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Biochar

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Benefits and limitations of biochar for climate-smart agriculture: a review and case study from China

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Abstract

Biochar has gained significant attention in agricultural and environmental research over the last two decades. This comprehensive review evaluates the effects of biochar on soil organic carbon (SOC), emission of non-CO_2 greenhouse gases, and crop yield, including related mechanisms and major influencing factors. The impacts of biochar on SOC, methane and nitrous oxide emissions, and crop yield are controlled by biochar and soil properties and management practices. High-temperature biochar produced from lignin-rich feedstocks may decrease methane and nitrous oxide emissions in acidic soils and strengthen long-term carbon sequestration due to its stable aromatic structure. In contrast, low-temperature biochar from manure may increase crop yield in low-fertility soils. Applying biochar to farmlands in China can increase SOC content by 1.9 Pg C and reduce methane and nitrous oxide emissions by 25 and 20 Mt CO_2-eq year^{-1}, respectively, while increasing crop yields by 19%. Despite the increasing evidence of the positive effects of biochar, future research needs to explore the potential factors that could weaken or hinder its capacity to address climate change and secure crop production. We conclude that biochar is not a universal solution for global cropland; however, targeted applications in fields, landscapes, or regional scales, especially in low fertility and sandy soils, could realize the benefits of biochar as a climate-smart measure.

Highlights

- The findings of research on biochar's effects on soil C sequestration, GHG mitigation, and crop production were summarized.
- The factors influencing the impact of biochar on soil functioning were reviewed.
- The effects of biochar on soil C sequestration and GHG mitigation in farmlands of China were quantified.

Keywords Biochar, Carbon sequestration, Greenhouse gas, Food security, Mitigation potential

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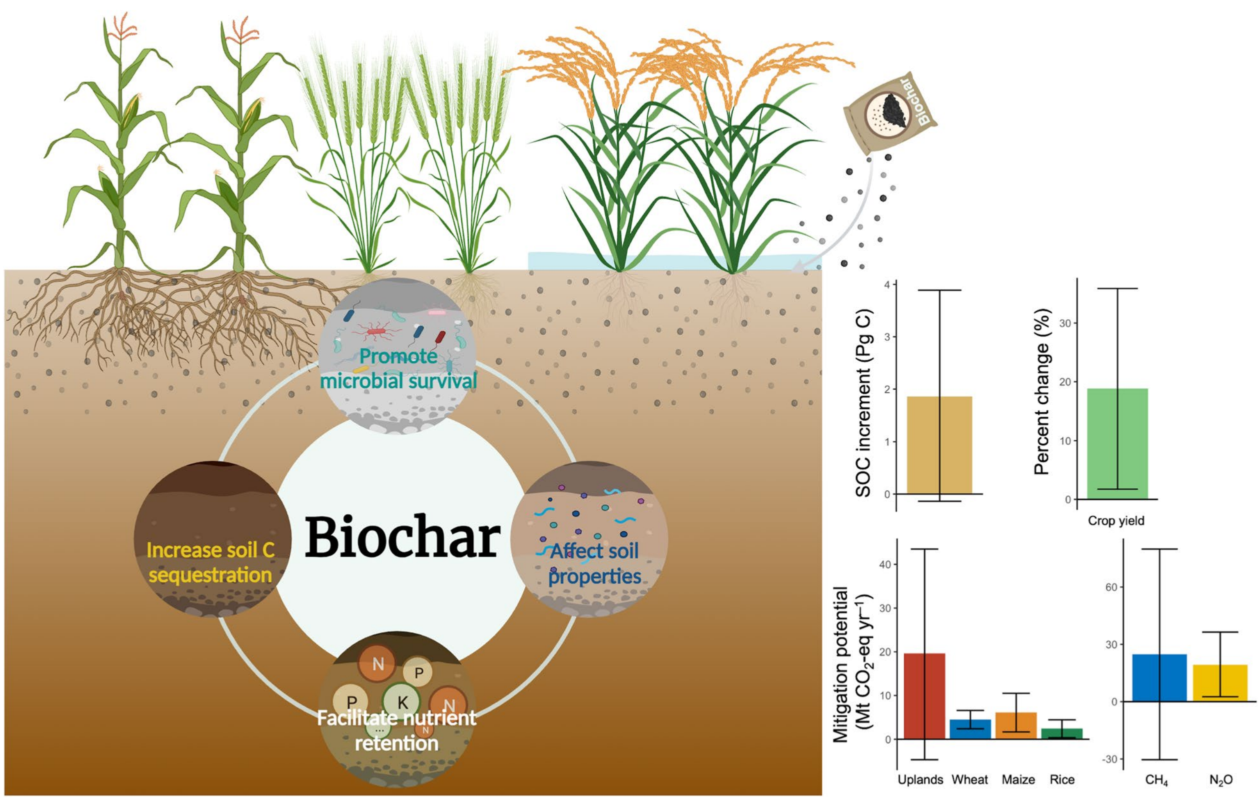
Graphical Abstract



1 Introduction

Biochar, a solid material with high carbon (C) content, is produced during high-temperature pyrolysis of biomass such as crop straw, wood, manure, and solid waste (Schmidt and Noack 2000; Lehmann 2007). It has a very porous structure with a large specific surface area covered with active functional groups, which gives it exceptional stability and long persistence in soil (Kuzyakov et al. 2014; Wang et al. 2016). Thus, biochar is key in mitigating global climate change and developing sustainable agricultural systems (Woolf et al. 2010) by linking extremely long C sequestration with raising crop yield nearly without adverse effects. Biochar can increase soil organic C (SOC) sequestration, alter crop nutrient uptake, and affect ecosystem productivity, ultimately enhancing socioeconomic value (Inyang et al. 2016). Additionally, the liming effect of biochar modifies soil microbial activity and composition, nutrient and substrate availability, and indirectly affects the production, consumption, and emissions of GHGs from the soil. A comprehensive life-cycle assessment has shown that biochar application combined with improved water and nitrogen (N) fertilizer practice would reduce life-cycle GHG emissions of Chinese staple crops by 434 Tg CO_2-eq year^{-1} and increase crop yield by 57 Tg year^{-1} (Xia et al. 2023). As a result, biochar has received widespread attention as a win–win strategy for mitigating climate change and ensuring food security. It is low-cost, easy to implement, and presents significant potential to increase C sinks compared to other negative emission technologies such as direct atmospheric capture, bioenergy, and enhanced weathering (Smith 2016; Obersteiner et al. 2018; Lehmann et al. 2021).

Anthropogenic activities, including fossil fuel combustion, land-use change, and agricultural intensification, have resulted in large emissions of greenhouse gases (GHGs), causing global warming and numerous environmental issues. Addressing the ongoing challenge of global climate change requires reducing GHG emissions and removing carbon dioxide (CO_2) from the atmosphere, ultimately achieving net zero emissions (Field and Mach 2017). Croplands are an important source of GHG emissions, accounting for approximately 43% and 25% of global methane (CH_4) and nitrous oxide (N_2O) emissions, respectively (Crippa et al. 2020). CH_4 is the second largest greenhouse gas after CO_2, accounting for 16–25% of the atmospheric warming (Etminan et al. 2016). Global CH_4 emissions have increased by 151% since 1750, and further increases are expected due to the growing demand for food from expanding population (IPCC 2013). CH_4 production is an exclusively anaerobic process (Conrad 2007), and rice cultivation is a crucial contributor to CH_4 emissions, accounting for about 10% of total CH_4 emissions during the 2000s (Tian et al. 2016). N_2O is a potent greenhouse gas and a long-lived substance that contributes to the depletion of the stratospheric ozone (Ravishankara et al. 2009). Croplands are now the most significant global anthropogenic N_2O emissions source, primarily due to the over-application of mineral fertilizers (Davidson and Kanter 2014; Cui et al. 2018). Atmospheric N_2O concentrations have risen from 269 ppb in 1750 to 331 ppb in 2018 (Tian et al. 2020). As such, agroecosystems have substantial potential to mitigate greenhouse gas emissions (Tian et al. 2016; Wollenberg et al. 2016; IPCC 2019), corresponding globally to 7.3 Pg CO_2-eq year^{-1} (Smith et al. 2007).

Soil organic matter content and composition is essential for sustainable agriculture, as it plays a crucial role in food production, climate change mitigation, and adaptation (Lorenz et al. 2019). SOC dynamics in croplands depend on the balance between C inputs and outputs under long-term environmental and management conditions. However, climate change has disrupted this balance, particularly warming, exacerbating organic matter decomposition and weakening the C sequestration (Wiesmeier et al. 2016). Croplands have limited potential to sequester CO_2 from the atmosphere due to tillage, mineralization, and erosion, resulting in lower C content than soils under natural vegetation (Hassink 1997). In the context of global climate change and sustainable agricultural development, increasing and stabilizing C in croplands is critical to achieving C neutrality in the agricultural sector (Kern et al. 2019). Moreover, the demand for food is expected to double by 2050 due to population growth and an increased need for animal products. Consequently, finding appropriate agricultural management practices to improve global crop productivity on limited arable land is an urgent need (Green et al. 2005; Tilman et al. 2011; Le Mouël and Forslund 2017).

Multiple meta-analysis studies have been performed to explore soil CH_4 and N_2O emissions (Cayuela et al. 2014; Jeffery et al. 2016; He et al. 2017; Zhang et al. 2020; Shakoor et al. 2021; Wang et al. 2022; 2023), SOC (Bai et al. 2019; Gross et al. 2021; Han et al. 2022), and crop yield (Xu et al. 2021; Yangjin et al. 2021; Zhang et al. 2022) in response to biochar additions. Most biochar-related meta-analyses or available reviews focus on the individual GHG, SOC, or crop yield effects and potential mechanisms, failing to emphasize two or more responses simultaneously to assess the influencing factors. For example, several comprehensive overviews explain the impacts of biochar amendment and the potential mechanisms for reducing paddy soil CH_4 emissions (Feng et al. 2012; Nan et al. 2021). Cayuela et al. (2014) outlined assumptions explaining the impacts of biochar on soil N2O production and consumption pathways. Biochar application rate is the most critical driver for SOC stock response (Han et al. 2022). Crop yield responses to biochar application reported in reviews have ranged from negative to positive due to differences in soil properties, biochar characteristics, and complex soil, biochar, crop, climate, and management interactions (Jeffery et al. 2015; Fidel et al. 2017). Consequently, there remains a need for systematic reviews that comprehensively elaborate on the effects of biochar on GHG emissions, C sequestration in soil, increased crop yields, and their underlying mechanisms.

The review is divided into three main aspects. First, we summarized the impact of biochar on climate change mitigation and yield increase. Second, we discussed the relevant mechanisms and key influencing factors. Finally, as a case study, we quantified the potentials of biochar application for C sequestration, GHG emissions reduction, and yield enhancement in Chinese croplands. Overall, this review aims to present a comprehensive understanding of the potential benefits and limitations of biochar in agricultural systems and highlight research gaps and future perspectives in this field. By doing so, we aspire to provide insights for policymakers, researchers, and practitioners to make informed decisions regarding the use of biochar in croplands.

