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Biochar for sustainable wastewater remediation

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ABSTRACT

Biochar, produced from industrial and agricultural waste, has evolved from an agricultural soil amendment to an effective and sustainable material for wastewater treatment. Engineered/modified biochar produced using chemical or physical methods possesses a high surface area, porosity, stability, and a variety of surface functional groups and active sites for removing a wide spectrum of organic and inorganic contaminants, including heavy metals, nutrients, and antibiotics. However, its performance depends on feedstock characteristics and pyrolysis conditions. Slow pyrolysis optimized with artificial intelligence and machine learning technologies is increasingly adopted because of its operational simplicity, high yield, and economic feasibility. Alkali treatment enhances surface functionality, whereas acidic modification promotes the formation of oxygen-containing groups. Experimental evidence indicates that the solution pH is a critical parameter influencing heavy metal adsorption onto biochar, and nanomaterial-based biochar composites can further improve treatment efficiency and cost-effectiveness. Recent advances are shifting toward tailor-made, regenerable biochar integrated into membranes, electrodes, photocatalysts, biofilters, and nutrient-recovery systems. Despite its potential, challenges remain in long-term performance, regeneration, spent-biochar management, and full-scale comparison with activated carbon. Overall, biochar supports environmental, social, and governance principles and sustainable water management while contributing to circular economy strategies and the United Nations' sustainable development goals.

Key words: adsorption, biochar, contaminant removal, sustainable technology, wastewater treatment

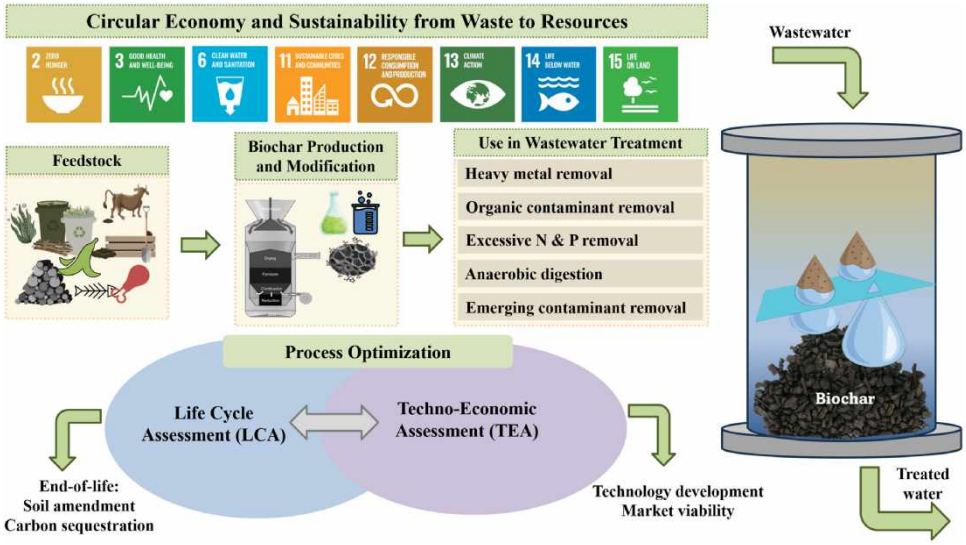
HIGHLIGHTS

- Engineered biochar enables tailored removal of diverse wastewater contaminants.
- Feedstock, pyrolysis conditions, and surface modification govern biochar performance.
- Wastewater complexity and competitive sorption hinder practical deployment.
- Life cycle and techno-economic assessments are critical for sustainable scale-up.

