications illustrate the potential to tailor BC properties, such as by enriching it with specific functional groups or minerals, for targeted remediation of particular pollutants.

4.2 Production Processes and Properties of BC

BC can be synthesized through various thermochemical processes, including pyrolysis, gasification, and hydrothermal carbonization (HTC). Pyrolysis is the predominant method for producing BC for soil amendment. In a typical slow pyrolysis process, biomass is subjected to temperatures ranging from 300 °C to 900 °C in an oxygen-limited environment, resulting in the formation of BC along with co-products such as syngas and bio-oil (Lehmann & Joseph, 2015). Gasification occurs at higher temperatures with increased oxygen availability, which favors the production of gaseous fuels and yields lower quantities of BC. HTC involves the reaction of biomass in hot compressed water (200–300 °C) to produce coal-like char and is particularly suitable for wet feedstocks such as manure or bagasse (Funke & Ziegler, 2010). Thus, the selection of a production technique is often aligned with feedstock characteristics: moisture-rich residues common in tropical regions may be processed via lower-temperature or hydrothermal routes, whereas dry lignin-rich residues (e.g., woody wastes or coconut shells) can be efficiently converted at higher pyrolysis temperatures to produce BCs with greater carbon stability. The conditions under which the BC is produced significantly influence its properties. Slow pyrolysis at moderate temperatures is often chosen to maximize the BC yield and preserve the nutrient content, which is beneficial for soil remediation applications (Babu et al., 2023). Conversely, fast pyrolysis or very high temperatures may enhance the surface area but result in reduced char yield and potentially degrade some functional groups.

The key production parameters include the temperature, heating rate, and residence time. Low pyrolysis temperatures (e.g., 300–400 °C) typically yield BCs with a higher concentration of surface functional groups, such as carboxyl (–COOH) and hydroxyl (–OH) groups, and an increased volatile matter content, which can enhance the CEC and chemical adsorption capacity (Parthasarathy et al., 2023; Şensöz & Can, 2002). For example, the retention of oxygen-containing functional groups at lower temperatures augments the capacity of BC to bind metal cations via surface complexation. Conversely, high pyrolysis temperatures (500–900 °C) result in more complete carbonization of the biomass, producing BCs with increased surface areas and porosities but reduced quantities of functional groups. Surface areas can increase significantly at elevated temperatures; for instance, a durian-wood BC produced at 650 °C exhibited a surface area of approximately 221 $m^2 g^{-1}$ (Chowdhury et al., 2016; Weber & Quicker, 2018). Employing activation techniques, such as physical activation with steam or CO_2 or chemical activation with KOH, can further enhance the surface area to the thousands of $m^2 g^{-1}$ (Bashir et al., 2018; Kumar et al., 2020), which is beneficial for pollutant adsorption. However, highly porous BCs may possess reduced nutrient content and fewer functional groups, indicating a trade-off between the surface area and chemical functionality. Prolonged residence times during pyrolysis can increase the aromatic carbon fraction and enhance the stability of BC, thereby increasing its persistence in soils. Thus, the optimal production parameters are contingent upon the intended application. For HM immobilization, a balance must be achieved to ensure sufficient adsorption sites (surface area and functional groups) and adequate alkalinity and CEC.

The physicochemical properties of BC play a crucial role in its interactions with HMs. The key properties include (i) pH, as BCs generally exhibit an alkaline pH due to ash content, which can neutralize acidic soils; (ii) CEC, which refers to the ability to exchange cations such as Ca^{2+} and K^+ with metal ions in the soil solution; (iii) surface area and porosity, which are related to the capacity for physical adsorption of metal ions or complexes; (iv) surface functional groups, which provide binding sites for metals through complexation; and (v) mineral content, which can result in the precipitation of metals as minerals. For instance, typical BCs have a pH ranging from approximately 6 to 12, depending on the feedstock and temperature, often imparting alkalinity that can precipitate metals as hydroxides or carbonates (Ahmad et al., 2014a; Zhang et al., 2013). BC also commonly exhibits significant CEC due to negative surface charges, enabling it to sequester metal cations by exchanging with benign cations such as Ca^{2+} or Mg^{2+} originally present on the BC (Shakoor et al., 2020; Zhang et al., 2020). The point of zero charge (PZC) of a BC, which is the pH at which the net surface charge is zero, determines whether its surface is negatively or positively charged at a given soil pH, thereby influencing its attraction or repulsion between metal cations and anions (Munera-Echeverri et al., 2018). By carefully controlling the feedstock choice and production conditions, it is possible to produce BCs optimized for high CEC, appropriate pore structure, and abundant functional groups, thereby facilitating the trapping of HMs.

4.3 Physicochemical Mechanisms of BC

BC mitigates the bioavailability of HMs in soil through a series of interconnected physicochemical mechanisms. The principal mechanisms encompass adsorption (both physical and chemical), ion exchange, surface complexation, and precipitation of metals into stable mineral forms (Fig. 1, Supplementary Table S3). These processes often operate synergistically when BC is introduced into contaminated soils (Guo et al., 2020). Following BC amendment, the soil pH generally increases, particularly in acidic soils, owing to the alkaline nature of many BCs. This elevation in pH can significantly reduce the solubility of numerous metal cations by facilitating the formation of insoluble hydroxides, carbonates, or phosphate minerals (Zhang et al., 2013). BC surfaces, which contain various oxygen-bearing functional groups (such as carboxyl, hydroxyl, and carbonyl groups), can bind metal ions through coordination bonds, a process referred to as surface complexation. Metals such as Cu^{2+} and Zn^{2+} readily form complexes with these functional groups, and low-temperature BCs with a greater abundance of such groups tend to exhibit higher capacities for this mechanism (Pan et al., 2022; Uchimiya et al., 2012; Zhu et al., 2018).

BC is characterized by high CEC levels, which enable it to carry exchangeable cations, such as Ca^{2+}, Mg^{2+}, and K^+, on its surface. These cations can be exchanged with HM ions present in the soil solution (Fahmi et al., 2018; Shakoor et al., 2020). This ion exchange process not only facilitates the removal of metals from the soil solution but also immobilizes them in less bioavailable forms within the BC matrix. Additionally, physical adsorption plays a significant

