

REVIEW

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Role of biochar toward carbon neutrality



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Abstract

Carbon neutrality by the mid-twenty-first century is a grand challenge requiring technological innovations. Biochar, a traditional soil amendment which has been used for fertility improvement and contaminant remediation, has revealed new vitality in this context. In this review we highlight the huge potential of biochar application in different fields to mitigate as high as 2.56×10^9 t CO₂e total greenhouse gas (GHG) emissions per year, accounting for 5.0% of the global GHG emissions. Soil applications of biochar as either a controlled-release fertilizer or an immobilization agent offer improved soil health while simultaneously suppressing the emissions of CH₄ and N₂O. Non-soil applications of biochar also contribute to carbon neutrality in unique ways. Firstly, biochar application as a ruminant feed decreases CH₄ emissions via physical sorption and enhanced activities of methanotrophs. Secondly, biochar can be used as a green catalyst for biorefinery. Besides, biochar as an additive to Portland cement and low impact development (LID) infrastructure lowers the carbon footprint and builds resilience to climate change. Furthermore, biochar can be used as novel batteries and supercapacitors for energy storage purposes. Finally, the high CO₂ adsorption capacity makes it possible for biochar being used as a sorbent for carbon capture, utilization, and storage (CCUS). We advocate that future research should further explore the effectiveness of biochar systems for climate change mitigation in large scale applications, and assess the economic and social viability of local biochar systems to combat climate change.

Highlights

- Biochar application mitigates 2.56×10^9 t CO₂e greenhouse gas emissions per year.
- Biochar contributes to carbon neutrality due to carbon negative nature, CO₂ sorption, and negative priming effects.
- Future research should explore the effectiveness in large scale applications.

Keywords Biochar, Sustainability, Climate change, Amendment, Remediation

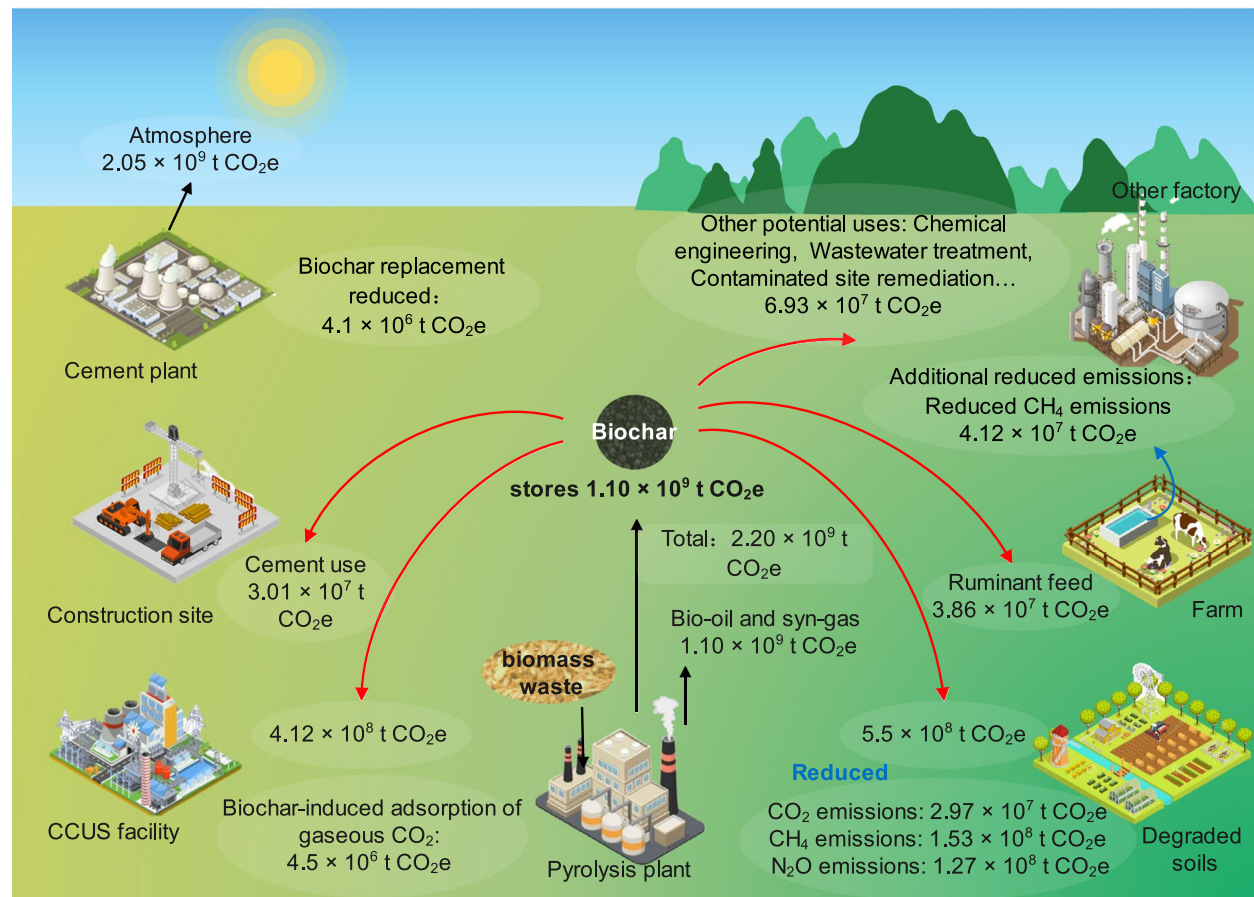
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Graphical Abstract



1 Introduction

Carbon neutrality is defined as a state where a balance between carbon emission and carbon absorption from the atmosphere is reached (European Parliament 2022; UNEP 2019). Carbon neutrality is essential to achieving the Paris Agreement Goal to limit global warming within 2°C (UNFCCC 2015). One hundred thirty-five countries have committed to achieve carbon neutrality by mid-twenty-first century (Statista 2022). However, grand challenges remain to reach that ambitious goal. For instance, decoupling economic growth from GHG emissions is the fundamental challenge of many developing countries including China toward carbon neutrality (Liu et al. 2022b). Technological innovation is an important aspect of achieving this aim.

Soil is the largest carbon reservoir in the terrestrial environment (Friedlingstein et al. 2022). Organic carbon stock and stability in the soil environment determine the effectiveness of our effort towards carbon neutrality

(Bradford et al. 2016; Lehmann et al. 2020; Schmidt et al. 2011). The non-renewable nature of soil also highlights the necessity of soil organic carbon preservation (Antoneli et al. 2022; Bombino et al. 2022; FAO 2015). However, degradation is occurring in billions of hectares of the world's land, resulting in irreversible carbon loss (Gibbs and Salmon 2015; Rickson et al. 2015; UN 2022). Human activity-induced land use change is a key process worsening the scenario (Asare et al. 2022; Borrelli et al. 2020; Siqueira-Neto et al. 2022; Vasilchenko et al. 2022). It is crucial that suitable soil use and management practices should be made to restore carbon in ground. In addition to soil use and management-related emissions, it is well acknowledged that various other human activities such as electricity generation, transportation, and industry emit a considerable amount of CO₂ into the atmosphere. For instance, the aforementioned three activities contribute to 25%, 27%, and 24% global GHG emissions in 2020, respectively (US EPA 2022).

In this case, biochar is a promising material for both soil and non-soil applications toward carbon neutrality. On the one hand, it is an intrinsically carbon-negative soil amendment, because its production converts labile biomass into recalcitrant carbon that persists in the environment on centennial timescales (Glaser et al. 2009; Lehmann et al. 2021). Around 60 billion metric tons of carbon are taken up via photosynthesis annually, of which 10% are able to be converted to waste biomass. Converting these 6 billion tons of carbon into biochar via pyrolysis, therefore, produces 3 billion tons of biochar annually (assuming the biomass yield to be 50%), thus reducing CO₂ emissions by the same amount (Kleiner 2009). On the other hand, several recent findings suggest that biochar also possesses huge potential to combat climate change in non-soil applications including aquaculture (Man et al. 2021), chemical engineering (Shen et al. 2014), construction (Lunt et al. 2022), carbon capture, utilization and storage (CCUS) (Shafawi et al. 2021), and energy storage (Cheng et al. 2017).

This review summarizes recent advances of biochar application toward carbon neutrality. Apart from its traditional use as a soil amendment, novel uses of biochar for non-soil applications are critically summarized. A roadmap of biochar toward carbon neutrality is provided based on quantitative estimation of biochar's climate change mitigation potential in different fields. Existing challenges and future research directions are also discussed.

2 An overview of biochar's role toward carbon neutrality

Biochar is an old material which has revealed new vitality in the context of carbon neutrality. As early as 500 to 2500 years B.P., Pre-Columbian Indians have made use of biological wastes to produce Amazonian Dark Earths with high fertility (Denevan 1992; Eden et al. 1984; Sanford Jr et al. 1985). The core idea of turning waste to highly fertile soils was termed pyrolysis, namely, thermal conversion of biomass into charcoal under oxygen-limited conditions (Figueiredo et al. 1989; Maschio et al. 1992; Mobarak et al. 1982). It was not until the twenty-first century that the concept of "biochar" became popular (Lehmann 2007; Marris 2006). It is defined as a carbon-rich material derived from thermal conversion (including pyrolysis and hydrothermal carbonization) of biomass feedstock under oxygen-limited conditions (IBI 2015). Various feedstocks can be used to produce biochar, including wood, grass, crop residues, animal waste, sewage sludge, anaerobic digestate, bone, etc. (Alkurdi et al. 2019; Wang et al. 2020g). Pyrolysis refers to the process where biomass feedstock was subjected to thermal conversion under oxygen-limited environment at

temperatures usually between 250 to 900°C. In contrast, hydrothermal carbonization is conducted via mixing biomass with water and heating in a closed reactor below 250°C (Masoumi et al. 2021; Yaashikaa et al. 2020). However, one should note that to produce biochar at large quantities for field applications, pyrolysis seems to be a more feasible one with higher technical maturity. It takes around 449~1847 dollars to produce 1 metric ton of biochar at a large scale, which is comparable with other soil fertilizers and immobilization agents (Nematian et al. 2021).