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GRAPHICAL ABSTRACT



1. INTRODUCTION

Biochar is a carbon-rich material produced via a thermochemical process of bio-based residues under a limited or oxygen-free environment. Its potential to remediate environmental pollutants and effectively sequester carbon has attracted increasing attention (IBI 2015). Biochar's versatility and its role in managing waste economically provide an alternative to the conventional linear 'take-make-waste' model. It can be obtained from a range of sources, including municipal and industrial waste, agricultural residues, forest byproducts, and animal-derived waste, which supports effective waste management (Hu *et al.* 2021). According to the UNEP Global Waste Management Outlook 2024, municipal solid waste generation is expected to reach 3.8 billion tons by 2050. The global annual cost of waste management is projected to rise to USD 640.3 billion (by 2050), compared with USD 361 billion in waste management costs in 2020 (UNEP 2024). These waste feedstocks, when converted to biochar, are used in a wide range of applications, including soil remediation, carbon sequestration, wastewater treatment, as well as bioenergy production, construction materials, catalysts, and electrode materials (Wang & Wang 2019; Tan & Yu 2024). In general, lignocellulosic waste, animal waste, and sludge waste are the most common feedstocks used for biochar production. Different feedstocks show varied performance for contaminant removal. For example, livestock-derived biochar has been reported to exhibit a high ammonia recovery capacity (Jagadeesh & Sundaram 2023).

Biochar, which was originally used as a soil amendment, can potentially enhance soil properties, including pH, organic matter content, nutrient retention, porosity, surface area, cation exchange capacity, microbial activities, and water retention capacity (Hossain *et al.* 2020; Singh *et al.* 2022). Biochar also acts as a sorbent for organic and inorganic contaminants in soil, water, and air, aiding environmental remediation (Gao *et al.* 2022; Qiu *et al.* 2022). Moreover, it contributes to climate change mitigation by capturing carbon and reducing greenhouse gas emissions (Kumar *et al.* 2022). Its production involves bioenergy generation and storage by cogenerating liquid (bio-oil) and gaseous (biogas) products, while also serving as a catalyst in biodiesel synthesis, further advancing renewable energy technologies (Laird *et al.* 2009) (Figure 1).

The production methods of biochar include torrefaction, gasification, pyrolysis, and hydrothermal/flash carbonization (Hu *et al.* 2021). Among these, slow pyrolysis is the most widely adopted due to its operational simplicity, high yield, and economic feasibility (Tripathi *et al.* 2016). In general, engineered or modified biochar performs better in contaminant removal because of improved physicochemical and biological properties (including specific surface area, porosity, cation exchange capacity, surface functional groups, and pH) compared with pristine biochar. Biochar can be modified through physical methods (e.g., ball milling, steam activation, microwave modification), chemical methods (e.g., acidic/alkaline treatment, metal impregnation), and biological methods (e.g., microbial inoculation) (Rajapaksha *et al.* 2016; Wang *et al.* 2017). In addition, artificial intelligence (AI) and machine learning (ML) technologies have been integrated into the process of selecting optimal feedstock for biochar production and identifying potential applications (Lakshmi *et al.* 2021). Emerging

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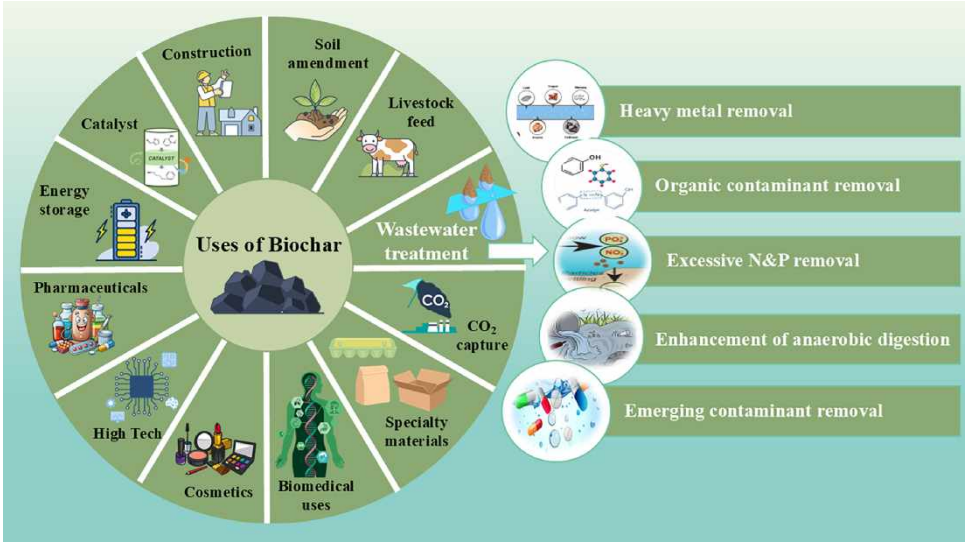