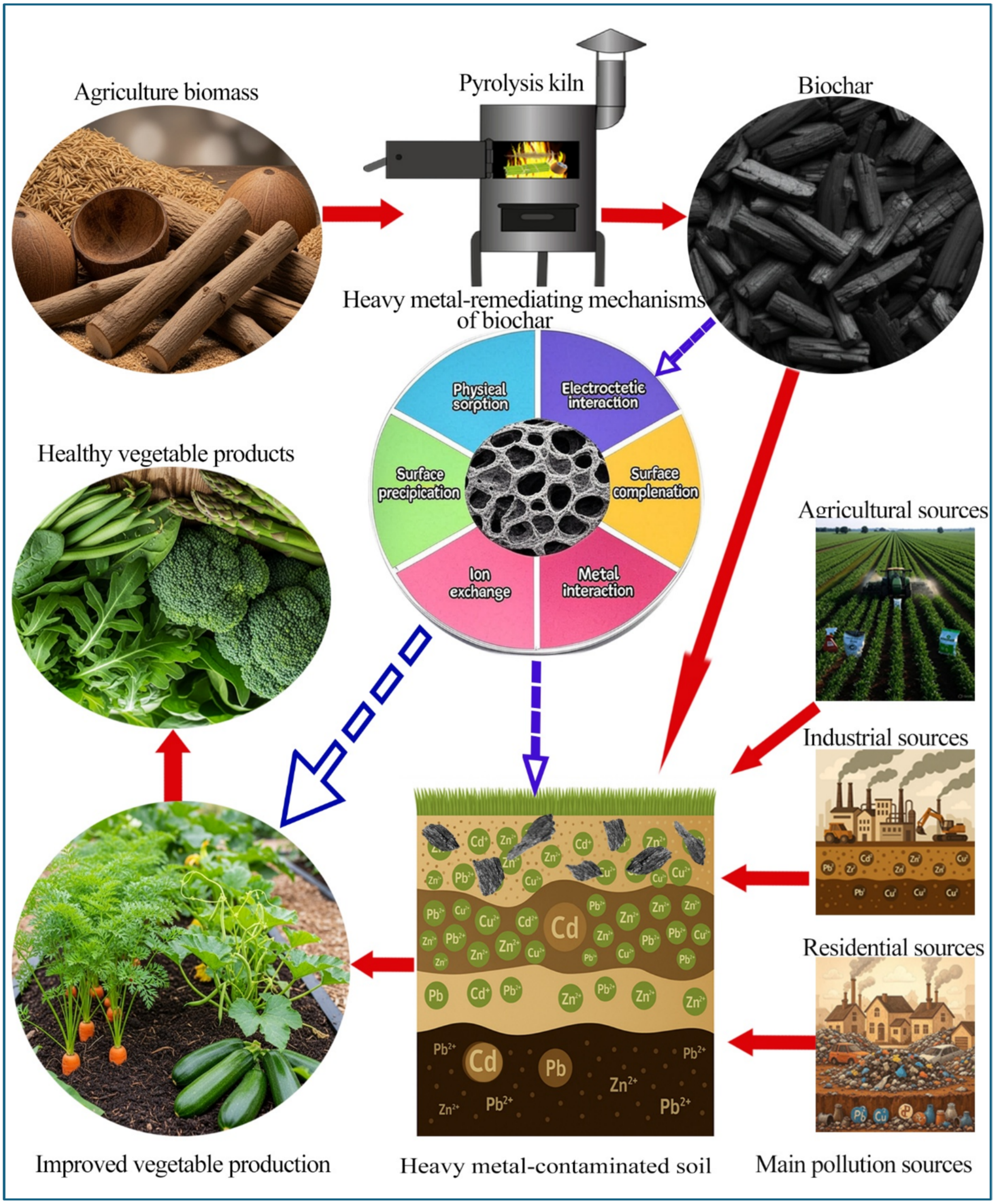

Fig. 1 The pathway of biochar from plant biomass and its impacts and mechanisms on remediating heavy metal-contaminated soil used for vegetable cultivation. Note that red arrows are pathways, and green arrows are the potential impacts or mechanisms related to biochar

role. The porous structure of BC offers extensive surfaces and pore spaces where metal ions or metal-bearing molecules, such as organometallic complexes, can adhere through van der Waals forces or electrostatic attraction (Fan et al., 2023a; Qi et al., 2017). The increased porosity and surface area at higher pyrolysis temperatures further enhance the physical trapping of metals. Furthermore, the mineral constituents of BC derived from the original biomass ash can react with metals to form compounds with low solubility. For example, BCs containing silica, carbonates, or phosphates can induce the precipitation of metals, such as silicates, carbonates, or phosphate minerals (Alozie et al., 2018; Wang et al., 2017).

The mechanisms described collectively diminish the proportion of exchangeable or dissolved HMs, which are readily absorbed by plants, and enhance the proportion that is bound or fixed within the soil matrix. It is important to note that metals do not respond uniformly to BC amendment. Typically, the efficacy of immobilization is most pronounced for Pb, which exhibits strong adsorption and precipitation, followed by Zn and Cd, with a somewhat reduced effect observed for Cu (Namgay et al., 2010; Yang et al., 2021). This hierarchy is attributed to the distinct chemical properties of each metal; for instance, Cu^{2+} often demonstrates a higher affinity for organic matter and may persist in organo-complex forms even after BC application. Additionally, it is crucial to consider that while cationic metals become less bioavailable under more alkaline conditions induced by BC application, anionic species such as arsenate (As(V)) or chromate (Cr(VI)) may exhibit increased mobility if the soil pH increases because of diminished adsorption or competition with other anions for binding sites (Guo et al., 2020; Lebrun et al., 2018).

5 Effects of BC on HM Uptake in Vegetables

The application of BC as a soil amendment has garnered growing attention for its potential to reduce the bioavailability of HMs in contaminated agricultural soils, particularly in vegetable production systems (Wu et al., 2025). Vegetables are among the most sensitive crops to HM uptake due to their rapid growth, high water content, and frequent consumption in fresh form, thereby posing significant risks to food safety and human health (Shetty et al., 2025). BC can mitigate these risks by altering soil properties and immobilizing HMs through multiple mechanisms, including surface adsorption, ion exchange, complexation with functional groups, and pH-induced precipitation (Chen et al., 2024a). Numerous studies have demonstrated that BC amendments can significantly reduce the accumulation of Cd, Pb, As, and other toxic metals in edible plant tissues (Irshad et al., 2022). The extent of metal uptake reduction depends on several factors, including BC feedstock, pyrolysis temperature, application rate, and the type of vegetable grown (Lahori et al., 2017). This section critically reviews experimental evidence on BC's effectiveness in lowering HM concentrations in commonly cultivated vegetables, with a focus on leafy greens, root crops, and fruiting vegetables. Special attention is given to the underlying mechanisms, crop-specific responses, and contextual variables that influence BC performance in real-world agricultural settings.

5.1 Metal Uptake Mechanisms in Vegetable Plants

Vegetable plants possess sophisticated biochemical systems that evolved over millions of years to acquire HMs from contaminated soils (Manegabe et al., 2025). Although these mechanisms evolved to regulate the uptake of essential micronutrients such as Fe and Zn, they can inadvertently transport toxic metals like Cd, Pb, and Hg into plant tissues (Mathur & Chauhan, 2020), thereby introducing them into the food chain. This process occurs through three primary phases: root uptake, intracellular processing, and systemic transport to aerial tissues (Zhang et al., 2024). Understanding these mechanisms is crucial not only for food safety assessment but also for developing phytoremediation technologies that utilize plants to decontaminate polluted soils.

