

Wood-based biochar: A sustainable remedy for soil and water contamination in the agricultural field

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Abstract

Wood-based biochar, produced through the pyrolysis of wood biomass in an oxygen-limited environment, is a highly porous carbon material with diverse environmental applications. Its unique physicochemical properties, including high surface area, porosity, and abundant functional groups, enable it to adsorb trace metals (e.g. Pb^{2+} , Cd^{2+}) and organic contaminants (e.g. PAHs, pesticides) effectively through mechanisms such as surface complexation, cation exchange, π - π interactions, and hydrogen bonding. These interactions make biochar a promising material for soil remediation, water purification, and carbon sequestration. In agriculture, wood-based biochar enhances soil quality by improving nutrient retention, water-holding capacity, and microbial activity, increasing crop productivity and reducing reliance on chemical fertilisers. Its ability to sequester carbon binding it into a stable form for centuries contributes to climate change mitigation by reducing greenhouse gas emissions. However, the efficacy of biochar depends on factors such as feedstock type, pyrolysis conditions, and soil properties, necessitating tailored production and application strategies. This article explores the production and properties of wood-based biochar, its interaction mechanisms with organic and inorganic contaminants, and its environmental applications in soil amendment, wastewater treatment, and carbon storage.

Keywords: biochar; wood-based biochar; pyrolysis; cation exchange; trace metals; organic contamination

Introduction

Biochar, produced from biomass via thermal conversion, aids climate change mitigation. Its carbon sequestration and emissions reduction potential holds significance. Human activities like fossil fuel use and deforestation elevate atmospheric CO_2 , the primary driver of climate change. So, global endeavours focus on capturing and storing CO_2 . Biochar offers promise benefits in carbon storage, soil enhancement, energy generation, and waste control. Beyond carbon sequestration, biochar effectively immobilises pollutants in tainted soils stemming from human actions such as waste disposal and mining. These contaminants jeopardise human well-being and ecosystems. Recent studies indicate the efficacy of biochar in remediating polluted soil and water, averting contaminant propagation through groundwater and streams (Lehmann and Joseph 2015, Liang et al. 2021).

In recent years, water treatment methods like chemical precipitation, ion exchange, membrane separation, coagulation/filtration, flotation, and electrochemical approaches have been developed (Song et al. 2018). However, drawbacks like high costs, incomplete contaminant removal, and energy requirements have led to interest in eco-friendly absorbents like biochar for water and soil pollution treatment. Notably, biochar recyclability sets it apart from non-recyclable absorbents like limestone (Iakovleva et al. 2015), zeolite (Shaheen et al. 2012), and coal fly ash (Hizal et al.

2013), which generate waste. Biochar derives from various waste sources like agricultural, municipal, and widely available wood waste, making it a prime option. This feedstock yields high-quality biochar, known for its greater surface area and lower ash content, possibly due to high lignin content in wood (Shaheen et al. 2013). Significant water pollutants, including copper (Cu), cadmium (Cd), mercury (Hg), arsenic (As), chromium (Cr), nickel (Ni), lead (Pb), selenium (Se), zinc (Zn), and vanadium (V), accumulate due to natural and human activities; for the record, the developing nations such as India and Bangladesh contributing significantly to global pollution. Current estimates point to over 249 million people impacted by water contamination and more than 3.4 billion facing water shortages by 2025 (Shaheen et al. 2019).

Enter wood-based biochar, a ray of hope for soil and water remediation. This review will take a journey to discover the transformative power of this sustainable biochar technology. This investigation paves the way for understanding its applications in critical domains such as soil enrichment and water purification, from its fundamental properties and production methods to its intricate interactions with contaminants. Finally, this review demonstrates the power of innovation in addressing pressing global issues, reminding us that even seemingly insignificant substances can hold the key to a brighter, more sustainable future.

Sustainable biochar technology

Sustainable biochar technology uses eco-friendly methods and materials to produce and apply biochar. This process involves using environmentally friendly biomass sources, such as agricultural and forest residues, and employing low-emission pyrolysis methods to reduce greenhouse gas emissions while reusing its by-products.

Historically, increases in greenhouse gases resulting from deforestation and industrial combustion, particularly the rise in atmospheric CO₂ levels since 1950, have exacerbated environmental issues and accelerated climate change. To address this, the Kyoto Protocol was ratified by 192 parties in 1997, aiming to mitigate these effects and spur global efforts in carbon capture and storage. Biochar technology has emerged as a promising method for carbon sequestration in soil within this framework. Additionally, research findings have inspired scientists and farmers to utilize charcoal for carbon storage and soil enhancement, leading to the coining of the term “biochar” in 2006 (Lehmann et al. 2006).

During photosynthesis, plants absorb atmospheric CO₂ and convert it into biomass, which eventually decomposes into soil organic matter. This process releases the stored carbon back into the atmosphere as CO₂ within a few years (Tiessen et al. 1994). However, biochar enables the long-term retention of carbon from plant biomass in soil, effectively reducing atmospheric carbon levels. This approach offers a wide range of benefits, including waste management, climate change mitigation, energy generation, and soil quality improvement (Beesley and Marmioli 2011, Beesley et al. 2011).

Despite these advantages, challenges remain. These include maintaining sustainable feedstock production amidst land use changes and competition for resources, ensuring consistent biochar quality, and

understanding its long-term impacts on soil, water, and ecosystems.

Biochar production

Pyrolysis is the process of heating biomass at a regulated temperature in an oxygen-free (or oxygen-limited) atmosphere with the addition of a protective gas (N₂ or CO₂) to produce biochar and by-products (syngas and bio-oil) (Figure 1; Amalina et al. 2022). To achieve the best possible results concerning economic viability, process efficiency, and environmental sustainability when producing biochar from agricultural waste feedstock, it is important to choose the appropriate pyrolysis technology carefully. This selection should be based on careful consideration of the specific characteristics of the feedstock in question. By choosing the right technology, it is possible to optimize the biochar production process and minimize any negative environmental impacts. Slow pyrolysis (SP), Flash pyrolysis (FLP), Fast pyrolysis (FP), Microwave pyrolysis (MWP), Hydro pyrolysis (HP), and Vacuum pyrolysis (VP) are the six forms of pyrolysis (Figure 2).