Biochar has long been used as an amendment to increase soil fertility. Its ability to immobilize heavy metals and organic contaminants also manifested itself as a green immobilization agent (Hou et al. 2022; Wang et al. 2022b; Wang et al. 2021b). Apart from its traditional application as a soil amendment to improve soil fertility and immobilize contaminants, non-soil applications of biochar have also emerged in recent years, which have been proven to exhibit much potential to mitigate climate change (Bartoli et al. 2020; Bolan et al. 2022). For instance, it has been used as an additive to animal feed to enhance its growth while reducing methane (CH₄) emissions simultaneously (Leng et al. 2012; Man et al. 2021). Besides, it has been applied in biorefinery as a green catalyst (Kumar et al. 2020a; Xiong et al. 2017). Biochar can also be added to cementitious materials to enhance hydration and reduce the use of traditional Portland cement with a high carbon footprint (Danish et al. 2021; Maljaee et al. 2021a). It is also used for novel batteries and supercapacitors to store energy (Cheng et al. 2017; Li et al. 2021). Finally, it can be directly applied as a sorbent for carbon capture, utilization and storage (CCUS) (Cao et al. 2022; Jung et al. 2019). The detailed discussion on the role of biochar toward carbon neutrality in these different soil and non-soil applications is provided in the following sections (Sections 3~4).

Here we provide a preliminary calculation of the potential for biochar utilization toward carbon neutrality based on available data (Table 1, Fig. 1). It is assumed that 10% of traditional practices of biomass waste management, animal feeding, cement production, CCUS, and land restoration are altered by biochar systems. Firstly, converting waste biomass into biochar, bio-oil, and syn-gas intrinsically stores 2.20×10^9 t CO₂e annually, because it turns labile carbon that will be rapidly mineralized into recalcitrant carbon whose half life exceeds 1000 years (Ippolito et al. 2020; Lehmann et al. 2021; Spokas 2010). Secondly, application of biochar to restore degraded land is the key area of carbon emission mitigation, consuming 5.50×10^8 t CO₂e of as-formed biochar annually even at a mild application rate of 1.5t/ha. Amending

Table 1 Calculating the highest potential of biochar towards carbon neutrality

Description	Value	Reference
Global annual biomass production	6×10^{10} t of C	(Kleiner 2009; UNEP 2009)
Rate of biomass to be discarded as waste	10%	(Kleiner 2009)
Rate of waste biomass to be converted to biochar	10%	Assumed
Biochar yield	50% ^a	(Kleiner 2009; Laghari et al. 2016; Wang et al. 2020g)
Bio-oil and syn-gas yield	50% ^a	(Kleiner 2009; Laghari et al. 2016; Wang et al. 2020g)
Global annual feed production	1.17×10^9 t	(IFIF 2021)
Proportion of ruminant feed	8%	(IFIF 2021)
Proportion of animal feed with biochar amendment	10%	Assumed
Biochar amending rate to animal feed	9%	(Hansen et al. 2012)
Global annual methane production via ruminant emissions	9.7×10^7 t CH ₄	(Chang et al. 2019)
Methane emission reduction rate following biochar addition to animal feed	17%	(Hansen et al. 2012)
Global annual cement production	4.1×10^9 t	(GCCA 2022)
Proportion of cement with biochar amendment	10%	Assumed
CO ₂ generated per ton of cement production	0.5 t CO ₂	(Ali et al. 2011; Andrew 2018)
Biochar application rate in cement	2%	(Chen et al. 2020; Gupta et al. 2018b; Maljaee et al. 2021a)
Annual capture capacity of CO ₂ in CCUS	4.5×10^7 t	(IEA 2022)
Proportion of CO ₂ captured by biochar	10%	Assumed
Adsorption capacity of biochar towards CO ₂	40 mg/g	(Cao et al. 2022; Huang et al. 2015; Karimi et al. 2022)
Total area of global degraded land that is committed to be restored	1×10^9 ha	(UN 2022)
Proportion of degraded land with biochar amendment	10%	Assumed
Biochar application rate	1.5 t/ha ^b	(An et al. 2022; Paneque et al. 2016; Peng et al. 2021)
Percentage of soil CO ₂ emission mitigation by biochar	1%	(Lyu et al. 2022)
Average CO ₂ emissions from soil without amendment	811 g C/(m ² -year)	(Bond-Lamberty and Thomson 2010)
Percentage of soil CH ₄ emission mitigation by biochar	7%	(Lyu et al. 2022)
Average CH ₄ emissions from soil without amendment	87.6 g CH ₄ /(m ² -year)	(Le Mer and Roger 2001)
Percentage of soil N ₂ O emission mitigation by biochar	31%	(Lyu et al. 2022)
Average N ₂ O emissions from soil without amendment	0.876 g N ₂ O-N/(m ² -year)	(Wu et al. 2021)

^a Slow pyrolysis, which is the most widely used biochar fabrication method

^b A relatively mild rate of biochar application at field

biochar also alters the GHG flux of the soil environment via altering the microbial activities, providing an additional mitigated emission of CO₂, CH₄, and N₂O, whose reduced emissions would be 2.97×10^7 t CO₂e, 1.53×10^8 t CO₂e, and 1.27×10^8 t CO₂e per year, respectively (Bond-Lamberty and Thomson 2010; Le Mer and Roger 2001; Lyu et al. 2022; Wu et al. 2021). Thirdly, it requires 3.86×10^7 t CO₂e biochar to be used as an additive to ruminant feed per year, which theoretically reduces 4.12×10^7 t CO₂e of CH₄ emissions from ruminants annually. Besides, 3.01×10^7 t CO₂e biochar can be incorporated into cementitious materials to improve their mechanical strength, reducing 4.10×10^6 t CO₂e emissions per year due to material replacement. Furthermore, application of biochar as a sorbent for CO₂ capture may directly sorb 4.5×10^6 t CO₂e from the atmosphere, assuming that biochar

accounts for 10% CO₂ sorption in CCUS. Finally, the remaining 6.93×10^7 t CO₂e biochar can be used for other purposes including catalyst in chemical engineering, supercapacitor for energy storage, wastewater treatment, contaminated site remediation, etc. The preliminary calculation provided here highlights the huge potential of biochar to mitigate as high as 2.56×10^9 t CO₂e total emissions per year, accounting for 5.0% of the global GHG emissions of 5.15×10^{10} t CO₂e (UNEP 2021). It is noteworthy that biochar production itself contributes 85.9% of the reduced emissions. A recent precise calculation provided by Lehmann et al. (2021) suggested that the highest potential of biochar's mitigation could reach 2.4×10^9 t CO₂e ~ 3.9×10^9 t CO₂e per year. Another calculation provided by Woolf et al. (2010) suggested that 1.8×10^9 t CO₂e annual net emissions of GHG can be reduced via biochar systems.

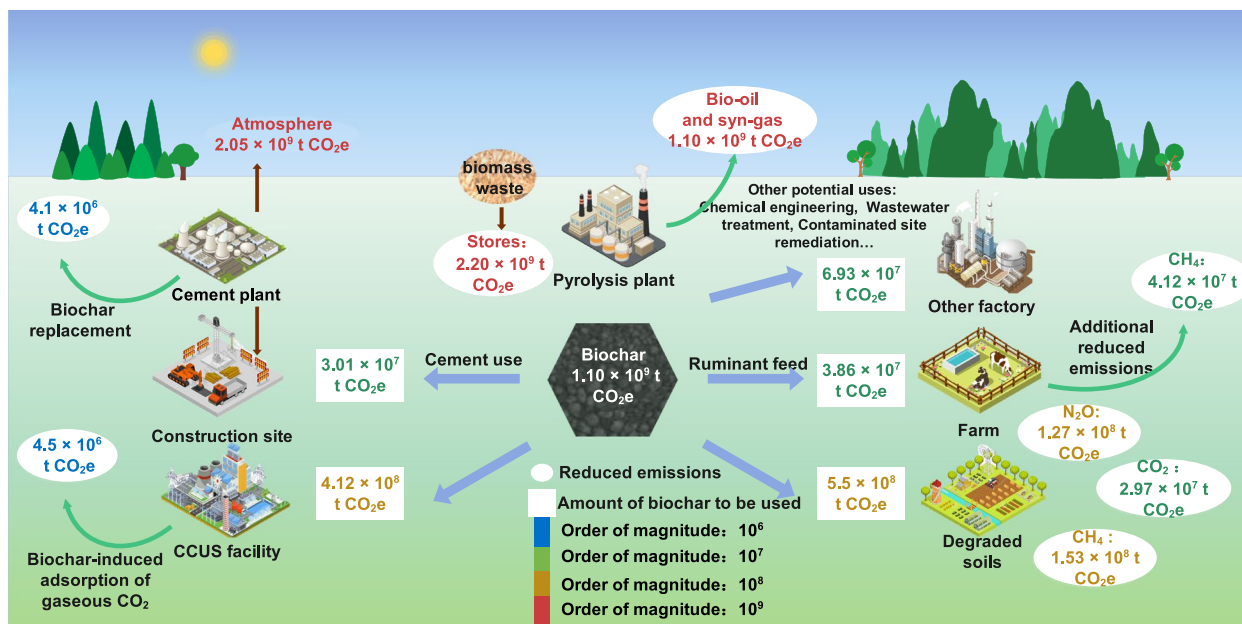


Fig. 1 Potential use of biochar for various applications toward carbon neutrality

3 Biochar as a soil amendment reducing GHG emissions

The promise of biochar as an organic fertilizer is tremendous. Firstly, it delivers necessary inorganic nutrients to plants via dissolution of its ash components (Silber et al. 2010; Wang et al. 2020f). In this case, ash-rich and animal waste-derived biochars, such as manure biochar may be the best candidate for inorganic nutrient delivery (Binh Thanh et al. 2022; Hossain et al. 2020; Subedi et al. 2016). Secondly, the high cation exchange capacity and anion exchange capacity of biochar manifest itself as a good sorbent preventing nutrient loss (Al-Wabel et al. 2018; Lawrinenko and Laird 2015). Low temperature biochars possess more ion exchange sites, which may have performed better in nutrient retention (Li et al. 2019). The slow release nature of biochar-based fertilizers makes it a better candidate for soil fertility improvement in the long term as compared with other traditional fertilizers (Sim et al. 2021; Yao et al. 2013). It could be that the well-developed pores of biochar serve as barriers that delay nutrient release from biochar matrix to the soil environment (Rombel et al. 2022). It's critical that inorganic nutrients can be released gradually at a slow rate to prevent water contamination by nitrate and phosphate (Corbett et al. 2022; Kumari and Maiti 2022; Yin et al. 2022; Zou et al. 2022). Thirdly, the organic matrix of biochar acts as promising habitat for plant growth promoting bacteria (PGPB) (Bertola et al. 2019). Again, low-temperature biochars with low aromaticity are good candidates because bacteria can utilize aliphatic moieties

much easier than aromatic ones (Luo et al. 2018; Zhong et al. 2020). Stimulated microbial activities within the charosphere account for improved soil health within the rhizosphere, thus leading to an elevated plant biomass yield (Dissanayake et al. 2022; Weralupitiya et al. 2022).