The root system serves as the primary interface for HM acquisition and represents the most critical control point in the entire uptake process (Jia et al., 2022). Specialized membrane transport proteins function as selective channels for metal entry, with the Zinc-Regulated Transporter/Iron-Regulated Transporter (ZIP/IRT) family serving as the predominant pathway for metal influx from soil solution into root cells (Ajeesh Krishna et al., 2020). These transporters operate through electrochemical gradients and exhibit broad substrate specificity, simultaneously transporting essential metals such as iron alongside toxic metals such as Cd and Pb (Williams et al., 2000; Zhao et al., 2022). Under iron-deficient conditions, plants upregulate these transporters, inadvertently increasing HM uptake capacity (Morrissey & Guerinot, 2009). Additionally, root systems actively modify the rhizosphere chemistry through secretion of low-molecular-weight organic acids including citrate, malate, and oxalate, which function to solubilize soil-bound metals and enhance their bioavailability (Oburger et al., 2009). The root cell wall matrix, particularly pectin components rich in galacturonic acid residues, provides initial metal-binding sites that sequester metal cations through electrostatic interactions before they reach cellular transport systems.

Upon metal entry into root cells, plants immediately activate sophisticated detoxification cascades to mitigate cellular damage and facilitate safe metal handling (Hasan et al., 2017). The primary defense mechanism involves the synthesis of phytochelatins, small peptides with the general structure (γ-glutamyl-cysteine)n-glycine, which are produced from glutathione by the enzyme phytochelatin synthase in response to direct exposure to HMs (Faizan et al., 2024). These peptides coordinate with metal ions through sulfur-metal bonds, forming stable complexes that neutralize metal toxicity while preparing metals for compartmentalization (Seregin & Kozhevnikova, 2023). Following chelation, metal-phytochelatin complexes are actively transported into the vacuolar compartment through specialized tonoplast transporters, including ATP-binding Cassette (ABCC) proteins and HM ATPases, where metals are maintained in chemically inert forms isolated from metabolically active cellular compartments (Park et al., 2012). Subsequently, HM translocation from roots to shoots occurs primarily through the xylem transport system, driven by transpiration-induced mass flow, with xylem loading representing a critical regulatory checkpoint mediated by HM ATPases HMA2 and HMA4 that actively pump metals into xylem vessels against concentration gradients (Ceasar et al., 2020; Wong et al., 2009). During transport, metals are typically complexed with organic ligands such as citrate, histidine, or nicotianamine to maintain solubility, while the Yellow Stripe-Like (YSL) transporter family facilitates the movement of nicotianamine-metal complexes through both xylem and phloem systems (Curie et al., 2009). This integrated transport system results in leafy vegetables generally exhibiting higher metal concentrations than root or fruit vegetables due to their extensive transpiring leaf surface areas and direct exposure to xylem-transported metals (Zhou et al., 2016), with significant implications for food safety and dietary exposure risks.

5.2 Mechanistic Interactions Between BC and Vegetable Metal Uptake Processes

The effectiveness of BC in remediating HMs contaminated soils depends fundamentally on complex interactions between BC's physical and chemical modifications to soil environments and the sophisticated physiological and molecular processes of vegetables to absorb HMs. These interactions create intricate feedback mechanisms that can either synergistically enhance or antagonistically reduce remediation efficiency, extending far beyond simple additive effects.

5.2.1 BC Modulation of Soil Chemical Environments and Plant Metal Transport Systems

BC application creates cascading changes in soil chemistry that directly interact with plant metal transport mechanisms at the molecular level. The alkaline nature of most BCs increases soil pH, which simultaneously decreases HM solubility and modifies the expression patterns of metal transport proteins in plant roots (Chen et al., 2023; Majewska & Hanaka, 2025). This pH-mediated mechanism operates through multiple pathways: elevated pH shifts metal speciation toward less bioavailable forms while concurrently down-regulating the expression of broad-spectrum metal transporters such as ZIP (Zinc-Regulated Transporter/Iron-Regulated Transporter) and NRAMP (Natural Resistance-Associated Macrophage Protein) families (Khan et al., 2025; Sun et al., 2023). Studies demonstrate that BC amendment can reduce bioavailable concentrations of Cd, Pb, Cu, and Zn while simultaneously suppressing the expression of metal uptake genes, creating a dual protective mechanism (Chen et al., 2018; Haider et al., 2024).

The high cation exchange capacity of BC creates nutrient competition effects that exploit the non-specific nature of plant metal transporters. Essential nutrients like Zn and Fe compete with toxic metals such as Cd and Pb for the same transport proteins, with increased availability of beneficial metals reducing toxic metal uptake through competitive inhibition (Ghassemi-Golezani et al., 2025). This mechanism proves particularly effective when BC derived from nutrient-rich feedstocks (such as manure) provides sustained release of competing cations (Chen et al., 2018). Simultaneously, BC's surface functional groups, including carboxyl, hydroxyl, and aromatic structures, directly adsorb metal ions through complexation, electrostatic attraction, and π-electron interactions, reducing the pool of metals available for plant uptake (Manikandan et al., 2023).

5.2.2 Rhizosphere Microbiome Interactions and Metal Bioavailability

The addition of BC fundamentally alters rhizosphere microbial communities in ways that create complex feedback loops affecting metal bioavailability. BC increases bacterial diversity and modifies community structure by providing colonization sites, nutrient sources, and pH buffering (Wang et al., 2022; Yan et al., 2022). These microbial changes interact with plant metal uptake through multiple mechanisms: BC-stimulated bacteria can immobilize metals through biosorption and biomineralization processes, while also producing organic acids that may either increase or decrease metal solubility depending on environmental conditions (Fan et al., 2023b). The interaction between BC and root exudates creates particularly complex dynamics. While plants naturally secrete low-molecular-weight organic acids (LMWOAs) such as citrate and malate to mobilize nutrients, BC can buffer the pH changes caused by these acids, potentially reducing their metal-mobilizing effects (Alozie et al., 2018). However, BC also provides electron transfer support between microorganisms and root exudates, enhancing the efficiency of microbial metal reduction and immobilization processes (Fan et al., 2023b). This creates a feedback system where BC simultaneously reduces the plant's need to exude organic acids while enhancing the efficiency of beneficial microbial processes.