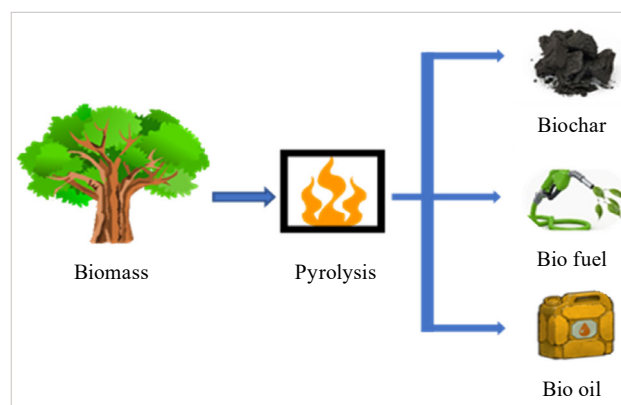


Figure 1. General concept of the pyrolysis process

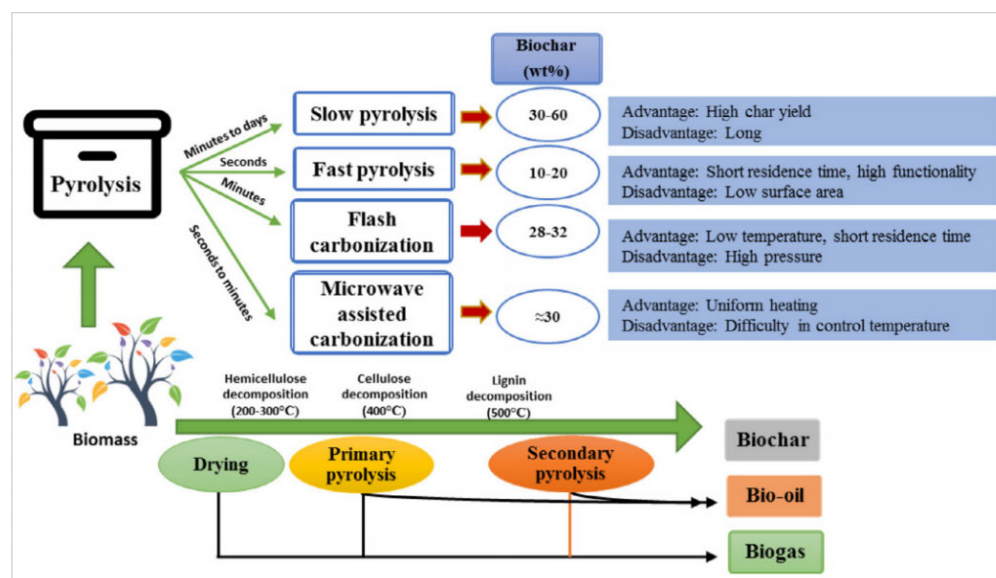


Figure 2. The differences of major biomass pyrolysis processes (Amalina et al. 2022)

Slow pyrolysis (SP)

Modern slow pyrolysis units are specifically designed to regulate the Highest Heating Temperature (HHT), Residence Time (RT), and Heating Rate (HR), which in turn determines the properties of biochar and captures the volatile components to produce bio-oil and syngas (Manyà 2012). The process of slow pyrolysis is conducted with a long RT of 5 to 120 minutes at a low HR of 0.1°C to 1°C per second and at 300°C to 600°C (Li and You 2022), which promotes biochar formation as the primary product. It reduces biomass secondary pyrolysis and thermal cracking (Tan et al. 2021). Slow pyrolysis is the common technique for producing biochar (Li et al. 2022). Compared to fast pyrolysis and flash pyrolysis, the HHT for slow pyrolysis is generally lower. It produces more charcoal and requires more time. Additionally, reducing HHT and increasing RT can improve biochar quality during the slow pyrolysis process. The heat and syngas generated during the process of slow pyrolysis can offset the energy consumed for biochar production (Manyà 2012).

Fast pyrolysis (FP)

Fast pyrolysis involves heating biomass at high temperatures (typically 500–1200°C) without oxygen supply to produce bio-oil or pyrolysis oil (Liu et al. 2020). The process occurs in a sealed reactor, and the biomass is rapidly heated at HR of 10–200°C/s with RT ranging from 0.5–10 s. The major disadvantage of fast pyrolysis is that heating feedstock can be converted to liquid products over biochar formation (Ghysels et al. 2019). The bio-oil produced through fast pyrolysis can be used as a feedstock for producing transportation fuels, such as diesel and gasoline. In addition, the high oxygen content of bio-oil makes it useful as a chemical feedstock for speciality chemicals.

Flash pyrolysis (FLP)

Flash pyrolysis involves heating feedstock at a higher temperature (typically greater than 1000°C) for a shorter RT (typically < 1 s) at HR (greater than 1000°C/s) to produce more bio-oil and less biochar (Li et al. 2013). Flash pyrolysis produces a liquid product similar to fast pyrolysis, but the bio-oil is usually of higher quality and contains less water and other impurities. The high temperatures in flash pyrolysis cause a complete breakdown of the biomass into its constituent parts, resulting in a bio-oil with a composition more like crude oil. Fast pyrolysis and flash pyrolysis are both detrimental to the production of biochar. However, fast pyrolysis and flash pyrolysis do not promote biochar formation.

Finally, fast pyrolysis and flash pyrolysis are important methods for converting biomass into valuable products like bio-oil. Because the solid matrix shrinks at higher temperatures, these methods produce a higher specific surface area than biochar produced through the slow pyro-

lysis method (De Mendonça et al. 2017). Fast pyrolysis is especially beneficial for generating stationary power and chemical feedstocks. On the other hand, flash pyrolysis is better suited for producing transportation fuels and higher-quality chemical feedstock. Both methods have advantages and disadvantages; the specific application and feedstock will determine the methods used.

Microwave pyrolysis (MWP)

Microwave pyrolysis is an innovative technology that utilises electromagnetic radiation in the microwave frequency range to convert biomass into value-added bio-products efficiently. This process involves heating wood chips in a microwave reactor, where non-intrusive electromagnetic radiation induces internal molecular vibrations, known as phononic oscillations (Ethaib et al. 2020). The rapid heating of wood chips occurs through conduction, convection, and radiation, releasing volatile compounds at temperatures between 400°C and 500°C and leaving behind a solid residue known as biochar. Key operating parameters, such as pyrolysis temperature, pressure, and heating rate (HR), typically range from 300°C to 700°C, 5 MPa to 10 MPa, and 0.2°C/s to 5°C/s, respectively (Li et al. 2022). Additionally, factors such as purge gas flow rate, microwave power, the amount and concentration of microwave absorbers, initial moisture content, and residence time (RT) significantly influence product yields (Morgan et al. 2017).

Numerous studies have explored how variations in these parameters affect the efficiency and outcomes of MWP-based processes. For instance, increasing the pyrolysis temperature reduces biochar yield but enhances its thermal stability. Specifically, biochar produced from wheat straw at 500°C exhibits high carbon stability, making it an excellent soil conditioner. In contrast, biochar generated at 300°C demonstrates high polarity, which enhances its ability to attract inorganic and polar organic pollutants, making it suitable for use as an adsorbent. These findings highlight how adjusting MWP parameters can tailor biochar properties for specific applications (Nzediegwu et al. 2021).

One of the primary advantages of MWP for wood biochar production is its ability to achieve higher yields compared to the traditional pyrolysis methods. Microwave heating is more energy-efficient and reaches the desired temperature faster, resulting in increased biochar output. Furthermore, MWP produces high-quality biochar characterised by a porous and homogeneous structure with a high surface area. This enhanced surface area improves the biochar capacity to adsorb and retain nutrients, making it an effective soil amendment for agricultural purposes.

