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REVIEW ARTICLE

Ecological consequences of biochar and hydrochar amendments in soil: assessing environmental impacts and influences

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Abstract
Anthropogenic activities have caused irreversible consequences on our planet, including climate change and environmental pollution. Nevertheless, reducing greenhouse gas (GHG) emissions and capturing carbon can mitigate global warming. Biochar and hydrochar are increasingly used for soil remediation due to their stable adsorption qualities. As soil amendments, these materials improve soil quality and reduce water loss, prevent cracking and shrinkage, and interact with microbial communities, resulting in a promising treatment method for reducing gas emissions from the top layer of soil. However, during long-term studies, contradictory results were found, suggesting that higher biochar application rates led to higher soil CO_2 effluxes, biodiversity loss, an increase in invasive species, and changes in nutrient cycling. Hydrochar, generated through hydrothermal carbonization, might be less stable when introduced into the soil, which could lead to heightened GHG emissions due to quicker carbon breakdown and increased microbial activity. On the other hand, biochar, created via pyrolysis, demonstrates stability and can beneficially impact GHG emissions. Biochar could be the preferred red option for carbon sequestration purposes, while hydrochar might be more advantageous for use as a gas adsorbent. This review paper highlights the ecological impact of long-term applications of biochar and hydrochar in soil. In general, using these materials as soil amendments helps establish a sustainable pool of organic carbon, decreasing atmospheric GHG concentration and mitigating the impacts of climate change.

Keywords Greenhouse gas emissions · Bioavailability · Adsorption mechanisms · Carbon storage · Life cycle assessment

Introduction

Soil contamination by toxic elements is a pressing global issue that has far-reaching implications for the environment, human health, and food security (Kong et al. 2021; Qin et al. 2021). While toxic elements can exist naturally, the influence of human activities (e.g., mining, industrial waste disposal, and burning of fossil fuels) also has a substantial negative impact (Palansooriya et al. 2020; De Almeida Ribeiro Carvalho et al. 2022). The high concentrations of toxic elements in industrial waste, including chromium (Cr), zinc (Zn), nickel (Ni), cobalt (Co), cadmium (Cd), copper (Cu), manganese (Mn), and lead (Pb), further exacerbate the problem (Li et al. 2021). Additionally, the contaminants can percolate into deeper layers, eventually reaching the groundwater. Contamination of soil poses a high risk to the ecosystem as the untreated waste released recklessly in the soil environment contains these persistent and non-biodegradable heavy metals (Sun et al. 2020b; Liu et al. 2021).

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Various methods have been developed to remove toxic substances from the soil and remediate soil pollution. These techniques include adsorption, photocatalytic degradation, advanced oxidation, and biodegradation (Yao et al. 2021). Adsorption has become increasingly popular due to its effectiveness and ease of operation (Zou et al. 2020; Luo et al. 2022). Many researchers considered using different materials, particularly biochar and hydrochar, to adsorb and immobilize soil pollutants.

Biochar and hydrochar possess a microporous structure, high aromaticity, plentiful oxygen-containing functional groups, and a substantial specific surface area, and are accessible economically, making them efficient adsorbents for the removal of organic pollutants (Mazarji et al. 2023; Ahmed and Hameed 2018). The economic efficiency of using biochar and hydrochar is determined by the benefits of waste processing, the advantages of carbon sequestration (Dickinson et al. 2015), and the cost savings associated with reduced irrigation expenses (Kroeger et al. 2020). Previous studies investigated the ability of biochar to mitigate organic pollutants such as pesticides, pharmaceuticals, polychlorinated biphenyls, dyes, and polycyclic aromatic hydrocarbons (Liu et al. 2019a). It was reported that charred materials are an effective adsorbent for removing these organic pollutants from wastewater (Ahmed and Hameed 2018). During biochar and hydrochar production, different feedstocks, durations of pyrolysis, and heating temperatures yield products with varying characteristics and adsorption capacities (Chen et al. 2021).

An analysis of published literature was conducted by performing a literature search on peer-reviewed articles published between 2010 and 2022. *Web of Science* database was queried using various keyword combinations: soil remediation; biochar & adsorption; hydrochar & adsorption; biochar & greenhouse gas or carbon dioxide; hydrochar & greenhouse gas or carbon dioxide. According to open data, 2843 articles were published on using biochar, and 341 were published on using hydrochar. Over the past few years, the number of research on biochar and hydrochar (either mechanistic study or practical application) has been rapidly increasing (Fig. 1a). Figure 1b represents a network based on crucial keywords of the literature. Through a thorough examination of the network, one can discern patterns associated with the volume of publications (indicated by the thickness of the lines), the clustering of similar topics, and the identification of influential literature sources. It can be observed that the most significant number of publications is dedicated to studying biochar as soil amendments for remediation more than hydrochar. Most review papers on biochar and hydrochar focused on their influence on pollutant removal and the resulting implications for agriculture and soil quality (Dan et al. 2023; Ji et al. 2022; Mei et al. 2022). Numerous reviews are dedicated to comparing biochar and hydrochar on their production characteristics (Masoumi et al. 2021; Cavali et al. 2022; Kumar et al. 2020; Fu et al. 2019).

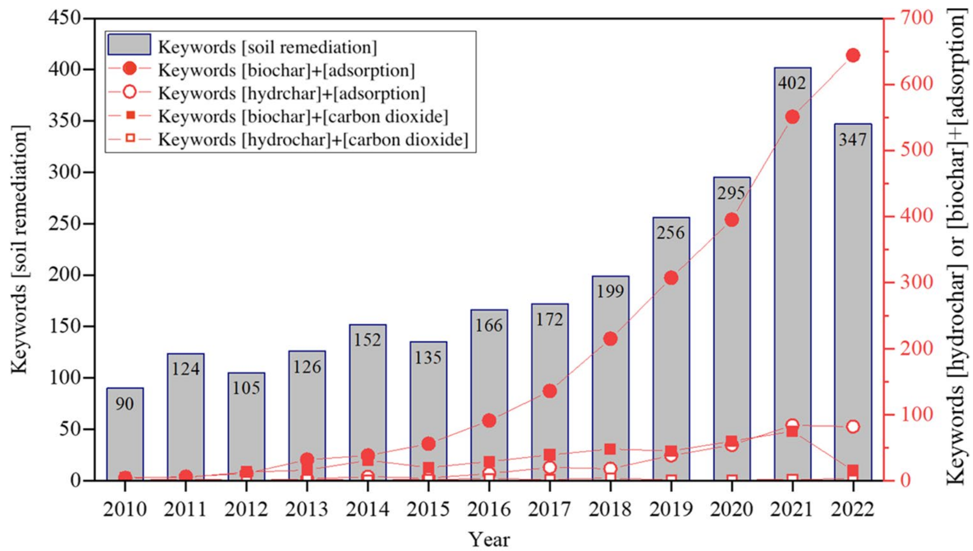
This surge in publications and interest is deeply connected to the ecological benefits of using biochar and hydrochar, particularly in soil applications. It is known that soil is critical in regulating the climate as it is both a source of greenhouse gas (GHG) emissions and a sink for carbon (Lal et al. 2021). The rise in the global average temperature is a consequence of the intensification of the greenhouse effect caused by the escalation of GHG concentration in the atmosphere. Since the carbon pool is susceptible to climate change, rising air temperatures increase GHG emissions from the soil (Crowther et al. 2016).

Biochar and hydrochar amended in soils have also been considered qualified candidates to capture more carbon and reduce GHG emissions. Biochar and hydrochar as soil amendments exert direct and indirect control over nitrification and denitrification processes by influencing various environmental factors involved in microbial nitrogen cycling, inorganic nitrogen uptake, and soil moisture (Chen et al. 2020). These alterations subsequently impact nitrogen transformation and the release of N_2O. Also, the soil amendments improve soil structure and accelerate the formation of soil aggregates, thus protecting soil organic carbon and potentially reducing CO_2 emissions.

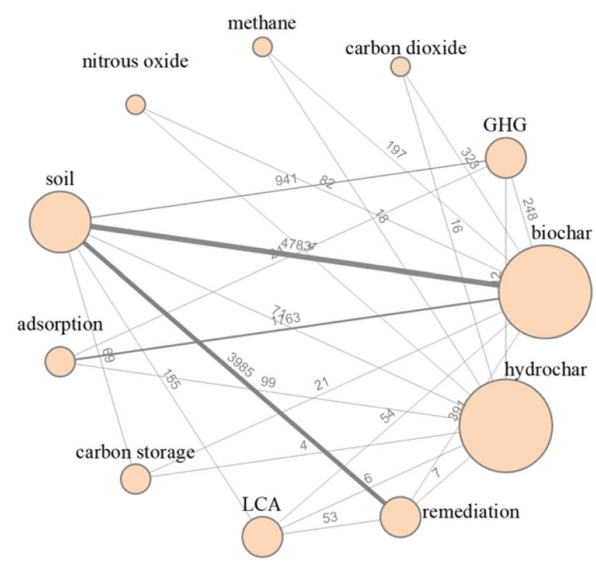
It is worth noting that soil aggregates are subject to change, and the rate of evaporation due to the introduction of biochar contributes to the reduction of cracks in the soil (Kravchenko et al. 2023). In the end, this also affects GHG emissions. Biochar is often used to reduce soil degradation and increase mesoporosity and water availability in the soil (Bordoloi et al. 2019). In addition, numerous studies have revealed that biochar enhances soil fertility, leading to sustainable crop yields (Yu et al. 2019; Palansooriya et al. 2019). This is attributed to various factors, including improved soil structure, increased water retention capacity, and the activation of microbial processes. Due to its inherent chemical stability and slow decomposition rate in soil, soil application of biochar has emerged as one of the most effective approaches to mitigate climate change.