5.2.3 Molecular-Level Interactions with Plant Detoxification Systems

At the cellular level, BC influences plant metal detoxification through complex interactions with phytochelatin synthesis and vacuolar sequestration systems. While BC reduces external metal exposure, it may also affect the expression of genes encoding phytochelatin synthase and ABC transporters responsible for vacuolar metal sequestration (Cobbett, 2000; Mendoza-Cózatl et al., 2011). Studies indicate that BC amendment can reduce the expression of HM transporter genes while maintaining or enhancing the plant's internal detoxification capacity (Haider et al., 2024; Khan et al., 2025). This suggests that BC allows plants to maintain their detoxification machinery while reducing the burden on these systems through external metal immobilization. Moreover, the tonoplast transporters, particularly ABCC-type transporters and HM ATPases (HMA), play crucial roles in vacuolar metal sequestration (Mendoza-Cózatl et al., 2011; Sharma et al., 2016). BC application may influence the efficiency of these transporters through changes in root cell ion concentrations and membrane properties, though the specific mechanisms remain under investigation (Sharma et al., 2016). The interaction between BC-mediated external metal reduction and internal cellular transport systems represents a critical area for optimizing remediation strategies.

5.3 Effects and Influencing Factors of BC on HM Behavior in Soil and Uptake by Vegetables

5.3.1 Introduction

Vegetables primarily absorb HMs from the soil through their root systems, with a lesser extent of absorption occurring via foliar deposition of airborne particles (Xu et al., 2022). Vegetables, such as lettuce (*Lactuca sativa*), spinach (*Spinacia oleracea*), amaranth (*Amaranthus spp.*), and Brassica species (e.g., *mustard greens, Brassica juncea, and cabbage*), are particularly susceptible to HM accumulation, due to their extensive root systems and large leaf surfaces that can intercept metal-containing dust (Xu et al., 2022; Zhou et al., 2016). Empirical evidence suggests that HM concentrations generally follow the trend of leafy vegetables > root/tuber vegetables > fruiting vegetables (e.g., tomatoes and peppers) in the same contaminated soil (Zhou et al., 2016). Consequently, reducing HM uptake in leafy vegetables is a priority to ensure food safety. In this context, BC has emerged as an effective soil amendment that mitigates HM accumulation in vegetables by decreasing metal bioavailability, improving soil chemical properties, and modifying soil–plant interactions. For example, in one study, chicken-manure-derived BC applied at 10% (w/w) to a highly Pb-contaminated soil (1,000 mg kg^{-1} Pb) reduced the Pb concentration in mustard green shoots to as low as ~ 2 mg kg^{-1}, whereas in untreated soil, the shoots contained toxic levels of Pb (Park et al., 2011).

5.3.2 Mechanistic Pathways of Biochar Influencing HM Uptake

Reduction of Metal Bioavailability in Soil One of the primary mechanistic pathways by which BC reduces HM uptake in vegetables is through decreasing metal bioavailability in soil. BC applications typically increase soil pH (Nguyen et al., 2021a), promoting the precipitation of metals such as Cd and Pb as carbonates and hydroxides, thus limiting their solubility and plant availability (Hasan et al., 2024). The surface of BC contains abundant functional groups (–COOH, –OH) that facilitate physical sorption and surface complexation, forming stable metal–ligand bonds that immobilize metals (Hasan et al., 2024). Furthermore, the porous structure of BC provides a large surface area for electrostatic interactions and ion exchange, particularly between metal cations (e.g., $Cd2^{+}$, $Pb2^{+}$, $Zn2^{+}$) and exchangeable base cations on BC surfaces (e.g., $Mg2^{+}$, K^{+}) (Qian et al., 2024). BC also alters the soil redox potential (Eh), leading to the reduction of mobile and toxic species such as As(V) and Cr(VI) into less bioavailable forms (Choppala et al., 2016). Collectively, these physicochemical processes—sorption, ion exchange, and precipitation—act synergistically to reduce HM mobility in soil, thereby decreasing their uptake by vegetable roots (Fig. 1).

Immobilization via Mineral and Organic Phases BC also immobilizes HMs through interactions with its mineral and organic components. BCs derived from mineral-rich feedstocks—such as rice husk, poultry manure, or sewage sludge—contain oxides and carbonates of Ca, Mg, Fe, and Al that facilitate surface precipitation and metal–mineral complexation (Wu et al., 2019a). These minerals promote the formation of metal–phosphate, metal–carbonate, and metal–oxide compounds, which are highly insoluble under most agricultural soil conditions (Basta & McGowen, 2004). Concurrently, organic functional groups on BC surfaces contribute to electrostatic binding and surface complexation with dissolved metal ions. For example, poultry-manure-derived BC exhibits strong metal–mineral interactions, effectively immobilizing Cd and Pb through co-precipitation and coordination with phosphate and carbonate phases (Nguyen et al., 2021a). Similarly, rice husk BC, enriched in silica and alkali metals, provides multiple binding sites for physical sorption and surface precipitation, enhancing long-term stabilization of HMs (Huang et al., 2020). Together, these mineral–organic mechanisms contribute to sustained reductions in HM mobility and plant uptake in vegetable systems.

Modification of Rhizosphere Chemistry BC modifies rhizosphere chemistry in ways that further restrict HM availability and root absorption. Through its buffering effect, BC stabilizes soil pH and redox potential, influencing metal speciation and promoting the conversion of toxic soluble forms into more stable, adsorbed states (Teng et al., 2025). The presence of BC stimulates microbial activity and diversity, enhancing microbially mediated transformations such as As(V) ↔ As(III) reduction–oxidation cycling and Fe/Mn oxide formation (Wu et al., 2019b), which favor metal immobilization through surface complexation and precipitation. BC also influences root exudation patterns (Gu et al., 2022), where organic acids and ligands interact with BC surfaces to form stable metal–organic complexes, further reducing metal uptake (Fan et al., 2023b). Additionally, the release of base cations ($Ca2^{+}$, K^{+}, $Mg2^{+}$) from BC competes with toxic metal ions (e.g., $Cd2^{+}$, $Pb2^{+}$) for adsorption and root binding sites via electrostatic interactions and ion exchange. These coupled biogeochemical processes in the rhizosphere significantly mitigate HM mobility and limit their translocation from soil to plant tissues.

Effects on Plant Physiology and Root Uptake Beyond soil interactions, BC affects HM uptake by influencing plant physiology and root processes (Wan et al., 2023). By improving soil structure, aeration, and moisture retention (Abbas et al., 2024), BC reduces oxidative stress and enhances root health (Chen et al., 2021), which strengthens the root barrier function and limits passive metal entry. The enhanced soil environment promotes root exudation of organic ligands and silicon release, which can bind or co-precipitate with metals at the root–soil interface, forming a protective barrier through surface complexation and metal interaction. Moreover, BC-induced improvements in nutrient status and microbial symbiosis lead to increased biomass (Feng et al., 2021), generating a dilution effect that lowers HM concentrations in edible tissues. Variations in BC effects are often observed between leafy and root vegetables, largely due to differences in metal uptake pathways and root morphology. Overall, the combination of electrostatic stabilization, ion exchange, and surface precipitation (Anbuganesan et al., 2024a) at the root interface explains BC's effectiveness in reducing metal accumulation in vegetables while simultaneously improving plant growth and soil health.