However, challenges remain in the widespread adoption of microwave pyrolysis for wood biochar production. These include the high capital cost of microwave reactors and the difficulty of scaling up the process for larger production volumes. Additionally, there is a risk of over-

heating wood chips, which can lead to excessive charring and negatively impact biochar quality.

Vacuum pyrolysis (VP)

Vacuum pyrolysis is a process that utilises a closed reactor operating under sub-atmospheric pressure to thermally decompose feedstock in oxygen-free conditions. Typical operating parameters for VP include pressure, pyrolysis temperature, and heating rate (HR), which generally range from 0.01 MPa to 0.20 MPa, 300°C to 700°C, and 0.1°C/s to 1°C/s, respectively. Vacuum pyrolysis is an effective method for producing biochar, as it minimises secondary degradation and generates high bio-oil yields. This is achieved by preferentially removing volatile matter at elevated temperatures, which enhances heating efficiency and promotes biomass decomposition. However, similar to other pyrolysis methods, increasing the pyrolysis temperature in vacuum pyrolysis can reduce biochar yield (Yousaf et al. 2021).

Vacuum pyrolysis offers several advantages over traditional pyrolysis methods. By preventing biomass oxidation, it produces biochar with high carbon content. As the temperature rises, the biomass undergoes a series of chemical reactions, producing char and releasing gases and liquids. These by-products can be collected and utilised for energy generation or chemical production. The resulting solid char is cooled, collected, and processed into biochar, which typically contains 70–95% carbon and low ash content, making it a valuable soil amendment for agriculture (Lam et al. 2020).

Despite its advantages, vacuum pyrolysis faces several challenges in wood biochar production: Scale-up: While vacuum pyrolysis has been successfully demonstrated on a laboratory scale, it has yet to be fully scaled up for commercial production. Impurities: Feedstocks such as municipal solid waste may contain impurities that compromise biochar quality. Energy consumption: The high temperatures required for vacuum pyrolysis result in significant energy demands. Cost: The equipment and operational costs associated with vacuum pyrolysis are relatively high. Future research on vacuum pyrolysis for wood biochar production should focus on addressing these issues to improve its feasibility and scalability.

Hydropyrolysis (HP)

Hydropyrolysis is a process that uses a high-pressure hydrogen atmospheric condition within the reactor. The process parameters usually fall within specific ranges, including a pressure of 10–17 MPa, pyrolysis temperature ranging from 350°C to 600°C, a heating rate ranging from 10°C/s to 300°C/s, and residence time greater than 60 s (Oh et al. 2021). Studies show that using high-pressure hydrogen-based gaseous conditions in this process can improve aromatic hydrocarbon and gas yields by 57% and 19%, respectively, compared to processes under inert atmospheric conditions. High hydrogen pressure can also

increase biochar yield and reduce light aromatic, and tar yields via secondary reactions. Co-loading of gallium (Ga) and zinc (Zn) in hydropyrolysis will help to increase further the aromatic hydrocarbon yield (Wang and Song 2018). The bio-oil generated by hydropyrolysis units cannot be employed as a fuel for transportation without further treatment due to its high content of oxygenated compounds, such as acids and aldehydes. As a result, hydrotreating is necessary to improve its quality, which adds complexity and expenses to the process. Consequently, hydropyrolysis is considered a less preferred technology for standalone applications (Kong et al. 2020).

Biochar properties and interactions with contaminants

Biochar properties

The effectiveness of biochar in soil remediation is influenced by several properties, including surface area (SA), particle size (PS), cation exchange capacity (CEC), pH, aromaticity, elemental composition (AEC), volatile matter content (VMC), and ash content (AC). Among these, surface area, CEC, and pH are the most critical.

Surface area, determined by biochar pore structure, directly impacts its ability to adsorb trace metals and organic contaminants (OCs), with more porous structures offering higher adsorption capacity. pH plays a significant role in trace metal adsorption, as higher pH levels enhance this process through various mechanisms. CEC, which reflects biochar ability to exchange cations with the environment, aids in retaining nutrients such as nitrogen, phosphorus, and potassium (Tag et al. 2016). This property is closely tied to the presence of exchangeable cations like Ca^{2+} , Na^{+} , K^{+} , and H^{+} .

Wood-derived biochar exhibits distinct physical, chemical, and mechanical properties. Physically, it is highly porous, providing a large surface area-to-volume ratio that improves soil water and nutrient retention. Its low density (0.1–0.4 g/cm³) facilitates easy handling and transport, while particle size varies from < 1 mm to several centimetres, depending on feedstock and production methods. Fine-particle biochar adsorbs trace metals more effectively than coarse-particle biochar (Mukherjee et al. 2011). Chemically, wood-based biochar is predominantly carbonaceous (50–90% carbon), making it valuable for carbon sequestration. Its pH (7–10) helps neutralise acidic soils, though variations occur based on feedstock and production conditions. Mechanically, it is brittle and has modest strength, but its durability can be improved by blending with binders.

Additionally, its high stability makes it ideal for long-term soil enhancement and carbon sequestration, reinforcing its role in sustainable agriculture (Domingues et al. 2011). Table 1 summarizes the physical-mechanical properties of selected wood-based biochars produced using different methods.

Table 1. Physical and chemical properties of biochar synthesised from diverse feedstocks and temperatures

Methods	Feedstock	Residence time (s)	Temperature (°C)	pH	Surface area (cm ² /g)	Volume (cm ³ /g)	References
SP	Pinewood	1800	500	8.7	380	0.15	Leng et al. 2021
FP	Pine saw-dust	3	400	4.2	6.2	0.011	Lee et al. 2020
	Saw-dust	1	400	6.35	83.5	0.012	Lee et al. 2020
	Pinewood	2	425	-	1.35	-	Gurav et al. 2021
	Pinewood	2	500	-	175	-	Gurav et al. 2021
	Saw-dust	3	500	6.42	36.60	0.015	Lee et al. 2020
	Pinewood	900	650	7.85	52.1	-	Nidheesh et al. 2021
MWP	Willow wood-chips	-	170	0	3.87	2.07	Ge et al. 2021

According to Jahiril et al. (2012), the composition of each feedstock varies depending on its cellulose, hemicellulose, and lignin content. For example, wood has higher lignin content than straw, while leaves may have higher hemicellulose content than straw or wood. The production of high-quality biochar with environmental benefits depends on the thermal decomposition of cellulose, hemicellulose, and lignin. Among these components, hemicellulose decomposes the most readily, followed by cellulose, whereas lignin is highly resistant to thermal decomposition, even at temperatures as high as 500°C. As a result, biochar often has an elemental composition similar to lignin, while bio-oil is derived from the decomposition products of cellulose, hemicellulose, and lignin (Axelsson et al. 2012).