The indirect effect of biochar on plant growth promotion should not be neglected. Improved water holding capacity (Bruun et al. 2022; Razzaghi et al. 2020), enhanced aggregation characteristics (Islam et al. 2021), suppressed acidity (Dai et al. 2017), reduced salinity (Wang et al. 2022f), suppressed activities of pathogens (de Medeiros et al. 2021), as well as enhanced activities of soil fauna (Lehmann et al. 2011) contribute to biochar's performance in promoting plant growth.

Biochar has also been widely used as a sorbent for metal and organic contaminant immobilization in soil, whose mechanisms have been thoroughly reviewed (Ahmad et al. 2014; Bandara et al. 2020). Here we stress that application of biochar in remedies aligns very well with the ongoing Green and Sustainable Remediation (GSR) movement due to the following reasons (Fig. 2) (Hou 2020; Hou 2021a; Wang et al. 2021b). Firstly, biochar material is a green amendment because it is waste-derived and carbon-negative (Fig. 2a) (Glaser et al. 2009; Hou 2021c, 2022). Besides, application of biochar to the soil environment promotes soil health, thus leading to multifaceted benefits including enhancing plant growth, and increasing soil diversity on contaminated land (Fig. 2b) (Ayaz et al. 2022; Baveye 2021; Bolan et al. 2022). The restored ecosystems may also serve as a good place

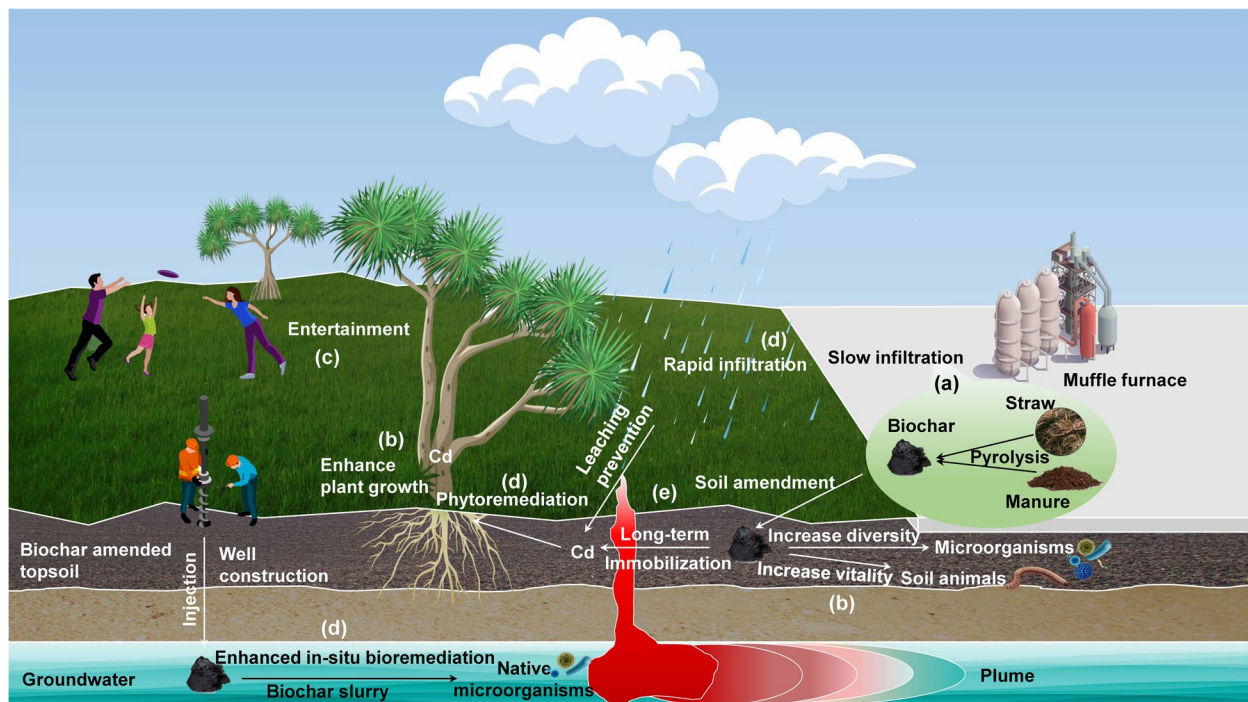


Fig. 2 How biochar application aligns with the ongoing GSR movement

for leisure and entertainment (Fig. 2c) (Ghosh and Maiti 2021). Furthermore, biochar application can increase the resilience and effectiveness of nature-based solutions (NBS) including phytoremediation, constructed wetland, and in-situ bioremediation with native soil microorganisms (Fig. 2d) (Wang et al. 2020a; Wang et al. 2020e). Finally, engineered biochar can achieve excellent long-term effectiveness, rendering elevated resilience to external environmental changes (Fig. 2e) (Wang et al. 2022d; Zhao et al. 2020).

Apart from the fact that biochar application to soil directly increases the soil organic carbon pool, soil application of biochar also provides an additional benefit for climate change mitigation, that is, reducing GHG emissions from soil to the atmosphere. The following sections discuss how biochar application reduces soil GHG emissions.

3.1 CO₂ emissions

Recent attempts have been made to reduce organic carbon loss from soils for climate change mitigation, whose measures include crop residue incorporation (Lupwayi et al. 2022; Zhu et al. 2022a), organic matter stabilization with clay and metal oxides (Baumann et al. 2022; Di et al. 2022), fertilizer application (da Silva et al. 2022; Gasser et al. 2022; Qiu et al. 2022; Thakur et al. 2022; Zhang et al. 2022b), reduced or conservational tillage (Ferrara et al.

2022; He et al. 2022a; Islam et al. 2022), rotational grazing (Abdalla et al. 2022; Dong et al. 2021), silvopastoral system (Aryal et al. 2019; Valenzuela Que et al. 2022), and ecological restoration (Howson et al. 2022; Wang et al. 2020b; Zhao et al. 2022). Among them, application of biochar is a promising one due to simultaneous achievement of waste management, nutrient delivery, contaminant immobilization, and climate change mitigation.

The role of biochar in regulating soil CO₂ emissions is quite complicated. Previous studies have shown that biochar amendment can increase (Johnson et al. 2017; Yuan et al. 2014), decrease (Gascó et al. 2016; Sun et al. 2014), or have negligible effect (Sackett et al. 2015; Zhou et al. 2017) on soil CO₂ emissions. Multiple mechanisms affect this process. Firstly, biochar itself contains certain proportions of labile carbon (i.e., aliphatic moieties) that can be easily metabolized by soil microorganisms (Wang et al. 2020f; Zhong et al. 2020). Secondly, biochar sorbs soil organic matter directly and form aggregates, thus preventing organic matter from being used by microorganisms (Weng et al. 2018; Zheng et al. 2021). Besides, gaseous CO₂ within soil pores can be directly sorbed by biochar (Dissanayake et al. 2020; Jung et al. 2019). Biochar application also alters the activities of certain soil microorganisms that are involved in the carbon cycle (Chen et al. 2016; Palansooriya et al. 2019). The crop utilization efficiency of carbon is also altered following

biochar application, leading to decreased CO₂ emissions (because of plant utilization of labile carbon) (Joseph et al. 2021). Furthermore, sorption of plant root exudates causes a diminished CO₂ emission mitigation effect due to biochar acidification (Weng et al. 2017).

Biochar is relatively stable in the soil environment, which may take over a thousand years to reach full mineralization (Lehmann et al. 2012). Assuming that the mineralization of biochar carbon only had a minor contribution to the total CO₂ flux annually, focusing on the mineralization of native soil organic matter is therefore a necessary step to predict CO₂ emissions from biochar amended soils. Hence it is crucial to judge whether and to what extent the mineralization process of native soil organic matter could be altered by biochar carbon.

Amending fresh organic amendments (e.g., manure, charcoal) to soil has long been acknowledged to alter the mineralization rate of native organic carbon, which is referred to as the priming effect (He et al. 2022c; Kuzyakov et al. 2000; Santoni et al. 2022). Both positive and negative priming could occur for biochar amended soils. The former one refers to the process where carbon mineralization is stimulated, while the latter one refers to the process where the mineralization rate is lowered (thus leading to suppressed CO₂ emissions) (Maestrini et al. 2015; Wang et al. 2022c). Positive priming is induced via

the addition of fresh labile carbon which can stimulate the activities of soil microorganisms, thus accelerating the carbon mineralization process (Fang et al. 2019; Keith et al. 2011; Zimmerman and Ouyang 2019). In contrast, negative priming is achieved via the physical entrapment of labile organic carbon by biochar matrix, thus preventing organic matter from being mineralized (Abbruzzini et al. 2017; Liu et al. 2018b; Lu et al. 2014).

The intrinsic carbon stability and bioavailability of biochar is the key determinant of the priming direction (Fig. 3). Amending Florida soils with a range of biochars produced at different temperatures, Zimmerman et al. (2011) found that low-temperature biochars (250 and 400°C) induced positive priming while high-temperature biochars (525 and 650°C) induced negative priming. A high lignin content of the feedstock also favors negative priming, because the resulting biochar should possess higher aromatic moieties which are much harder to be metabolized as compared with aliphatic ones (Wang et al. 2020g). A low ash content is also favorable for negative priming, because of the limited introduction of inorganic nutrients for microbial growth (Murray et al. 2015). Besides, it is believed that the introduction of certain toxic substances of biochar such as polycyclic aromatic hydrocarbons (PAHs) also contributed to negative priming because of suppressed microbial activity (Lyu et al.