Figure 1 | Applications of biochar with a focus on wastewater treatment.

contaminants (ECs) are synthetic or naturally derived chemical and biological agents that have recently been detected in environmental compartments, yet are largely unregulated and insufficiently characterized. The ecological threats posed by these contaminants range from ecosystem disruption to human health hazards, often due to the lack of comprehensive risk assessment frameworks and regulatory coverage (Cheng *et al.* 2021). By adapting the biomass source and adjusting the pyrolysis parameters, biochar can be designed to display specific surface characteristics that enable it to effectively target particular pollutants during wastewater treatment. This is achieved through adsorption, surface redox reactions, precipitation, and catalytic degradation. Additionally, multifunctional biochar can improve sludge settling, enhance biological treatment, and promote the recycling of resources by utilizing sludge as a raw material in municipal wastewater treatment (He *et al.* 2022). Although biochar is effective for many contaminants, its use is limited by a naturally negative surface charge, limited anion-exchange capacity, and hydrophobicity, which make the removal of anionic pollutants (such as phosphates) and hydrophilic organics and persistent organic pollutants particularly challenging (Beljin *et al.* 2024). Moreover, while pristine biochar has shown promise in laboratory settings, it frequently falls short in field applications due to limitations in its adsorption efficiency, selectivity, and overall stability under varying environmental conditions.

In this study, we conducted a bibliometric analysis to identify the main research trends in biochar application in wastewater treatment and identify deficiencies in the existing literature from 2009 to 2026. This analysis has laid the foundation for future studies (Supplementary Information). We observed a rapid increase in the number of publications related to wastewater treatment (Figure 2). These studies mainly discuss biochar production techniques, contaminant removal (such as heavy metals, antibiotics, etc.), anaerobic digestion (AD), fermentation processes, microbial community dynamics, denitrification mechanisms, as well as kinetics, isotherms, and thermodynamic aspects relevant to wastewater treatment. Recent research has moved beyond generic adsorption to include pollutant-specific surface engineering, targeted removal of ECs such as per- and polyfluoroalkyl substances (PFAS), and the development of biochar-supported catalytic and single-atom systems (Xue *et al.* 2023; Zeng *et al.* 2024). This trend highlights the growing recognition of biochar as an effective remediation tool by repurposing biomass that may otherwise be disposed of through incineration or landfilling (Xie *et al.* 2015), and emphasizes the increasing demand for environmentally sustainable solutions to address current environmental challenges.

Biochar can promote a circular economy by reintegrating effective waste materials into wastewater treatment systems. This approach supports the achievement of net-zero emission goals set by numerous nations and plays a crucial role in advancing sustainable development worldwide (He *et al.* 2022; Li *et al.* 2022). While previous reviews have explored the use of biochar in wastewater treatment, they have predominantly focused on aspects such as production methods, surface modification strategies, and contaminant removal efficiencies. However, systematic evaluations incorporating life cycle assessment