Nevertheless, the long-term potential of biochar and hydrochar in mitigating climate change by reducing gas emissions from the soil has not been adequately evaluated. In addition, the long-term effects of using biochar and hydrochar have not been comprehensively assessed in terms of nutrient loss, biodiversity loss, invasive species, soil health, and fertility. To enable the

Fig. 1 Publication trends and network analysis: from 2010 to 2022 (according to the search results of *Web of Science*): **a** Number of publications by keywords and titles, **b** network of using biochar and hydrochar for soil amendments in selected literature



(a) Number of publications by keywords and titles



(b) Network of using biochar and hydrochar for soil amendments in selected literature

practical application of these materials, it is necessary to comprehensively assess their potential for soil remediation and their ability to emit or sequester GHG from the soil. This presents a significant research challenge that must be addressed to fully understand the ecological consequences of biochar and hydrochar amendments in soil. By thoroughly examining the available literature, this paper provides valuable insights into the adsorption capacity, selectivity, and mechanisms of action of biochar and hydrochar, for soil remediation and climate change mitigation. Additionally, it assesses the environmental impacts and influences associated with biochar and hydrochar amendments.

Production and properties of biochar and hydrochar

Production technologies

Biochar can be obtained from dry raw materials with a moisture content of less than 10% using several well-known pyrolysis technologies, such as slow pyrolysis, fast pyrolysis, intermediate pyrolysis, and microwave (Safarian et al. 2022). The initial biomass must be pre-dried, which increases the production cost by incorporating additional equipment into the production cycle. However, the energy requirement for biomass drying can be met through the energy and heat generated during the pyrolysis process, making the system less energy-intensive (Safarian 2023). During the pyrolysis process, biochar's chemical and physical characteristics undergo influence from several factors, such as the maximum temperature applied, the duration of heating, and the rate at which the maximum temperature is attained (Tomczyk et al. 2020). According to Blenis et al. (2023), the pyrolysis temperature can reach up to 1100 °C, with the combustion time of biomass depending on the assigned temperature, ranging from 1 to 5 s for fast pyrolysis to several days for slow pyrolysis.

Hydrothermal carbonization (HTC) has been developed to address the energy efficiency issue, which eliminates the need for pre-drying biomass and allows the production of hydrochar at lower temperatures and in a shorter period. HTC is a process distinct from pyrolysis and gasification methods commonly employed for biomass with low moisture content (Zhang et al. 2019). It is better suited for treating biomass with high moisture content (Lee et al. 2018). HTC offers a cost-effective means of producing char, as it involves low temperatures of 180–300 °C and operates under pressure in water (Zhang et al. 2019). The key factors influencing the product characteristics are reaction temperature, pressure, residence time, and water-to-biomass ratio. The desired outcome of the HTC process is hydrochar, which typically yields around 50–84 wt% (Hu et al. 2023a). At lower temperatures of HTC, mineral substances in the feedstock can dissolve in the aqueous phase, decreasing ash concentration in the hydrochar (Gao et al. 2018).

However, as the temperature increases, more organic compounds are transferred to the water phase, leading to a higher concentration of inorganic substances in the hydrochar, thus increasing the ash content (Djandja et al. 2023). The temperature-dependent surface properties of the hydrochar are also influenced. For instance, elevated temperatures promote the dissociation of water into OH^- and H_3O^+ ions, which can impact the colloidal stability and surface tension of the hydrochar spheres. This, in turn, may affect the presence of specific functional groups, the fusion of spheres, and the smoothness of the surface (Ischia et al. 2022). In summary, previous studies have consistently demonstrated that the hydrothermal carbonization temperature has a significant effect on various properties of the hydrochar. Polymerization and recombination reactions are likely initiated around 185 °C, while a temperature of approximately 220 °C is suggested to achieve improved hydrochar properties and control the reaction.

Physicochemical properties of biochar and hydrochar

Morphological properties and surface area

The morphological properties of biochar and hydrochar play a crucial role in nutrient adsorption, hydrologic processes, and soil density when used for land application. In the study of Fu et al. (2019), the physical alterations of food wastes and their corresponding biochar and hydrochar showed varying patterns depending on the feedstock, carbonization method, and peak temperature. The biochar produced at 200 °C showed a smooth surface structure, but as the temperature increased, obvious pore and crack structures were developed. In contrast, hydrochar exhibited a spherical-shaped structure, which differed among feedstocks. For instance, hydrochar produced from fish residue showed spherical-shaped structures under low temperatures, which fused as the temperature increased. The reaction mechanism for the formation of these carbonaceous particles in hydrochar is highly complex due to the presence of subcritical water, which initiates the hydrolysis of biomacromolecules, followed by dehydration, condensation, polymerization, and other reactions (Kambo and Dutta 2015).

The surface area and porosity of biochar are crucial factors that determine the quantity and quality of active sites available in biochar (Leng et al. 2021). The availability of active sites is essential for biochar's ability to interact with the environment and improve soil health. As stated by Kumar et al. (2020), the volatilization of organic materials during the production process of biochar creates voids, resulting in a porous structure with a high surface area and large pore volume. Zhang et al. (2021b) determined that the surface area of biochar derived from peanut shells at 700 °C can reach 448 m^2g^{-1}. The hydrochar exhibits a relatively lower surface area due to the deposition of depolymerized products during the production process (Kumar et al. 2020).

Ash content, volatile matter, and fixed carbon

Fixed carbon in biochar refers to the portion of carbon that remains in the material after volatile matter has been driven off through processes such as pyrolysis. During this process, volatile compounds like water, gases, and some organic substances are released, leaving behind a more carbon-rich and stable residue known as biochar. The ash content of biochar and hydrochar can vary due to differences in their production processes. Depending on the proposed application, the ash content of both chars will play an essential role in the adsorption processes, as it affects cation exchanges and electrostatic interactions (Fu et al. 2019). According to Fu et al. (2019), the ash content of biochar typically rises with higher peak temperatures due to the concentration of minerals and combustion residues from organic matter after pyrolysis. In contrast, hydrochar produced under the same peak temperatures showed similar or lower ash contents, as the inorganic components would be solubilized and leached out of the solid product, reducing the overall ash content.

Moreover, as summarized by Kumar et al. (2020), the volatiles range of biochar and hydrochar is 12.3–60.6% and

49.3–88.6%, respectively, while a fixed carbon content of 10.7–86.3% in biochar based on a dry basis and 3.0–47.1% in hydrochar was observed. Both char materials have a low amount of volatiles and a high amount of fixed carbon compared to their feedstock. A decrease in volatile matter was also observed for both chars as temperature increased. This can be attributed to the degradation of organic components, the release of volatiles, and their further transformations (Fu et al. 2019).

Elemental composition

The main elemental composition of chars includes carbon (C), hydrogen (H), oxygen (O), nitrogen, silicon, phosphorus, sulfur, and iron. These elements are present in both hydrochar and biochar in different amounts. The level of carbonization and aromaticity in biochar and hydrochar can be assessed using the char's H/C ratio, where lower H/C ratios typically suggest a higher degree of aromatic structure within the char (Li et al. 2023). The H/C ratio is commonly used as a proxy for biochar and hydrochar stability due to its association with increased resistance to microbial decomposition in soil, which is linked to higher aromatic structure (Burgeon et al. 2021). Due to higher reaction rates of decarboxylation during HTC, higher H/C and O/C ratios are observed for hydrochar (Kambo and Dutta 2015). Furthermore, it was observed that as the peak temperature increased, the H/C ratio decreased, indicating enhanced aromaticity and stability of both biochar and hydrochar produced at higher temperatures (Fu et al. 2019). According to Wang et al. (2018) and Zhang et al. (2020b), the H/C molar ratio for biochar and hydrochar is > 1.5 and > 2.3, respectively. Masoumi et al. (2021) mentioned that the O/C molar ratio for biochar and hydrochar is > 0.7 and > 1.7, respectively. Additionally, the presence of a substantial number of oxygen-containing groups was also observed in hydrochar, attributed to its affinity for water, making it beneficial for increasing soil water retention capacity (Zhang et al. 2019).

Functional groups

Functional groups play a crucial role in determining the properties of biochar and hydrochar and are essential for removing contaminants by affecting sorption and electron transfer (Sun et al. 2018). High temperatures (500–700 °C) have lower O-containing functional groups for pyrolysis. This leads to increased stability of biochar towards microbial and chemical degradation compared to hydrochar (Khosravi et al. 2022). Studies have shown that different food wastes and their corresponding biochar and hydrochar have similar functional group types but differ in some featured functional groups. O-containing functional groups, such as hydroxyls, carboxyls, ketones, and ethers, are primarily derived from the hydrolysis, dehydration, condensation, and polymerization of organic components, such as carbohydrates and lignins, during HTC (Fu et al. 2019; Sun et al. 2018; Liu et al. 2019a). This is evident in the study of Zhang et al. (2020b), in which a hydrochar derived from coffee ground waste at 160 °C contained 13.1–104.0% higher O-containing functional groups than the biochar derived from the same feedstock at 400 °C and 500 °C.