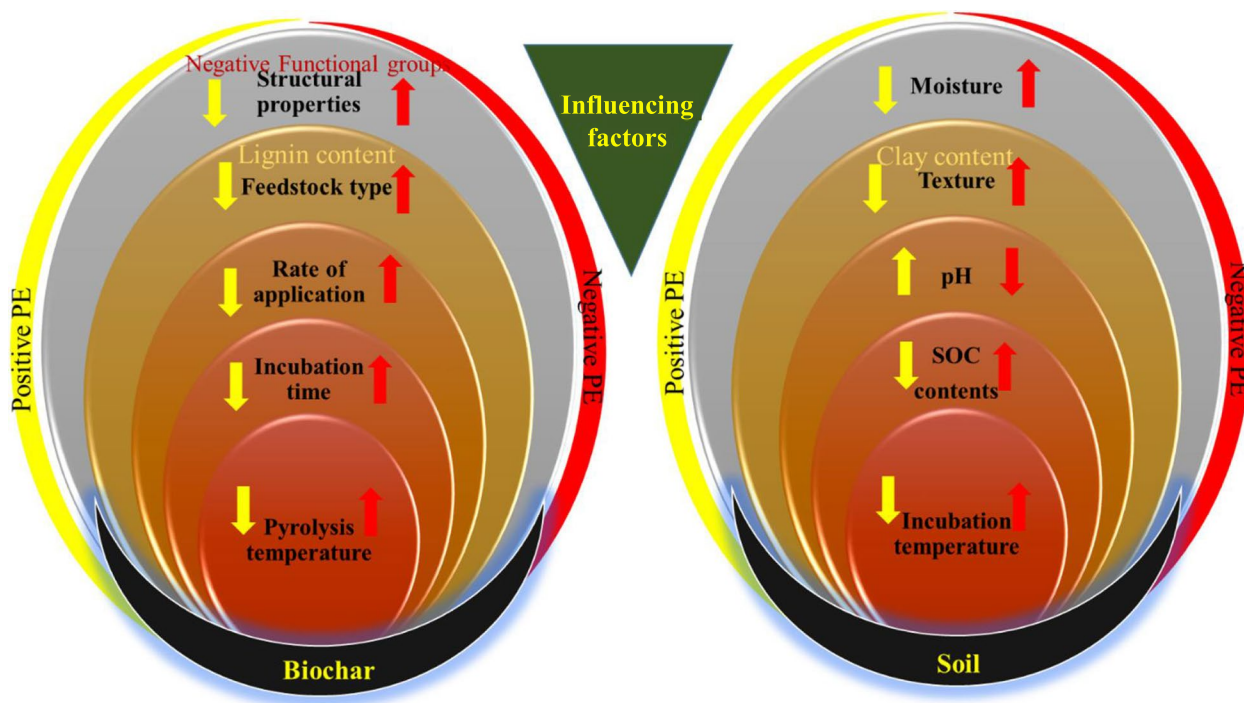


Fig. 3 Effects of biochar properties and soil characteristics on priming directions, affecting BC priming effects. Reproduced with permission from Rasul et al. (2022). Copyright 2022 Elsevier

2016). Soil type is also an important factor affecting the direction of priming (Fig. 3). Wood biochars caused positive priming in clay-poor Inceptisol, but induced negative priming for clay-rich soils including Oxisol and Vertisol (Fang et al. 2015). A high organic carbon content, a low soil pH is also favorable for negative priming (Rasul et al. 2022). Previous meta-analysis suggested that biochar application generally tended to induce negative priming (average retardation of soil organic matter mineralization by 3.8%) (Wang et al. 2016).

It is of note that at the initial stage of biochar application positive priming may be the dominant process because of the microbial utilization of labile carbon fractions from biochar, followed by decreased or even reversed priming intensity following several months of biochar application (Rasul et al. 2022). The initial stimulated positive priming upon biochar application is attributed to the activities of r-strategist microorganisms that respond rapidly to the fresh carbon sources amended to soil (Chen et al. 2021a). The long-term aging of biochar at field benefits negative priming via different mechanisms. A 9.5-year long-term field trial of biochar priming suggested that field aged biochar led to negative priming (via slowing down soil organic matter mineralization by 5.5%) (Weng et al. 2017). Formation of microaggregates protected native soil organic matter from mineralization. Sorption of root exudates, and enhanced organo-mineral protection also accounted for negative priming (Weng et al. 2017). A previous study developed a boosted regression trees machine learning model to depict biochar-induced soil priming effects (Ding et al. 2018). Interestingly, the modeling result suggested that incubation conditions affected priming more (accounting for 36.5%) as compared with biochar properties (33.7%) and soil properties (29.8%). The major factor among incubation conditions was the incubation time (27.1%). Biochar application for the initial 2 years were modeled to exhibit positive priming, after which this trend was reversed to negative priming (Ding et al. 2018).

It should be noted that although negative priming stabilizes soil organic carbon effectively, it may also suppress plant growth via decreasing the bioavailability of nutrients, therefore reducing plant absorption via photosynthesis (Kuppusamy et al. 2016; Qayyum et al. 2017). Finding a suitable biochar amendment that can simultaneously stabilize intrinsic soil organic carbon while simultaneously provide carbon source for plant growth is the key to use negative priming for carbon sequestration in soil.

3.2 CH₄ emissions

Biochar application has great potential for the mitigation of soil CH₄ emissions. The meta-analysis provided

by Lyu et al. (2022) found that biochar reduces soil CH₄ emissions by 7% on average. The meta-analysis provided by Shakoor et al. (2021) suggested that biochar application was more effective for CH₄ (natural log response ratio -0.399) emission mitigation as compared to CO₂ (natural log response ratio -0.108). Wood biochars possessed highest potential to mitigate CH₄ emissions (-1.198), followed by biosolids (-0.544) and herbaceous (-0.263) biochars (Shakoor et al. 2021). Besides, a high pyrolysis temperature is also favorable for CH₄ emission mitigation due to lower available organic carbon content suppressing methanogenesis (Jeffery et al. 2016; Ji et al. 2020). The meta-analysis by Shakoor et al. (2021) also suggested that neutral soil pH (6.6~7.3) favored CH₄ emission mitigation, whereas an acidic nature of the amended soil promoted CH₄ emissions following biochar application (Shakoor et al. 2021). However, there's still controversy over whether an elevation of soil pH should necessarily lead to a reduction in CH₄ emissions. The activities of methanogens may be enhanced at neutral or alkaline environments, whereas the methanotrophs are more tolerant to soil pH changes. Therefore, an elevation of soil pH following biochar application should theoretically stimulate CH₄ emissions (Yu et al. 2013). Another meta-analysis by Jeffery et al. (2016) also drew a contradictory result that biochar has much higher potential to mitigate CH₄ emissions in acidic soils.

Soil moisture content greatly affects CH₄ emissions of the biochar amended soil. A moderate water content is favorable for the reduction of CH₄ emissions, whereas a nearly saturated water content greatly stimulates CH₄ emissions (Yu et al. 2013). This is because a decreased soil Eh suppresses the activities of methanotrophs while stimulates the growth of methanogens. This is the reason why rice paddies release CH₄ at a higher rate as compared with upland soils (Feng et al. 2012). Numerous attempts have therefore investigated the effectiveness of biochar application on CH₄ emissions in rice paddy. A decreased bulk density, along with an improved soil aeration, is the main reason for CH₄ emission reduction in such reduced environments. From a more mechanistic perspective biochar application regulates the abundance of certain functional genes including *mcrA* (methanogen, CH₄ synthesis, suppressed) and *pmoA* (methanotroph, CH₄ oxidation, enhanced) (Lyu et al. 2022).

The time following biochar application may have also altered the activities of methanogens and methanotrophs. A 4-year field trial conducted in rice paddy suggested that release of labile organic carbon accounted for the stimulation of both methanogens and methanotrophs in the first year of biochar application, whereas the stimulation effect was much more significant for the latter group. However, biochar suppressed the activities

of methanogens (via increased soil porosity) while exhibited little impact on methanotrophs in the next 3 years (Wang et al. 2019). The findings highlighted the fact that changes in soil physical and chemical properties by biochar require much time. Aging serves as a pivotal factor affecting biochar's performances in CH₄ emission mitigation at field. A faded liming effect, a gradual decrease of labile organic carbon content, as well as a slow improvement of soil aggregation and aeration are all potential influencing factors (Nan et al. 2021a; Nan et al. 2021b).

3.3 N₂O emissions

The potential of biochar to mitigate soil N₂O emissions is tremendous. The global meta-analysis by Borchard et al. (2019) suggested that biochar application could mitigate soil N₂O emissions by 38% on average. Another meta-analysis by Cayuela et al. (2014) reached more inspiring results of 54% mitigation on average. The recent work by Lyu et al. (2022) found that the GHG mitigation potential of biochar followed the order N₂O > CH₄ > CO₂. From a regional perspective, Lee et al. (2021) suggested that biochar application to East Asian soils had the potential to mitigate soil N₂O emissions by 21.1%. Biochar application rapidly reduces N₂O emissions. The lab incubation experiment using seven different biochars suggested that short-term application of biochar was effective to reduce N₂O emissions dramatically by 52%~84% within only 14 days (Nelissen et al. 2014). Besides, N₂O emission is not necessarily associated with limited N bioavailability in biochar amended soil, suggesting that chemical fertility can be remained apart from a reduction of GHG emissions (Case et al. 2015).

Biochar and soil characteristics are also key factors affecting the mitigation effect. Biochar produced from wood is the best candidate (Borchard et al. 2019). A high pyrolysis temperature over 400 °C is also favorable for N₂O emission mitigation (Lee et al. 2021). Low C/N biochars (below 30) have limited mitigation potential, whereas high C/N biochars are effective for emission mitigation (Cayuela et al. 2014). The underlying mechanism may be that higher aromaticity and adsorption capacity (toward soil organic matter) of high C/N biochars reduce the bioavailable carbon content, thus suppressing denitrification which is a facultative process that requires external organic carbon as the electron donor (Cayuela et al. 2014). Another reason is that biochars with higher aromaticity are more conductive, thus favoring the electron transfer to denitrifiers (Lyu et al. 2022). The H:Corg ratio can also be applied to predict biochar's mitigation potential. Biochar with a high aromaticity possesses a H:Corg ratio below 0.3, which lowers N₂O emissions by 73 ± 7% on average. In contrast, biochar with a high H:Corg ratio over 0.3 is much less effective (i.e., lowers N₂O emissions

by 40 ± 16%) (Cayuela et al. 2015). The strongest reduction in N₂O emissions was observed for paddy soils and sandy soils via meta-analysis (Borchard et al. 2019). Besides, a low soil organic matter content usually results in higher N₂O emission mitigation (Cayuela et al. 2014). N₂O emissions in upland soils are affected more following biochar application as compared with waterlogged rice paddy (Lyu et al. 2022).