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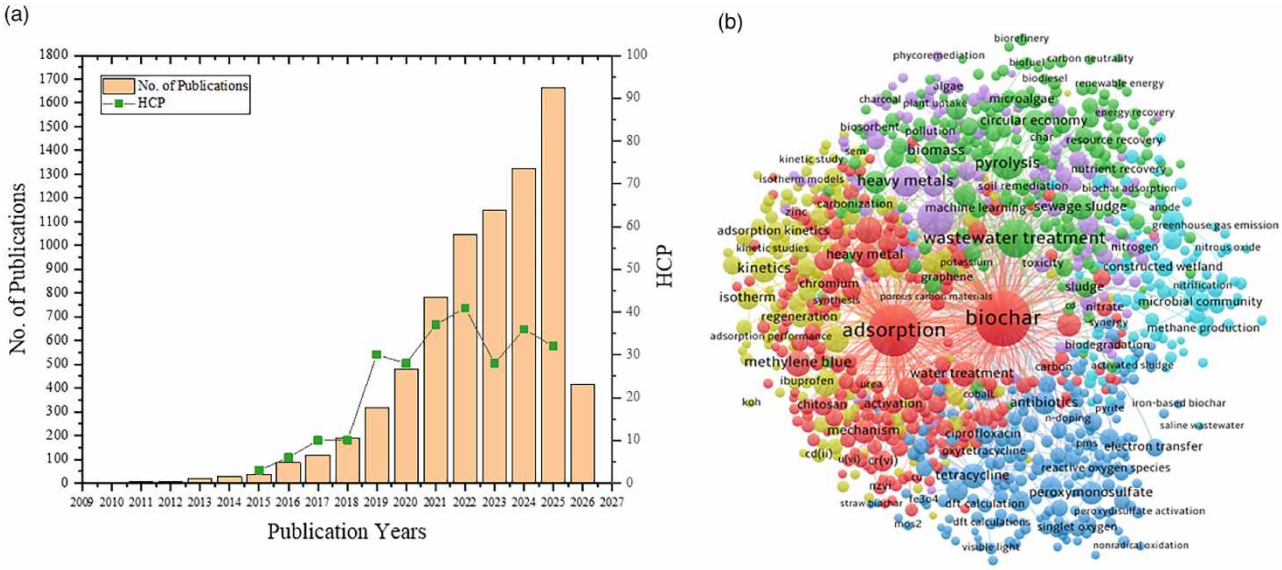


Figure 2 | (a) Number of publications on biochar and wastewater treatment from 2009 to 2026. HCP: Highly Cited Papers (b) Visualization of author keyword analysis from VOSviewer software.

(LCA) and techno-economic analysis (TEA) remain limited. This review addresses this gap by providing a focused examination of the environmental and economic dimensions of biochar-based treatment systems. Particular emphasis is placed on the influence of feedstock selection, pretreatment and post-treatment processes, and material characterization on overall performance, including the removal efficiency of persistent pollutants such as PFAS (Ihsanullah *et al.* 2025).

2. TECHNOLOGIES FOR BIOCHAR PRODUCTION

Feedstock selection, biochar production technologies, and modification strategies play a key role in determining the yield, physicochemical characteristics, and adsorption performance of biochar. A variety of materials have been reported to serve as feedstock sources for producing biochar for environmental remediation. Commonly used feedstocks include lignocellulosic residues, animal waste, and sewage sludge, each of which differs in availability and composition. These differences influence the surface area, porosity, mineral content, ash content, and functional groups, thereby affecting the suitability of the resulting biochar for specific wastewater contaminants. For example, wood-derived biochar reportedly has high biochar yield and stability compared with other types of biochar (Hassan *et al.* 2020). In addition, feedstock conversion technologies and modification techniques account for biochar properties as discussed in the following sections.