5.3.3 Biochar Influencing Transfer Factors

The influence of BC on HM uptake by vegetables essentially reflects its impact on the crop's transfer factor (TF), which indicates the efficiency of metal transfer from soil to plant tissues. Transfer factors (TFs), defined as the ratio of HM concentration in plant tissue to that in soil, provide a vital measure of metal mobility and the risk of entry into the food chain. BC influences TFs by altering key soil–plant processes: by reducing metal bioavailability, modifying root-zone chemistry, and altering plant uptake pathways (as discussed above), BC can significantly lower the proportion of metals translocated into vegetables. For example, specific studies report that BC-based amendments significantly reduced the transfer factors of Pb and Cd, with the extent of reduction generally increasing with applied BC rates (Antonangelo & Zhang, 2019). A meta-analysis found that BC application reduced HM concentrations in plant tissues by approximately 17–39% compared to untreated soils (Joseph et al., 2021). The TFs also vary among elements, with Pb typically exhibiting higher TF values than Cd (Letey et al., 2025). BC application has been shown to substantially reduce the uptake of both metals in lettuce, highlighting its effectiveness in limiting their translocation from soil to plant tissues (Letey et al., 2025). In lettuce cultivated in soil contaminated with Cd and Pb, the addition of an appropriate BC resulted in a 57% reduction in shoot Cd concentration, a 75% reduction in Zn, and a 63% reduction in Pb, with concomitant decreases in metal levels in the roots as well (Zheng et al., 2017). Nonetheless, considerable variability exists, as TF reductions are influenced by BC feedstock type, pyrolysis temperature, application rate, soil properties, and application duration. For instance, the effectiveness of BC in reducing HM uptake declines over successive harvests due to surface protonation and metal desorption, with weaker effects observed for Cu, Zn, and Pb than for Ni and Cd (Qin et al., 2022). Consequently, understanding how BC alters TFs across metals and crop systems is essential for designing effective remediation strategies.

5.3.4 Factors Influencing BC Performance

BC effectiveness in reducing HM uptake is strongly influenced by several interrelated factors (Table 3, Supplementary Table S3) and feedstock types and pyrolysis conditions among the important factors determining BC performance (Zhao et al., 2020). Feedstock type determines key BC properties such as mineral composition and surface functional groups. For example, wood-based biochar has higher surface area, enhancing soil physical structure while crop- and grass-based BCs improve nutrient retention via greater cation exchange capacity (Ippolito et al., 2020). The second factor can be involved in pyrolysis temperature, which further shapes BC structure. High-temperature BCs exhibit larger surface area, greater microporosity, and higher aromaticity, which improve adsorption through surface complexation and pore filling (Ahmad et al., 2014b); whereas low-temperature BCs retain abundant oxygen-containing functional groups that facilitate metal binding via electrostatic attraction, ion exchange, and precipitation (Ahmad et al., 2014b; Hu et al., 2022). A meta-analysis shows BC produced at moderate temperatures (400–550 °C) achieved higher adsorption for Cd, Pb and Cu (Ghorbani & Amirahmadi, 2025). Among seven tested BCs (from cotton stalks, rice straw, poultry manure, lawn grass, vegetable peels, maize straw, and rice husks), rice husk, poultry manure, and maize straw BCs most effectively reduced Cd (by 33–34%), Pb (by 41–51%), and Ni (by 63%) concentrations in wheat grain, while rice straw BC enhanced plant P and K uptake (Amin et al., 2023). Garden waste-derived BC produced at a higher pyrolysis temperature (600 °C) resulted in lower concentrations of Cd, Pb, Zn, and Cu in the shoots

Table 3 Effects of different biochar types, pyrolysis temperature, and application rates on heavy metal concentrations in soil and vegetables

No	Biomass type	Pyrolysis Temp. (°C)	biochar rate	Vegetable species	Heavy metals	Soil metal concentrations (mg kg^{-1})	Vegetable metal concentrations (mg kg^{-1})	Effects on vegetables and metal uptake	Key conclusions	Ref
1	Garden waste (GB400)	400	6% (w/w)	Brassica juncea (Mustard greens)	Cd, Pb, Zn, Cu	Before: Cd:2.5; Pb:979.75; Zn:1018.61; Cu:33.27 After: Cd:0.73; Pb:265.25; Zn:77.56; Cu:4.06 (DTPA)	Shoot: Cd:1.21; Pb:1.63; Zn:218.15; Cu:4.88 Root: Cd:1.61; Pb:146.14; Zn:204.22; Cu:9.93	Increased biomass; reduced metal uptake	GB400 reduced bioavailability of Cd, Pb, Cu effectively, enhancing plant growth	Awad et al., 2020)
2	Garden waste (GB600)	600	6% (w/w)	Brassica juncea (Mustard greens)	Cd, Pb, Zn, Cu	Same as GB400; After: Cd:0.84; Pb:267.23; Zn:77.36; Cu:4.07 (DTPA)	Shoot: Cd:0.93; Pb:1.17; Zn:10.95; Cu:4.34 Root: Cd:1.32; Pb:166.97; Zn:231.72; Cu:11.20	Increased biomass; better metal reduction	GB600 was more effective than GB400 in reducing Pb, Zn, Cu availability	
3	Paulownia wood	700–800	6% (w/w)	Brassica juncea (Mustard greens)	Cd, Pb, Zn, Cu	Same as GB400; After: Cd ~ 0.98; Pb ~ 280.12; Zn ~ 85.23; Cu ~ 5.12 (DTPA)	Shoot: Cd:1.29; Pb:2.28; Zn:193.15; Cu:4.64 Root: Cd:2.12; Pb:222.07; Zn:194.28; Cu:9.58	Increased biomass; moderate metal reduction	Paulownia biochar less effective than GB in Cd and Pb reduction	
4	Bamboo	700–800	6% (w/w)	Brassica juncea (Mustard greens)	Cd, Pb, Zn, Cu	Same as GB400; After: Cd ~ 1.10; Pb ~ 300.45; Zn ~ 90.34; Cu ~ 6.20 (DTPA)	Shoot: Cd:3.61; Pb:11.60; Zn:424.90; Cu:10.11 Root: Cd:4.69; Pb:172.78; Zn:600.77; Cu:13.88	Increased biomass; limited metal reduction	Bamboo biochar was least effective in reducing metal bioavailability	

Table 3 (continued)