2 Effects of biochar amendment on SOC

The process of C sequestration involves transforming organic C into a fixed form to prevent its mineralization and release into the atmosphere. The "carbon−neutral" and "carbon-negative" cycles, proposed by Lehmann (2007), illustrate how biochar could reduce atmospheric CO_2 and sequester C in the soil. In the "carbon neutral" cycle, plants absorb CO_2 through photosynthesis, with half of it being returned to the atmosphere by respiration, while the other half is used for plant growth, stored in the plants, and later returned to the soil as plant residues. These residues are mineralized and decomposed to CO_2, which returns to the atmosphere without any C reduction. In contrast, if the plant residues are pyrolyzed to biochar and added to the soil, as biochar is orders of magnitude more stable than plant biomass, only a negligible portion of C (5%) is released into the atmosphere through biochar decomposition. In addition, bioenergy produced during biomass pyrolysis can reduce emissions from fossil fuels, thus reducing the CO_2 release into the atmosphere. This entire process is referred to as “carbon negativity,” highlighting the potential of biochar application to sequester C in the soil. Applying biochar to croplands can primarily contribute to C sequestration by increasing input and slowing down the SOC mineralization rate. Meta-analysis studies have shown that SOC content can be increased by 30–85% under biochar application compared to soil without biochar (Table 1). However, various factors, such as biochar feedstocks, pyrolysis conditions, and soil properties, influence the response of SOC to biochar amendment in croplands.

2.1 Biochar increases C inputs

2.1.1 Biochar stability

Biochar directly increases SOC content and stores it for a long time, primarily because it consists of highly recalcitrant C compounds and remains stable in the soil (Kuzyakov et al. 2014; Wang et al. 2016) (Fig. 1). Biochar stability is influenced by various factors, including raw materials, pyrolysis conditions (temperature and residence time), and soil characteristics (Wang et al. 2016). Compared to other feedstocks, biochar derived from wood has a larger C sequestration potential because of the high content of stable lignin, resulting in a higher aromatic C content and slower mineralization rate. The pyrolysis temperature also plays a crucial role in determining biochar stability, with higher temperatures leading to decreased surface reactivity and increased stability (Leng and Huang 2018). As the pyrolysis temperature rises to 650 °C, the influence of raw materials on the labile C content of biochar strongly decreases (Crombie and Mašek 2015). Longer pyrolysis duration leads to a higher degree of carbonization of the biochar, resulting in less unstable organic matter and increased resistance to microbial and enzymatic attacks (Zornoza et al. 2016).

In addition to the biochar internal factors, the external factors, mainly soil texture and SOC content, crucially define its oxidation and decomposition. The mineralization rate of biochar in clayey soils is lower than in sandy soils (Yang et al. 2022a, b) due to the potential limit on biomineralization caused by fine clay particles filling biochar pores. Biochar surface functional groups may also interact with soil clay minerals, reducing microbe accessibility to biochar and increasing its stability (Yang et al. 2016). With an increase in labile organic matter content (e.g., sugars, carboxylic acids), biochar mineralization accelerates due to their role as an indispensable source of C and energy for microorganisms. The co-metabolism that arises from this process further spurs the biochar decomposition (Kuzyakov et al. 2009; Keith et al. 2011).

2.1.2 Biochar raises the inputs of organic C from plant sources into the soil

In agroecosystems, plant residues and rhizodeposits are the main C inputs into soil. Biochar application increases plant growth by altering soil physicochemical properties, leading to the increased return of plant residues (Gul et al. 2015; Buss et al. 2018) (Fig. 1). For example, biochar application raises soil pH due to its abundant ash and base cations, such as K^+, Mg^{2+}, and Ca^{2+}, which substitute for exchangeable soil H^+ and Al^{3+} (Hailegnaw et al. 2019). Soil pH increased from 5.4 to 6.2 after adding biochar at 11 t ha^{-1} (Matsubara et al. 2002), close to the optimal range for nutrient availability (5.5–7.0). After this nutrient input, plants grow better and leaf litter and rhizodeposits increase (Biederman and Stanley Harpole 2013; Hardy et al. 2017). Increased soil pH helps to counteract Al^{3+} damage to crop roots in acidic soil. Biochar application also reduces the Al and Fe concentration in the soil solution, allowing plants easier access to previously bound P and increasing SOC input from plants. Additionally, biochar amendment reduces soil bulk density and improves soil aeration because of its rich pore structure, resulting in increased input of root secretions by increasing plant root elongation and deeper growth (Cui et al. 2011). The increased input of litter or rhizodeposits is delivered to the soil via decomposition or modification by extracellular enzymes secreted by microorganisms. Small-molecule organic substrates are directly consumed by soil microorganisms and incorporated into biomass by assimilation. This contributes to microbially derived organic C in the soil through iterative processes involving microbial cell growth, population proliferation, and production and accumulation of dead residues (Liang et al. 2017).

2.1.3 Biochar increases the input of organic C from microbial sources into the soil

Microbial biomass C (MBC) accounts for 1–4% of SOC, but it is sensitive to many factors and is the driver of most soil biochemical processes. Additionally, MBC is a reservoir of soil-active nutrients for plants and can effectively increase soil nutrient turnover, playing a pivotal role in soil fertility and plant nutrition. Biochar addition increases MBC content by providing the necessary nutrients and habitat to stimulate microbial growth and reproduction and change their community structure (Fig. 1).

Soil microbial communities are crucial in providing ecosystem functions, including C sequestration, which forms stable SOC through their necromass (Gougoulias et al. 2014). Microbial necromass, a necessary precursor for SOC formation, is essential for long-term C sequestration and stabilization, accounting for over 50% of SOC

Table 1 Results of the meta-analysis of biochar application on SOC, CH_4, and N_2O emissions and data for carbon sequestration and emission reduction response estimation in Chinese croplands

Variables	Crop type	Mean (%, 95% CI)	No. of studies (obs.)	Experimental type	References	Data (Source)
SOC	Total	28.0 (23.0 to 33.0)	33 (130)	Field	**Bai et al.** (2019)	5.2 Pg C (Yang et al. 2022a, b)
		32.9 (21.7 to 44.8)	27 (148)	Field	**Xu et al.** (2021)	
		45.8 (39.3 to 52.7)	70 (389)	Field	**Han et al.** (2022)	
		40.0 (32.0 to 51.0)	27 (148)	Field/Pot/Incubation	Liu et al. (2016)	
		39.0 (33.1 to 44.8)	56 (222)	Field/Pot/Incubation	Bai et al. (2019)	
		84.3 (68.4 to 101.6)	54 (166)	Field/Pot/Incubation	Chagas et al. (2022)	
CH_4	Total	− 9.5 (− 15.6 to − 3.4)	40 (155)	Field	Wu et al. (2019)	
		− 7.3 (− 23.1 to 8.4)	50 (97)	Field	Xu et al. (2021)	
		− 3.0 (− 29.5 to 25.9)	32 (121)	Field/Pot/Incubation	He et al. (2017)	
		− 61.0 (− 81 to − 47.0)	43 (160)	Field/Pot/Incubation	Ji et al. (2018)	
		11.6 (− 11.3 to 40.5)	NA (29)	Field/Pot/Incubation	He et al. (2021)	
	Paddy	− 9.3 (− 16.5 to − 2.5)	23 (95)	Field	**Wu et al.** (2019)	8.16 Tg CH_4 year^{-1} (Wang et al. 2023)
		− 13.0 (− 22.0 to − 4.0)	33 (150)	Field	**Liao et al.** (2021)	
		2.0 (− 8.2 to 9.9)	NA (204)	Field	Zhang et al. (2020)	
		− 12.0 (− 26.0 to − 3.0)	24 (77)	Field/Pot/Incubation	Ji et al. (2018)	
	Upland	− 72.0 (− 97.0 to − 44.0)	19 (83)	Field/Pot/Incubation	Ji et al. (2018)	
N_2O	Total	− 12.4 (− 18.3 to − 7.0)	43 (122)	Field	**Verhoeven et al.** (2017)	239 Gg N yr^{-1} (Aliyu et al. 2019)
		− 18.7 (− 22.1 to − 14.8)	54 (182)	Field	**Wu et al.** (2019)	
		− 38.0 (− 45.0 to − 27.0)	70 (468)	Field	**Zhang et al.** (2020)	
		− 14.9 (− 19.3 to − 10.8)	47 (195)	Field	**Zhang et al.** (2021)	
		− 14.7 (− 16.2 to − 13.2)	NA (45)	Field	**Yangjin et al.** (2021)	
		− 17.8 (− 27.6 to − 6.6)	75 (146)	Field	**Xu et al.** (2021)	
		− 15.9 (− 19.5 to − 12.2)	NA (173)	Field	**Zhang et al.** (2022)	
		− 49.0 (− 54.0 to − 44.0)	56 (1375)	Field/Pot/Incubation	Cayuela et al. (2015)	
		− 16.0 (− 21.0 to − 11.0)	51 (177)	Field/Pot/Incubation	Song et al. (2016)	
		− 30.9 (− 38.1 to − 24.4)	56 (371)	Field/Pot/Incubation	He et al. (2017)	
		− 32.0 (− 36.3 to − 27.2)	70 (478)	Field/Pot/Incubation	Liu et al. (2018)	
		− 14.6 (− 25.6 to − 2.7)	18 (79)	Field/Pot/Incubation	Xiao et al. (2019)	
		− 38.0 (− 42.3 to − 32.4)	NA (435)	Field/Pot/Incubation	Borchard et al. (2019)	
		− 14.7 (− 22.1 to − 3.0)	45 (143)	Field/Pot/Incubation	He et al. (2021)	
		− 8.1 (− 19.7 to − 0.7)	41 (186)	Field/Pot/Incubation	Shakoor et al. (2021)	
	Upland	− 18.0 (− 26.0 to − 10.0)	17 (28)	Field	**Song et al.** (2016)	310 Gg N year^{-1} (Li et al. 2001)
		− 11.5 (− 17.5 to − 5.1)	27 (70)	Field	**Verhoeven et al.** (2017)	
	Rice	− 20.0 (− 27.0 to − 13.0)	9 (27)	Field	**Song et al.** (2016)	33 Gg N year^{-1} (Aliyu et al. 2019)
		− 14.0 (− 26.2 to − 2.6)	16 (50)	Field	**Verhoeven et al.** (2017)	
		− 15.6 (− 20.6 to − 10.7)	19 (88)	Field	**Wu et al.** (2019)	
		− 22.0 (− 36.4 to − 12.7)	NA (51)	Field	**Zhang et al.** (2020)	
		− 14.4 (− 30.5 to 1.7)	NA (125)	Field	**Liao et al.** (2021)	
		− 5.3 (− 14.1 to 3.6)	NA (46)	Field/Pot/Incubation	Shakoor et al. (2021)	
	Maize	− 19.2 (− 27.7 to − 9.8)	7 (22)	Field	**Verhoeven et al.** (2017)	66.8 Gg N year^{-1} (Aliyu et al. 2019)
		− 19.8 (− 27.4 to − 12.1)	17 (41)	Field	**Wu et al.** (2019)	
		− 25.0 (− 39.0 to − 11.0)	NA (43)	Field	**Zhang et al.** (2020)	
		− 28.2 (− 34.7 to − 21.8)	NA (30)	Field/Pot/Incubation	Shakoor et al. (2021)	
	Wheat	− 11.7 (− 27.5 to 20.8)	3 (6)	Field	**Verhoeven et al.** (2017)	49.3 Gg N year^{-1} (Aliyu et al. 2019)
		− 18.9 (− 28.1 to − 9.8)	11 (33)	Field	**Wu et al.** (2019)	
		− 33.0 (− 42.0 to − 24.0)	NA (67)	Field	**Zhang et al.** (2020)	