Biochar application is known to promote the expression of *nosZ* gene of the denitrifiers, leading to a complete reduction of NO₃⁻ to N₂ instead of N₂O (Harter et al. 2014; Lyu et al. 2022; Van Zwieten et al. 2014). An elevation of soil pH stimulates this process. A higher application rate of biochar also leads to improved mitigation effect (Cayuela et al. 2014). For instance, applying biochar at 1%~2% reduces N₂O emissions by 27% on average, whereas a higher dosage over 10% reduces its emissions dramatically by 87% (Cayuela et al. 2014). Application rate and type of N fertilizers also affect this process (Vangeli et al. 2022). Lee et al. (2021) found that a moderate N fertilizer application rate of 500~600 kg/ha is favorable for N₂O emission mitigation, whereas higher application rate beyond this value leads to reduced mitigation potential. Ammonium nitrate (reduction 32.3%) is better than urea (22.7%) and ammonium sulfate (15.2%) to assist biochar's N₂O emission mitigation (Lee et al. 2021). Physical immobilization of NO₃⁻ is another important mechanism (Case et al. 2012). Formation of a distinct "charosphere" that is locally anaerobic and alkaline also promotes complete denitrification (Ameloot et al. 2016).

A diminished mitigation effect may take place during field aging of biochar. The meta-analysis by Borchard et al. (2019) suggested that the effectiveness would remain effective only in the first year of biochar application. A 4-year field trial also found that biochar's mitigation potential was 31.5%~42.4% during the first year but diminished gradually to 6.9%~21.3% in the 3rd and 4th years (Liao et al. 2020). Another long-term field trial also found that biochar did not significantly reduce N₂O emissions after 7 years of in-situ field aging (Liao et al. 2021). With progressive aging the surface of biochar will be more oxidized, thus reducing biochar's liming effect (Cayuela et al. 2014; Yang et al. 2022). Aged biochar may have also stimulated both ammonia-oxidizing archaea and ammonia-oxidizing bacteria induced N₂O production (Zhang et al. 2019). Direct application of artificially aged biochar provides further support for the faded mitigation potential. In a recent work biochar suffered from proton attack were applied to a Luvisol, exhibiting no mitigation potential toward N₂O (Zhang et al. 2022a). It is notable that certain types of biochar may remain its mitigation potential with aging. For instance, wood biochar with a high aromaticity can continuously reduce

N₂O emissions for 3 years in a field trial (Hagemann et al. 2017).

4 Biochar as sustainable material with low carbon footprint in non-soil applications

4.1 Bioenergy generation

Biomass pyrolysis generates biochar, bio-oil, and syngas. Bio-oil is the liquid byproduct generated during biochar pyrolysis. It is a dark brown liquid with a pungent smell (Gupta et al. 2021a). Major components of bio-oil include alcohols, phenols, hydroxyketones, esters, carboxylic acids, etc. (Brassard et al. 2017; Sekar et al. 2021). Owing to the high energy density, bio-oil is a promising energy for combustion engines and boilers. However, relatively high oxygen and moisture content (due to release of internal moisture of the biomass and the dehydration process) affects its performances when directly using it as an energy source (Kung et al. 2022; Lee et al. 2020a). Upgrading is therefore a necessary step. Physical upgrading using density differences, or chemical upgrading via a catalytic cracking and catalytic hydrotreating are feasible ways to improve the quality of bio-oil to meet the quality of hydrocarbon grade fuels (Gupta et al. 2021a). Syngas consists of H₂, CO, CH₄, C₂H₄, etc., which can also be used as a clean energy source (Amenaghawon et al. 2021; You et al. 2018). In particular, it can be directly used for a supplementary energy source of the reactor during pyrolysis. Typical yield of syn-gas is below 40% (Amenaghawon et al. 2021). Higher pyrolysis temperature favors syn-gas generation due to enhanced release of volatile matters and tar formation.

Bio-oil and syngas formed during biomass are sustainable energy sources contributing to carbon neutrality. This is primarily because GHG emissions generated during biomass pyrolysis are regarded as carbon neutral because of their biogenic origin (Lu and El Hanandeh 2019; US EIA 2021). Previous calculations suggested that effective utilization of bio-oil and syn-gas as clean energy could greatly contribute to carbon neutrality. Ideally, a rough calculation assumes that one third of the aforementioned 6 billion tonnes of biomass could be converted to bio-oil used for transport, thus reducing carbon emissions by an additional 1.8 billion tonnes per year (Kleiner 2009). The life cycle analysis by Roy and Dias (2017) suggested that global warming potential (GWP) of bio-oil utilization could be 54.5% lower than that of fossil fuels. Fan et al. (2011) provided a life cycle analysis of bio-oil utilization for electricity generation. Results suggested that bio-oil combustion led to life cycle GHG savings ranging from 77%~99% as compared with fossil fuel combustion. Another life cycle analysis of biochar systems suggested that biomass conversion into biochar, bio-oil, and syngas yielded a positive net energy while achieving negative

net GHG emissions (Roberts et al. 2010). Slow pyrolysis of one ton of plant biomass yielded a net energy value ranging from 3044~4899 MJ, whereas syngas heat energy production (3507~5787 MJ) was the primary contributor to the net positive energy. In the meantime, net GHG emissions were negative within biochar production system, whose values ranged from -885~-793 kg CO₂e per ton of dry biomass.

4.2 Animal feed

From the early 19th to 20th, charcoal had been applied widely to treat digestive disorders in animals, including horses, pigs, and dogs (Bolan et al. 2022; Man et al. 2021). In recent years, a few attempts have been made using modern biochar as a feed additive for animals. An application rate of <10% in daily diet could achieve significant weight gain, egg weight, and survival rate (Man et al. 2021). Biochar has multifaceted benefits in aquaculture. Firstly, it is a promising nutrient source which benefits animal growth and immunity enhancement (Evans et al. 2017; Gerlach and Schmidt 2012; Saleem et al. 2018). Secondly, it is an excellent sorbent to remove toxic elements and organic substances from animal gut (Jandosov et al. 2017). Third, biochar suppresses the activities of certain pathogens (such as *Clostridium tetani*) (Prasai et al. 2016).

A stringent feedstock selection is crucial for biochar application in aquaculture. For instance, European Biochar Foundation proposes that the feedstock should be free of paint, solvent, plastic, and rubber, and that only those feedstock on the “positive list” can be pyrolyzed for animal feeding (Man et al. 2021). Other criteria include: carbon content should reach at least 80% of dry matter, polycyclic aromatic hydrocarbons (PAHs) concentration should below 4 mg/kg. As, Pb, Cd, and Hg concentration should below 2, 10, 1, and 0.1 mg/kg, respectively (Man et al. 2021).

Several evidences have shown that biochar addition to animal diet could effectively reduce the emissions of methane by ruminants (e.g., cattle and goat). The emission of methane from enteric fermentation of livestock reached 171 million tons CO₂e (Man et al. 2021). Amending biochar in animal diet may serve as a potent measure to mitigate this source of GHG emissions. The reason why biochar suppresses the emission of methane lies in the fact that biochar provides a suitable habitat for gut methanotrophs. Therefore, the oxidation of methane (which is generated by methanogens) is greatly enhanced (Black et al. 2021; Hansen et al. 2012; Leng et al. 2012) (Fig. 4). Besides, biochar itself also directly sorbs methane, which also accounts for reduced methane emissions from ruminants (Fig. 4). However, a recent study

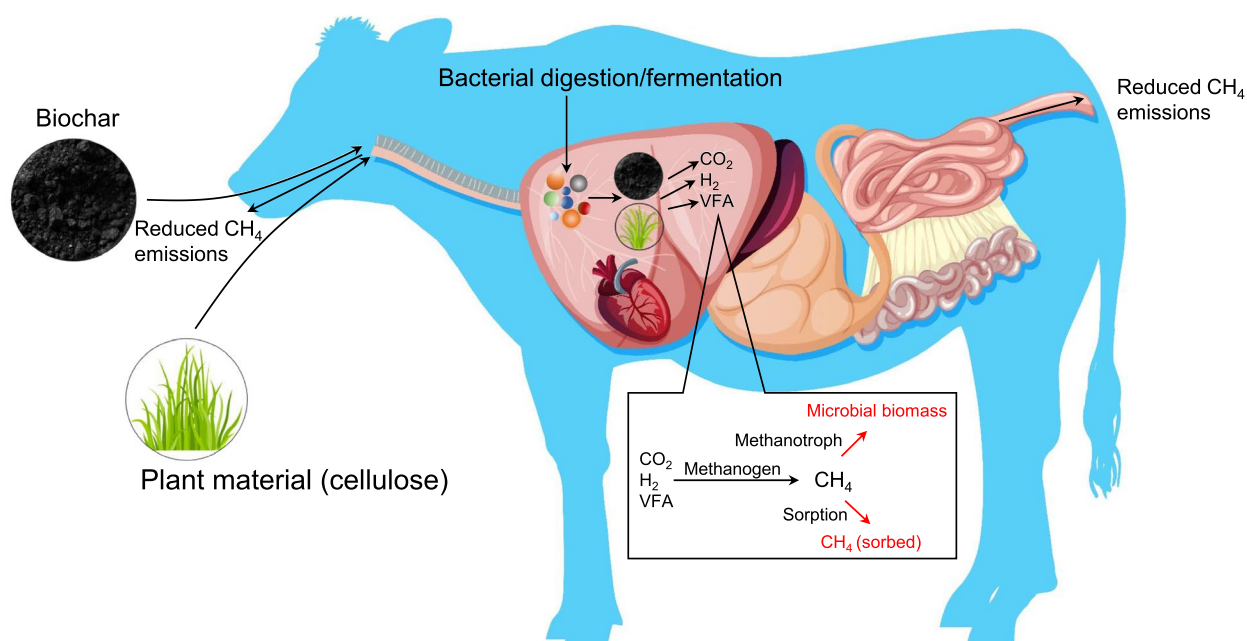


Fig. 4 Possible processes leading to CH₄ emission mitigation by ruminants after incorporating biochar into animal feed

by Romero et al. (2022) found that biochar addition to manure as a cattle diet did not significantly reduce CO₂, N₂O or CH₄ emissions. More evidences are needed to test whether and which types of biochar addition could successfully reduce ruminant GHG emissions.