2.1. Thermal processes of biochar production

Various thermochemical pathways, including torrefaction, pyrolysis (including microwave-assisted), hydrothermal carbonization (HTC), and gasification produce biochar (Wang *et al.* 2017; Fang *et al.* 2018). Pyrolysis is a thermochemical process for converting biomass in an oxygen-deprived or limited oxygen environment (Cha *et al.* 2016). In a typical procedure, the biomass feedstock is first collected, pretreated, and then placed in a reactor and heated, usually at a controlled rate and at a desired temperature for a specific residence time. After the thermal process, the material is cooled under an oxygen-limited atmosphere to prevent combustion, and the resulting solid product is collected as biochar. The physicochemical properties of the biochar produced are affected by operational factors such as temperature, residence time, heating rate, and inert gas flow rate (Ahmad *et al.* 2012; Suliman *et al.* 2016). Pyrolysis methods include slow and fast pyrolysis. These differ in terms of the temperature, residence time, heating rate, and pressure. Slow pyrolysis (with moderate temperatures (typically 350–600 °C) and longer residence times (ranging from several hours to days)) yields a higher biochar content. This is because it minimizes condensable liquid products through increased cracking reactions (Inyang & Dickenson 2015). Fast pyrolysis, in contrast, favors bio-oil production up to a maximum of 75% from biomass (Qian *et al.* 2015). Studies have revealed that increasing the pyrolysis temperature beyond 500 °C yields biochar with increased hydrophobicity, surface area, and micropore volume. This makes it highly effective in eliminating organic pollutants (Keiluweit *et al.* 2010).

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Torrefaction is a conventional mild pyrolysis thermal technique that can be used to enhance the physical and chemical characteristics of biomass to yield a higher energy density and greater uniformity. This process typically involves a gradual heating of biomass at temperatures between 200 and 300 °C (generally without oxygen or with limited oxygen) under atmospheric pressure (Rousset *et al.* 2011). The resulting material retains more oxygen-containing functional groups, enhancing its adsorption capabilities (Zhang *et al.* 2019a; Li *et al.* 2019b). For example, previous studies on the removal of methylene blue (MB) and uranium (VI) from aqueous solutions showed that torrefaction char had an enhanced adsorption capacity compared with pyrolysis-derived biochar (Salapa *et al.* 2018; Li *et al.* 2019b).

Microwave-assisted pyrolysis is a sustainable approach for producing solid biochar, bio-oil, and biogas (Mutsengerere *et al.* 2019). This method utilizes electromagnetic radiation to rapidly heat biomass, accelerating decomposition reactions and reducing energy demands (Durán-Jiménez *et al.* 2018; Nhuchhen *et al.* 2018; Kadlimatti *et al.* 2019). Compared with conventional systems, it offers faster reaction rates, improved thermal efficiency, and higher selectivity (Durán-Jiménez *et al.* 2018). Moreover, it provides a uniform heat distribution and better control over product characteristics. Studies have revealed that utilizing a microwave system operating at 500 W power for 3 min at a frequency of 2,450 ± 25 MHz yields biochar with high specific surface area. However, further microwave treatment yields a lower surface area owing to the degradation of the micropore structure from overheating (Duran-Jimenez *et al.* 2017; Durán-Jiménez *et al.* 2018).

The HTC process transforms wet biomass into carbon-rich materials under temperatures ranging from 120 to 260 °C. This generally yields hydrochar or similar carbonaceous products (Kambo & Dutta 2015; Fang *et al.* 2018). The inclusion of a liquid (generally water) accelerates the carbonization process by functioning as both solvent and medium for reactions, which facilitates hydrolytic breakdown of lignocellulosic biomass (Libra *et al.* 2011). Typically, wet biomass undergoes heating and pressurization (2–10 MPa) within a confined system for durations of 5–240 min (Zhang *et al.* 2020d). As the process temperature increases, the hydrochar tends to exhibit a higher abundance of acidic functional groups on its surface. This enhances its capacity for adsorbing contaminants. Moreover, increases in both operating temperature and residence time can increase the hydrochar porosity. This expands its potential utility as an adsorbent (Saha *et al.* 2019). Many studies involving hydrochar derivatives have demonstrated notable efficacy as adsorbents in the removal of diverse inorganic and organic pollutants (Sun *et al.* 2011; Regmi *et al.* 2012; Bai *et al.* 2017; Zhou *et al.* 2017; Wu *et al.* 2020). HTC is a highly appealing thermochemical conversion technique for producing materials with high energy densities, oxygen-containing functional groups, and porous frameworks. These enhance the capability of biochar to adsorb contaminants from wastewater (Bridgwater 2012; Enaime *et al.* 2020). A TEA of HTC of sewage sludge revealed that the cost of producing HTC (USD 38 per ton) was lower than that of conventional methods for managing sewage sludge, such as AD, incineration, landfilling, and composting (Xiao *et al.* 2022).