No., number; obs, observations; NA, not available. References in bold are used for quantitative estimates

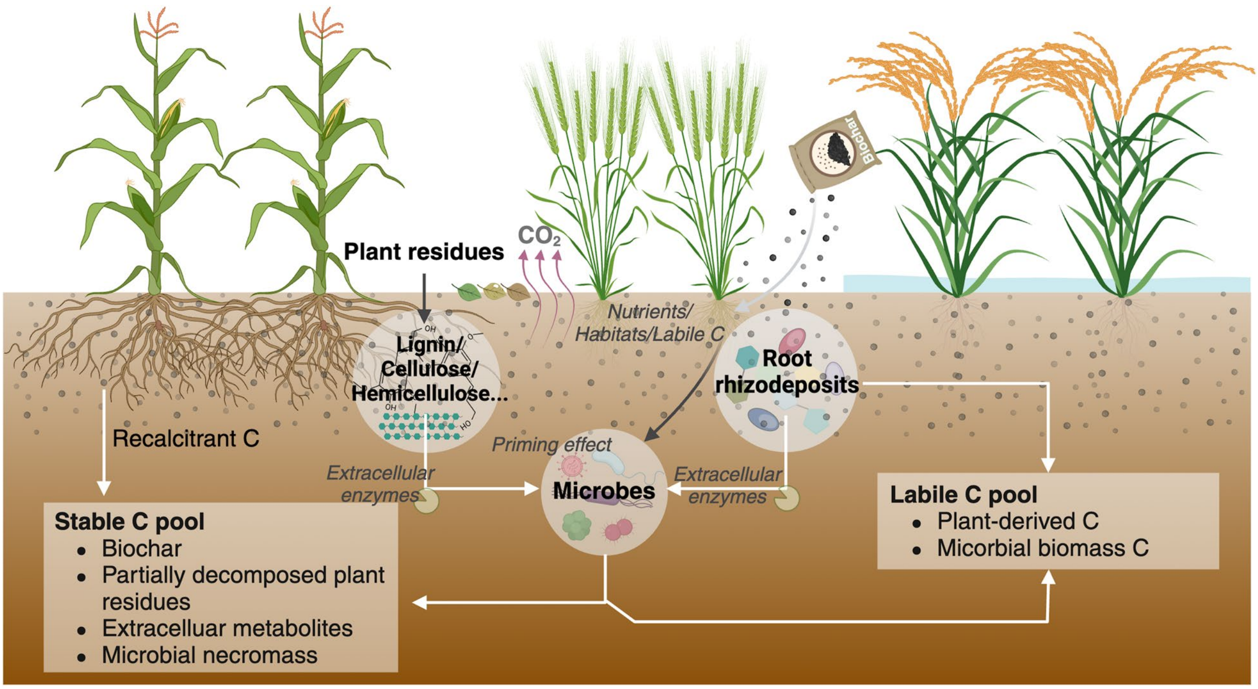


Fig. 1 Main mechanisms of biochar effects on soil organic C pool. Biochar increases plant residues, roots, and root rhizodeposits by promoting plant growth, and then lignin, cellulose, and hemicellulose are converted to a labile or stable soil C pool by microorganisms or extracellular enzymes. Microorganisms are affected by biochar because it provides nutrients, habitat, and labile C. Biochar can also facilitate the input of organic C from microbial sources. The priming effect induced by biochar affects the rate of SOC mineralization, and the recalcitrant C of biochar directly increases the stable C pool

in croplands. Biochar provides substrates that accelerate microbial anabolism and assimilation processes of internal turnover, leading to the continuous formation of microbial necromass and stable SOC through entombing effects (Liang et al. 2017). The balance between microbial necromass formation and decomposition substantially affects its contribution to the SOC accretion (Hu et al. 2020). Biochar provides readily available N sources for microorganisms and reduces their reutilization of necromass, resulting in its accumulation in the soil.

Microbial necromass may be adsorbed on the biochar surface, leading to rapid retention and stabilization, and thus increasing the SOC stock. Fungi and bacteria respond differently to increases in soil pH biochar is added, as most bacteria grow best at neutral pH. Thus, changes in the relative abundance and activity of bacteria and fungi may affect C cycling and storage. Fungi, in contrast, have a relatively wide tolerance to acidity, allowing them to dominate in extremely low pH conditions (Lehmann et al. 2011). Increased acidity raises the ratio of fungi to bacteria, resulting in a fungus-dominated microbiota that slows the SOC turnover rate. This slowdown can be attributed to the creation of fungal residue tannin complexes and the extracellular polysaccharides and mycelium of fungi, which facilitate the formation of soil macroaggregates and shield soil organic matter from decomposition (Six et al. 2006; Adamczyk et al. 2019).

2.2 Biochar affects the rate of SOC mineralization

2.2.1 Positive priming effect

Adding biochar to soil can directly and indirectly affect soil physicochemical properties and microbial activities, ultimately impacting SOC mineralization (Fig. 1). The increased SOC mineralization by biochar can be attributed to three primary factors discussed below. First, the well-developed pore structure of biochar can protect the existence and habitat of soil microbes and stimulate their growth and reproduction, thus increasing SOC mineralization. The biochar structure with many large pores creates a favorable microenvironment for soil microbes, sheltering them from several biotic and abiotic stresses that may otherwise reduce their metabolic activities and increase enzyme activity involved in the breakdown of SOC (Buss et al. 2018). Moreover, the porous structure of biochar increases aeration and water-holding capacity, creating an optimum environment for heterotrophic microbes to utilize the available organic matter rapidly (Laird 2008).

Secondly, the labile C in biochar is a good C source for soil microorganisms, stimulating SOC mineralization (Fig. 1). Biochar contains organic compounds that undergo rapid decomposition upon application to soil, releasing CO_2 and other greenhouse gases (Wang et al. 2016). Low-temperature biochar provides more labile C sources for soil microbes than high-temperature biochar. The easily mineralized C in low-temperature biochar can stimulate SOC mineralization and release more CO_2 into the atmosphere (Major et al. 2010). Finally, biochar delivers multiple nutrients to the soil, increasing co-metabolism among soil microorganisms and leading to increased microbial biomass and activity; this, in turn, promotes native SOC decomposition. Nutrients contained in biochar increase the metabolic activity of soil microbes by providing elements required for their growth and reproduction (Kammann et al. 2015; Zhu et al. 2019). Furthermore, biochar can potentially increase soil microbial diversity and modify microbial community structure, stimulating SOC decomposition by fostering co-metabolic activity among microorganisms. In brief, applying biochar can increase SOC mineralization through various mechanisms, such as creating a favorable microenvironment for soil microbes, releasing easily degradable C sources, and providing numerous nutrients that encourage collaborative metabolic relationships among soil microorganisms.

2.2.2 Negative priming effect

The negative priming effect of biochar pertains to its capacity to decrease the rate at which SOC mineralizes once introduced into the soil. SOC mineralization is mainly performed by the soil biota, particularly through the catabolic activities of microbial communities. The mechanisms responsible for reducing SOC mineralization include microbial access limitation and metabolic limitation. Microbial access limitation involves physical barriers that prevent microorganisms and their exoenzymes from reaching the soil organic substrates and metabolizing them, while metabolic limitation refers to the conditions that regulate microbial catabolism.

The SOC persistence is primarily governed by its accessibility to microbes in the long term, with protection inside micropores and organic-mineral binding being the primary spatial constraints on SOC microbial mineralization. After adding biochar to the soil, SOC can be adsorbed and encapsulated by biochar because of its well-developed pore structure (Totsche et al. 2018), which isolates microorganisms and the extracellular enzymes they produce from contacting SOC within the pores. This isolation increases SOC resistance to microbial degradation, reducing the SOC mineralization rate. Moreover, the rich surface morphology of biochar also adsorbs SOC to its external surface, reducing accessibility and suppressing the degradation of adsorbed SOC, known as adsorption protection. Encapsulation and adsorption protection may also weaken respiration by reducing the C source required for soil microbial activities, leading to lower rates of SOC mineralization.

Biochar facilitates the development of organic–inorganic complexes in the soil and safeguards SOC from microbial degradation by providing stabilization. Biochar also may contribute to SOC stabilization by increasing the binding of labile C and organic–inorganic complexes in montmorillonite-rich soils via the sorption of SOC and minerals, forming new organic–inorganic complexes and soil aggregates. In clay soils containing high amounts of minerals, exogenous C addition will contribute to the compound formation and, consequently, to the SOC stabilization (Wang et al. 2016). Conventional views held that root exudates, like simple sugar, stimulate microbial activity through co-metabolism, increasing the SOC mineralization (Kuzyakov et al. 2000). However, exudates serve as ligands (e.g., organic acids) that can effectively release C via complexation and dissolution reactions with protected mineral phases, thereby increasing microbial accessibility (Keiluweit et al. 2015). Also, biochar has been found to build SOC by stabilizing rhizodeposits for over a decade and act as a ligand to strengthen the organic-mineral interactions, thus stabilizing the new C (Han Weng et al. 2017).

However, biochar contains toxic substances such as dioxins, furans, and phenols that may suppress microbial growth. Biochar also adsorbs enzymes derived from microorganisms and soil inorganic nutrients (e.g., N), thus reducing microbial activity and lowering the SOC mineralization (Lu et al. 2014). In contrast, biochar addition increases the abundance of soil arbuscular mycorrhizal fungi (AMF), and the entanglement of their mycelium, and the secretion of glycoproteins (Glomalin) and polysaccharide substances facilitate soil aggregate formation and structural stability (Peng et al. 2013). Oxygen limitation is an essential mechanism for soil C sequestration in microsites of highly structured soils, where organic C blockage in aggregates restricts the availability of oxygen required for microbial utilization. Biochar addition also increases soil N immobilization with the presence of exogenous organic C, inducing the preferential use of exogenous organic C by soil microbes and reducing the mineralization of native SOC. While biochar can positively or negatively contribute to the decomposition of native SOC, the increase or decrease in decomposition is usually slight compared to the carbon sequestration effect of biochar. Consequently, biochar has a significant potential to increase SOC sequestration.