4.3 Green catalyst for biorefinery

Biochar can act as a proxy for traditional activated carbon as a novel support for metal catalysts. For instance, a novel Ru-ReOx catalyst was loaded onto biochar support pyrolyzed from rice straw, which can produce value added chemicals from furan (such as tetrahydrofuran and 1,4-butanediol) (Lee et al. 2020b). The biochar supported catalyst was exhibited three times more active than the one supported by traditional activated carbon (Lee et al. 2020b). Presence of alkali minerals in the ash component of rice straw biochar contributed to the catalytic performance of biochar-supported catalysts (Lee et al. 2020b). It was suggested by Ramos et al. (2022) that biochar supported metal catalysts can be used for biorefinery, including transesterification (Hazmi et al. 2021; Zhang et al. 2016a), hydrogenation/hydrodeoxygenation (Liu et al. 2018a; Santos et al. 2020), reforming and gasification (Shen et al. 2014; Yao et al. 2016), pyrolysis (Nejati et al. 2020; Richardson et al. 2010), and hydrolysis reactions (Wei et al. 2020; Zhang et al. 2014). Successful incorporation of metal and metal oxides is the key to catalyst fabrication. Therefore, an abundant oxygen-containing

functional group on the biochar matrix favors metal decoration (Xiong et al. 2017). Besides, a relatively high ash content assures the presence of alkali minerals (i.e., K), which have been proven to exhibit additional catalytic performances toward certain reactions such as transesterification (Ramos et al. 2022).

4.4 Low-carbon infrastructure

Portland cement contributes to 8% of the global CO₂ emissions (Andrew 2018). To mitigate the carbon footprint of this conventional building material, several attempts have been made to use biochar as a green additive to traditional cement (Table 2). It was found that adding biochar application at a dosage below 10% will not cause detrimental impacts on the mechanical properties of cements (Danish et al. 2021). Actually, previous study suggested adding merely 1% of biochar to the cement matrix could increase its compressive strength dramatically by 8.9% (Wang et al. 2020d). Elevated elastic modulus (Gupta et al. 2018b), decreased water absorption behavior (Gupta et al. 2020a), and improved shrinkage properties (Gupta et al. 2018a) were also observed. The stability of biochar incorporated into the cementitious matrix is excellent as compared with direct incorporation of fresh biomass, which is because of the same alkaline nature of cement matrix and pyrolyzed biochar.

Hydration is a key process of cement-based materials, during which silicate phases of portland cement react with water to form calcium silicate hydrate (C-S-H).

Table 2 Biochar as an additive to cementitious materials

Biomass feedstock	Pyrolysis temperature (°C)	Cement type	Addition rate (%)	Key findings	Reference
Wheat straw	650	Magnesium phosphate cement	1.5	Biochar addition improved the resistance of cement in water.	(Ahmad et al. 2020)
Rice husk and bagasse	700	Portland cement	5, 10	Biochar addition increased the compressive and tensile strength.	(Asadi Zeidabadi et al. 2018)
Wood	700	Portland cement	10, 20, 30	Biochar addition at 30% sequestered 59 kg CO ₂ per ton of cement.	(Chen et al. 2022a)
Corn straw	300	Portland cement	1, 3, 5	Biochar addition compensated for the strength loss caused by carbonation curing.	(Chen et al. 2022b)
Sewage sludge	500	Portland cement	2	Biochar addition accelerated hydration.	(Chen et al. 2020)
Sewage sludge	700	Portland cement	1, 2, 5, 10	Biochar addition accelerated hydration.	(De Carvalho et al. 2022)
Wood	500	Portland cement	2, 5, 8	Biochar addition accelerated hydration.	(Dixit et al. 2019)
Wood waste and coconut shell	500	Portland cement	5	Biochar addition reduced autogenous shrinkage.	(Gupta et al. 2020a)
Wood	300, 500	Portland cement	2	Biochar addition promoted mechanical and permeability properties.	(Gupta and Kua 2018)
Wood	500	Portland cement	0.25, 0.50, 1, 2	Biochar addition accelerated hydration.	(Gupta and Kua 2019)
Wood	300, 500	Portland cement	1, 2, 5, 8	Biochar addition increased the compressive strength, but did not significantly affect tensile strength.	(Gupta et al. 2018a)
Food waste, rice waste, wood	500	Portland cement	1, 2, 5	Biochar addition increased the compressive and tensile strength.	(Gupta et al. 2018b)
Wood	300	Portland cement	2	Biochar addition increased the compressive strength and improved permeability.	(Gupta et al. 2018c)
Wood	500	Portland cement	0.5, 1, 2	Biochar addition increased the compressive strength.	(Gupta et al. 2020b)
Rice husk, wood	500	Portland cement	1, 2	Biochar addition increased the compressive strength and decreased permeability.	(Gupta et al. 2021b)
Corn straw	400	Magnesium oxychloride cement	5, 10, 15, 20	Biochar addition formed an internal network with excellent cohesion strength.	(Han et al. 2022)
Bagasse, coconut husk, peanut husk, rice husk, wheat husk	500	Portland cement	1, 2, 3, 4, 5	Biochar addition promoted hydration and decreased the settling time.	(Haris Javed et al. 2022)

Table 2 (continued)

Biomass feedstock	Pyrolysis temperature (°C)	Cement type	Addition rate (%)	Key findings	Reference
Bamboo	650	Portland slag cement	0.2, 0.4, 1, 2, 3, 4	Biochar addition increased the compressive strength and built resistance to cracking.	(Liu et al. 2022a)
Olive stone, rice husk, wood	500	Portland cement	0.5, 1, 2, 4	Biochar addition promoted hydration.	(Maljaee et al. 2021b)
Oilseed rape, wood	700	Portland cement	2, 4, 6, 8	Biochar addition increased the compressive strength.	(Park et al. 2021)
Poultry litter	450	Portland cement	10, 20, 40	Biochar addition decreased the density and increased the compressive strength.	(Praneeth et al. 2021)
Poultry litter	450	Portland cement	20, 40	Crushed biochar-cement mortar effectively sorbed Zn, Cu, and Pb in aqueous media.	(Praneeth et al. 2022)
Wood	500	Portland cement	0.65, 3.2, 6.5, 9.5, 13.5	Biochar addition promoted hydration.	(Qin et al. 2021)
Sugarcane bagasse	200	Portland cement	2	Biochar addition promoted hydration.	(Rodier et al. 2019)
Wood	400	Portland cement	2, 5	Biochar addition improved the microstructure of the concrete.	(Sirico et al. 2022)
Wood	900	Portland cement	1, 2.5	Biochar addition did not affect mechanical properties.	(Sirico et al. 2020)
Wood	400, 500, 600, 700	Portland cement	0, 1, 3, 5, 10	Biochar addition decreased fluidity.	(Tan et al. 2020)
Wood	500	Portland cement	1, 3, 5, 8	Biochar addition increased the compressive and tensile strength.	(Tan et al. 2022)
Wood	Gasification biochar	Magnesia cement and magnesia cement-Portland binary cement	2	Biochar addition promoted hydration.	(Wang et al. 2021a)
Wood	500, 700	Portland cement	1, 2, 5	Biochar addition increased the compressive strength.	(Wang et al. 2020d)
Rice husk	550	Portland cement	2, 5	Biochar addition decreased the compressive strength.	(Yang and Wang 2021)

Evidences have shown that biochar addition accelerates the hydration process (Lv et al. 2022) (Table 2). This is because fine biochar particles fill the voids between cement particles and aggregates, and promote the formation of clusters due to electrostatic interaction between their negatively charged surfaces and the positively charged cement particles. This results in the formation of nucleation clusters, favor the hydration

process by attracting more surrounding particles as compared with the unamended cement (Fig. 5) (Danish et al. 2021; Gupta et al. 2021b). Besides, biochar addition also shortens the setting times, which is possibly due to the high hydrophobicity of biochar that reduces the segregation of concrete (Chen et al. 2020).

Carbonation is another major process which affects the longevity of cementitious materials. It was found

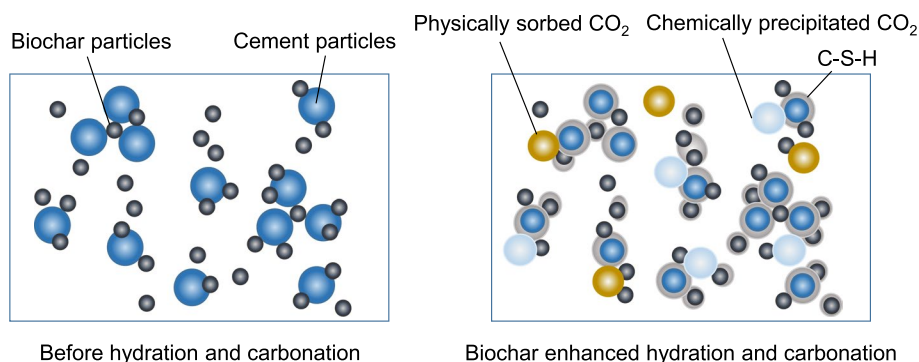


Fig. 5 Biochar incorporation accelerates the hydration and carbonation processes of cementitious materials. Key information obtained from Gupta et al. (2021b), Legan et al. (2022), and Danish et al. (2021)

that biochar addition could facilitate this carbonation process, resulting direct CO_2 capture from the atmosphere ($0.033 \text{ mmol CO}_2/\text{g biochar} \sim 0.138 \text{ mmol CO}_2/\text{g biochar}$) (Fig. 5) (Legan et al. 2022). In this case higher specific surface area led to higher CO_2 absorption performance of biochar amended concrete. However, one should not neglect the detrimental impact of carbonation on mechanical properties of the concrete material. C-S-H reacts with CO_2 to form calcite, which precipitates within the pores of concrete (Fig. 5), thus reducing its mechanical strength. Besides, CO_2 diffusion causes decrease in pH, which results in corrosion (Ekolu 2016; Qiu 2020). It deserves further explorations concerning whether mechanical improvements following biochar addition outplay the detrimental impact resulting from accelerated carbonation.