Heating biomass causes the breakdown of carboxylate groups on the edges of aromatic rings, facilitating condensation reactions that allow small aromatic units to combine and form larger aromatic structures. This process increases the pH and aromaticity of biochar due to the transformation of alkali and alkaline earth metals into their oxidised forms, such as K₂O. Generally, increasing the heating temperature leads to higher pH and aromaticity. Increased aromaticity enhances trace metal adsorption by strengthening cation interactions, while a higher volatile matter content (VMC) indicates more acidic surface functional groups, improving adsorption via cation exchange and complexation (Mukherjee et al. 2011).

However, the impact of heating temperature on CEC is complex. At temperatures below 400°C, HHT can create acidic functional groups, increasing the concentration of H⁺ ions in biochar. Above 400°C, higher HHT leads to the decomposition of acidic functional groups and the formation of mineral phases, resulting in a complex interplay of factors that influence CEC. For instance, HHT at high temperatures can create minerals (e.g. Ca²⁺, K⁺, and Na⁺), which increase the concentration of exchangeable cations. Thus, the effect of HHT on CEC is a balance between the decomposition of acidic groups and the formation of mineral phases. Additionally, higher HHT often leads to a more porous structure and increased surface area, although the extent of these changes depends on the biomass type and pyrolysis conditions (Suliman et al. 2016).

Lehmann (2007) investigated the effects of varying heating treatment temperatures (HHT – Highest Heating Temperatures) on the properties of wood-based biochar. As the HHT increased within the range of 100–800°C, the pH of the biochar also increased. The cation exchange capacity (CEC) reached its maximum at 500°C within the 300–600°C range, followed by a secondary increase in the 600–800°C range. This dual trend in CEC is likely due to the contrasting effects of acidic functional group decomposition and the formation of mineral phases at higher temperatures. The surface area of the biochar peaked at 500°C, with a slight reduction observed as the temperature increased from 500°C to 800°C. This reduction may be attributed to the destruction of macropores and the simultaneous formation of micropores. The surface area of biochar is typically determined using the Brunauer-Emmett-Teller (BET) N₂ adsorption method (Yaashikaa et al. 2020). Generally, there is a positive correlation between the BET surface area and micropore volume (Lehmann and Joseph 2015).

Interactions between biochar and contaminants

Biochar has a high surface area, porosity, and functional groups that enable it to adsorb contaminants physically and chemically, as well as promote the biodegradation of contaminants by providing a habitat for microorganisms. However, the interactions between biochar and microorganisms are still not fully understood (Mukherjee et al. 2022). The effectiveness of biochar in removing contaminants varies depending on the type of contaminant. The effectiveness of sustainable biochar technology in soil remediation is significantly influenced by the interactions between contaminants and biochar. These interactions, particularly sorption, depend on factors such as surface chemistry, surface area, porosity, functional groups, and the physicochemical properties of both the contaminants and biochar (Schreiter et al. 2018). However, the long-term sustainability of biochar technology also depends on its stability in soil, its impact on soil health, and the environmental and economic feasibility of its production and application.

Biochar and trace metal interaction mechanisms

Physical adsorption

The process of physical adsorption involves the attraction of trace metal ions to the surface of biochar through intermolecular forces. Two primary forces govern trace metal adsorption onto biochar surfaces: Van der Waals forces and electrostatic forces. Van der Waals forces are weak, non-covalent interactions arising from temporary fluctuations in electron density. These forces are generally attractive in the context of trace metal adsorption, helping to hold metal ions near the biochar surface. In contrast, electrostatic forces are stronger and result from the attraction or repulsion between charged particles. In biochar systems, electrostatic forces drive the interaction between charged functional groups on the biochar surface and trace metal ions (Wang et al. 2022b).

The adsorption of trace metals onto biochar typically involves a combination of Van der Waals and electrostatic forces. Positively charged trace metal ions (e.g. Pb^{2+} , Cd^{2+} , Zn^{2+}) are attracted to negatively charged functional groups on the biochar surface, while negatively charged ions (e.g. CrO_4^{2-} , AsO_4^{3-}) are repelled (Singh et al. 2023). This charge-based selectivity can be exploited to target specific trace metal contaminants in soil and water remediation (Dixit et al. 2015).

The extent of physical adsorption depends on several factors, including the surface area, porosity, and functional groups of the biochar, as well as the physicochemical properties of the trace metal ions (Wang et al. 2022b). Biochar with a high surface area and porosity provides more sites for trace metal capture (Inyang et al. 2016). Additionally, functional groups on the biochar surface, such as carboxyl and hydroxyl groups, play a critical role in adsorbing trace metals. The physicochemical properties of the trace metal ions, such as ionic charge, ionic radius, and solubility, also influence adsorption. Ions with higher charges and smaller radii exhibit greater affinity for biochar surfaces due to stronger electrostatic interactions (Wang et al. 2022b).

However, physical adsorption is a weak and reversible process. This limits its effectiveness in reducing the mobility and bioavailability of trace metals in soil and water, as adsorbed ions can desorb and become available to living organisms. Despite this limitation, physical adsorption is a viable approach for wastewater treatment, where biochar can be reused after trace metal removal. For example, animal bone biochar has been shown to effectively adsorb trace metals such as Zn^{2+} , Cu^{2+} , and Cd^{2+} through physical adsorption (Choy and McKay 2005).

Cation exchange

Biochar can immobilize trace metals through cation exchange, where its exchangeable cations (e.g. Ca^{2+} , Na^+ , H^+ , and K^+) are replaced by trace metal ions. These exchangeable cations originate from two primary sources in biochar: alkaline minerals (e.g. carbonates and oxides) and acidic functional groups (e.g. carboxyl and hydroxyl groups) formed during pyrolysis. Trace metals bound to biochar through cation exchange are generally considered bioavailable and can be readily taken up by plants (Filgueiras et al. 2002).

While cation exchange is effective for immobilizing trace metals in contaminated soils, it may not provide a long-term solution. Changes in soil pH or environmental conditions can trigger the release of adsorbed trace metals, posing environmental risks. For example, a decrease in pH can protonate functional groups on the biochar surface, reducing its cation exchange capacity and releasing previously bound trace metals. Despite this limitation, cation exchange is a promising mechanism for water treatment applications. The adsorbed trace metals can be removed from biochar by adjusting the pH or through competitive exchange with other cations, allowing the biochar to be regenerated and reused. This reusability makes cation exchange a practical approach for treating trace metal-contaminated water, similar to physical adsorption mechanisms.

Cation- π interactions

Cation- π interactions are a type of electrostatic interaction between positively charged ions (cations) and the π -electron cloud of aromatic structures in biochar. These interactions can also occur through metal coordination with oxygen-containing functional groups, such as phenolic and carboxylic acids (Keiluweit et al. 2010). Biochar, rich in aromatic compounds and functional groups, facilitates cation- π interactions with trace metals. In this process, cations are attracted to the π -electron cloud of aromatic rings, forming a bond that does not require a functional group capable of donating an electron pair. This distinguishes cation- π interactions from ion exchange mechanisms.