3 Biochar effect on CH_4 emissions from soil

Using biochar as an amendment to reduce soil GHG emissions has gained significant attention. However, there is still controversy regarding its effect on CH_4 emission. Recent meta-analysis results indicate a wide range of differences, from − 72% to 11.6% (Table 1). These differences are related to several mechanisms, including modifying anaerobic conditions by increasing soil aeration (Karhu et al. 2011), providing available organic compounds for methanogenic bacteria, increasing soil pH, and increasing CH_4 sorption by soil particles and/or biochar (Qin et al. 2016; Sial et al. 2019). Biochar can also retain cations (e.g., Fe^{3+}, Al^{3+}, or NH_4^+) in paddy soils by adsorption, and these cations, when reduced, may compete with CH_4 for oxidation sites of methanotrophs (Fig. 2). Furthermore, biochar application reduces CH_4 emissions in rice fields (Wang et al. 2023), possibly through its inherent redox-active functionalities mediating the anaerobic oxidation of CH_4 (AOM) (Fig. 2). There exist three explanatory mechanisms for this phenomenon, including direct electron transfer from ANME-2d (an anaerobic methanotroph *Methanoperedenaceae*) to biochar, indirect extracellular electron transfer between ANME-2d and *Geobacteraceae*, and indirect extracellular electron transfer from ANME-2d to *Geobacteraceae*, possibly through intermediates like acetate (Cai et al. 2019; Zhao et al. 2021) (Fig. 2). All of these mechanisms (except for AOM) are influenced by five factors including feedstock, pyrolysis temperature, soil properties, cropland management, and biochar aging.

3.1 Feedstock

Feedstock selection for biochar is a crucial factor affecting soil CH_4 emissions. Biochar from manure or straw has a low C/N ratio and contains many low-molecular-weight organic compounds, leading to increased available C for methanogenesis and ultimately increasing CH_4 production (Lehmann et al. 2011). In contrast, wood/cellulose-derived biochar decreases CH_4 emissions due to the greater porosity resulting from preserving the pore structure during pyrolysis due to lignin's stability (Fungo et al. 2014). Excellent porosity can better improve soil aeration and increase the activity of methanotrophs. Biowaste biochar amendments may also decrease CH_4 emissions as available compounds have already been utilized during

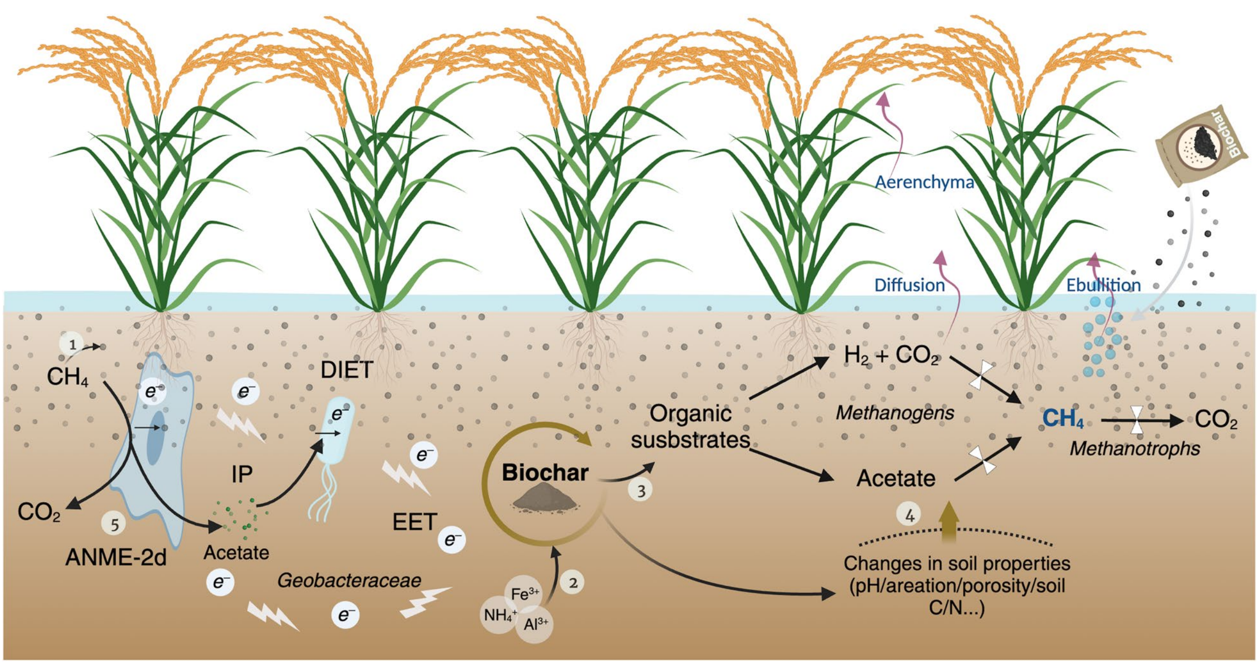


Fig. 2 The main mechanisms of biochar effects on CH_4 emissions from paddy soils. The conceptual diagram illustrates five main mechanisms by which biochar addition to paddy soils affects CH_4 emissions, including direct and indirect effects. The direct effect refers to the direct adsorption of CH_4 on the surface of the biochar. Indirect effects include adsorption of cations by biochar (e.g., Fe^{3+}, Al^{3+}, or NH_4^+) in rice soils, which may compete with CH_4 for oxidation sites in the methanotrophs when they are reduced; provision of a usable organic substrate for CH_4 production; changes in soil properties (e.g., increase in pH, porosity and aeration) that affect methanogenesis or oxidation; and the influence of biochar on the anaerobic oxidation of CH_4. There are three gas transport pathways for CH_4 emitted from rice fields: plant-mediated root aerenchyma transport, ebullition, and diffusion. DIET, EET, and IP denote direct interspecies electron transfer, extracellular electron transfer, and intermediate pathways, respectively

biochar preparation and may not provide sufficient substrates for microbial CH_4 production (Ji et al. 2018).

3.2 Pyrolysis temperature and duration

Pyrolysis temperature and duration strongly influence the physicochemical properties of biochar, affecting soil CH_4 fluxes. High-temperature (>600 °C) pyrolysis biochar reduces soil CH_4 emissions more than low-temperature biochar due to its higher pH, larger surface area and pore volume, and fewer microbial substrates for CH_4 production (Jeffery et al. 2016; Qin et al. 2016; He et al. 2017). Higher pH increases the activity of methanotrophs to oxidize CH_4 (Qin et al. 2016), and higher surface area and pore volume provide favorable conditions for the growth of methanotrophic bacteria (Karhu et al. 2011). As the pyrolysis temperature increases, the content of labile non-aromatic structures decreases; thus, little organic substrate is available for methanogenesis. Dissolved organic carbon (DOC) is also an essential substrate for methanogenic microorganisms. Biochar produced at high pyrolysis temperatures can adsorb DOC to reduce substrate for CH_4 production by methanogenic microorganisms (Nan et al. 2021). Furthermore, biochar with longer pyrolysis duration is more effective in reducing soil CH_4 emissions due to its lower content of bioavailable organics for the methanogenesis (Bruun et al. 2012). In summary, the application of high-temperature pyrolysis biochar is effective in reducing CH_4 emissions due to the suitable pH, lack of substrate required for CH_4 production, and better conditions for CH_4 oxidation, and the longer the biochar pyrolysis duration, the more effective it is.

3.3 Soil properties

The impact of biochar application on CH_4 emissions is greatly influenced by soil properties, particularly moisture content, texture, and pH. Soil moisture content directly affects methanogenic and methanotrophic activities, reducing CH_4 emissions from flooded soils or paddies after biochar addition. This is because biochar increases O_2 fluxes by promoting high porosity, which makes the soil environment toxic to methanogens. The decrease in CH_4 production due to increased oxygen secretion was associated with stimulated rice root growth following nutrient-rich biochar addition. Furthermore, DOC, an essential compound for methanogen growth, is probably reduced by biochar surface adsorption. However, ${NH_4}^+$, Al^{3+}, and Fe^{3+} concentrations are higher in flooded soils than in upland soils, and these ions might be adsorbed by biochar and compete with CH_4 for oxidation sites of methanotrophs, reducing CH_4 oxidation and failing to alleviate CH_4 emissions from paddy soils (Mosier et al. 1991) (Fig. 2). Under extreme moisture conditions or complete waterlogging, a reduction of CH_4 emissions may occur due to reduced substrate availability, which can lead to the introduction of toxic compounds (Spokas 2013) or decreased methanogenic activity (Lin et al. 2015).

Soil texture is another important factor affecting CH_4 emissions. Soils with fine texture, high SOC content, and susceptibility to waterlogging may increase methanogenesis by controlling substrate and product transport. Biochar application in such cases helps maintain low redox conditions, thus resulting in less reduction in soil CH_4 emissions. In contrast, biochar increases soil aeration for fine-textured soils in uplands with moderate SOC content by introducing O_2 to previously anaerobic sites, reducing CH_4 generation. For uplands with coarse texture, biochar addition reduces CH_4 emissions to a great extent due to new habitats that favor methanotroph growth and adsorb active organic C for microbial metabolism, thereby increasing methanotrophic activity by an increased positive priming effect (Fig. 1).

Additionally, soil pH is a significant factor affecting CH_4 production and oxidation, with the optimum pH for methanogens and methanotrophs being 6–8 (Semrau et al. 2010). Biochar addition can neutralize acidic soil due to its generally higher pH than the soil, providing a favorable environment for both methanogens and methanotrophs (Jeffery et al. 2016). Methanotrophs are more susceptible to changes in soil pH than methanogens in size or structure, making CH_4 reduction more effective when biochar is applied to acidic soils. Biochar could reduce the release of Al^{3+} from cation exchange sites by raising soil pH, reducing toxic effects on methanotrophs, and thus increasing CH_4 oxidation. In acidic soils, biochar provides organic C compounds for CH_4 production by fostering plant growth, which may increase CH_4 production (Cross and Sohi 2011; Liu et al. 2013; Ji et al. 2018).

3.4 Cropland management

The magnitude of CH_4 emission reductions from biochar application is related to the amount of biochar added (Kang et al. 2016; Qi et al. 2021). In a 4-year field experiment, the strongest decrease in CH_4 emissions was observed when biochar was applied at a rate of 20 t ha^{-1} (Qin et al. 2016). However, this study did not evaluate the effect of higher biochar application rates on CH_4 fluxes. As mentioned above, biochar protection of SOC leading to reduced methanotrophic activity and CH_4 adsorption onto the hydrophobic surface of biochar could make it more readily utilized by methanotrophic bacteria, thereby decreasing CH_4 emissions (Kang et al. 2016; Kubaczyński et al. 2022; Sadasivam and Reddy 2015). Other aspects, such as biochar feedstock and soil aeration, also impact the CH_4 abatement potential of biochar. For example, woody biochar contains less available organic matter than manure or straw biochar, which limits its ability to reduce CH_4 emissions, even at high application rates (Singh et al. 2012).