Biochar and hydrochar stability in soil

Biochar aging can lead to acidification that mobilizes soil metals and increases their bioavailability to soil organisms and plants. Furthermore, Wang et al. (2020) suggested that aging can increase the mobility of small biochar particles in the subsurface, potentially resulting in nutrient loss and contaminant migration in soils amended with biochar. Biochar can either positively or negatively affect agriculture in the long term. According to Wang et al. (2021), the natural aging process of hydrochar resulted in altered physicochemical properties, including lower hydrophilic/polarity indices, higher porosity, more significant ash content, and better stability than pristine hydrochar. When hydrochar is directly added to soil, it may act as a slow-release fertilizer, releasing plant-available nutrients during mineralization (Gronwald et al. 2016). This poses the importance of pro-longed biochar and hydrochar assessment, especially those that may harmfully affect the environment and humans. The production parameters and physicochemical properties of biochar and hydrochar are summarized in Table 1.

Soil property change after biochar and hydrochar amendments

Soil density

Numerous research studies have demonstrated the beneficial effects of biochar on soil, including enhancing the physicochemical characteristics, preserving soil organic matter (SOM) levels, improving the efficiency of fertilizer utilization, and ultimately boosting crop yield (Deenik et al. 2011; Jien and Wang 2013). Biochar and hydrochar have been proven effective in reducing soil bulk density in various agricultural soils with different textures, as demonstrated in studies conducted by Abel et al. (2013), Castellini et al. (2015), and Mau et al. (2020).

Mau et al. (2020) emphasized that the treatment temperature of hydrochar influenced the bulk density of soil. It significantly increased from 180 to 220 °C but remained unchanged with further temperature rises to 250 °C. This implies that the particle density of hydrochar increased with temperature treatment ranging from 180 to 220 °C and then

Table 1 Production parameters and physicochemical properties of biochar and hydrochar

Properties	Biochar	Hydrochar	Reference
Highest reaction temperature	1100 °C	300 °C	Blenis et al. 2023; Zhang et al. 2019
Moisture content of feedstock	Dry biomass	Wet biomass	Lee et al. 2018
Total carbon content	58.1–90.1%	39.6–57.6%	Bargmann et al. 2013
Proximate: Volatiles	12.3–60.6%	49.3–88.6%	Kumar et al. 2020
Fixed carbon	10.70–86.37 wt%	2.66–47.1 wt%	Kambo and Dutta 2015
Ash content	0.45–40%	0.23–49.7%	Kambo and Dutta 2015; Zhang et al. 2019; Kumar et al 2020
pH	3–11.3	2–12	Kambo and Dutta 2015; Sun et al. 2020a
Aromaticity	Contains aromatic groups	Contains alkyl moieties	Masoumi et al. 2021
H/C molar ratio	> 1.5	> 2.3	Wang et al. 2018; Zhang et al. 2020b
O/C molar ratio	> 0.7	> 1.7	Masoumi et al. 2021; Wang et al. 2018
Physical: surface area (m^2 g^{-g})	35–448	4–12	Zhang et al. 2021b; Jain et al. 2016
Shape	Non-defined	Spherical and core–shell type	Kumar et al 2020

remained constant up to 250 °C. Additionally, the study conducted by Ghorbani et al. (2023) underscored that corn straw biochar, produced at higher pyrolysis temperatures (450–550 °C), had a notable impact on decreasing soil bulk density and increasing porosity compared to biochar produced at lower temperatures (350 °C). One of the primary factors behind this observation was the limited volatilization of material at lower temperatures, resulting in fewer pores forming and, consequently, reducing the internal volume of the biochar.

Soil hydraulic conductivity and soil organic matter

Adding porous hydrochar and biochar enhances hydraulic conductivity, often leading to increased water retention capacity and a reduced risk of nutrient leaching (Kalderis et al. 2019). Li et al. (2018a) claim that particles of biochar derived from apple branches induce a low intrapore entry pressure within the biochar's pores, resulting in water-permeable intrapores' formation. This phenomenon contributes to an increase in saturated hydraulic conductivity. Additionally, it is argued that the effect of biochar on saturated hydraulic conductivity is not primarily due to its internal porosity but rather to the tortuosity of the pores, which refers to the variations in particle arrangement (Lim et al. 2016). Therefore, the shape of the pores and their interconnectivity can better explain the influence of biochar on soil hydraulic conductivity. The hydrochar may have a similar impact on soil hydraulic conductivity due to its comparable microstructural arrangement of pores. However, there is a lack of specific research examining this aspect.

Zhang et al. (2020a) found that adding biochar reduces quantitative parameters such as the crack ratio, the soil mass, and the fractal dimension of clay soil. It was found that the biochar amendments to the soil led to a 16% decrease in the overall volume fraction of cracks compared to the untreated soil (Kravchenko et al. 2023). This decline in crack volume can be attributed to the water retention capabilities of biochar, which aid in mitigating soil dehydration and shrinkage. Hydrochar and biochar amendments contribute to the increase in the size of soil aggregates, and a dosage of 5% of the soil weight is considered optimal for preventing erosion in highly weathered soils (Jien and Wang 2013). Kravchenko et al. (2024) conducted a laboratory incubation experiment with soil treated with hydrochar to explore its impact on soil crack development. The study demonstrated that adding wood hydrochar at concentrations of 2% and 4% reduced the crack intensity factor by 22% and 43% compared to the untreated control soil. In contrast to the control soil, which showed significant expansion of large cracks and extensive void spaces, the hydrochar-treated soils displayed a greater crack length density with a network of finer cracks. This modification reduced water loss in hydrochar-amended soils due to lower evaporation rates compared to the control. Adding hydrochar to soil creates a secondary peak in pore size distribution (Abel et al. 2013). This peak falls within a range where water is held by capillary and adsorptive forces, making it accessible to plants. Introducing hydrochar increases medium-sized pores while reducing the number of wide pores (Mau et al. 2020). This occurs because the hydrochar fills up the wide pores and has smaller pore sizes (Abel et al. 2013). This was also confirmed by Zhang et al.'s study (2020c) for biochar. However, despite these extensive laboratory incubations aimed at understanding soil crack development, there remains a significant gap in our understanding of how these amendments will perform under field conditions. This limitation highlights the need for comprehensive field trials to validate laboratory findings and ensure the practical applicability of hydrochar in real-world agricultural settings.

SOM turnover can be influenced by storing SOM within the porous network of biochar and hydrochar, which reduces accessibility to microbial decomposers (Bernard et al. 2022). Biochar and hydrochar can enhance microbial proliferation and growth by creating a favorable habitat for microorganisms (Giannetta et al. 2023; Deenik et al. 2011). It is a retention hotspot for labile C, N, phosphorus, and other micronutrients, accelerating SOM decomposition.

Soil salinity

Biochar mitigates the harmful impacts of salinity by enhancing antioxidant functions, improving photosynthetic efficiency, optimizing plant water relations, and boosting the accumulation of osmolytes, hormones, and secondary metabolites (Huang et al. 2023). On the other hand, it has also been demonstrated that biochar can potentially increase soil salinity and reduce soil fertility (Dahlawi et al. 2018). This is attributed to the rise in alkalinity of the soil pH caused by biochar, leading to the precipitation of nutrients (Brtnicky et al. 2021).

In the study conducted by Qin et al. in 2023, it was found that the use of microbial-aged hydrochar significantly impacts the reduction of soil salinity. According to their experimental results, this type of hydrochar reduces salinity by approximately 35% compared to the use of fresh hydrochar. The risks of soil quality deterioration when applying biochar and hydrochar should be minimized through comprehensive long-term field studies and the determination of optimal application rates for practical agricultural use.

Enhancing soil health and plant growth: the role of soil amendments and rhizobacteria

Biochar and hydrochar addition to soil can effectively immobilize heavy metals through various mechanisms. These include increases in soil pH, cation exchange, direct adsorption, functional group complexation, and metal-hydroxide precipitation. Reducing the availability of chemical-extractable heavy metals can enhance their extraction through the high ash content of biochar that exposes organic matter on the surface of the biochar (Zhang et al. 2021a, 2016).

Most root-associated microbes can enhance plant growth and are commonly studied for their capacity to boost plant yield, nutrient absorption, stress tolerance, and control of soil-borne diseases (Rasool et al. 2021). Bacteria such as plant growth-promoting rhizobacteria (PGPR) are the most abundant among the various soil microbial communities. PGPR, such as *Bacillus subtilis*, *Pseudomonas fluorescens*, *Burkholderia phytofirmans*, and *Azospirillum* spp., play a vital role in enhancing nutrient availability to plants, as well as in suppressing diseases and abiotic stresses that plants may face. PGPR presents an eco-friendly alternative for enhancing soil fertility, controlling plant diseases, and promoting plant growth in agriculture. It has been suggested that the combined application of biochar and compost can modify the physical and chemical properties of the soil, resulting in improved plant growth and production (Trupiano et al. 2017). This co-application of biochar and compost also has a synergistic effect on managing soil-borne diseases and increasing the activity of beneficial soil microbial populations, including arbuscular mycorrhizal fungi and PGPR, along with other bio-control agents. Biochar has been identified as an effective carrier for the inoculation of PGPR due to its ability to stimulate the growth and activity of microorganisms, creating favorable planting conditions for agricultural products. Its slow-release effect can also benefit long-term biological control (Tao et al. 2018). Bacteria such as *Bacillus subtilis* SL-44, combined with biochar, can effectively control plant diseases caused by pathogenic fungi. The biofilm formed by biochar and SL-44 can improve the survival of SL-44 in the soil and increase its competitiveness against Fusarium wilt (*Rhizoctonia solani*), as observed in the study of radish plants, ultimately resulting in effective prevention of damage by *Rhizoctonia solani* (Chen et al. 2023a).