The strength of cation- π interactions is influenced by soil pH and cation concentration (Wang et al. 2022a). Higher pH values enhance the binding capacity of biochar by deprotonating acidic functional groups, increasing the availability of π -electron clouds for interaction. Conversely, high cation concentrations in the soil solution can compete with trace metals for binding sites, reducing the effectiveness of cation- π interactions.

Cation- π interactions contribute to the immobilisation of trace metals in water and soil, with adsorption strength intermediate between physical adsorption and chemisorption (Bian et al. 2014). These interactions also enhance the selectivity of biochar for specific trace metals. For example, copper ions exhibit stronger interactions with phenolic

groups in biochar compared to zinc ions, leading to preferential adsorption of copper over zinc (Xiang et al. 2020). However, cation- π interactions have limitations. They can be reversible under certain soil conditions, potentially releasing adsorbed trace metals back into the environment. Additionally, the availability of binding sites for cation- π interactions may be limited, particularly if biochar is saturated with other cations (Mukherjee and Lal 2013). Therefore, the efficiency of cation- π interactions in immobilising trace metals depends on specific soil conditions, metal properties, and biochar characteristics.

Surface precipitation

Surface precipitation is a mechanism by which biochar adsorbs trace metals by forming insoluble precipitates on its surface. Surface precipitation can occur through various means, including coprecipitation, adsorption-induced precipitation, and ion exchange-induced precipitation (Gupta 2023). The pH of biochar is often high due to alkaline minerals formation, creating an alkaline environment on the biochar surface that aids the trace metals precipitation. Compounds such as CO_3^{2-} and PO_4^{3-} from biochar can also help (Yang et al. 2014).

Metals with intermediate ionization potentials, such as Pb^{2+} , Cu^{2+} , Zn^{2+} , and Ni^{2+} , have a higher propensity to undergo precipitation on the surface of biochar in comparison to other elements (Gupta 2023). The precipitation occurs when these metals react with the biochar and form solid particles that are insoluble in the solution. This phenomenon can be attributed to the intermediate ionization potentials of these metals, which make them more prone to undergo hydrolysis and form stable complexes with functional groups present on the surface of biochar. Pb^{2+} is a metal that has been extensively studied in the context of biochar precipitation, with several studies reporting on its effective removal from contaminated environments using biochar. For instance, Yang et al. (2014) demonstrated that Pb^{2+} can be efficiently precipitated as cerussite and hydro cerussite by *Alternanthera philoxeroides* biochar. Similarly, Cao et al. (2011) found that dairy manure biochar has the potential to precipitate Pb^{2+} as $\text{b-Pb}_9(\text{PO}_4)_6$ in aqueous solutions and hydroxy pyromorphite $\text{Pb}_5(\text{PO}_4)_3(\text{OH})$ in soil.

Surface precipitation can contribute to trace metal immobilisation in water and soil but may also have limitations. The effectiveness of surface precipitation depends on the specific soil and metal conditions. It can also be reversible, which may lead to trace metals leaching back into the water or soil, resulting in environmental contamination. Therefore, further study is required to comprehend better knowledge of surface precipitation mechanisms and limitations in biochar.

Surface complexation

The surface complexation mechanism is a chemical process in which trace metals are adsorbed onto biochar

surfaces through the formation of chemical bonds, resulting in metal-biochar surface complexes (Bian et al. 2014). This process is influenced by the properties of biochar, trace metals, and environmental conditions. Biochar surface properties, such as surface area, surface charge, and functional groups, play a critical role in surface complexation. Biochar with a larger surface area and more functional groups (e.g. carboxyl, hydroxyl, and amine groups) exhibits a greater capacity for metal adsorption (Qiu et al. 2021). The properties of trace metals, including their type, concentration, and chemical speciation, also affect the strength and stability of the metal-biochar complexes (Ahmed et al. 2017). Additionally, environmental conditions such as pH, temperature, and ionic strength influence the formation and stability of these complexes (Kumar et al. 2021).

Surface complexation can result in two types of complexes: inner-sphere and outer-sphere complexes. Outer-sphere complexes involve the adsorption of metal ions onto the biochar surface through electrostatic interactions, without forming covalent bonds. These complexes are relatively weak and depend on the attraction between metal ions and charged functional groups on the biochar surface (Zhang et al. 2021). In contrast, inner-sphere complexes involve the formation of covalent bonds between metal ions and functional groups (e.g. $-\text{C}=\text{O}$, $-\text{OH}$, or $-\text{RNH}_2$) on the biochar surface. These complexes are stronger and more stable, as the metal ions are directly coordinated to the functional groups of biochar. The formation of inner-sphere complexes depends on the biochar surface characteristics, the chemical properties of the metal ions, and the pH of the solution (Yu et al. 2021).

Trace metals on the surface of biochar can form ligand complexes, particularly with oxygen-containing functional groups such as carboxylic ($-\text{COOH}$), phenolic ($-\text{C}_6\text{H}_5\text{OH}$), and lactonic ones. These functional groups exhibit a strong affinity for trace metals, facilitating complexation (Gupta 2023). Biochar produced at lower temperatures typically retains a higher concentration of oxygen-containing functional groups, making it more effective at adsorbing trace metals through surface complexation.

When applied to contaminated soil or water, biochar undergoes slow oxidation, which generates additional carboxyl groups and enhances trace metal complexation over time. However, soil microbes can gradually decompose the organic fraction of biochar, reducing its capacity to immobilize trace metals through complexation. Despite this, oxygen-containing functional groups on the aromatic fraction of biochar are expected to have the most significant long-term impact on trace metal immobilisation. Trace metals adsorbed onto biochar through complexation are often considered non-bioavailable, posing fewer environmental risks compared to metals immobilised through weaker mechanisms like physical adsorption.

The surface complexation mechanism of trace metal and biochar interaction is influenced by surface charge,

which determines the attraction or repulsion between metal ions and biochar. The surface charge of biochar depends on its functional groups and the pH of the solution, while the surface charge of metal ions is also pH dependent. Metal ions can exist in different chemical forms depending on the pH of solution. For example, in acidic conditions, metal ions are typically positively charged, whereas in alkaline conditions, they may become negatively charged. The interaction between metal ions and biochar is governed by their respective charges: ions with the same charge as the biochar surface will repel each other, while oppositely charged ions will be attracted to the biochar surface (Ahmed et al. 2017).

Another critical factor influencing surface complexation is the presence of competing ions in the solution. Competing ions can occupy binding sites on the biochar surface, reducing the adsorption of trace metals. High concentrations of competing ions can significantly decrease the adsorption capacity of biochar, while their absence can enhance it (Ahmed et al. 2017). The interaction between biochar and trace metals through surface complexation has been extensively studied in environmental and agricultural contexts. This mechanism is being explored as a promising strategy for remediating trace metal-contaminated water and soil. Understanding the factors that influence surface complexation, such as pH, surface charge, and competing ions, is essential for optimising biochar effectiveness in trace metal immobilisation.