The effect of the combined application of biochar and N fertilizer on CH_4 emissions was influenced by the type of fertilizer used. Combined application of biochar and slow-release fertilizer resulted in lower CH_4 emissions than combined application of biochar and urea (Kim et al. 2017). Slow-release fertilizers gradually release inorganic N as rice grows, and this slow release of NH_4^+ not only reduces N loss through the NH_3 volatilization (Zhang et al. 2016) but also promotes CH_4 oxidation. High NH_4^+ content in the soil inhibits CH_4 oxidation and increases CH_4 emission because NH_4^+ competes with CH_4 for the binding sites (Jeffery et al. 2016). The relationship between CH_4 fluxes and N fertilization depends on the amount of fertilizer applied. Higher N fertilization rates typically result in higher CH_4 uptake, while lower rates typically increase the CH_4 flux (Aronson and Helliker 2010). This may be due to soil N status: addition of N sources to N-limited soils can increase CH_4 uptake by increasing the abundance or activity of methanotrophs (Bodelier et al. 2000; Nazaries et al. 2013), whereas biochar application can further increase soil carbon sequestration by reducing the substrate required for CH_4 production (Ji et al. 2018).

3.5 Biochar aging

There is a lack of consensus on how biochar aging impacts soil CH_4 emissions. This section discusses the current understanding of the impact of biochar field aging on soil CH_4 emissions, including the physicochemical changes that occur over time and their influence on CH_4 fluxes. Research has shown that long-term aging of biochar can reduce soil CH_4 fluxes by four main mechanisms. First, there is less C available from the biochar itself, leading to a decrease in the substrate produced by CH_4. Second, with gradual aging, the benefits of biochar amendment on soil health, including increased aeration and methanotroph colonization, become evident. Third, the ratio of methanogens to methanotrophs is reduced due to the increase in soil porosity and air introduction, which leads to a rise in the oxidation–reduction potential (Eh), thereby reducing the abundance of methanogens (Wang et al. 2019). Fourth, biochar surface oxidation occurs, forming carboxyl, hydroxyl, carbonyl and other active functional groups that increase the CH_4 oxidation as they can act as electron acceptors (Mia et al. 2017; Zhang et al. 2019).

However, several studies have revealed that the CH_4 reduction potential of biochar decreases over time due to the loss of its liming effect, lower ability to increase soil aeration, and increased biochar surface area. The liming effect of biochar disappears gradually due to reduced ash content during the aging process, reducing CH_4 consumption by methanotrophic bacteria (Nan et al. 2020). Additionally, biochar pore structure could be blocked by flocculent oxides from root secretions or microorganisms, reducing its capacity to increase soil aeration (Gu et al. 2017). As biochar ages, its surface area increases because new pores are formed by the aggregation of biochar and biochar-derived organic materials (Spokas et al. 2014). This can increase CH_4 adsorption and N retention (Dong et al. 2017; Nan et al. 2021). However, the long-term impact of biochar application on soil CH_4 emissions is intricate and still a topic of intense discussion. The impact of biochar aging on CH_4 fluxes also depends on several factors, including biochar feedstocks, soil properties, and management practices. Further exploration is thus needed to fully understand the impact of biochar aging on soil CH_4 emissions.

4 Effect of biochar on soil N_2O emissions

Soil N_2O production pathways are highly variable spatially and temporally, making reduction efforts challenging. Since its initial report in 2005 (Rondon et al. 2005), biochar has been suggested as a soil amendment to mitigate N_2O emissions. Recent meta-analysis studies demonstrated that biochar application reduces soil N_2O emissions from − 49% to − 8.1% (Table 1). However, the mechanisms that cause this reduction are still poorly understood and subject to ongoing debate. Several hypotheses have been suggested to explain the possible mechanisms involved, including an increase in N_2O reductase activity and N_2O reduction to N_2 by increasing soil pH (Woolf et al. 2018), increased soil moisture conditions and aeration inhibiting denitrification, adsorption of C and N compounds in the soil reducing C sources and N substrates required for N_2O production, and toxic substances such as polycyclic aromatic hydrocarbons (PAHs), heavy metals, polychlorinated dibenzo dioxins and furans contained on the surface directly altering microbial composition and community function. However, conflicting results have been found in individual experiments, with some showing that biochar addition increases soil N_2O emissions (e.g., Chen et al. 2017; Duan et al. 2018). This increase may be due to the priming effect or release of biochar embodied-N, which provides inorganic N/C substrates for soil microorganisms, increases soil water content, and creates a denitrifying environment (Elbasiouny et al. 2021). A conceptual diagram of the main mechanisms of biochar effects on N_2O emissions is shown in Fig. 3.

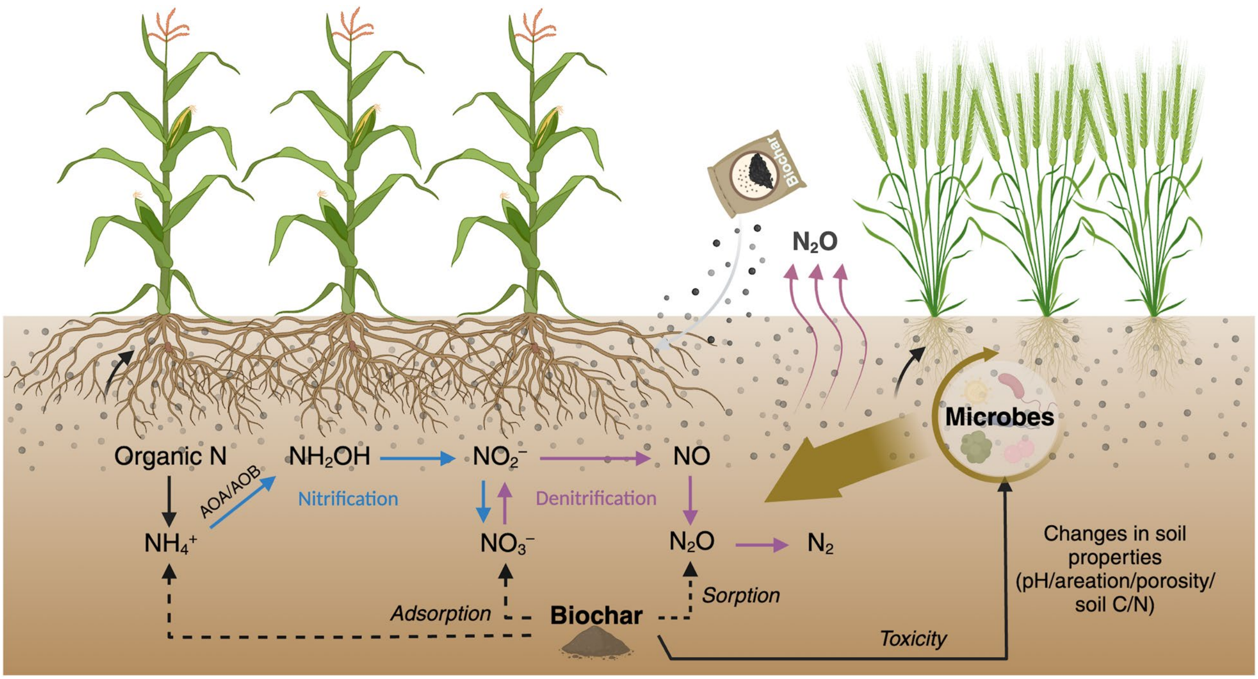


Fig. 3 The main mechanisms of biochar effects on soil N_2O emissions from croplands. The main mechanisms by which biochar affects soil N_2O emissions include: biochar regulates soil nitrification and denitrification process-driven gas emissions through adsorption by directly affecting the effectiveness of the substrates (NH_4^+ and NO_3^-); biochar influences the activity of functional microorganisms associated with the N cycle as well as the structure and composition of their communities through changes in the soil environment (e.g., pH, aeration) and inputs of toxic substances to control the strength of soil N transformation processes and the proportion of gaseous products; and biochar application can also affect soil N_2O emissions indirectly by affecting crop growth and its uptake of nutrients

4.1 Feedstock

The physicochemical properties of biochar are primarily determined by the feedstock used for its production. Plant-derived biochar has a greater potential to reduce N_2O emissions than manure biochar, primarily due to its higher C/N ratio (Cayuela et al. 2014; Liu et al. 2018; Borchard et al. 2019). Plant-derived biochar with a high C/N ratio can limit soil N availability and reduce the substrate required for N_2O production, while manure biochar with a low C/N ratio can increase N_2O production by releasing N into soils through mineralization. Plant-derived biochar has more condensed aromatic structures, which cannot be used as a C source for microbial activity. In contrast, manure biochar contains more labile C, which can induce microbial N immobilization as an energy source.

4.2 Pyrolysis temperature and duration

Apart from feedstock, pyrolysis temperature and duration also play a critical role in determining the potential of biochar to mitigate N_2O emissions. High-temperature biochar had greater N_2O reduction potential than low-temperature biochar due to its unique properties, such as high pH, large surface area, and low H:Corg ratio (Lee et al. 2021). High-temperature pyrolysis increases ash content and primary functional groups, forming mineral deposits and increasing *nos*Z gene abundance, which promotes the complete reduction of N_2O to N_2 (Xiao et al. 2018). The increase in micropore numbers with high-temperature pyrolysis leads to an increase in the surface area and adsorption capacity of biochar, enabling it to restrict N availability by adsorbing NO_3^- and suppress N_2O formation (Mukherjee and Zimmerman 2013). Biochar with a low molar H:Corg ratio (<0.3) is particularly effective in mitigating N_2O emissions due to its high aromatic C ring ratio that increases redox activity and sorption capacity (Weldon et al. 2019). Similarly, the mitigation effect of biochar application is only correlated negatively with the total C content of biochar (Wang et al. 2022). Furthermore, the longer the pyrolysis duration, the higher the ash content and pH, thus making the biochar more effective in reducing N_2O emissions.

4.3 Soil properties

The effect of biochar application on soil N_2O emissions is closely dependent on soil properties. The balance between nitrification and denitrification is crucial in determining the overall impact of biochar on soil N_2O

emissions (Ji et al. 2020), and this balance is affected by soil texture and moisture content. Biochar tends to mitigate N_2O emissions more on fine-textured loamy soils than on sandy soils (Thomson et al. 2012; Cai et al. 2018) due to more capillaries that can firmly hold soil moisture and support denitrification in the former (Saxton et al. 1986). Biochar addition can restrain denitrification by removing water from the soil, reducing compaction, and increasing soil aeration. Under high soil moisture conditions (> 80% WFPS), biochar tends to mitigate soil N_2O primarily by acting as an "electron shuttle" to facilitate electron transfer to soil denitrifiers for complete denitrification (Sun et al. 2017). Biochar's hydrophilic nature and binding to micro-aggregates may also protect soil microsites from the air, creating conditions for complete denitrification to reduce N_2O to N_2 (Chapuis-lardy et al. 2007). However, biochar may increase N_2O emissions under low moisture conditions because nitrification is the primary process of N_2O production.