When corn straw and pig manure-derived biochars were combined with a mutant species from *Bacillus subtilis* (B38), they demonstrated the ability to reduce the bioavailability of heavy metal contaminants in soil. This effect was notably enhanced by the pig manure-derived biochar, which had approximately twice the surface area of the corn straw-derived biochar, providing more sorption sites for heavy metal ions. This increased surface area, coupled with a threefold higher ash content in the pig manure-derived biochar than the corn straw-derived biochar, significantly improved its capacity to immobilize heavy metals such as Cd, Hg, Pb, and Cr. The high ash content, primarily composed of mineral impurities, acted as the primary adsorption sites for these heavy metals, enhancing the biochar's effectiveness in remediation. Additionally, the enriched nutrient content in the pig manure-derived biochar supported more robust growth and reproductive activities of B38, further utilizing its remedial properties. The combination of biochar and B38 not only improved the efficiency of heavy metal remediation but also led to notable enhancements in plant biomass and significant decreases in heavy metal concentrations in the plants. Polymerase chain reaction-denaturing gradient gel profiles revealed the pig manure-derived biochar's ability to enhance both the exotic B38 and native microbes, indicating their potential for the remediation of soils contaminated by multiple heavy metals (Wang et al. 2017). Using biochar and beneficial microorganisms presents a promising and eco-friendly strategy for addressing agricultural and environmental issues, with the combined effects of biochar and PGPR leading to both environmental and economic benefits.

In the literature, there are also negative aspects of using biochar. Baronti et al. (2010) conducted laboratory incubations and observed that the highest increase in dry matter of perennial ryegrass (*Lolium perenne*) was obtained at a biochar rate of 60 t ha^{-1}. In addition, it was found that applying biochar at a rate of 72 t·ha^{-1} caused a significant decline in corn and wheat yields, with reductions of 46% and 70%, respectively. This decline in yield at high biochar application rates is due to a micronutrient deficiency induced by the increasing soil pH. In addition, Sun et al. (2020a) showed that at the amendments of 0.5% and 1.5% hydrochar derived from different wheat sources led to increases at 0.5% and 1.5% hydrochar derived from different wheat sources, which led to the rise in soil pH.

In the study of Egamberdieva et al. (2020), the hydrochar produced from maize silage was shown to be a suitable carrier for the bacterial inoculant since there was no decrease in bacterial populations after 6 and 8 weeks. The hydrochar-based *Bradyrhizobium japonicum* increased soybean's symbiotic performance and agronomic traits, especially in watered conditions. The porous structure and nutrient provision of hydrochar can provide a suitable environment for PGPR, sustaining growth and survival. However, due to evidence of many macropores for certain types of hydrochar, it can be a suitable carrier of PGPR and accounts for further exploration (Thunshirn et al. 2021).

Current methods of deploying beneficial microbes into the soil, often involving the use of biochar and hydrochar as carriers, prove to be ineffective due to a lack of field studies and the absence of mechanized application methods, highlighting a critical gap in agricultural biotechnology research (Belcher et al. 2019). This limitation is concerning because ineffective deployment can severely diminish the potential benefits of these microbes, such as enhancing plant growth and improving nutrient uptake. The inconsistency of these methods makes it challenging for researchers to evaluate the effectiveness of microbial treatments and for farmers to adopt these biological solutions confidently. Consequently, resources invested in developing and applying microbial products might not yield the expected benefits, leading to economic inefficiencies and potentially deterring further investment in this area. Therefore, there is a pressing need for focused research to develop more reliable and efficient methods for introducing beneficial microbes into various soil environments.

Biochar and hydrochar for carbon sequestration

Biomass conversion as a route for carbon sequestration

Biorefining is a practical approach in the bio-economy for utilizing biomass on a large scale, as it enables the cost-effective production of bioenergy and bio-based products. This strategy ensures favorable socio-economic and environmental outcomes (International Energy Agency 2022). Biomass is frequently considered a byproduct or waste that eventually decomposes and emits GHG. Existing agricultural practice of burning biomass also results in the release of GHG. This practice releases harmful substances, such as particulate matter, carbon monoxide, and volatile organic compounds, significantly threatening our environment (Abdurrahman et al. 2020). However, converting biomass into biochar can decrease CO_2 emissions by trapping a significant amount of carbon in biochar. As a result, producing biochar and hydrochar is considered a cleaner process, as waste that would have otherwise decomposed is now transformed into biochar and utilized for sustainable purposes.

In addition to biochar and hydrochar, biomass materials, through thermochemical decomposition processes, are converted to syngas, a mixture primarily composed of carbon monoxide and hydrogen, and bio-oil, a complex mixture of organic compounds derived from biomass pyrolysis (Sri Shalini et al. 2021). Biomass supplied 10.5% of the world's energy needs in 2019 (Global Bioenergy Statistics 2021). In terms of volume, the global market of biochar was sized at 394.09 kilotons in 2021 and is expected to reach 781.09 kilotons by 2028 (Market Research Report 2022). Biomass-based fuels, using appropriate techniques and procedures, can be less harmful to the environment than fossil fuels, making bio-energy an essential option for short- and medium-term substitution of fossil fuels and the mitigation of GHGs (Vamvuka 2011). Pyrolysis has benefits for treating food waste but can be more beneficial to low-moisture content biomass, avoiding the need for pre-drying. Differences in the carbonization processes of solid products, biochar, and hydrochar may result in variations in their physical and chemical properties, such as ash content, elemental compositions, and morphological characteristics. Hydrochars can also be produced from various waste (wet and dry) materials, allowing waste valorization and reducing the need for landfilling or incineration.

Using charred biomass as a soil amendment in agriculture may be a viable solution to combat climate change by sequestering atmospheric carbon. Applying biochar in soil has gained worldwide interest as a strategy for CO_2 mitigation. Hydrochar produced via HTC may not be stable in soil, leading to increased greenhouse gas emissions due to easier carbon degradation and microbial activity. According to Kambo and Dutta (2015), biochar through pyrolysis may be a better option for carbon sequestration, while hydrochar through HTC may be more suitable as a gas adsorbent. Biochar was a highly recalcitrant carbon storage medium, with no significant decomposition observed during a 19-month field incubation study and a continuous trend of low carbon losses per year (Gronwald et al. 2016). Biochar addition may affect native soil carbon and its decomposition via priming similar to hydrochar, but no priming effects due to biochar were detected in the study. Hydrochar amendments can also have positive and negative priming effects on soil carbon decomposition, depending on the rate of hydrochar decomposition (Rasul et al. 2022; Fatima et al. 2021). While positive priming can occur in the short term, the protective impact of hydrochar carbon on native soil carbon decomposition may prevail over long term. The capacity of biochar and hydrochar to adsorb GHGs is significantly influenced by their physicochemical characteristics, including specific surface area, microporosity, aromaticity, hydrophobicity, and the presence of basic functional groups (Fig. 2). These properties are determined by the type of feedstock used and the conditions under which the char is produced. Both biochar and hydrochar have the potential for carbon storage in soil, but further assessment is needed to understand their long-term effectiveness fully. It is essential to evaluate the environmental impact of biochar and hydrochar production entirely and ensure that their application benefits in enhancing carbon stock outweigh any adverse effects.

Another beneficial use of biochar is its potential to influence gas fluxes from the soil. Biochar has a porous structure with a high specific surface area and thus high CH_4 and volatile organic compound adsorption. Liu et al. (2011) claimed that biochar treatment reduced CO_2 emissions from the soil through its effect on carbon cycles in the soil–water-gas system. In addition, biochar can stabilize the microbial biomass carbon content by lowering the mineralization rate. However, Xu et al. (2020) proved that total greenhouse gas emissions increased by 19% and 21% when the soil was treated at 5% and 15% biochar dosages, respectively. This is because CO_2 fluxes significantly correlate with soil temperature and moisture, while the temperature sensitivity value decreases with increasing biochar application rates. Applying biochar increases the water-holding capacity, significantly correlating with soil CO_2 emissions. Therefore, the higher content of water-soluble organic carbon in the soil during biochar treatment contributes to the release of CO_2 from the soil. Other studies have shown that applying biochar to the soil can reduce CO_2 emissions or not affect gas fluxes (Sackett et al. 2015). Such inconsistency may be caused by the difference in the application rate, soil type, feedstock, and pyrolysis temperature of the biochar (Brassard et al. 2016) and needs to be investigated.

Impact of biochar or hydrochar amendments on GHG emissions from soil

Reducing GHG emissions through evaporation control

Soil carbon reservoirs are vast and constantly changing, and alterations in soil carbon content can significantly influence the concentration of atmospheric CO_2. Climate change is expected to pose more significant risks and vulnerabilities to arid and semi-arid areas, as highlighted by Golla (2021). Arid lands cover about 41% of the earth's terrestrial surface and are home to more than a third of all human beings (Mortimore et al. 2009). In these areas, agricultural production is hindered by various natural and human factors, leading to food insecurity. In arid regions, shrink-swell processes cause soil cracking during drying, promoting increased gas emissions from the topsoil layer and rapid moisture loss through the soil profile.

Additionally, the above procedures enhance the vertical redistribution of dissolved substances in the soil environment. Biochar is commonly applied in agricultural soil in arid and semi-arid regions to avoid soil cracking. Furthermore, hydrochar has recently been used in agricultural soil amendments (Ebrahimi et al. 2022; Joshi et al. 2022; Adjuik et al. 2020).