Biochar and the mechanisms of adsorbing organic contaminants (OCs)

Biochar is highly effective in adsorbing organic contaminants (OCs) due to its unique surface chemical properties and pore structure. Compared to non-pyrolytic materials or soils, biochar exhibits significantly higher bond strength and adsorption capacity for OCs (Zhang et al. 2013). The adsorption of OCs onto biochar primarily depends on surface area and pore size distribution; although cation exchange capacity (CEC) also plays a role in the adsorption of trace metals (Shen et al. 2015). Other properties influencing sorption include pH, ash content and composition, and polarity and functionality. Biochar can exhibit sorption capacities one to three orders of magnitude higher than those of the original feedstock material or conventional soils (Zhang et al. 2013).

The sorption of OCs on biochar is controlled by multiple simultaneous processes, as summarised in Table 2. Partitioning refers to the distribution of contaminant molecules between the solid (biochar) and aqueous phases, contributing to overall sorption. Pore filling involves transporting adsorbate molecules to sorption sites within biochar pores. Since partitioning is distinct from adsorption, the term sorption encompasses both processes. The three primary adsorption mechanisms are: π - π electron donor-acceptor interactions, hydrophobic effects, and hydrogen bonding. This section explores the relative contributions of

Table 2. The interaction mechanisms of biochar with trace metals and organic contaminants (OCs) (Mukherjee et al. 2022)

No	Biochar and trace metal interaction mechanisms	Biochar and organic contaminant interaction mechanisms
1	Physical adsorption	Partitioning
2	Cation exchange	Pore filling
3	Cation- π interactions	π - π interactions
4	Surface precipitation	Hydrophobic effects
5	Surface complexation	Hydrogen bonding

these mechanisms to OC sorption on biochar and evaluates their significance in remediation by analysing their response to biochar ageing in the soil profile.

Partitioning

The mechanism of partitioning in biochar sorption refers to the dissolving of organic contaminants in the amorphous organic phase of biochar. The proportion of the amorphous phase present in the biochar influences this process. Higher ratios of the amorphous phase led to a higher dominance of partitioning in sorption, particularly in non-carbonised materials and low-temperature biochars (Chiou et al. 2015). Partitioning, in contrast to adsorption, is typically linear regarding water concentration. Partitioning can be characterised using a partitioning coefficient (KP), which defines the relationship between the amount of an adsorbate partitioned to a solid phase and its aqueous concentration. The nonlinearity of isotherms increases as the HHT increases, associated with eliminating the amorphous phase and a rising prevalence of adsorption over partitioning at higher pyrolysis temperatures. Studies on various biochar feedstocks and contaminants have shown this relationship (Zhang et al. 2013).

The partitioning coefficient, KP, is used to determine the degree of partitioning in the sorption of contaminants to the amorphous organic phase of biochar. The linear relationship between the amount of adsorbate in the solid phase and its concentration in the aqueous phase is described by KP (Jung et al. 2013). The dual-mode sorption model can isolate the partitioning component from isotherms and determine distinct partitioning and adsorption isotherms (Huang and Chen 2010). The isotherm slope at high concentrations represents KP because the dual-mode sorption model predicts a slight increase in adsorption at high concentrations.

To evaluate the effectiveness of biochar in soil remediation, it is necessary to differentiate the partitioning mechanism from adsorption. The partitioning mechanism, in which organic contaminants dissolve in the amorphous organic phase of biochar, is not likely to significantly impact biochar-based soil remediation. Therefore, it is important to use a tool like the dual-mode sorption model to isolate partitioning from the isotherm and accurately assess the ability of biochar for soil remediation. Although partitioning can contribute to overall sorption, it may not provide significant environmental benefits (Zhang et al. 2011).

Pore filling

Pore filling is a key mechanism for the sorption of organic contaminants (OCs) on biochar, particularly due to its porous structure (Lattao et al. 2014). Sorption profiles for biochar typically exhibit nonlinear behaviour, indicating that pore filling plays a more significant role at low contaminant concentrations (Zhang et al. 2013). This occurs because contaminant molecules interact strongly with micropores that are comparable in size to the molecules, leading to adsorption on pore walls (Nguyen et al. 2007). At low concentrations, contaminant molecules are preferentially adsorbed into high-energy micropores. However, as concentration increases, these micropores become saturated, reducing further adsorption. This results in high sorption at low concentrations but diminished sorption at higher concentrations, contributing to the observed nonlinear behaviour (Zhang et al. 2013).

Pore filling significantly contributes to OC sorption, but its effectiveness depends on the pore network configuration. Micropores, which are similar in size to contaminant molecules, provide the majority of sorption sites and contribute most to total sorption. For example, Lattao et al. (2014) found that the sorption of 1,4-dinitrobenzene, benzene, and naphthalene onto wood chars was best predicted by a weighted combination of micropore and mesopore volumes. Similarly, Ahmad et al. (2013) observed that trichloroethylene is initially held in the mesopores of pine needle biochar before being drawn into micropores via capillary forces.

However, pore filling may be limited for larger contaminant molecules due to the size-exclusion effect. Bulky molecules can experience steric hindrance, preventing access to smaller micropores. Studies have shown that steric effects play a critical role in regulating the sorption of trace metals and polycyclic aromatic hydrocarbons (PAHs) onto biochar. Although pore filling is associated with low reversibility and high binding strength, its remediation potential may be constrained by several factors: a significant decline in adsorption of molecules larger than 1 nm, slow diffusion of contaminants into micropores, dependence on biochar with both micropores and mesopores, and potential obstruction of pores by soil organic matter. While biochars with high microporosity, produced at elevated temperatures, are effective at capturing contaminants via pore filling, they may lead to short-term leaching due to the slow diffusion of contaminants into micropores.

π - π interactions

One mechanism by which biochar adsorbs organic contaminants (OCs) is through π - π interactions. These interactions occur between the delocalized π -electrons in the aromatic rings of the organic contaminants and the carbon rings in the biochar structure (Zhang et al. 2013). As Lattao et al. (2014) explain, " π - π interactions refer to the chemical

interactions between the aromatic surface of the carbonised component of biochar and the π -electrons of aromatic adsorbate compounds. Biochar can act as both a π -electron donor and a π -electron acceptor, while aromatic OCs typically function as either donors or acceptors". During its production, biochar undergoes thermal decomposition, forming a complex, porous structure with a negatively charged and hydrophobic surface. This makes it highly attractive to hydrophobic organic compounds. The delocalized π -electrons in the aromatic rings of OCs interact with the carbon rings of biochar, leading to the adsorption of contaminants onto the biochar surface. The effectiveness of π - π interactions in adsorbing OCs depends on the aromaticity of contaminants and electron properties, biochar composition, porosity, and surface area, and environmental factors like pH, ionic strength, and temperature. These factors collectively influence adsorption capacity, highlighting the need to optimise biochar for environmental remediation.