Soil pH influences microbial-driven nitrification and denitrification pathways for soil N_2O production. Alkaline biochar is more effective in reducing N_2O emissions from acidic soils. Its liming effect increases soil pH and promotes complete denitrification by increasing the abundance of microorganisms carrying the *nos*Z gene (Ji et al. 2020; Han et al. 2021). Reducing (*nir*K + *nir*S)/*nos*Z in the acidic soil by biochar also increases N_2O reduction to N_2 (Han et al. 2021). However, the negligible correlation between soil pH and the response ratio of N_2O flux suggests that the mechanisms involved still need to be fully understood (Song et al. 2016). This means that alkaline biochar applied in acidic soils may cause N_2O reduction, which is probably not true for neutral or alkaline soils (Cayuela et al. 2014). Hence, the impact of biochar on soil N_2O emissions is influenced by intricate interplays between soil properties and biochar attributes.

4.4 Cropland management

The effectiveness of biochar in reducing N_2O emissions depends not only on feedstock type and pyrolysis conditions but also on its application rate. Higher rates of biochar application result in a greater decrease in soil N_2O emissions due to an increase in soil pH, leading to increased NH_3 volatilization and less substrate available for N_2O production (Cayuela et al. 2014; Liu et al. 2018), and higher capacity of biochar to adsorb NO_3^- and/or NH_4^+ also reduces its efficacy for nitrification and denitrification (Heaney et al. 2020). However, the threshold of application rates varies across studies, with the most significant N_2O reduction observed at 20–40 t ha^{-1} of biochar in field trials (Huang et al. 2023). In other meta-analysis studies, the greatest N_2O reduction is achieved when 40–50 t ha^{-1} (Liu et al. 2018; Borchard et al. 2019; Lee et al. 2021) or 20 t ha^{-1} (Li et al. 2020) of biochar is applied. Such discrepancies may be attributable primarily to differences in the individual studies used for these meta-analyses.

The interaction between biochar and exogenous N fertilizer also affects N_2O emissions from amended soils, but the results are inconsistent. The adsorption capacity of biochar for N may immobilize excess N and reduce N_2O emissions. Still, under some conditions, such as co-application with urea or pig manure, N_2O emissions increased when biochar was added (Troy et al. 2013). The potential of biochar to stimulate NH_3 volatilization can contribute to reducing N_2O production, but only in soils where biochar and urea or nitrate-based fertilizers are co-applied (Cayuela et al. 2014; Nelissen et al. 2014). N fertilizer application rates also influence the extent of N_2O reduction, with the greatest reduction observed at higher rates until a limit is reached (Borchard et al. 2019). Biochar has a limited ability to interact with heavily applied N, resulting in weaker N_2O emission reductions at high N application rates (Hagemann et al. 2017).

4.5 Biochar aging

Biochar aging is an additional factor that can impact soil N_2O emissions by altering microbial activity associated with nitrification and denitrification. Biochar aged for five years increased N_2O emissions from alkaline and acidic soils by 43% and 78%, respectively (Duan et al. 2018). This is attributable to various factors, including increased carboxyl and hydroxyl functional groups on the biochar surface due to oxidation, resulting in greater nutrient retention and higher microbial activity (Glaser et al. 2001a). The increased SOC mineralization also provides more usable C or N substrates for N_2O production (Duan et al. 2018). In alkaline soils, an increase in AOA-*amo*A gene abundance increases nitrification, while in acidic soils, increased fungi lacking N_2O reductase due to aged biochar providing more organic C may lead to N_2O production (Xu et al. 2017). Moreover, the desorption of nitrification or denitrification inhibitors adsorbed by biochar due to its blocked pores and decreased adsorption capacity may decrease the suppression of the N_2O production (Spokas 2013). These findings could explain the gradual weakening or even disappearance of the N_2O abatement effect of biochar found in the meta-analysis as the duration of the experiment increased (Guo et al. 2023). However, opposite results have also been observed, possibly because the labile C of aged biochar (6 years) decreased significantly, with the remaining recalcitrant C unavailable to most denitrifiers (He et al. 2019). Soil N_2O emissions after seven years of biochar application were lower than those of control in an acidic tea plantation soil, probably due to the increased effectiveness of soil substrate, nitrification and denitrification activities, and facilitated N_2O reduction (Guo et al. 2023). The natural aging of biochar is a complex process involving both physical, chemical and biological mechanisms. Therefore, long-term studies are needed to investigate the aging process of biochar and its impact on non-CO_2 greenhouse gas emission reductions.

5 Effect of biochar on crop yield

Ensuring sustainable crop productivity on limited arable land to feed a growing global population remains one of the most significant challenges facing global agriculture. The appropriate use of biochar has emerged as a potential solution to this challenge by improving soil nutrient cycling, increasing water and nutrient retention, and ultimately boosting crop yields (Osman et al. 2022). The beneficial effects of biochar on crop productivity can be largely attributed to its influence on soil physicochemical and biological properties, such as bulk density, porosity, cation exchange capacity, and soil microbial and enzyme activity (Fig. 5). A comprehensive analysis of crop yield responses to biochar showed that average yield increases ranged from 4.9% to 48.4% on a global scale (Table 2). These positive effects can be attributed to various factors, including direct nutrient delivery to plants, increased soil pH and cation exchange capacity, enhanced nutrient uptake and fertilizer use efficiency, and improved soil water holding capacity. It is important to note that the impact of biochar on crop growth can vary greatly depending on several factors, including the biochar feedstock and pyrolysis conditions used, soil characteristics, agricultural management practices implemented, and the degree of aging of the biochar. Therefore, it is essential to carefully consider these variables and adjust biochar use accordingly to ensure optimal yields are obtained.

5.1 Feedstock and pyrolysis temperature

The efficacy of biochar in increasing crop yield is influenced by the type of raw materials and the pyrolysis temperature, as these are considered critical determinants of biochar properties. Animal manure biochar is particularly effective in enhancing crop yield, followed by crop residue biochar, while woody biochar has a limited impact on crop performance (Woolf et al. 2018). The high C/N ratio observed in woody biochar may cause N immobilization and limit plant N availability, ultimately suppressing crop growth. In contrast, manure-based biochar contains higher levels of mineral nutrients and is more conducive to higher nutrient uptake and crop yields (Maru et al. 2015).

Table 2 Summary of results of the meta-analysis of biochar application on crop yield

Crop type	Mean (%, 95% CI)	No. of studies (obs.)	Experimental type	References
Total	16.2 (2.0 to 31.0)	33 (58)	Field	**Biederman and Stanley Harpole** (2013)
	9.0 (5.1 to 11.6)	NA (115)	Field	**Wu et al.** (2019)
	28.7 (19.0 to 40.5)	23 (150)	Field	**Ye et al.** (2020)
	1.7 (0.5 to 2.9)	NA (42)	Field	**Yangjin et al.** (2021)
	15.1 (12.1 to 18.3)	278 (648)	Field (without fertilizer)	**Xu et al.** (2021)
	48.4 (41.8 to 55.3)	177 (430)	Field (with fertilizer)	**Xu et al.** (2021)
	12.8 (10.7 to 14.9)	NA (409)	Field	**Zhang et al.** (2022)
	8.3 (6.6 to 10.7)	NA (286)	Field	Liu et al. (2013)
	4.9 (0.5 to 9.0)	86 (301)	Field/Pot	Jeffery et al. (2011)
	13.0 (10.7 to 15.3)	109 (1125)	Field/Pot	Jeffery et al. (2017)
	25.3 (22.6 to 28.0)	NA (105)	Field/Glasshouse/Pot (without fertilizer)	Bai et al. (2022)
	25.7 (22.2 to 29.2)	NA (68)	Field/Glasshouse/Pot (with fertilizer)	Bai et al. (2022)
Maize	13.9 (8.3 to 18.5)	12 (29)	Field	**Wu et al.** (2019)
	12.9 (9.6 to 16.5)	NA (53)	Field/Pot	Liu et al. (2013)
Wheat	7.3 (0.0 to 15.0)	5 (14)	Field	**Wu et al.** (2019)
	9.9 (6.8 to 12.8)	NA (76)	Field/Pot	Liu et al. (2013)
Rice	9.4 (7.3 to 11.6)	16 (72)	Field	**Wu et al.** (2019)
	8.3 (6.4 to 10.3)	NA (103)	Field	**Liu et al.** (2021)
	9.1 (5.8 to 13.1)	33 (150)	Field	**Liao et al.** (2021)
	6.9 (4.5 to 9.4)	NA (104)	Field/Pot	Liu et al. (2013)

No., number; obs., observations; NA, not available. References in bold are used for quantitative estimates

Biochar generated under low-temperature (< 600 °C) conditions produce the greatest response in terms of crop yield increase (Liu et al. 2013; Ye et al. 2020). Several factors primarily explain this outcome. First, low-temperature biochar is generated from partially pyrolyzed biomass, making more C available for microbial growth and greater retention of N and organic C for crop growth. Second, more inorganic elements (e.g., P, K, and Mg) are present in low-temperature biochar, increasing soil nutrient availability. Third, low-temperature biochar possesses a higher specific surface area and thus increases nutrient retention. Finally, low-temperature biochar contains a higher concentration of carboxyl and hydroxyl functional groups that serve as nutrient-binding sites (Glaser et al. 2002).

In contrast, obtaining biochar through pyrolysis at excessively high temperatures (> 600 °C) can have negative impacts on crop growth for several reasons (Gao et al. 2019). First, excessively high-temperature pyrolysis biochar may possess a higher C/N ratio, leading to soil N fixation and reduced N availability for crop growth. Second, biochar generated from pyrolysis above 750 °C may form calcium and phosphorus precipitates that plants cannot uptake (Zwetsloot et al. 2016). Finally, slurry biochar produced at excessively high temperatures may contain elevated levels of heavy metals such as lead, chromium, and arsenic, which can harm crop growth. Therefore, biochar from pyrolysis of animal manure at a relatively low temperature may be more effective in improving soil nutrients and increasing crop yield.

5.2 Soil properties

Soil texture strongly influences the impact of biochar on soil nutrient retention, leading to variations in crop yield following biochar application (Zhang et al. 2021). Observations suggested that biochar increased crop yields in sandy and loamy textured soils, while fine-textured clay soils did not yield marked effects (Farhangi-Abriz et al. 2021). Increased crop yield in sandy soils is mainly attributed to improved soil–water relations, decreased hydraulic conductivity, and increased binding agents provided by biochar, contributing to more remarkable aggregation improvement than in clayey soils. Conversely, the lack of a marked effect in clayey soils is likely due to improved nutrient retention in soils with insufficient aeration, reduced bulk density, and an enhanced water–air exchange, facilitating nutrient transfer to the root-soil layer (Cui et al. 2011; Peake et al. 2014). In general, the benefits of using biochar to promote crop yield are greater in soils with poor structure than those with well-textured soils.