The rise in impervious surfaces in cities increases surface runoff, which leads to more frequent flooding events in urban areas with climate change (He et al. 2022b; Wang et al. 2022a). In this context, low impact development (LID) (also known as Sponge City in China) has emerged as an effective urban runoff prevention philosophy (Eckart et al. 2017; Jia et al. 2017). Apart from acting as an additive to cementitious materials, biochar can also be used in infiltration-based LID to treat stormwater as a bioretention system. To be used for bioretention, a filter material should possess a high hydraulic conductivity plus a high storage volume. The porous nature of biochar makes it an excellent candidate in this case as compared with other geomedia such as clay. It also offers multiple benefits such as the removal of potentially toxic elements (PTEs) from stormwater, increased groundwater replenishment, improved growth of plants, and an elevated resilience to the changing environment (Mohanty et al. 2018; Tsang et al. 2018). In addition, it can also be applied as green roof to reduce urban runoff by increasing water holding capacity (Werdin et al. 2021).

4.5 Novel batteries and supercapacitors

The application of biochar for energy storage applications is a very impressive route of creating additional value apart from long-term carbon storage. Biochar has been recently used as the anode material for the conventional lithium-ion batteries due to its well developed porous structure. The interconnected pores promote Li^+ transfer, introduce heteroatoms, making biochar a good proxy to graphite (Chen et al. 2021b; Ryu et al. 2016; Salimi et al. 2017; Wang et al. 2022g). Biochar doping with certain elements (e.g., Sn, Ni) can further improve its discharge capacity and reusability (Nie et al. 2021; Zhang et al. 2021). It can also be used as novel cathode materials for other types of batteries. For instance, Lei et al. (2021) fabricated a Fe_3C -biochar composite as the cathode for lithium sulfur battery, which remained a steady discharge capacity of 555 mAh/g even after 250 cycles. Zhong et al. (2022) synthesized biochar composite co-doped with N, P, and O from waste biomass feedstock generated following phytoextraction of heavy metals. The as-formed composite was used as the cathode for lithium sulfur battery, which had 80% of its discharge capacity remained following 200 cycles. Qiao et al. (2021) synthesized KCl and heat modified biochar and used it as a cathode for zinc-air battery. Compared with the conventional Pt/C based zinc air battery, biochar-based battery exhibited higher specific capacity (767 mAh/g Zn vs 684 mAh/g Zn) and peak power density (141 mW/cm^2 vs 126 mW/cm^2) (Qiao et al. 2021). Furthermore, biochar has also been used as a pigment in solar absorber coatings owing to its possibility to reduce the reflectance of the material (Gonzalez-Canche et al. 2021). It can also be utilized as a proxy to traditional Pt catalyst in dye solar cells (Tiihonen et al. 2021).

Another novel type of energy storage technology, namely, supercapacitor, has gain much attention because of its rapid charge-discharge characteristics and high reusability (Salanne et al. 2016).

Supercapacitors can be divided into two categories. The first one is the electrochemical double-layer capacitor (EDLC) which physically sorbs and desorbs the electrolyte ions on the electrodes under external voltage (Jeanmairet et al. 2022). In this case, a high specific surface area and electrical conductivity of the biochar material as the electrode favors energy storage (Osman et al. 2022). Biochar or biochar-derived material with a 3D interconnected hierarchical porous structure performs well in an EDLC (Fig. 6) (Gao et al. 2020; Li et al. 2021). Micropores below 2 nm provide much specific surface area for physical sorption, whereas mesopores (2 ~ 50 nm) and macropores (> 50 nm) favor ion transport because the ion diffusion distance is shortened (Cuong et al. 2021). Steam activation (Kim et al. 2019), alkaline modification (Liu et al. 2015), metal modification (e.g., ZnCl_2) (Hou et al. 2015), and organic modification (e.g., EtOH) (Zhang et al. 2016c) were reported to create hierarchical porous structures (Cuong et al. 2021). Besides, a high pyrolysis temperature produces a graphite-like structure with high electrical conductivity, which is also favorable for biochar application as an EDLC (Keiluweit et al. 2010; Rawat et al. 2022). The other type of supercapacitor is a pseudocapacitor (PC) whose electrode consists of redox materials. A PC stores energy via reversible faradaic reactions (redox reactions) near the electrode surface (Choi et al. 2020). Although biochar-based supercapacitors mostly belong to the former group, several works have successfully fabricated biochar-based EDLC/PC composite supercapacitors (Fig. 6). Transition metal oxides, including MnO_2 (Nirmaladevi et al. 2021), NiO (Paravannoor 2018), and Co_3O_4 (Liu et al. 2016), and certain organic polymers such as polyaniline (Thines

et al. 2016) have been used to modify biochars to fabricate EDLC/PC composite supercapacitors. Reported specific capacitance of biochar supercapacitors ranged from 106 to 1950 F/g, whose reusability was all above 1000 cycles (Cheng et al. 2017). 6 M KOH is the most widely used type of electrolyte for a biochar supercapacitor (Cheng et al. 2017).

4.6 Sorbent for CO_2 capture, utilization, and storage

Carbon capture, utilization, and storage (CCUS) is an important strategy toward carbon neutrality, which may contribute nearly 12% of global mitigation to achieve the goal of limiting global temperature rise within 2°C (Wei et al. 2021). As a porous sorbent itself, biochar can be directly used for CO_2 capture. Physical sorption plays a vital role in this process, suggesting that higher specific surface area should result in higher adsorption capacity (Fig. 7). For instance, Cao et al. (2022) fabricated wood and straw biochars for the adsorption of CO_2 . Wood biochars possessed much higher specific surface areas (which were 2.7 ~ 4.4 times larger than that of the straw biochars), thus resulting in higher adsorption capacities (41.2 ~ 45.8 mg/g vs 26.5 ~ 41.5 mg/g). High reversibility was observed for CO_2 adsorption by biochar. The adsorption capacity of biochar could remain above 96.5% following 10 adsorption-desorption cycles, as observed by Cao et al. (2022). In addition, CO_2 adsorption is an exothermic process. An increase in operating temperature led to decreased adsorption capacity (Cao et al. 2022). Biochar modification with KOH is a popular way to increase its physical adsorption capacity, because KOH removes the impurities and enhances the evaporation of volatile matter, thus leading to a widened pore size, increased pore volume and increased specific surface area (Coromina et al. 2016; Deng et al. 2014; Manyà et al. 2018). However,

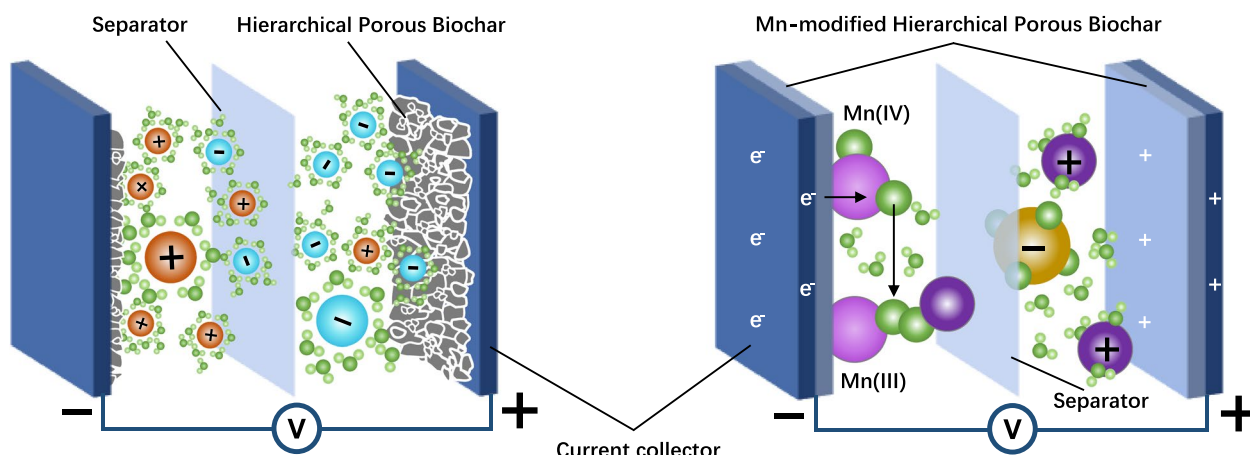


Fig. 6 Schematic diagram of biochar-based EDLC (left) and EDLC/PC composite (right) supercapacitors

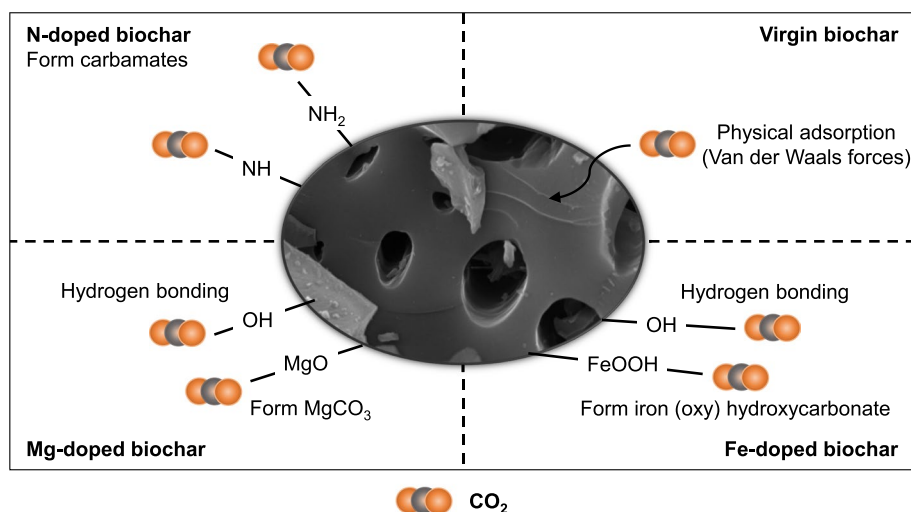


Fig. 7 Mechanisms involved in CO₂ capture by biochar

one should note that the concentration of KOH should be selected with care, because a high KOH concentration may cause damage to the porous structure (due to swelling of lignin structure) (Wang et al. 2020c).