No	Biomass type	Pyrolysis Temp. (°C)	biochar rate	Vegetable species	Heavy metals	Soil metal concentrations (mg kg^{-1})	Vegetable metal concentrations (mg kg^{-1})	Effects on vegetables and metal uptake	Key conclusions	Ref
5	Rabbit manure	550	10–30% (compost mix)	Lactuca sativa, Spinacia oleracea, Valerianella locusta, Brassica spp. (Leafy vegetable)	Cu, Cr, Cd, Pb	N/A	Leaf only: reduced Cd, Pb uptake; potential Cr increase	Improved growth; reduced Cd/Pb uptake, potential Cr risk	Rabbit manure biochar reduces Cd/Pb but may increase Cr	Medyńska-Juraszek et al., 2022)
6	Sugarcane bagasse*	300–600	5–10%	N/A	Pb, Zn (Fraction F1)**	Pb:3928.4 → 285.3–350 (F1); Zn:1789.8 → 120.5–182.9 (F1)	N/A	Reduced exchangeable Pb/Zn fraction	Sugarcane biochar reduces exchangeable (F1) Pb, Zn fractions	Vuong et al., 2025)
7	Wheat straw	550	5–10%	Raphanus sativus, Lactuca sativa, Anethum graveolens, Spinacia oleracea, Petroselinum crispum (Leafy vegetables)	Cu, Pb, Zn, Cd, Cr, Ni	Before: Cu:321; Pb:174; Zn:32.4; Cd:6.2; Cr:8.9; Ni:9.2	N/A	Reduced vegetable metal uptake, more effective at higher dosage	Wheat straw biochar reduced heavy metal uptake, 10% dose most effective	(Kim et al. 2015, Medyńska-Juraszek et al. 2020)
8	Chicken manure	550	1–15%	Brassica juncea (Mustard greens)	Cd, Cu, Pb	Before: Cd:5; Cu:160; Pb:1000 After: N/A	Shoot: Cd:0.88–3.5; Cu:8.2–10; Pb:1.8–7.5 Root: Cd:2.5; Cu:50–106; Pb:112–282	Increased biomass; reduced metal uptake	Chicken manure biochar enhanced biomass significantly, effective metal immobilization	Park et al., 2011)

Table 3 (continued)

No	Biomass type	Pyrolysis Temp. (°C)	biochar rate	Vegetable species	Heavy metals	Soil metal concentrations (mg kg^{-1})	Vegetable metal concentrations (mg kg^{-1})	Effects on vegetables and metal uptake	Key conclusions	Ref
9	Unidentified biochar	N/A	0–20%	Spinacia oleracea (Spinach)	As, Cd, Cr, Cu, Ni, Pb, Zn	Before: As:28.79; Cd:1.56; Cr:115.31; Cu:71.2; Ni:61.97; Pb:25.39; Zn:124.28; After: Reduced concentrations	Shoot: As:0.2–0.8; Cd:0.1–0.3; Cr:0.5–1.5; Pb:0.1–0.3; Root: N/A	Metal reduction; health risk remains	Reduced soil/vegetable metals; supplementary measures required	Xiang et al., 2021)

DTPA extraction: Bioavailable soil metal fraction determined by diethylenetriaminepentaacetic acid analysis; *N/A*: Data unavailable from original studies

of mustard greens compared with BC produced at a lower temperature (400 °C) (Table 3). Moreover, BCs derived from crop residues and produced at moderate pyrolysis temperatures often provide an optimal balance between surface functionality and structural stability, thereby enhancing their effectiveness in immobilizing and adsorbing a broad range of HMs.

Application rate is also an important factor determining the effectiveness of BC in remediating HMs in soil and reducing their uptake by vegetables. Beyond a certain application rate, the incremental benefit of additional BC often diminishes as metal uptake reduction plateaus (Antonangelo et al., 2023), likely because the bioavailable metal pool becomes largely depleted or other growth-limiting factors emerge. Excessive BC can even lead to nutrient imbalances, such as nitrogen immobilization, which may stress plants (Nguyen et al., 2023) and indirectly affect metal uptake. For instance, in one experiment, adding more than 5% BC did not further reduce vegetable metal concentrations; instead, it led to signs of nutrient deficiency in plants (Rees et al., 2015).

Metal characteristics influence outcomes and BC amendments generally exert the most significant effect on reducing Pb uptake in vegetables, followed by Cd and Zn, whereas Cu sometimes exhibits smaller reductions or even slight increases in plant tissue under specific conditions. The pronounced efficacy of Pb is attributed to robust sorption and precipitation mechanisms, whereas the complex behavior of Cu may be due to the influence of BC on soil organic matter dynamics (Rees et al., 2015). Finally, plant variety is also significant: cultivars bred for high HM uptake, such as those used in phytoremediation, may accumulate more metals even with the addition of BC. This is evidenced in metal-hyperaccumulating Brassica cultivars, which absorb more Zn and Cd when BC is present, possibly because BC mitigates toxicity, allowing the plant to grow more and continue accumulating (Nehnevajova et al., 2007). In contrast, common cultivars of lettuce can exhibit a wide range of responses to BC, with some showing a strong reduction in metal uptake and others showing less so (Lamb et al., 2010).

5.3.5 Indirect and Secondary Effects

BC amendments can influence plant growth and quality, which in turn can indirectly affect metal uptake

(Wang et al., 2020). Enhancements in soil fertility and structure, such as improved water retention, aeration, and nutrient supply resulting from BC, often lead to increased plant biomass (Kang et al., 2022). This growth can lead to a dilution effect on contaminants within plant tissues, as there is more biomass per unit of metal absorbed. For example, BC has been reported to significantly increase the yield of certain vegetables; one study documented a 116% increase in Swiss chard biomass with BC application (Rivelli & Libutti, 2022). In many instances, plants cultivated in BC-amended soils not only contain less HMs per gram but also produce more edible biomass, thereby amplifying the benefit.

5.3.6 BC Co-Application with other Soil Amendments

Co-application of BC with complementary amendments often outperforms single-agent treatments for immobilizing HMs and protecting vegetable safety. Co-application of BC with compost has shown strong synergistic effects—while compost enriches soil organic matter and nutrients, BC's porous structure and high cation exchange capacity help retain these nutrients and immobilize HMs (Chafik et al., 2025). The application of BC combined with compost reduced Pb uptake in *Brassica napus* while simultaneously enhancing its growth, yield, and oil content by improving soil properties and plant physiological and biochemical functions (Jiang et al., 2024). Adding alkaline minerals (lime, ash, crushed rock) to BC raises pH and promotes metal precipitation as carbonates/hydroxides, substantially strengthening immobilization of Pb and Cd and inhibit it uptake be vegetables in Acidic red soil of the Oxisol type (Huang et al., 2024). Targeted BC modifications or blends (Fe-impregnated BC) markedly improve retention of As and Cd cations by promoting the formation of iron plaque in paddy fields (Wei et al., 2024). A meta-analysis shows that the combinations of BC with zeolites can further lock metals into insoluble phases and reduce the availability of Cd, Cu, Pb, and Zn in soils by 32.6%, 54.3%, 35.4%, and 18.3%, respectively (Viana et al., 2025). Potential trade-offs include nutrient imbalance, salinity from low-quality composts/BCs, and pH overcorrection, so co-amendment design must consider contaminant speciation, vegetable species, and local soil properties.