In clayey soils, biochar and hydrochar can effectively inhibit the width of desiccation cracks (Zhang et al. 2020c) by reducing the rate of water evaporation through increased soil porosity and hydraulic conductivity (Liu et al. 2019b). Using biochar and hydrochar contributed to the reduction

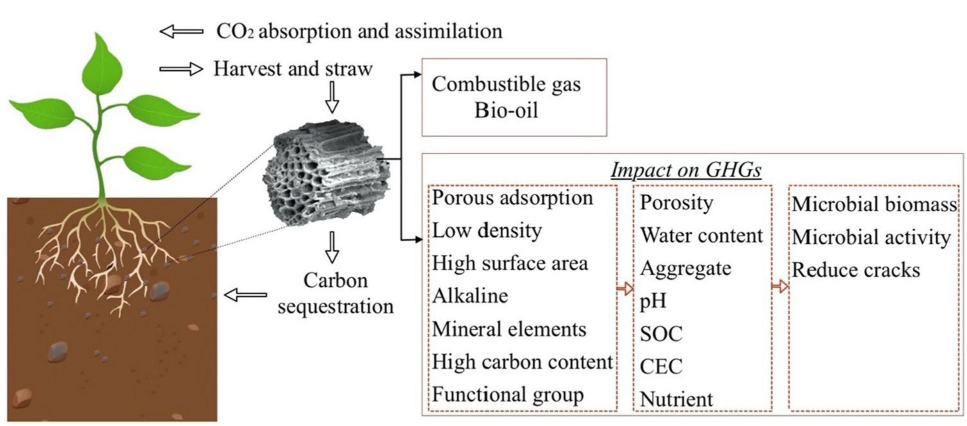


Fig. 2 Conceptual scheme of the impact of biochar/hydrochar application on GHG reductions

of crack formation in the soil (Bordoloi et al. 2019), which was caused by the formation of soil and coal aggregates that prevent the development of cracks. This leads to at least two positive effects on the environment. Using biochar increases water retention, reducing irrigation costs for urban green infrastructure and agricultural land (Mohamed et al. 2016; Bordoloi et al. 2019). Secondly, this potentially reduces the physical release of GHG from the soil by lowering dry shrinkage and cracking. This was confirmed in the study by Kravchenko et al. (2023), which stated that biochar amendment reduced cumulative CO_2 fluxes from the soil by 5% compared to unamended clay soil. This reduction was attributed to a decrease in crack opening. Furthermore, biochar and hydrochar have a highly porous structure with a large specific surface area and the ability to adsorb CH_4 and volatile organic compounds.

Nevertheless, the effect of biochar or hydrochar amendments largely depends on the soil mineralogy and the conditions of biochar and hydrochar pyrolysis. Jing et al. (2022) conducted incubation experiments using different kaolinite biochar types. They observed a significant increase in the formation of micropores in walnut shells and corn straw biochars. This phenomenon can be attributed to the interaction between kaolinite and the surface functional groups of the biochar, facilitated by the low expansibility and strong hydrogen bonding of kaolinite. When biochar and hydrochar come into contact with minerals in the soil, the water present causes the pores to expand (Oleszczuk et al. 2016) continuously. Additionally, kaolinite slows down the decomposition rate of biochar (Wang et al. 2020). In soils, Ca^{2+}, Fe^{3+}, and Al^{3+} ions act as cationic bridges, forming organo-mineral microaggregates with kaolinite on the biochar surface. This also has a positive effect on reducing the dynamics of soil cracking formation.

Laboratory incubation and field monitoring of GHG emissions from amended soil

Table 2 shows the data on the effect of biochar or hydrochar amendments on GHG emissions from the soil. Over the past 5 years, laboratory incubations and field experiments have been conducted to study biochar and hydrochar's impact on the soil's GHG emissions. All methodologies are based on mixing the top layer of soil (up to 25 cm) with amendments and conducting short-term (around 40 days) and long-term monitoring (1 year). The laboratory incubations conducted by Fidel et al. (2019), under constant moisture and temperature conditions, showed no influence on the CO_2 flux from the biochar-amended loess-derived soil. During incubation, the total N_2O emissions from the biochar-amended soil were reduced by 50% at 20 °C and 31% soil moisture. Furthermore, a long-term field monitoring study conducted by Ginebra et al. (2022) on andisol soil did not reveal any influence on CO_2 flux, which may be associated with soil acidity (Bian et al. 2014). However, in soil laboratory incubations, bamboo biochar produced at a temperature of 300 °C decreased cumulative CO_2 emissions by 30% (Zheng et al. 2023), likely due to biochar weakening the bacterial network complexity, possibly caused by increased environmental stress.

Field experiments conducted by Guo et al. (2020) have provided further evidence that cattail biochar, produced at 300 °C, can potentially reduce CO_2 and N_2O emissions from the soil while increasing CH_4 fluxes. The biochar's high surface pH and alkaline metal content could result in the precipitation of CO_2 as carbonates. Moreover, biochar's organic matter absorption may hinder its subsequent conversion into CO_2 (Pokharel et al. 2018). Additionally, biochar amendments have been found to decrease the abundance of two enzymes, glucosidase and cellobiosidase, which are involved in carbohydrate mineralization and contribute to reduced CO_2 emissions. Hu et al. (2023b) conducted a soil laboratory incubation with the addition of dairy processing sludge biochar obtained at temperatures of 450 °C and 700 °C. They found that biochar reduced CO_2 flux from sandy loam soil by 59% and 50%, respectively. Since the experiments were conducted under identical conditions, it can be concluded that a lower pyrolysis temperature leads to a more significant reduction in gas flux. However, in a similar experiment with clay loam soil, it was found that biochar reduced CO_2 flux by 94% and 62% at pyrolysis temperatures of 450 °C and 700 °C, respectively. Furthermore, in the long-term investigation conducted by Cui et al. (2021), it was observed that biochar amendment in soil led to a decline in CH_4 oxidation. This reduction was attributed to utilizing organic compounds, instead of CH_4, as growth substrates by facultative methanotrophs. The increased availability of soil nutrients and carbon influenced this shift. The study also noted the occurrence of N_2O production, which was influenced by various biotic and abiotic factors.

This suggests that clayey soils are more susceptible to the positive influence of biochar in reducing CO_2 flux. This finding was corroborated in a comprehensive analysis of priming effects, which indicated that while the application of biochar to sandy soils appeared to promote the degradation of organic matter significantly, the use of biochar in other types of soils might potentially impede the process (Wang et al. 2016). However, according to a 3-year field monitoring study carried out by Abagandura et al. (2019), it was observed that biochar is effective in reducing N_2O and CO_2 emissions in sandy soil but does not exhibit the same effectiveness in clayey soil. This disparity could be attributed to the higher water-holding capacity of clay-rich soils, which can create anaerobic conditions and promote increased denitrification activity (Shakoor et al. 2021).

Table 2 Effect of biochar or hydrochar amendments on GHG emissions from soil

Feedstock for biochar/hydrochar	Duration	Pyrolysis conditions	Soil type	Experiment conditions	GHG emissions	Remark	References
Biochar: mixture of hardwood and softwood feedstocks (Quercus, Ulmus, and Carya)	Incubation study: 60 days Field study: 248 days	550–600 °C	Loess-derived soils	Laboratory incubations at 10, 20, and 30 °C Field experiments	No effect of biochar on long-term (> 60 days) soil CO_2 emissions. During incubation, the total N_2O emissions were reduced by 50% at 20 °C and 31% soil moisture CO_2 emissions were significantly affected by cropping system and spatial (block) and temporal variables but not by biochar. Biochar reduced N_2O emissions from soils under continuous corn by an average of 27%	Biochar was amended at 0.5% (wt%/wt) rate	Fidel et al. 2019
Rice straw biochar		600 °C for 90 min	Rice paddy	Field experiments	Biochar amendments decreased the cumulative CO_2 flux in the late paddy and for the entire year	Biochar rate was 8 Mg ha^{-1}	Wang et al. 2019b
Cattail biochar (harvested from the wetland)	-	300 °C for 5 h	Constructed wetlands	Field experiments	The CO_2 fluxes in the biochar-amended constructed wetlands were lower than those without biochar, indicating that biochar could mitigate CO_2 emissions		Guo et al. 2020
Bamboo biochar		300 °C for 6 h	Typical acid red soil	Laboratory incubations	Biochar amendments decreased cumulative CO_2 and N_2O emissions by 30% and 69%, respectively	Biochar was amended at 3% (wt%/wt) rate	Zheng et al. 2023

Table 2 (continued)

Feedstock for biochar/hydrochar	Duration	Pyrolysis conditions	Soil type	Experiment conditions	GHG emissions	Remark	References
Farm waste biochar (poultry, pig, and dairy)	365 days	500–550 °C for 2 h	Andisol	Field experiments	Biochar did not affect CO_2 fluxes from the soil	Biochar/soil ratio was 1:100 (w/w)	Ginebra et al. 2022
Dairy processing sludge biochar		450 °C and 700 °C	Sandy loam Light-textured clay loam	Laboratory incubations at 20 °C	Biochar (450 °C) reduced CO_2 flux by 59% when applied to sandy loam soil; from clay loam soil, emissions were reduced by 94%. Biochar amendments showed no effect on CH_4 emissions Biochar (700 °C) reduced CO_2 flux from loam and clay loam soil by 50% and 62%, respectively		Hu et al. 2023b
Pure spruce woodchips	49 days	550 °C	Dystric Cambisol	Laboratory experiment	Biochar amendment was found to decrease CH_4 consumption and N_2O production in soil by 106% and 94%, respectively. However, it had no impact on soil CO_2 production but can contribute to climate change mitigation by increasing the total soil carbon content by 26% with presence of litter		

Table 2 (continued)