Hydrophobic effects

Biochar can adsorb organic contaminants (OCs) through hydrophobic interactions. During biochar production, the material surface develops hydrophobic characteristics due to the presence of aliphatic and aromatic functional groups. Since many OCs are also nonpolar and hydrophobic, they can interact with the hydrophobic surface of biochar through van der Waals forces and other non-polar interactions, leading to the adsorption of contaminants onto the biochar surface. The effectiveness of biochar in soil remediation depends on its contribution to sorption and the long-term effects of biochar oxidation. The efficiency of this adsorption mechanism is influenced by several factors, including the surface area, porosity, and chemical composition of the biochar, as well as the type and concentration of the organic contaminant and environmental conditions such as temperature, pH, and ionic strength.

The hydrophobicity of biochar plays a crucial role in OC adsorption. High temperature biochar types, characterised by low polarity and high aromaticity, are particularly hydrophobic. For example, Liao et al. (2012) found that the hydrophobicity of bamboo charcoal was strongly correlated with its ability to adsorb dyes, highlighting the importance of hydrophobic interactions in the adsorption of non-polar compounds. These findings suggest that hydrophobic effects are especially significant in biochar types with high aromaticity and low polarity, which are typically produced at higher pyrolysis temperatures.

Hydrogen bonding

Hydrogen bonding is a crucial mechanism in the adsorption of organic contaminants (OCs) by biochar (Ahmad et al. 2012). Biochar contains functional groups such as hydroxyl ($-\text{OH}$), carbonyl ($> \text{C} = \text{O}$), and carboxyl ($-\text{COOH}$), which can form hydrogen bonds with OCs. These functional groups are typically located on the surfa-

ce of biochar particles, where they facilitate the adsorption of organic molecules.

Hydrogen bonding is a specific but relatively weak intermolecular force that arises from the attraction between a hydrogen atom bonded to an electronegative atom (e.g. oxygen or nitrogen) and another electronegative atom in a nearby molecule. In the context of biochar, hydrogen bonding occurs between the functional groups on its surface and OCs, including compounds such as phenols, polycyclic aromatic hydrocarbons (PAHs), and pesticides.

Studies have demonstrated that hydrogen bonding significantly enhances the adsorption of OCs. For example, the adsorption of PAHs to biochar is primarily driven by hydrogen bonding interactions, with hydroxyl groups on the biochar surface playing a key role (Devi and Saroha 2015). Similarly, the adsorption of phenols is influenced by hydrogen bonding, where $-OH$ and $-COOH$ groups on the biochar surface contribute to the process (Yaashikaa et al. 2020).

Ren et al. (2016) describe hydrogen bonding as an interaction where an atom of fluorine (F), nitrogen (N), or oxygen (O) in one molecule forms a bond with a hydrogen atom (H) that is bonded to another F, N, or O atom in a different molecule. In biochar, functional groups such as carboxyl ($-COOH$) and phenol ($-OH$) can participate in hydrogen bonding with contaminant molecules, leading to strong adsorption. The effectiveness of hydrogen bonding in soil remediation depends on its contribution to the overall sorption process, its interaction with water molecules, and the impact of biochar oxidation in the soil. For polar organic compounds, hydrogen bonding plays a significant role in regulating their sorption onto biochar (Ahmed et al. 2017). Studies have shown that hydrogen bonding can tightly bind contaminants, with binding strength exceeding that of hydrophobic interactions, particularly for polar compounds (Lima et al. 2010). Therefore, hydrogen bonding should be a key consideration when designing biochar for environmental remediation applications.

Environmental application of wood-based biochar

Due to its superior properties, biochar is increasingly being used as a sustainable alternative to activating carbon for removing contaminants from water. Beyond water treatment, biochar has diverse applications, including its use as a catalyst in wastewater treatment, a soil amendment, a composting agent, a material for energy storage, and a tool for carbon sequestration (Figure 3).

In soil remediation, biochar is commonly used as an *in situ* amendment. *In situ* remediation involves treating contaminants directly in the soil without excavation or transportation to a treatment facility. Biochar can be applied directly to contaminated soil, where it enhances remediation through mechanisms such as adsorption, chemical reactions, and stimulation of microbial activity. Additionally, biochar improves soil properties, including nutrient

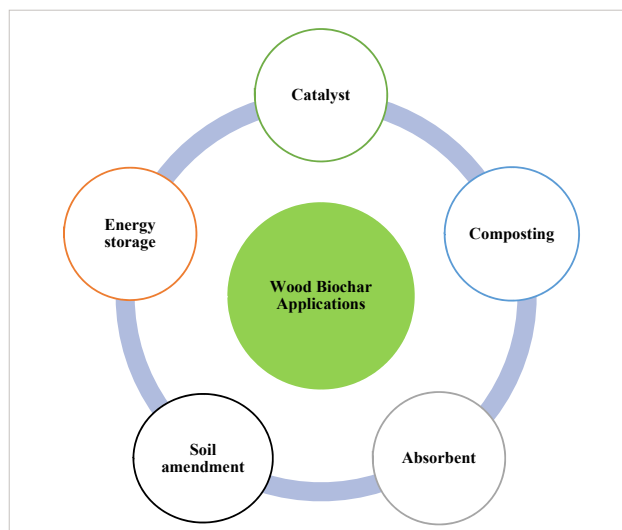


Figure 3. Applications of wood-based biochar

retention, water-holding capacity, and microbial activity, which can boost soil productivity and plant growth. Due to its stability and long-lasting nature, biochar provides long-term benefits to soil health and function. As a result, using biochar as an *in situ* amendment offers a sustainable and effective approach to remediating contaminated soil while improving overall soil quality.

Biochar also shows significant potential in phytoremediation, a process that uses plants to remove, degrade, or contain (pollutants) xenobiotic compounds in contaminated soils. As a soil amendment, biochar positively influences plant growth and enhances the uptake of contaminants such as polycyclic aromatic hydrocarbons (PAHs) and trace metals. Research indicates that biochar-amended soil can reduce the bioavailability of contaminants, making them less toxic to plants. Furthermore, biochar promotes the growth of contaminant-degrading microorganisms, enhancing the degradation of pollutants in the soil. Its high surface area and porosity also support the colonisation by beneficial microorganisms, further improving the phytoremediation process. For example, Li et al. (2019) found that biochar addition increased the growth of and metal uptake by *Solanum nigrum*, a plant species known for accumulating trace metals. Similarly, Zhang et al. (2020) demonstrated that biochar enhanced the growth of and PAH uptake by *Festuca arundinacea* while promoting the degradation of phenanthrene, a common PAH.