Apart from physical sorption, chemical binding may have also played an additional role. Nitrogen-containing functional groups, such as $-NH_2$ and $-NH$, react with CO₂ directly to form carbamates (Fig. 7) (Chatterjee et al. 2018; Shafawi et al. 2021; Zhang et al. 2016b). Therefore, chemical modification of biochar by amine serves as an effective means to improve its performance in CO₂ adsorption (Chatterjee et al. 2018). NH₃ ambient pyrolysis is also a simple way to introduce nitrogen-containing functional groups (Wang et al. 2020g). But the reversibility of sorption may be impeded by elevated proportion of chemisorption. In particular, aliphatic C-N/C-O groups cannot be regenerated following CO₂ sorption (Jung et al. 2019). Another strategy to improve CO₂ adsorption is to introduce metal oxides onto biochar (Fig. 7). For instance, Liu et al. (2013) fabricated MgO-modified biochar with a high adsorption capacity of 5.45 mmol/g. This is due to the fact that CO₂ also reacts with MgO on the biochar surface to form MgCO₃, and interacts with surface hydroxyl via hydrogen bonding apart from physical adsorption (Liu et al. 2013). CO₂ adsorption onto MgO modified biochar is also highly reversible (Lahijani et al. 2018). Modification by other metals can also increase the adsorption capacity as compared with the virgin biochar, but they may not be as effective as Mg loading. Lahijani et al. (2018) found that the sequence of the adsorption capacity for metal-biochar composites followed the order Mg > Al > Fe > Ni > Ca. An elevated abundance of basicity of biochar following metal doping (due to substitution of proton by metal ions) also accounted for increased CO₂ adsorption (Jung et al. 2019).

5 Selection of a suitable biochar for specific applications

To assure biochar's performances for specific applications, a wise selection of feedstock and pyrolysis condition is crucial. Figure 8 provides a practical guide for the selection of suitable feedstock and biochar fabrication conditions for specific purposes. Plant biomass as the feedstock material renders a low ash content as compared with animal waste or sludge feedstocks (Wang et al. 2020g). A high pyrolysis temperature results in elevated aromaticity and specific surface area (Keiluweit et al. 2010; Zhu et al. 2022b). Therefore, high ash and low temperature biochars are suitable for soil fertility improvement due to nutrient delivery and high labile carbon content (Hossain et al. 2020). High ash and high temperature biochars favor chemical catalysis due to the presence of alkali minerals as well as high conductivity (Ramos et al. 2022). Low ash and low temperature biochars are suitable to be applied for animal feeding because of high carbon content as well as high bioavailability of aliphatic carbon (Man et al. 2021). Low ash and high temperature biochars are suitable to be used for soil GHG emission mitigation (Lyu et al. 2022), CCUS (Jung et al. 2019), cement additive (Danish et al. 2021), and energy storage (Rawat et al. 2022) due to high carbon stability and high adsorption capacity. Biochars pyrolyzed at a medium temperature favor contaminant remediation because the co-existence of well-developed pores and abundant oxygen-containing functional groups (Ahmad et al. 2014; Hou et al. 2020; Hou et al. 2021; Xie et al. 2015). High ash biochars favor metal immobilization due to enhanced precipitation (Wang et al. 2021b), whereas low ash biochars favor organic contaminant retention due to enhanced hydrophobic interactions (Dai

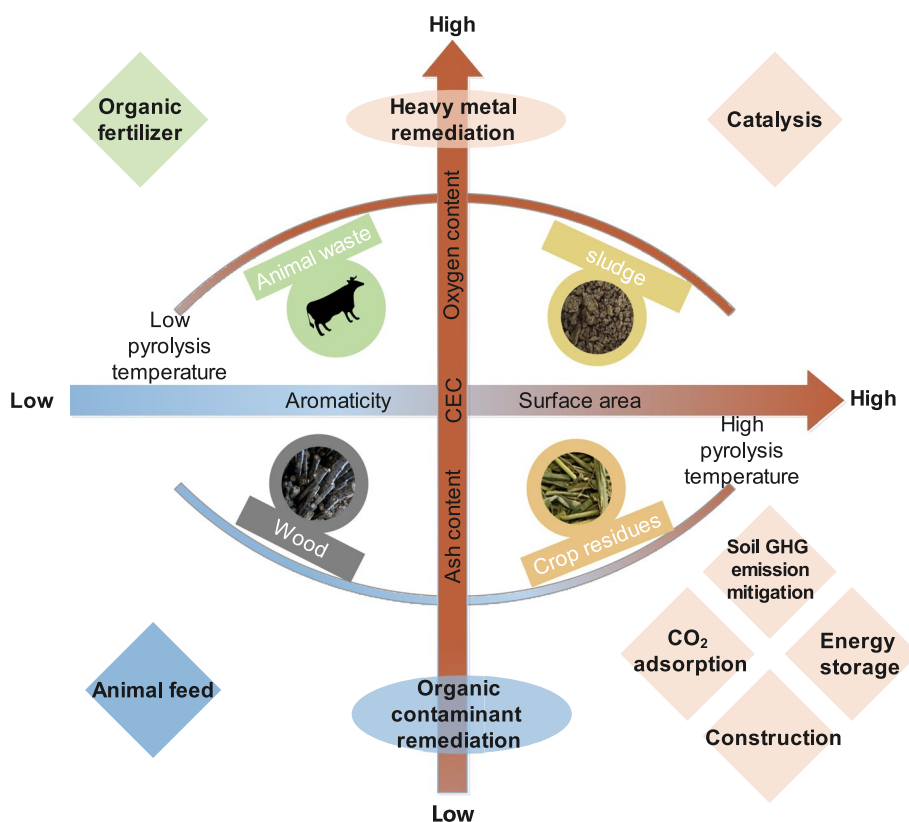


Fig. 8 Optimum biochar properties for different applications towards carbon neutrality

et al. 2019). Besides, engineered biochars may also be fabricated, whose engineering methods are summarized by our previous works (Wang et al. 2022e; Wang et al. 2020g). For instance, nano-biochar with extremely high sorption capacities can improve biochar performances in CCUS (Kumar et al. 2020b).

6 Challenges and future directions

In this study it was found that biochar application mitigates as high as 5% of the global annual GHG emissions. Application of biochar to restore degraded land exhibits the highest potential for GHG emission reduction, whereas other potential non-soil applications, including aquaculture, green catalysis for biorefinery, cement additive, energy storage, and CCUS also hold much promise towards carbon neutrality. Despite the huge potential of biochar toward carbon neutrality, several challenges remain prior to its large-scale application for climate change mitigation, where future studies should work on.

For soil applications of biochar to mitigate GHG emissions, one crucial factor that future work should focus on is the long-term effectiveness. Evidence is mounting that biochar ages following field application, leading to a diminished or reversed priming effect. Biochar

acidification, mineral dissolution, and dissolved organic carbon release in the long term also alter the microbial communities of the rhizosphere, leading to unpredictable GHG flux from soil to the atmosphere. The release and sorption of plant root exudates, and plant uptake of labile C and N also add much complexity to this system. Another challenge is that the biochar’s performances for GHG emission mitigation are highly case specific. Dynamic and tunable physicochemical property of biochar on the one hand provides much opportunity to make use of this material for different soil applications. Nevertheless, it adds much uncertainty to the effectiveness of biochar for climate change mitigation in specific cases (Fig. 8). For instance, misuse of low temperature biochar for soil GHG emission mitigation would result in positive priming instead of negative priming, thus leading to stimulated loss of intrinsic soil organic carbon stock. In addition, the unneglectable role of soil matrix on biochar performances should also be taken into account. Previous meta-analyses have well proven that soil pH, texture, taxonomy, and organic matter content greatly alter biochar’s performances in the soil environment. The effectiveness of biochar for GHG emission mitigation is also highly dependent on soil properties. Although existing reviews

and meta-analyses provide very useful information for the selection of a suitable biochar amendment, both lab and pilot-scale demonstrations are recommended prior to field implementation of biochar for GHG emission mitigation.

For non-soil applications of biochar towards carbon neutrality, more evidences from pilot scale demonstrations rather than lab experiments should be acquired. Biochar uses as animal feeding, energy storage, and CCUS are still in their infancy with only proof-of-concept studies at the lab scale. Underlying mechanisms are also poorly understood. For instance, there is still controversy over how biochar as an animal feed reduces methane emissions in ruminants. It is also still in doubt whether biochar can really improve the mechanical strength and carbon absorption characteristic simultaneously, because accelerated carbonation deteriorates the hydration products. To expand biochar's application towards carbon neutrality beyond traditional soil amendment, pilot- or full-scale demonstrations are required. Besides, engineered biochars, such as activated biochar, nano-biochar, and biochar composite, can be fabricated for specific purposes.

It's notable that using biochar as a "sustainable" amendment does not necessarily mean that the entire life cycle impact on ecosystem is always lower than that without biochar. Technical, economical, and social concerns should be carefully resolved (Hou 2021b; Jin et al. 2021; Wang et al. 2020e). Apart from aforementioned technical problems, several economical and social concerns should also be considered. The cost of the biochar system also affects the overall sustainability. Waste biomass should be obtained locally without a long traveling distance. The bio-oil and syn-gas reuse systems should be affordable for local communities and stakeholders. Besides, the farmer's motivation of using biochar against other soil amendments, and adding biochar to ruminant feed, should also be stimulated to assure biochar's successful fight against climate change.

Abbreviations

CCUS	Carbon capture utilization and storage
C-S-H	Calcium silicate hydrate
EDLC	Electrochemical double-layer capacitor
GSR	Green and Sustainable Remediation
GHG	Greenhouse gas
LID	Low-impact development
PAH	Polycyclic aromatic hydrocarbon
PC	Pseudocapacitor
PGPB	Plant growth promoting bacteria

Acknowledgements

Not applicable.

Authors' contributions

Liuwei Wang: conceptualization, investigation, data analysis, writing – original draft; Jiayu Deng: investigation, writing – review & editing; Xiaodong Yang: data analysis, investigation; Renjie Hou: writing – review & editing; Deyi Hou: conceptualization, supervision, funding acquisition, writing – review & editing. The author(s) read and approved the final manuscript.

Funding

This work was supported by the National Natural Science Foundation of China (Grant No. 42077118).

Availability of data and materials

All data generated or analyzed during this study are included in this manuscript.

Declarations

Competing interests

The authors have no competing interests to declare that are relevant to the content of this article.

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Received: 21 October 2022 Revised: 2 January 2023 Accepted: 2 January 2023

Published online: 16 January 2023

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