Feedstock for biochar/hydrochar	Duration	Pyrolysis conditions	Soil type	Experiment conditions	GHG emissions	Remark	References
Spruce biochar Willow biochar	2 years after biochar amendment, 105-day gas sampling	450 °C	Clayey soil	Field experiment	Biochars exhibited a tendency to decrease N_2O fluxes during peak emission periods. They also resulted in an increase in plant biomass, plant N uptake, nitrogen use efficiency, and crop yield. Furthermore, biochars contributed to a reduction in NO3^–N leaching		Kalu et al. (2022)
Spruce	7 years after biochar amendment, 105-day gas sampling	500–600 °C	Loamy sands		The long-term effects of biochar in loamy sands resulted in a substantial increase in crop yield by 65% and a significant reduction in greenhouse gas intensity (GHGI) by 43%		
Hydrochar from sludge–biomass mixtures	60 days	180 °C, 210 °C, and 240 °C	Sandy soil	Laboratory incubations at 28 °C	Hydrochar (240 °C)-amended soil generated the lowest total CO_2. N_2O, CH_4 emissions from the hydrochar (180 °C)-amendment soil were almost negligible		Ebrahimi et al. 2022
Food waste hydrochar	119 days	200 °C	Silty clay loam	Field experiments	Hydrochar amendments reduced mean CO_2 emissions by 34% compared to the control treatment	Hydrochar rate was 100 kg N ha^{-1}	Adjuik et al. 2020

Table 2 (continued)

Feedstock for biochar/hydrochar	Duration	Pyrolysis conditions	Soil type	Experiment conditions	GHG emissions	Remark	References
Primary sludge hydrochar; activated sludge hydrochar and straw hydrochar	44 days	180 °C and 230 °C	Loamy sand	Laboratory incubations	The total CO_2 flux from the soil decreased after the amendments of hydrochar. CO_2 emissions were higher for hydrochars processed at 180 °C than for those processed at 230 °C		Joshi et al. 2022

The laboratory incubations conducted by Ebrahimi et al. (2022) and Joshi et al. (2022) revealed that the most significant reduction in CO_2 emissions from the hydrochar-amended soil occurred when sludge hydrochar was produced at higher pyrolysis temperatures (up to 240 °C). Ebrahimi et al. (2022) attribute this to the finer dispersive structure of hydrochar obtained at higher pyrolysis temperatures, leading to a decrease in the overall porosity of the amended soil with hydrochar. Therefore, the reduced emissions of N_2O and CO_2, accompanied by increased CH_4 emissions from soils containing hydrochar, can be attributed to more anaerobic soil conditions resulting from decreased soil porosity. The field experiment conducted by Adjuik et al. (2020) also confirmed that hydrochar derived from food waste obtained at 200 °C reduced CO_2 emissions from silty clay loam soil by 34% compared to the soil without amendments. This result of CO_2 reduction contradicts the findings of Malghani et al. (2015) and Kammann et al. (2012), which demonstrate that hydrochar amendment leads to increased gas emissions from the soil. This inconsistency arises due to differences in the rate of biomass decomposition used in hydrochar production (Adjuik et al. 2020). Another possible explanation for this discrepancy is the variation in temperature and processing time of hydrochar, resulting in differences in the amount of carbon mineralized in the soil (Adjuik et al. 2020).

Under actual conditions of the practical application of hydrochar in agricultural soils, moisture variability can influence the rate at which microbial organisms decompose carbon in hydrochar, leading to lower carbon emissions during low soil moisture conditions. Additionally, hydrochar exhibits varied effects on fertility and plant growth due to its unique interactions with different soil types (Cavali et al. 2023). For instance, when maize germination experiments were conducted, it was observed that hydrochar derived from vinasse and sugarcane bagasse had a more favorable interaction with sandy soil compared to clay soil (Fregolente et al. 2021). Although biochar remains stable over the long term, its chemical, physical, and biological composition changes over time (Ghadirnezhad Shiade et al. 2023), and as stated, the aging of biochar in soil enhances the availability of nutrients and promotes plant growth (Mia et al. 2017). Recent meta-analyses have shown limited results from long-term field experiments on the impact of biochar on GHG emissions (Zhang et al. 2020a). Kalu et al. (2021), conducting an 8-year field monitoring, determined that nitrous oxide emission decreased by one-third when biochar was added to the soil. However, they were unable to control the impact of CO_2. The existing literature does not sufficiently analyze the long-term use of biochar and hydrochar, highlighting the need for research in this area.

Long-term potential eco-environmental impacts

Effect of biochar and hydrochar on the bioavailability of heavy metals

Adding biochar to soil can reduce the bioavailability of heavy metals through specific functional groups on the biochar surface. These functional groups included carboxyl, hydroxyl, phenol, alcohol, carbonyl, or enol, which can chelate metals and facilitate their complexation onto the surface and inner biochar pores (Ibrahim et al. 2022). Heavy metals can be grouped in declining order Hg > Cu > Zn > Ni > Pb > Cd > Cr > Sn > Fe > Mn > based on their toxicity (Zwolak et al. 2019). Zhang et al. (2021a) reported that the inclusion of tobacco stem biochar resulted in decreased availability of Cu, Ni, and Pb in the soil, along with a reduction in the overall content of these metals in plants. Additionally, biochar has been shown to significantly lower the uptake of Cu, Pb, Ni, Zn, and Cd by plants compared to control treatments (Zhang et al. 2016). In a series of experiments conducted by Cui et al. (2012), they investigated the effects of different rates of biochar (10 t/ha, 20 t/ha, and 40 t/ha) on Cd-contaminated soil for 2 years. The results showed that in the first year, the biochar content reduced Cd levels in rice and wheat grains, ranging from 16.8 to 45.0% and 24.8 to 44.2%, respectively. However, these reductions were more substantial in the second year, ranging from 39.9 to 61.9% for rice and 14.0 to 39.2% for wheat. Moreover, Cui et al. (2016a) reported the results of a 5-year experimental period, indicating that the bioavailability of Cd and Pb was significantly reduced by 8 to 45% and 14 to 50%, respectively. This reduction was attributed to mechanisms such as surface adsorption, oxygen-containing functional groups adsorption, mineral phases (Al, Fe, and P) precipitation, and cation exchange, as identified in the study of Cui et al. (2016b).

Liu et al. (2023) investigated the accumulation of heavy metals in mixed sludge hydrochar. They found that heavy metals were accumulated in the hydrochar in the following order: Zn > Cu > Cr > Pb > Ni, among others. Interestingly, this ranking was consistent with a previous study conducted by Wang et al. (2019c), which also explored the accumulation rate of heavy metals in hydrochar. A critical outcome of Liu et al. (2023) study is that the hydrochar's heavy metal content, particularly Zn and Mo, exceeded certain required limits. Specifically, the concentration of Zn in the hydrochar was more significant than 1850 mg/kg, and Mo was above 20 mg/kg. These findings have implications for the potential use of hydrochar in agricultural applications. According to Liu et al. (2023), the high levels of Zn and Mo in hydrochar may limit its suitability for land application due to concerns about environmental safety and potential impacts on soil and plants. Additionally, the researchers raised a concern about the long-term application of hydrochar to the soil, as it may lead to an increased accumulation risk of heavy metals in the food chain.

Biochar and hydrochar can adsorb or bind heavy metals in the soil through specific functions such as complex formation, anionic and cationic metal attractions, ion exchange, and precipitation (Fig. 3). However, in immobilizing heavy metals, biochar, and hydrochar may inevitably change the solubility and availability of some specific soil micronutrients such as Fe, Mn, Cu, and Zn (Xu et al. 2022a). This suggests that long-term use of a high dose of biochar and hydrochar can lead to soil micronutrient deficiencies, thus decreasing crop yields. It was found that the transfer of electrons between microorganisms and minerals by biochar and hydrochar can regulate redox-mediated reactions, consequently influencing various biogeochemical cycles in soils, such as Fe and Mn cycles (Kappler et al. 2014). The attachment of Cr(VI) to the biochar or hydrochar surface occurs by

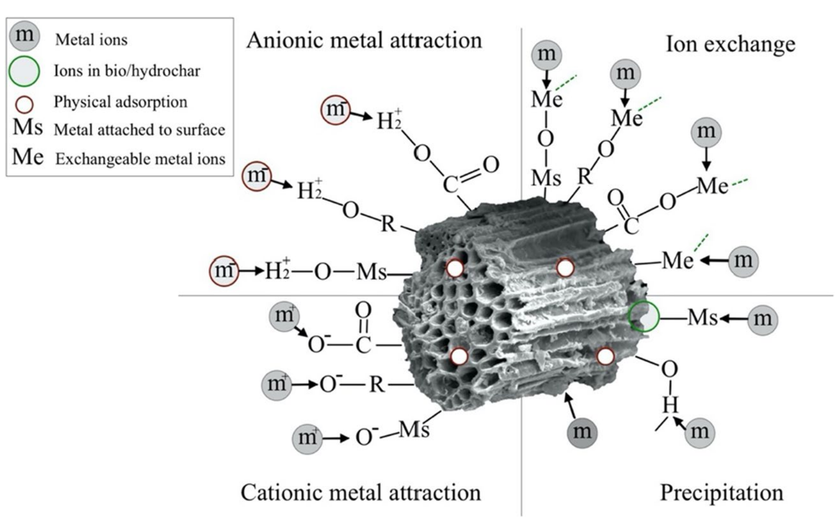


Fig. 3 Interaction mechanism between biochar particles and heavy metals in soil (adopted from Ahmad et al. 2014)

binding with negatively charged active sites located on biochar and hydrochar, which becomes possible after the reduction of Cr(VI) to Cr(III) and involves functional groups containing oxygen (Bolan et al. 2013). Biochar and hydrochar with greater cation exchange capacity can release cations like Ca(II) and Mg(II), improving their adsorption capacity by exchanging them with heavy metal ions. Furthermore, biochar and hydrochar with abundant functional groups can offer ample binding sites for heavy metals to form complexes, as noted by Yang et al. (2019). Recent research on biochar and hydrochar for remediating heavy metal-contaminated soil is summarized in Table 3.