Biochar has been shown to increase tropical crop yield but not temperate crop yield. Tropical soils are generally acidic, and their high acidity, Al^{3+} toxicity, and poor micro- and macro-nutrient availability are obstacles to plant growth. Biochar application increases soil pH, relieves Al^{3+} toxicity, and frees phosphorus bound to metals such as iron, promoting crop growth (Farhangi-Abriz et al. 2021). Additionally, biochar increases soil cation exchange capacity, thereby allowing for greater availability of nutrients such as P, Fe, Mn, and other base cations for plant use, and increases available water capacity due to its high porosity, providing a favorable water environment for plant growth (Jeffery et al. 2011). Nutritional biochar produced from manure or biosolids is highly beneficial for fertilization because of its high yield-limiting nutrients (Kätterer et al. 2019). However, limited factors can impede crop growth in temperate soils, indicating that crop yields are almost at their maximum potential (Glaser et al. 2001b; Loveland and Webb 2003). Biochar may adversely affect crop yield in temperate climates due to over-liming, leading to the immobilization of essential nutrients. Therefore, biochar applications in nutrient-poor tropical soils are more effective at increasing crop yields than in temperate soils, with the effect highly dependent on the biochar feedstock used.

5.3 Cropland management

The optimal application rate for biochar in agroecosystems is still debatable, with recommended rates varying widely (Farhangi-Abriz et al. 2021). However, recent studies have narrowed the range to 20–100 t ha^{-1} based on soil and biochar properties (Van Zwieten et al. 2010; Intani et al. 2019). A lower rate of wood biochar addition is more suitable for coarse soils, while a higher application rate is recommended for fine-textured soils. This may be due to coarse soils having a considerably lower buffering capacity than fine-textured soils and a higher rate of biochar not necessarily resulting in a noticeable increase in yield in subsequent crop seasons (Butnan et al. 2015). Observations have shown that the greatest yield increase in major crops, such as maize, wheat, rice, and soybean, occurs in the 1–10 t ha^{-1} range for biochar applications. Biochar is considered over-applied when the amount exceeds 20 t ha^{-1}, and its positive effect on yield is reduced (Farhangi-Abriz et al. 2021). Excessive biochar application (e.g., > 20 t ha^{-1}) may lead to the suppression of crop growth due to its high volatile and toxic content, which causes soil N immobilization and suppresses microbial activity and nutrient uptake (Ding et al. 2016). However, inconsistent results indicated that yields increased with increasing biochar application, with the highest yields occurring when the application exceeded 30 t ha^{-1} (Bai et al. 2022). This may be due to differences in the analytical methods used in the two studies. Nevertheless, since the high cost of biochar production is currently the biggest obstacle to its large-scale use, it is critical to select the optimal amount of biochar to be applied according to local conditions.

Co-application of biochar and mineral or organic fertilizers has the potential to improve N use efficiency and increase crop yields. On the one hand, this is related to the fact that biochar stimulates microbial activity, reduces the loss of mineral N fertilizers and mineralization of organic fertilizers, improves cation exchange capacity, forms organic-mineral complexes, increases SOC content, improves nutrient adsorption, and stabilizes phytotoxic elements in the soil to create good root conditions. On the other hand, due to the low effective N release rate of biochar, it contributes less to soil N availability for crops. The addition of 20 and 50 t ha^{-1} of biochar provided 64 and 160 kg ha^{-1} of total N to the soil, respectively, but when 50 t ha^{-1} of biochar was added, only a small amount of N (0.1 kg ha^{-1}) was available to the crop (Lehmann and Joseph 2012). Therefore, adding exogenous N can complement the advantages of biochar as a soil amendment and maximize the efficiency of N use. Thus, applying biochar in combination with mineral or organic fertilizers can improve N fertilization efficiency and take full advantage of biochar's ability to improve soil conditions such as cation exchange capacity, water retention, and water holding capacity, thus increasing crop yields.

5.4 Biochar aging

Biochar aging modifies its physicochemical properties, which can significantly affect nutrient bioavailability and delivery in soil, thereby influencing crop yield. However, how biochar aging affects crop yield remains to be determined. Previous findings suggested that the ability of biochar to increase crop yield may decline over time, which may be attributed to nutrients being taken up by plants or leached out of the biochar (Lin et al. 2017) and the microporous structure of the biochar becoming rougher and gradually collapsing (Rafiq et al. 2020). This may lead to lower rates of NH_4^+-N fixation and nitrification, offsetting the ability of biochar to alleviate Al^{3+} release, which is toxic to plants.

Contrary results have shown that aging biochar can promote plant growth due to improvements in soil fertility from certain aspects. First, the hydrophilicity of the aged biochar surface increases due to oxidation, thus increasing water retention. Second, soil aggregate sustainability over long time scales can be maintained by mucilage excretion and mycelial attachment of colonized bacteria and fungi. Third, aging biochar is characterized by surface oxygen-containing functional groups and negative charges, which increase the cation exchange capacity. Fourth, soil cations may sorb strongly to aged biochar, resulting in superior nutrient retention. Lastly, the mineral dissolution of aged biochar can serve as a plant nutrient source. Overall, the effect of biochar aging on crop yield is complex and multifaceted, dependent on various factors such as biochar properties, soil type, crop type, and management practices. Further research is needed to understand these interactions better and identify the optimal conditions for utilizing biochar as a soil amendment to maximize its benefits for crop production.

6 The negative effects and limitations of biochar application

Despite the advantages above of biochar in agricultural applications, such as C sequestration, reduction of GHG emissions, and increased crop yields, it can also have some negative impacts and limitations (Mukherjee and Lal 2014; Hussain et al. 2017). Biochar application can have specific negative effects such as toxicity introduction (Hospido et al. 2005; Van Zwieten et al. 2010), heavy metal retention (Beesley et al. 2010; Chen et al. 2018), and other limitations (Hol et al. 2017; Safaei khorram et al. 2018). PAHs, dioxins, and furans are produced during the pyrolysis of biochar, which means that biochar contains varying degrees of these organic pollutants (Garcia-Perez 2008; Hale et al. 2012). In addition, since biochar stimulates soil microbial activity, its application may promote the degradation of naturally occurring PAHs (Steinbeiss et al. 2009; Quilliam et al. 2013). Therefore, applying biochar to agricultural soils might concurrently contaminate the soil and, if so, would pose a potential risk to human health.

In addition to organic contaminants, biochar may carry a variety of heavy metals (Cu, Cd, Cr, Ni, and Zn) (Hospido et al. 2005), which may affect crop growth and even human health. Biochar can be effective in reducing the bioavailability of heavy metals through adsorption (Ahmad et al. 2014), plant uptake (Chen et al. 2018), and food chain transfer (Khan et al. 2013). However, Cu and As levels in soil increased more than 30-fold after biochar application (Beesley et al. 2010). Biochar application increases pH in acidic soils, which may induce Cu^{2+} complexation with DOM, increasing Cu mobility and availability to alkaline soils and enhancing Cu retention in alkaline soils even though there is less effect on soil pH after long-term biochar application (Chen et al. 2018). Biochar aging occurs due to biotic and abiotic factors, which can make its ability to adsorb toxic compounds, such as heavy metals, potentially diminish over time (Hale et al. 2011). Therefore, the potential contamination of soils with organic contaminants and heavy metals from biochar application can negatively impact soil health and the functioning of the ecosystem services it provides. It is worth noting that these negative impacts are usually related to the pyrolysis process and feedstock of biochar, as well as the environmental conditions of its application. Thus, further research on these aspects is needed for a better biochar application.

The beneficial effects of biochar vary from soil to soil, meaning that biochar addition does not have positive impacts on all types of soil. For example, biochar is more beneficial in degraded and less fertile soils in increasing crop yields (Lehmann et al. 2014). The effect of biochar on agricultural productivity is also related to the target site of the plant. According to Vaccari et al. (2015), applying 14 t ha^{-1} of biochar increased the asexual growth of tomato plants but not fruit yield. Biochar application may also delay plant flowering (Hol et al. 2017). If soils are dominated by nitrification processes, biochar tends to increase soil N_2O emissions due to enhanced nitrification; however, when denitrification is the dominant pathway for controlling N_2O, N_2O emissions from soils are usually reduced after biochar application (Liu et al. 2018). Regarding soil biology, biochar application may interfere with the decomposition of organic matter, thereby reducing the abundance of fungal species such as Ascomycetes and Ascomycetes by 11% and 66%, respectively (Zheng et al. 2016). Biochar application may also contribute to weed problems. Biochar application at 15 t ha^{-1} resulted in a two-fold increase in weed growth during lentil cultivation, and repeated biochar applications may be detrimental to the weed control (Safaei khorram et al. 2018). In conclusion, these negative impacts and limitations of biochar highlight the need for us to be more cautious in applying biochar, as it may not be a panacea for achieving sustainable agricultural development globally.

7 C sequestration and yield increment potential of biochar applied to croplands of China

Reducing the impact of climate change and ensuring food security are essential components of the United Nations' Sustainable Development Goals for 2030 (Lal 2020). Meeting these goals is a significant challenge for humanity. China aims to reach peak CO_2 emissions by 2030 achieve C neutrality by 2060, indicating China's commitment to sustainable development. With only 135 million hectares of arable land, approximately 14% of its territory, it has been challenging to feed almost 20% of the world's population. Intensive agriculture can result in greenhouse gas (GHG) emissions, making C sequestration and emission reduction from croplands critical to achieving sustainability and meeting global climate change goals. Climate-smart agricultural practices, such as conservation tillage, crop residue return, and adding amendments, are crucial in achieving higher yields and lower emissions (Paustian et al. 2016; Bai et al. 2019). Furthermore, biochar, an emerging low-cost soil amendment with a high potential for enhancing C sinks, is essential to climate change mitigation and sustainable agriculture. Although meta-analyses have been conducted by integrating the findings of independent experiments, as mentioned above, the results are still inconclusive (Ji et al. 2018; Wu et al. 2019; Zhang et al. 2020; Xu et al. 2021; Chagas et al. 2022). Therefore, this study aims to quantitatively assess the impact of biochar application on C sequestration, GHG emissions, and crop yield in China, thereby providing generalizable insights.

We complied meta-analyses on the effects of field application of biochar on SOC, CH_4, and N_2O emissions and crop yield to evaluate the impact of biochar application on C sequestration, GHG emissions, and crop yield. We extracted the overall and categorical effect values, sample sizes, and their 95% confidence intervals (CIs) of biochar application on these variables from these meta-analysis studies (Tables 1 and 2). To estimate the effects of biochar application on crop production, soil C sequestration, and GHG emission reductions in croplands of China, we considered only the effects from the field trials in these meta-analyses (shown in bold in the tables) because we thought that doing so would increase the representativeness of the data. The latest reported data on C stocks in farmland in China and total CH_4 and N_2O emissions from agricultural soils were used (Table 1). We conducted 10,000 Monte Carlo simulations using Microsoft Excel 2016 embedded in Crystal Ball software (Oracle Corporation, CA, USA) to derive 95% CIs for the impacts of biochar application on soil C stocks, GHG emission reductions, and crop yield increases in croplands of China.