Gasification involves transforming biomass into gaseous fuel with the aid of gasification agents, usually at temperatures surpassing 800 °C (You *et al.* 2017). The generated gas mixture (known as 'syngas') mainly consists of CO_2, H_2, CO, and possibly marginal amounts of CH_4, in conjunction with solid char (~10%) and a liquid phase referred to as 'tar' (~ 5%) (Guo *et al.* 2020). Biochar typically contains high levels of alkali salts and alkaline earth minerals. These facilitate the precipitation of numerous heavy metal contaminants, thereby enabling their direct use as remediation agents in problematic soils (Yu *et al.* 2019).

2.2. Pretreatment technologies of feedstock

Pretreatment of feedstocks can be used to optimize the conversion process and improve the quality of the final product. Pretreatment techniques can generally be categorized into physical, chemical, and biological.

Physical pretreatment techniques consist of a series of steps including drying, crushing, sieving, and/or washing of the biomass feedstock. For example, lignocellulosic or plant-based feedstocks are generally dried to a consistent weight, typically at ~105 °C. Subsequently, they are crushed into fine particles using a hammer mill before being cut into different shapes and sizes according to specific requirements (Wang *et al.* 2016; Zhang *et al.* 2017). To obtain mineral-free sludge from paper mill sludge, an acid-washing process is employed, followed by rinsing with deionized distilled water (Cho *et al.* 2017). Dewatered sewage sludge is typically oven-dried overnight. This is followed by crushing, sieving, and storage in sealed containers before utilization (Agrafioti *et al.* 2013). Algal biomass is alkaline. It is typically washed with fresh water and subsequently dried, crushed, or flaked before pyrolysis (Roberts & de Nys 2016).

Chemical pretreatment methods typically involve the application of chemical reactions to modify the compositions of feedstock materials. In general, during the treatment process, biomass is submerged in a chemical solution or colloidal

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suspension. Then, it is dried before biochar production (Tan *et al.* 2016). Following pretreatment with metal ion solutions such as $AlCl_3$, $FeCl_3$, and $MgCl_2$, metal-loaded biochar composites with embedded oxyhydroxide nanoparticles (e.g., FeOOH and AlOOH) can be obtained (Zhang & Gao 2013; Son *et al.* 2018). Pretreatment of biomass for engineered biochar production also involves the use of corrosive chemicals such as acids, alkalis, and oxidants. These treatments aim to create biochar with an increased surface area, unique pore structure, and enhanced specific functional groups (Zhao *et al.* 2018). However, chemical treatment has several disadvantages, such as the use of corrosive reagents, generation of secondary wastewater, increased chemical and water consumption, and the possible formation of hazardous byproducts. These issues may reduce the overall sustainability of the process unless chemical recovery and waste management are properly addressed (Tan *et al.* 2015).

Biological pretreatment technology is a novel approach used to enhance biomass feedstock for engineered biochar production (Wang *et al.* 2017). The bacterial treatment of biomass feedstocks (notably through AD or biofuel processes) has been validated as an efficient method for generating 'biologically activated' biochar. This process involves the action of microorganisms that degrade organic matter within the biomass. It yields biochar with improved physicochemical properties and increased surface area. This potentially enhances its adsorption performance for various applications (Yao *et al.* 2011, 2015).