After sewage sludge hydrochar is incorporated into the soil, there is a significant increase in Cd stability due to changes in soil properties (Ren et al. 2017). Subsequent research revealed that adding hydrochar to soil resulted in a significant reduction (15.4%) in the availability of Cd to plants, particularly at lower temperatures. Furthermore, hydrochar was found to decrease the uptake of Cd in both aboveground and underground plant parts. In addition, sludge hydrochar was observed to immediately increase the total and oxidable heavy metal fractions in the soil. Therefore, caution is needed when amending sludge hydrochar in the soil due to the potential increase in heavy metal levels and long-term environmental impacts. While biochar and hydrochar have shown promise in reducing the bioavailability of heavy metals in soil, caution is necessary to avoid unintended consequences when used as pest and weed control (Eibisch et al. 2015). Since pesticides and herbicides mainly function in water, sorption by biochar may adversely affect pest control efforts (Graber et al. 2011).

Holistic ecological function changes

Many studies have shown the positive impacts of biochar on soil fertility and crop productivity. However, it is essential to note that some limitations and potential adverse effects are still associated with its use. Harmful components from biochar can be produced if the chosen parameters for production are not appropriate for the desired application (Chen et al. 2023b; Xiang et al. 2021). Increased soil pH as a result of biochar or hydrochar application may limit the supply of certain nutrients to the original soil, leading to micronutrient deficiency and negatively impacting crop yields (Brtnicky et al. 2021; Xiang et al. 2021). When biochar produced at 700 °C from poultry litter manure was applied to acidic sandy loam, it resulted in an excessively raised soil pH. This led to excessive phosphorus (P) concentrations and leachate enriched with dissolved P (Novak et al. 2013). In addition, the increase in soil pH caused by biochar application may also promote the hydrolysis of N-acyl-homoserine lactone (AHL), which can result in a decrease in the bioavailability of AHL. AHL is a signaling molecule used for cell-to-cell communication. This reduction in AHL bioavailability may affect plant signaling processes, potentially disrupting normal physiological and developmental pathways.

In addition to increasing pH, a study by Andrés et al. (2019) using corn cob rachis biochar produced at 450–500 °C applied to sandy loam significantly reduced soil microbial biomass, while functional microbial diversity remained unchanged. Furthermore, the application of *Panicum virgatum* biochar produced through two-stage-pyrolysis to fine Aridisol soil resulted in a significant decline in fatty acid methyl ester and fungi ratio, alteration in soil microbial community composition, and a decrease in crop shoot (Kelly et al. 2015). Although biochar may increase fungal abundance and function in soil, a negative impact on arbuscular mycorrhizal fungal abundance has been observed by El-Naggar et al. (2019). Biochar may also contain toxic substances threatening human health and the environment. This includes the presence of heavy metals, volatile organic compounds (VOCs), polycyclic aromatic hydrocarbons (PAHs), dioxins and furans, and persistent free radicals (Brtnicky et al. 2021; Chen et al. 2021).

The impact of hydrochar on soil microbial communities has been sparsely documented in the literature. Nonetheless, a few studies have highlighted its positive effects on the growth and abundance of various soil microbes. Lang et al. (2023) determined that soil microbial metabolism, including carbohydrate exchange and amino acid metabolism, was enhanced by applying hydrochar derived from cow manure and corn stalks. Additionally, they demonstrated that the hydrochar amendment accelerated the removal of oxytetracycline from the soil and reduced its uptake by plants in the soil-Chinese cabbage system. These effects were attributed to changes in soil properties, an increase in the number of bacteria degrading oxytetracycline, and the stimulation of microbial metabolism. When examining the association between hydrochar and ectomycorrhizal fungi in seedling growth, Eskandari et al. (2019) found a higher abundance of ectomycorrhizal fungi with the application of hydrochar produced from paper mill biosludge compared to the control.

On the other hand, Andert and Mumme (2015) reported contrasting results, showing that hydrochar application significantly reduced the presence of *Acidobacteria*, up to 5–6 times more than the control. Meanwhile, the abundance of *Firmicutes* was less than one-third compared to the control group. Interestingly, the abundance of *Bacteroidetes* and *Proteobacteria* increased by 2.4 and 1.7 times more than the control, respectively. These changes in the microbial community are likely a result of the easily degradable carbon and the low pH characteristics of hydrochar.

Table 3 Biochar and hydrochar for heavy metal adsorption

Feedstock for biochar/hydrochar	Biochar/hydrochar preparation temperature	Functional groups	Approximate content for the main elemental composition (%)				Heavy metal, which is adsorbed	Remark	References
			C	N	O	H			
Rice straw biochar	400–700 °C 500 °C 700 °C	Carboxyl Hydroxyl	47.07	2.53	17.5	1.96	Ni Cd Pb	Mechanisms for Ni stabilization such as: higher alkalinity of biochar and calcite, which increased the soil pH; porous structure of biochar, surface electronegative charges and functional groups on the biochar surface	Ali et al. 2020; Deng et al. 2019; Tan et al. 2022; Xu et al. 2022b
Rice straw hydrochar	200 °C 250 °C	Oxygen-containing	40.69	0.81	39.07	5.12	Pb, Cu Cr	The adsorption process is endothermic and spontaneous 98–100% of loaded Cr(VI) was found as less toxic Cr(III) form in the solid fraction	Nadarajah et al. 2021; Kavindi et al. 2022; Li et al. 2018b
Avocado seed hydrochar	250 °C	Carboxyl, phenolic	61.68	2.47	32.11	3.75	Ni, Pb, Cu	Removal performance: $Ni^{2+} > Cu^{2+} > Pb^{2+}$	Dhaouadi et al. 2021
Waste plant—*Satureja khuzistanica* Jamzad biochar	400 °C	Carboxyl					Cd, Pb	Biochar application at 6% led to a decrease of Cd and Pb accumulation in aerial parts of peppermint	Mumivand et al. 2023
Hardwood biochar	700 °C	Oxidation	99.9 ± 0.6				Hg, Zn	> 80% Hg stabilized on the biochar without promoting methylation reactions	Wang et al. 2019a; Waqas et al. 2014; Jiang et al. 2016
Corn stover biochar	400–450 °C	Chemisorption/oxidation Phenolic -OH	70.4	1.92	16.8	3.9	Hg, Cd, Pb	Biochar addition reduced Hg migration from soil to spinach. The best addition rate of corn straw biochar was 7%	Zhao et al. 2021; Xia et al. 2021
Leaves and wood of *Platanus acerifolia* hydrochar Leaves and wood of *Platanus acerifolia* biochar	200 °C 450 °C	Oxygenated functional groups Hydroxyl Aliphatic carboxyl	55.9 51.6 64.7 70.4	1.0 0.1 1.2 0.2	33.5 39.3 26.3 21.2	5.9 6.1 3.9 4.1	Cr		Chen et al. 2021

According to Buss et al. (2022), toxic PAHs in biochar depend on the uneven heat distribution and vapor trapping during pyrolysis and chilly zones in the post-pyrolysis area. Sobol et al. (2023) discovered that dioxins and furans in biochar were predominantly below 20 ng total toxic equivalence per kilogram, with processing temperature and feedstock type being the primary causes of high dioxin concentrations in both biochar and hydrochar. It has been discovered that the most significant risk of elevated dioxin levels is associated with biochar and hydrochar produced within the temperature range of 200–300 °C, mainly through methods like torrefaction and hydrothermal carbonization. Furthermore, it has been emphasized that avoiding biomass and organic waste contaminated with chlorinated compounds or preservatives is crucial to obtaining dioxin-free biochar and hydrochar. However, it is worth noting that trace amounts of dioxins may still potentially remain in biochar and hydrochar matrices, even in small quantities. Chen et al. (2021) also reported that biochar-induced persistent free radicals could inhibit rice germination and growth and poison soil organisms due to free radical-induced oxidative damage. Additionally, persistent free radicals in biochar could trigger neurotoxic effects in soil organisms, such as nematode *Caenorhabditis elegans*, limiting its movement and defecation in soil.

The weathering of biochar surfaces and pore edges in soil may lead to an enrichment of more oxidized functional groups on the biochar surfaces, facilitating interactions between biochar and soil minerals (El-Naggar et al. 2018). In a field experiment conducted by El-Naggar et al. (2018), the particulate organic matter fraction of biochar physically interacted with soil minerals in the coarse sand fraction, while the biochar formed organo-mineral complexes with soil minerals in the clay/silt fraction due to the higher presence of exchangeable cations (e.g., Ca, Mg, Na, and K) compared to the coarse sand fraction. The formation of organo-mineral complexes, coating, and pore interactions between biochar and soil minerals or other amendments significantly influence the dynamics of nutrient release and retention in soils (El-Naggar et al. 2019). However, this area requires further investigation using integrated spectroscopic techniques to understand the associated mechanisms and effects on soil nutrients.

Therefore, it is crucial to carefully consider the feedstock type, pyrolysis temperature, and pyrolysis unit design when producing biochar to minimize the presence of toxic substances. Moreover, as the use of hydrochar gains increasing attention as a sustainable soil amendment, it is imperative to conduct further assessments to evaluate the potential adverse effects on soil and the surrounding environment, considering the various production parameters involved in comparison with biochar production.