Biochar has emerged as a promising and sustainable alternative to activated carbon for wastewater treatment due to its high porosity, large surface area, and cost-effectiveness (Enaime et al. 2020). Its exceptional ability to adsorb nutrients and pollutants from wastewater makes it not only an effective treatment material but also a valuable soil amendment when repurposed after use (Shokry et al. 2020). The continuous adsorption capacity of biochar for pollutants, combined with its surface functional groups, enhances its efficiency in wastewater treatment;

Table 3. Adsorption of organic pollutants using wood biochar

Absorbate	Absorbent	Biochar dosage	Removing efficiency (%)	References
Polychlorinated	Wood	1%	32	Qiu et al. 2022
	Mixed wood	1-4%	63	Silvani et al. 2019
	Mixed wood shavings	1-10%	37	Bielská et al. 2018
	Bamboo	2%	79	Huang et al. 2018a
	Bamboo	2%	66	Huang et al. 2018b
	Pinewood	5%	99	Gomez-Eyles and Ghosh 2018
	wood waste	3–11%	-	Denyes et al. 2012
Thiamethoxam	wood	1.5	23	You et al. 2021
Carbofuran	Wood	1%	51	Qiu et al. 2022
Phenanthrene	Wood	5%	50	Qiu et al. 2022
Salicylic acid	Pinewood	-	40	Essandoh et al. 2015
Ibuprofen	Pinewood	-	41.7	Essandoh et al. 2015
Sulfamethoxazole	Wood	40 mg/L	100	Zhu et al. 2018
Organic micropollutant	Beechwood	10–20 mg/L	-	Hagemann et al. 2020

often surpassing that of activated carbon despite its larger surface area and reliance on pore-filling mechanisms of the latter (Gwenzi et al. 2017). Additionally, biochar production generates significantly lower greenhouse gas emissions and requires less energy than activated carbon, making it a more environmentally friendly option (Kołtowski et al. 2017). These advantages position biochar as a cost-effective and sustainable solution for removing hazardous pollutants from wastewater while minimizing energy demand and environmental impact (Xiang et al. 2020).

The effectiveness of biochar in wastewater treatment depends on its properties, which are influenced by factors such as pyrolysis temperature and feedstock source (Tomczyk et al. 2020). Biochar produced at temperatures above 500°C typically exhibits high surface area, microporosity, hydrophobicity, and a high carbon-to-nitrogen (C/N) ratio, making it highly effective at adsorbing organic contaminants (OCs). In contrast, biochar produced at lower temperatures (< 500°C) often contains more dissolved organic carbon and oxygen-containing functional groups, making it better suited for removing residual contaminants (Panahi et al. 2020). Furthermore, biochar derived from forest biomass and agricultural waste generally has a larger surface area compared to that produced from livestock manure or municipal solid waste (Amalina et al. 2022). These properties make biochar a versatile and efficient adsorbent for wastewater treatment, as illustrated in Table 3, which details the adsorption of organic pollutants by wood biochar.

Permeable Reactive Barriers (PRBs) are a widely used technology for remediating contaminated groundwater. A PRB is a subsurface barrier installed perpendicular to the flow of groundwater, composed of a permeable material mixed with a reactive substance that removes contaminants as groundwater passes through it. Biochar has emerged as a highly effective reactive material for PRBs due to its high adsorption capacity, porous structure, and ability to adsorb a wide range of contaminants, including trace metals, polycyclic aromatic hydrocarbons (PAHs), and other organic

compounds. For instance, Casas et al. (2023) demonstrated that a biochar-based PRB removed up to 99% of nitrate from groundwater. These findings underscore the potential of biochar-based PRBs as a cost-effective and environmentally friendly solution for groundwater remediation.

The escalating impacts of climate change have intensified the urgency to reduce atmospheric CO₂ emissions. Within this context, the soil carbon cycle plays a critical role in regulating global warming, as soil carbon sequestration offers a viable strategy to mitigate CO₂ levels. Biochar, a carbon-rich material produced through pyrolysis, enhances this process due to its resistance to microbial decomposition, thereby stabilising carbon in soils over long timescales (Talaiekhzani et al. 2021). While studies have explored the potential of biochar for carbon sequestration, results remain inconsistent, with both positive and negative impacts on net carbon emissions observed under varying conditions (Bunce et al. 2018). Biochar contains labile carbon (readily metabolised by microbes) and recalcitrant carbon (resistant to degradation). Initial microbial consumption of labile carbon increases short-term mineralization, but the persistence of recalcitrant carbon results in a net gain of sequestered carbon over time (Nasrullah et al. 2020). Notably, mineralization rates are higher in infertile soils and soils with high native carbon content compared to fertile or carbon-poor soils. However, the efficacy of biochar depends on feedstock type and pyrolysis parameters (e.g. temperature, duration), which dictate its physical and chemical properties (Lu et al. 2019). Establishing clear relationships between pyrolysis conditions and the carbon sequestration potential of biochar is essential for optimising its application in climate mitigation strategies.

The exponential growth in demand for technological and electrical products has heightened the need for efficient energy storage solutions. Supercapacitors, known for their rapid charge/discharge cycles, high power density, and long-term durability, are among the most promising technologies. In contrast, rechargeable batteries like lithium-ion cells prioritise high energy density but suffer from slower charge rates. The performance of these systems

hinges on electrode materials such as carbon nanotubes, activated carbon, and graphene, which offer large surface areas and porous structures. However, their prohibitive cost limits are widespread. To address these limitations, biochar, a low-cost, porous carbonaceous material derived from biomass, has emerged as a viable alternative for electrodes in supercapacitors and microbial fuel cells, leveraging its high surface area and sustainable production (Thomas et al. 2019).

Beyond energy storage, biochar demonstrates versatility in animal husbandry. When added to animal feed, biochar enhances digestive efficiency, boosts immunity, reduces methane emissions, and improves growth rates by adsorbing toxins and optimising nutrient absorption (Ishak et al. 2019, Chiappero et al. 2020). Studies also highlight its role in improving manure quality, soil fertility, and feed conversion efficiency while lowering greenhouse gas emissions and nutrient losses (Schmidt et al. 2019). Synergised with sustainable farming practices, biochar significantly enhances livestock productivity and meat quality without adverse health effects, as evidenced by trials across poultry, swine, and cattle (Amalina et al. 2021). While some studies report neutral outcomes, none indicate detrimental impacts on animal health.

Conclusion

Biochar, a carbon-rich material produced through the pyrolysis of biomass, is a versatile tool for soil amendment in agriculture and water purification. When applied to soil, biochar enhances soil fertility, water retention capacity, and erosion resistance, leading to improved crop yields and reduced dependency on chemical fertilisers. Additionally, biochar contributes to environmental sustainability by sequestering carbon dioxide in a stable form for hundreds to thousands of years, thereby reducing greenhouse gas emissions. However, the environmental impact of biochar varies depending on the feedstock type (e.g. wood, crop residues) and production conditions (e.g. pyrolysis temperature, reactor efficiency). To maximise its benefits, biochar must be produced from sustainable biomass sources using energy-efficient technologies. A comprehensive Life Cycle Assessment (LCA) is essential to evaluate its global environmental impact, encompassing stages such as biomass collection, biochar production, field application, and by-product utilisation. The LCA should assess key impact categories, including climate change, acidification, ozone depletion, land use, and ecotoxicity. Furthermore, long-term field and greenhouse studies are critical to validate the effectiveness of biochar across diverse soil types, climatic conditions, and cropping systems. These studies will ensure that biochar implementation in agriculture delivers sustainable environmental and agronomic benefits.

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