5.3.7 The Potential Adverse Impacts of BCs

Despite its benefits in HM immobilization, BC can unintentionally increase soil HM load or impair vegetable safety if feedstock and production conditions are not carefully managed. For example, BC derived from sewage sludge contains elevated concentrations of Cd, Cr, Ni, and Cu, which may become further concentrated during the pyrolysis process as organic mass is lost and ash content rises (Kujawska et al., 2024). Similarly, studies report that sludge-based biochars may enrich soil with trace metals upon application (Kujawska et al., 2024; Vali et al., 2025), posing risks to plant and human health. Feedstocks such as animal manure, biogas digester residues, and municipal biosolids tend to contain higher HM loads than clean lignocellulosic materials (e.g., wood chips or crop residues), making BC from waste streams more hazardous without proper treatment. In another greenhouse study, rabbit-manure-derived BC combined with compost reduced Cd and Pb uptake in lettuce and spinach but unexpectedly increased Cr uptake in some vegetables (Medyńska-Juraszek et al., 2022), implicating BC ash constituents in converting Cr into more plant-available forms. This underscores the importance of selecting uncontaminated feedstocks, applying rigorous pretreatment or co-pyrolysis strategies (e.g., biomass and sludge blends) to reduce HM content, and adhering to certification standards for BC use in vegetable production systems (Mohamed et al., 2023).

In addition, several studies have shown that, under certain environmental conditions, BC can increase the availability of specific metals and metalloids in soil, although BC is generally recognized for its ability to immobilize HMs. When BC—especially high-ash or alkaline types derived from waste feedstocks—raises soil pH or alters redox conditions, it may trigger the desorption or re-solubilization of HMs such as As and Cd. For example, a meta-analysis found that certain BCs at high application rates increased As availability under field conditions (Mandal et al., 2024). The effectiveness of biochar in reducing HM uptake tends to decline after several cropping cycles due to gradual surface protonation, which leads to the desorption of previously adsorbed metals. Research indicates that this weakening effect is more pronounced for Cu, Zn, and Pb than for Ni and Cd, suggesting that over time, metals once immobilized by biochar may be remobilized, thereby increasing their bioavailable concentrations in soil (Qin et al., 2022). Biochar application effectively reduces the mobility of As and Cd in paddy soils; however, in upland conditions, it may increase As availability (Wei et al., 2024), possibly due to higher soil aeration and oxidative conversion of As(III) to the more mobile As(V) form. These cases highlight the dynamic nature of BC–metal interactions as they age in soil and illustrate the risk that BC may inadvertently enhance HM mobility under fluctuating conditions (e.g., pH shifts, moisture changes, redox swings).

6 Scientific and Technical Challenges in BC Applications

6.1 Biochar-Based Circular of Vegetable Production

Overall, Fig. 1 illustrates the conversion of agricultural biomass into BC through pyrolysis and its subsequent application as a soil amendment for HM-contaminated agricultural systems. The BC intervention strategy demonstrates how this carbon-rich material effectively immobilizes HMs in soil through multiple physicochemical mechanisms including surface adsorption, cation exchange reactions, and metal–organic surface complexation. By reducing metal mobility and bioavailability in the rhizosphere, BC application significantly limits HM uptake by plant roots and subsequent translocation to edible tissues. This remediation approach simultaneously improves soil physicochemical properties while creating safer growing conditions for vegetable production in contaminated environments.

6.2 Variability of BC Properties

Although BC exhibits considerable potential in mitigating the uptake of HMs by crops, several practical challenges and limitations warrant attention. A primary challenge is the inherent variability of BC materials. Given that BC can be produced from a diverse array of feedstocks under varying pyrolysis conditions, its properties are inconsistent. These BCs may exhibit substantial differences in pH, surface area, nutrient content, and sorption capacity for contaminants. This variability complicates standardization efforts and can result in inconsistent outcomes in field applications (Almutairi et al., 2023; Gezahegn et al., 2019). For instance, a farmer utilizing BC derived from rice husks at 350 °C may experience different effects than those using hardwood BC at 700 °C. This variability complicates efforts to standardize BC products, hindering the establishment of uniform quality criteria and predictable performance in reducing HM bioavailability. Developing standardized protocols, potentially through organizations like the International BC Initiative, could help ensure that BCs meet minimum safety and efficacy standards for agricultural use.

6.3 Environmental and Agronomic Considerations

BC's high adsorption capacity, while beneficial for immobilizing HMs, can also lead to unintended consequences, such as the immobilization of essential nutrients, potentially causing deficiencies in crops (Buss et al., 2018). For example, BC can adsorb nitrogen in the form of ammonium or nitrate, thereby reducing its immediate availability to plants, which may necessitate adjustments in fertilization practices. Some studies have reported initial reductions in crop yield when BC is applied without adequate nutrient management, although yields often improve in subsequent seasons as the system reaches equilibrium (Biederman & Harpole, 2013). Another environmental consideration is the potential for BC dust or particulate matter to contribute to air quality issues during application or be lost through erosion if not adequately incorporated into the soil. Appropriate handling techniques, such as moistening the BC before field application, can help mitigate this risk.

6.4 Long-Term Stability and Persistence

A significant challenge is the long-term stability and persistence of BC effects under field conditions. Although BC is generally considered stable, it is not inert; over extended periods, it can undergo gradual oxidation, altering its surface properties. Field experiments have shown that the benefits of BC, including its reduced metal bioavailability, may diminish over time. Sui et al. (Sui et al., 2018) found that in a three-year trial, during wetter periods, the uptake of Cd and Pb by wheat increased despite the presence of BC, suggesting that fluctuating conditions, such as wet-dry cycles, can influence BC performance. These findings suggest that the capacity of BC to immobilize metals may decline as it weathers and that periodic reapplication or complementary measures may be necessary to maintain its long-term effectiveness. More long-term field data, beyond the 1–3 year experiments, are needed, particularly in tropical climates where high temperatures and moisture could accelerate BC aging. The lack of long-term data in regions such as southeast Asia creates uncertainty about how often BC needs to be reapplied to sustain its benefits (Hagemann et al., 2017; Nguyen et al., 2021b).