We found that biochar application had significantly increased the C content of cropland soils in China by 1.86 Pg C (95% CI: − 0.14 to 3.89 Pg C) and crop yield by 19%, with wheat, maize, and rice yields increasing by 7.3%, 14% and 8.9%, respectively (Fig. 4). Moreover, biochar application could contribute to reducing CH_4 emissions from paddies and N_2O emissions from croplands by 24.7 Mt CO_2-eq $year^{-1}$ (95% CI: − 30.4 to 80.0 Mt CO_2-eq $year^{-1}$) and 19.4 Mt CO_2-eq $year^{-1}$ (95% CI: 2.6 to 36.4 Mt CO_2-eq $year^{-1}$), respectively. The GHG reduction potential of biochar application for the three major cereal crops in the order of rice (27.2 Mt CO_2-eq $year^{-1}$), wheat (4.5 Mt CO_2-eq $year^{-1}$), and maize (6.1 Mt CO_2-eq $year^{-1}$). Consequently, biochar application had considerable potential to mitigate GHG emissions while simultaneously increasing crop yields. Additionally, the C sequestration potential of biochar is primarily based on its C content and persistence in the soil, i.e., how long the added soil C pool can be retained (Woolf et al. 2021). Biochar embedded-C and the negative priming effect are the two primary mechanisms of SOC accumulation. The authors assume that the increased C stock originates

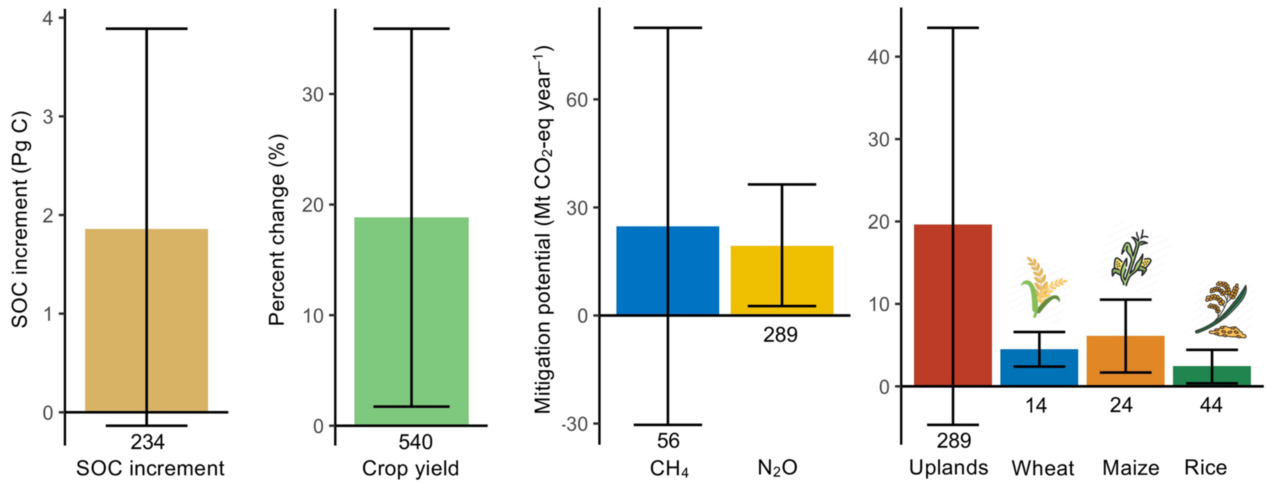


Fig. 4 Quantitative estimation of biochar effects on SOC increment, mitigation potentials of emissions of non-CO_2 GHG emissions, and crop yield from croplands of China. For each variable, meta-analytic data from field observations based on biochar application (shown in bold in tables) and the most recently reported estimates (the last column of Table 1) are used for quantitative estimation. Values are means and 95% confidence intervals. The numbers below the bars indicate the number of studies. Uncertainty analysis is implemented by Monte Carlo simulation ($n = 10{,}000$). See the text for the detailed calculations

entirely from biochar embodied-C and that the added biochar is highly stable, difficult to decompose by soil microorganisms, and remains around 70% after 100 years (Budai et al. 2013). Thus, biochar application over 100 years can increase SOC by approximately 13.0 Tg C $year^{-1}$. However, while this value is theoretical and based on several assumptions, the feasibility of using biochar for soil C sequestration requires further investigation.

Several limitations exist in this analysis. First, the database of meta-analysis studies used in this study is based on a global scale, and the results may be different from the actual situation in China due to differences in geographic locations and climatic conditions. Second, biochar mainly affects SOC by providing inert C and influencing SOC mineralization, and the priming effect induced by biochar was not considered in this study, so the annual carbon sequestration of biochar may be biased. Finally, this analysis included studies without long-term experimental data and did not consider the effects of biochar aging.

8 Summary and outlook

Biochar is a promising tool for mitigating climate change and increasing crop yields by enhancing multifaceted soil features. To reap the benefits of biochar, it is important to select the appropriate biochar feedstock and pyrolysis process conditions according to specific needs and site conditions (Fig. 5). For instance, wood-based biochar produced at high temperatures and long residence times tends to have a high potential for reducing GHG emissions but does not have a significant effect on increasing crop yields (Jeffery et al. 2016; He et al. 2017; Cayuela et al. 2014; Borchard et al. 2019; Liu et al. 2018). In contrast, low-temperature biochar derived from manure is rich in nutrients and can increase crop yields in low-fertility soils (Woolf et al. 2018; Gao et al. 2019; Ye et al. 2020). The simultaneous application of biochar and manure can improve soil nutrient management and utilization, increasing crop yield. In addition to these benefits, biochar amendment has some negative impacts and limitations, including biochar toxicity, heavy metal retention, and stimulated weed growth. Since biochar cannot be removed from the soil once it has been added to the soil, its toxicity must be determined before using biochar as a soil amendment to minimize long-term risks to the soil and crops. The rational application of biochar depends on various factors, including biochar feedstock, pyrolysis conditions, soil characteristics, and crop management. Therefore, how to weigh the trade-offs of biochar application on multiple service functions is an issue that needs to be addressed for using biochar as a climate-smart practice. In addition, the current lack of a robust C credit market and the high cost of using biochar make farmers less motivated to use it, as they have yet to benefit from it alone.

How to make better use of the positive impacts of biochar in combating climate change and ensuring sustainable agricultural development can be better utilized is an urgent issue for the future. As the knowledge-based agricultural nutrient management practices that scientists worldwide have explored only after unremitting efforts

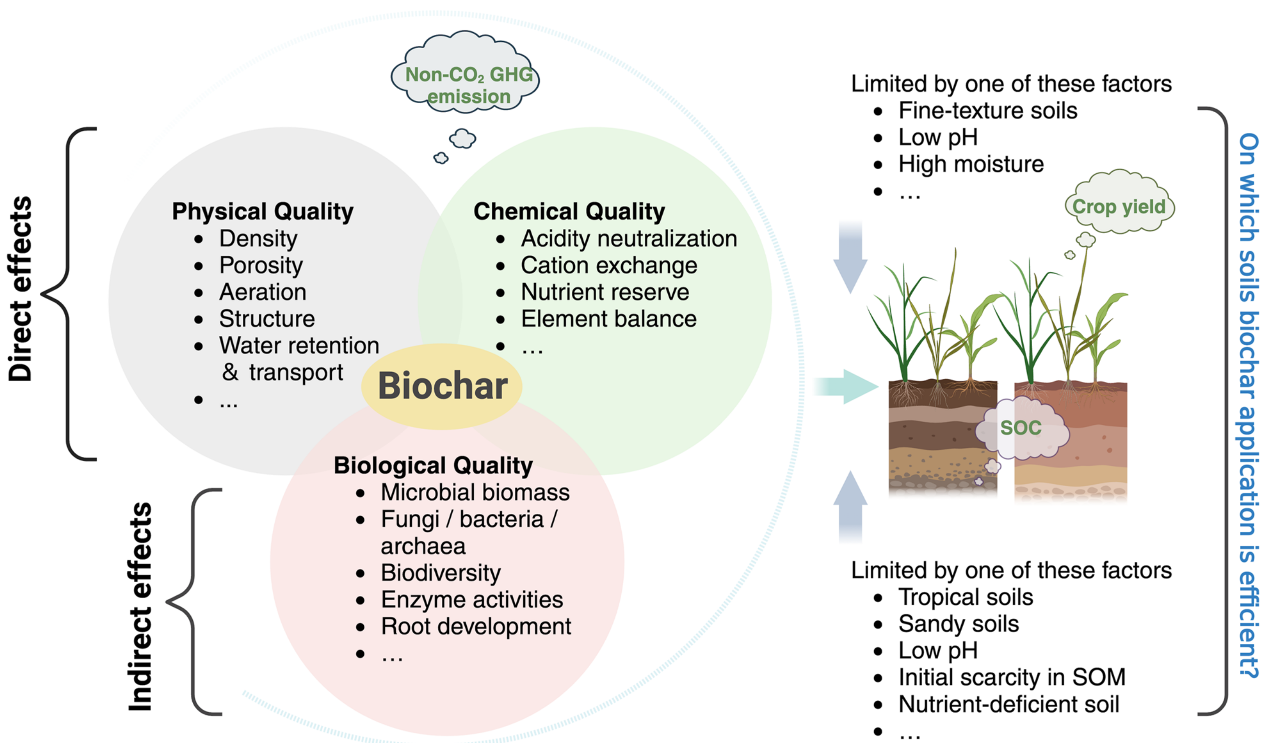


Fig. 5 Conceptual diagram illustrating changes in soil properties after biochar application and its effects on non-CO_2 GHG emissions, SOC, and crop yield. The conceptual diagram depicts how biochar application affects non-CO_2 GHG emissions through direct or indirect effects on multiple physical, chemical, and biological properties of the soil. It also illustrates the conditions under which the impact of biochar application on the positive effects on crop yield or SOC is limited

and practices over the past decades, biochar application also requires in-depth research and promotion regarding scientific understanding, technological application, and policy support. At the research level, researchers need to explore how to prepare biochar suitable for application under specific conditions based on different feedstocks, pyrolysis processes, crop species, and soil characteristics with the support of national and corporate programs to achieve a win–win situation for both agricultural production and sustainable development. At the application level, climate change mitigation and food security are the result of the joint efforts of the government, the market, and the consumers, so relevant incentives and publicity and promotion need to be established to help promote biochar. In conclusion, biochar has great potential for climate change mitigation and sustainable agriculture, but more research is required to fully understand its effects and maximize its benefits in practice.

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Author contributions
XB: Data curation, Visualization, Writing-original draft. ZZ: Writing-review & editing; JW: Conceptualization, Data curation, Visualization, Writing-review & editing. SG: Writing-review & editing. ZL: Writing-review & editing. HL: Writing-review & editing. YH: Writing-review & editing; ZH: Writing-review & editing. YK: Visualization, Writing-review & editing. JZ: Conceptualization, Writing-review & editing.

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Availability of data and materials
The datasets used or analyzed during the current study are available from the corresponding author upon reasonable request.

Declarations

Competing interests
The authors have no relevant financial or non-financial interests to disclose.

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