Life cycle assessment of biochar and hydrochar for soil amendments

Biochar and hydrochar contain a significant amount of carbon and act as carbon sinks to mitigate the effects of climate change. Life cycle assessment (LCA) is a well-established standardized method for assessing the efficiency of products or processes. In recent years, many articles have been published on the LCA of biochar and hydrochar, and several articles have been dedicated to these by-products as soil amendments. All studies were performed following ISO 14040:2006. However, functional units or system boundaries vary in different works, complicating the comparison of results (Matustik et al. 2020). Methodology unification would be beneficial for making LCA results from different authors comparable.

The results of LCA of biochar production in China indicate a strong preference for using waste biomass materials as biochar feedstocks rather than wood or other virgin biomass (Clare et al. 2015). These findings can be explained by the high accumulation of waste biomass resulting from agricultural activities, particularly straw. By diverting waste biomass from traditional waste management practices and converting it into biochar, it is possible to reduce greenhouse gas emissions and other environmental impacts associated with waste disposal. Additionally, the processing of waste biomass is supported by policies and regulations, as China actively promotes using renewable energy sources and sustainable waste management methods. However, waste from the wood processing industry can be a valuable resource for biochar production, as this type of biochar has a high carbon content and stability.

Roy et al. (2020) conducted the LCA of peat moss and miscanthus biomass. They noted that using hydrochar for soil improvement carried fewer environmental risks compared to HTC's energy application. However, the advantages of this approach hinged upon the rate at which biomass decomposes. Brassard et al. (2018) demonstrated a negative balance of GHG emissions −2110 and −2561 kg CO_2-eq Mg^{-1} biochar for scenarios with a lower pyrolysis temperature and a shorter solid residence in the reactor, respectively. The authors also confirmed that pyrolysis of switchgrass (*Panicum virgatum*) for biochar production could be a negative emission technology. However, the pyrolysis operating parameters should be chosen carefully. Some studies have shown that the carbon sequestration effect when using 1 tonne of biochar can reduce CO_2 emissions by 1153–3769 kg (Matustik et al. 2022). Table 4 presents the results of several LCA studies on using biochar and hydrochar as soil amendments. Applying biochar or hydrochar in agricultural soils must be combined with other carbon-capturing strategies such as crop rotation, zero tillage, and reforestation to achieve sustainable development.

Table 4 LCA results of biochar and hydrochar for soil amendments

Feedstock for biochar/hydrochar	Conversion technology (temperature)	Biochar/hydrochar application rate	Global warming potential	Reference
Pinewood biochar	Pyrolysis (500 ℃)	25 t/ha	1.4 Mg CO_2-eq/Mg feedstock	Field et al. (2012)
Rice straw biochar	Slow pyrolysis	1–2% (by soil weight)	− 1.35 Mg CO_2-eq/odt straw	Clare et al. (2015)
Food waste biochar	Pyrolysis	30 t/ha	1.07 t CO_2-eq/t feedstock	Ibarrola et al. (2012)
Wood waste biochar	Pyrolysis	30 t/ha	1.25 t CO_2-eq/t feedstock	
Hybrid poplar biochar	Pyrolysis	50 t/ha	− 17.73 t CO_2-eq/ha − 1.22 t CO_2-eq/t dry biomass feedstock	Peters et al. (2015)
Peat moss and miscanthus hydrochar	HTC (240 ℃)	-	321 kg CO_2-eq/t feedstock	Roy et al. (2020)
Miscanthus hydrochar	HTC (240 ℃)		-830 kg CO_2-eq/t feedstock	
Peat moss hydrochar	HTC (240 ℃)		79.51 kg CO_2-eq/t feedstock	

Future directions and challenges

Firstly, the conflicting results of increasing/decreasing GHG emissions from the biochar and hydrochar-amended soil must be systematized to determine the factors and mechanisms of influence on GHG fluxes. The current literature only lists these factors, such as soil structure, biochar and hydrochar ratio, and environmental conditions, such as air temperature, soil moisture, freezing–thawing cycles, and drying-rewetting. Still, it does not provide a precise and systematic understanding of how these factors affect GHG emissions from amended soils. By understanding these factors, optimizing soil management practices to reduce GHG emissions will be possible. One possible solution to this issue could be creating a unified model for predicting GHG emissions from soil with biochar or hydrochar amendments, which would systematize the factors above.

In addition, further investigation is required to determine the appropriate feedstock sources for producing hydrochar for various applications. Hydrochars' properties from various feedstock sources can influence how effective they are for carbon sequestration and other soil improvement applications. It is crucial to identify suitable feedstock sources and optimize the production process to ensure the production of efficient and sustainable biochars and hydrochars. This optimization should encompass not only the pyrolysis method but also factors such as temperature, humidity of the feedstock, and processing time. However, due to the limited information available, further research is required to determine acceptable feedstock sources for achieving the desired outcomes. In addition to other environmental management techniques, this will help to maximize the potential advantages of hydrochar in soil carbon sequestration.

Furthermore, there is a knowledge gap regarding GHG emissions during the cracking of agricultural soil. In arid and semi-arid regions, biochar and hydrochar are commonly used to increase soil water retention. However, they also reduce cracks, thus lowering GHG emissions. Therefore, a series of studies are necessary to determine the correlation between the following parameters under stable environmental conditions: measured GHG emission, microbial community activity, water evaporation rate, crack intensity factor, and crack volume fraction.

Lastly, more studies on the synergistic mechanism of biochar and hydrochar and PGPR are required to establish a theoretical framework for comprehending the dual effects of this strategy. Such a framework will make it easier to understand how PGPR biochar and hydrochar interact to improve soil fertility and health and how they affect plant growth, nutrient cycling, and soil microbial populations. A comprehensive understanding of the underlying mechanisms will allow for the full utilization of this method for soil remediation and sustainable agriculture. This, in turn, will optimize the long-term effects of biochar/hydrochar-PGPR in effectively dealing with soil contamination. The final comparison of biochar and hydrochar is summarized in Table 5.

Conclusions

This article reviews published peer-reviewed literature on using biochar and hydrochar as soil amendments. As by-products of green energy biomass processing, the application of biochar and hydrochar provides undeniable benefits to society by increasing land productivity, adsorbing heavy metals, and reducing GHG emissions. It was found that applying biochar or hydrochar can have unintended consequences, such as limiting the supply of essential nutrients to the soil, leading to micronutrient deficiency, and negatively impacting crop yields. While both biochar and hydrochar show promise for carbon storage in soil, their long-term effectiveness requires further assessment. It is crucial to thoroughly evaluate the environmental impact of their production and application, ensuring that the benefits of enhanced carbon stock outweigh any potential adverse effects.

Table 5 Comparison between biochar and hydrochar

Comparison aspect	Biochar	Hydrochar	Reference
Ash content	• Ash content rises with higher peak temperatures due to the concentration of minerals and combustion residues from organic matter after pyrolysis	• Shows lower ash content, as inorganic components dissolve and leach out during hydrothermal carbonization, reducing the total ash content	Fu et al. 2019
Soil fertility	• Can improve soil fertility by increasing soil pH, but may lead to micronutrient deficiencies due to excessive pH increases, potentially harming crop yields	• Enhances soil microbial metabolism and aids in the removal of contaminants like oxytetracycline, suggesting potential benefits to soil fertility	Brtnicky et al. 2021; Xiang et al. 2021 Lang et al. 2023
Environmental impact	• Production via pyrolysis is cleaner than traditional biomass burning, reducing CO_2 emissions by stabilizing carbon • Can increase GHG emissions under certain conditions due to interactions with soil carbon cycles and water-holding capacity • Environmental impact varies based on application rates, soil types, feedstock types, and pyrolysis temperatures	• Less stable in soil compared to biochar, potentially increasing GHG emissions due to easier carbon degradation and enhanced microbial activity • May protect native soil carbon from decomposition over the long term, exhibiting positive environmental effects	Liu et al. 2011; Xu et al. 2020; Brassard et al. 2016 Kambo and Dutta 2015; Rasul et al. 2022; Fatima et al. 2021
Heavy metal adsorption	• High adsorption capacity for heavy metals due to surface adsorption, oxygen-containing functional groups, and cation exchange • Reduces bioavailability of heavy metals, decreasing their uptake by plants • Repeated applications lower concentrations of toxic metals (e.g., Cd, Pb) in crops over time, benefiting plant health and human consumption	• May immobilize harmful metals like Cd, reducing their bioavailability • Safety concerns for agricultural use due to potential increases in total and oxidable metal fractions in soil	Ibrahim et al. 2022; Zhang et al. 2021a; Liu et al. 2023; Ren et al. 2017

Additionally, the effectiveness of biochar in reducing heavy metal availability is not universally consistent and can vary based on factors such as soil pH, application rate, method, and feedstock. This review covers these aspects of the benefits of using biochar and hydrochar, making it a promising strategy for reducing carbon emissions. However, the subsequent impact on soil and crop yields is still the subject of future research.

Author contribution Ekaterina Kravchenko: investigation, methodology, visualization, writing—original draft.

Trishia Liezl Dela Cruz: conceptualization, writing—original draft.

Xun Wen Chen: methodology, visualization, writing—original draft.

Ming Hung Wong: supervision, writing—original draft.

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Data availability The data used in this study are available from the corresponding author on reasonable request.

Declarations

Ethical approval No ethical approval was necessary for this study.

Consent to participate All participants in this study consent to participation.

Consent to publish All authors consent to this publication.

Competing interests The authors declare no competing interests.

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