6.5 Soil Microbial Communities and Long-Term Soil Health

Soil microbial communities play a fundamental role in regulating nutrient cycling, contaminant transformation, and soil resilience (Chen et al., 2024b), especially in vegetable systems where HM contamination suppresses microbial activity and reduces soil fertility (Manegabe et al., 2025). BC has been widely recognized for improving microbial abundance, diversity, and enzymatic activity in HM-contaminated soils through enhanced habitat structure, pH buffering, and the provision of labile carbon substrates (Anbuganesan et al., 2024b). For example, Moradi and Karimi (Moradi & Karimi, 2021) reports that raw biochar and Fe-modified biochar increased microbial biomass carbon by 40.5–75.1% and dehydrogenase activity by 25.5–102.1% in Cd-contaminated soil, while Zhu et al. (Zhu et al., 2022) found that biochar treatment significantly enhanced microbial diversity and network complexity—particularly among rare and abundant taxa—thereby strengthening soil microbial community resilience to Cd stress. Over the long term, these positive shifts may improve soil structure, organic carbon sequestration, aggregation, and resilience, contributing to healthier soil ecosystems. Nevertheless, it is essential to acknowledge that excessive BC application can lead to ecological imbalances. While moderate BC levels promote beneficial biota, very high rates may alter soil pH and disrupt native microbial communities. Jia et al. (Jia et al., 2024) observed reduced AMF root colonization at BC rates exceeding 5%, potentially negating some plant benefits. Such outcomes underscore the need to balance BC's metal-immobilizing capacity with its biological effects.

Despite these advances, challenges remain: many studies are short-term, and the aging of BC in soil, potential for metal remobilization, and long-term dynamics of microbial succession are poorly understood. For example, after decades, soil charcoal effects on microbial communities may be overridden by land-use changes. Other major knowledge gaps remain regarding BC's long-term effects on microbial succession, enzyme dynamics, and the linkages between microbial processes and HM uptake in vegetables. Addressing these uncertainties is essential to ensure that BC amendments not only immobilize metals but also sustain long-term soil microbial health and vegetable productivity and quality.

7 Economic and Environmental Challenges and Future Perspectives for Global Adoption

7.1 Economic Feasibility for Global Adoption

Despite its demonstrated potential, the large-scale adoption of BC in vegetable production remains constrained by economic, logistical, and policy-related challenges, requiring context-specific strategies to enhance feasibility and sustainability. Biochar production costs vary widely with feedstock, reactor type, scale, and energy recovery; recent analyses report typical production costs from about USD 116–197 per tonne for optimized systems (Saharudin et al., 2024). Logistical constraints—including the labor-intensive transport and field application of BC—further restrict its widespread use (Bergman et al., 2022). The agronomic co-benefits (increased crop yield by 14.45%, water use efficiency by 14.28% and nitrogen use efficiency by 13.97% (Han et al., 2023)) and reduced fertilizer needs can offset initial investment over several seasons, improving the return on investment particularly when biomass residues are locally available. Developing decentralized pyrolysis systems for on-site conversion of local crop residues, as demonstrated in UNIDO pilot projects, can substantially reduce costs and promote community participation (Koumpakis et al., 2025). However, the scalability of these systems depends on initial capital investment, technical capacity building, and local infrastructure. Strong policy integration is also crucial, particularly through the inclusion of BC within national soil health programs, carbon sequestration initiatives, and sustainable agriculture policies, which can incentivize adoption and align BC use with climate mitigation goals. Financial instruments such as subsidies, low-interest credit, and carbon credit schemes could further improve economic viability (Pourhashem et al., 2019). Establishing international quality and safety standards—for example, those modeled after the European Biochar Certificate—would ensure consistency and foster confidence among users and investors. Major barriers remain: high upfront capital, variability in biochar quality and price, limited access to affordable pyrolysis technology for smallholders, and insufficient policy recognition—issues that must be addressed for scalable deployment in vegetable production.

7.2 Life-Cycle Environmental Impacts

From a life-cycle perspective, BC production and application offer both environmental benefits and potential trade-offs. During pyrolysis, BC systems emit CO_2 and trace CH_4 but sequester carbon in a stable form, achieving a net greenhouse gas (GHG) reduction potential of -1.12 to -1.2 t CO_2-eq per tonne of BC produced (Gamaralalage et al., 2025). This balance depends on feedstock, technology, and energy efficiency. Co-production of bioenergy (syngas, bio-oil) enhances energy recovery and can displace fossil fuel use (Shoudho et al., 2024), improving the environmental return of BC systems. Feedstock sourcing plays a critical role in overall sustainability. Using agricultural and forestry residues avoids land-use change and supports circular bioeconomy principles (Samuel et al., 2024), while various organic wastes—such as animal manures, sewage sludge, and food-processing residues—can also be utilized for biochar production (Jagadeesh & Sundaram, 2023), thereby reducing environmental pollution pressure and contributing to sustainable waste management. However, pollution risks persist: volatile organics (PAHs, NO_x) and trace metals may be released from low-quality feedstocks such as sewage sludge or manure (Dong et al., 2025). Over time, BC contributes to soil carbon accumulation, microbial diversity, and nutrient retention. Nonetheless, aging processes may alter surface chemistry, leading to metal remobilization and potential secondary contamination under fluctuating pH or redox conditions (Qin et al., 2022). Thus, long-term field monitoring and life-cycle assessments are essential to balance BC's carbon sequestration benefits with possible ecological trade-offs.

8 Conclusions

This review underscores the urgent need to address heavy metal (HM) contamination in agricultural soils, particularly in vegetable production systems where dietary exposure poses immediate public health risks. Leafy and root vegetables are especially vulnerable, as they tend to accumulate metals such as cadmium (Cd), lead (Pb), and zinc (Zn) at levels that often exceed food safety thresholds. These challenges are exacerbated in regions where wastewater irrigation, agrochemical overuse, and industrial activities are prevalent. Biochar (BC) has emerged as a promising remediation strategy to mitigate these threats. Through mechanisms such as adsorption, ion exchange, and precipitation, BC can effectively immobilize HMs in soils, thereby reducing their uptake by vegetables and enhancing soil quality. Its remediation performance is strongly influenced by feedstock type, pyrolysis conditions, and resultant properties such as pH, surface area, and functional groups. Experimental evidence shows that BC amendments can reduce metal concentrations in edible plant parts, while improving soil pH, microbial activity, and nutrient retention. However, translating these benefits into field-scale vegetable systems requires overcoming several challenges. The variability of BC materials, risks of nutrient immobilization, uncertain long-term stability, and high production and logistics costs remain key barriers. Inconsistent results across studies also point to the need for crop-specific and soil-specific application strategies. To realize BC's full potential for safer vegetable production, future efforts should focus on long-term field trials, development of BCs tailored to vegetable systems, and integration with complementary practices such as balanced fertilization and microbial inoculants. Policy support, economic incentives, and quality certification standards will be crucial for enabling adoption, particularly in low-resource settings. With coordinated scientific, technical, and institutional efforts, BC can play a pivotal role in transforming contaminated lands into productive and food-safe vegetable farming systems.

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