2.3. Post-treatment technologies of biochar

The modification process and the nature of the applied agents critically shape the physicochemical properties of biochar and its capacity for pollutant adsorption. Widely employed post-treatment strategies include acid–base activation, incorporation of metal oxides or salts, integration with clay minerals, and mechanical processing such as ball milling (Cheng *et al.* 2021) (Figure 3).

Acid–base treatment is widely employed to alter the surface characteristics of adsorbents, primarily by enhancing the surface area and pore structure of biochar. Additionally, the formation of functional groups such as C–OH through acid–base modification significantly influences the chemical adsorption process, thereby impacting the adsorption capacity of biochars (Hu *et al.* 2018). In phosphoric acid pretreatment, the pretreated biochar exhibited a substantial increase in surface area. This enhanced its adsorption capacity owing to the induction of acid catalysis and cross-linking, thereby yielding larger

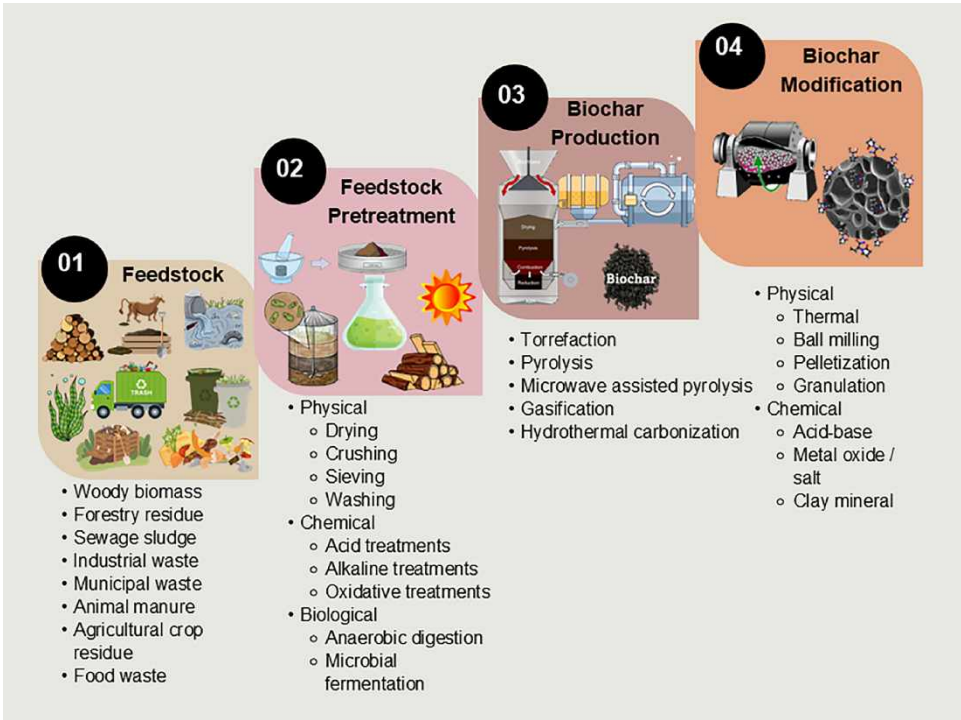


Figure 3 | Feedstock pretreatment and biochar production and modification.

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micropores (Chu *et al.* 2018). Wang *et al.* observed that the surface area of biochar increased significantly following acid and alkali treatment (a 10-fold increase with acid treatment and 14-fold increase with alkali treatment). This enhancement improved the adsorption capacity for enrofloxacin (an antibiotic), with a 27.80% increase after acid modification and 54.08% increase after alkali modification (Wang *et al.* 2019b). In addition, biochar modified with acids and alkalis has been employed for the removal of various ECs, including tetracycline and ciprofloxacin (CIP) (Huang *et al.* 2017; Li *et al.* 2018a; Jang & Kan 2019).

Metal oxide and metal salt modifications alter the biochar's adsorption capacities depending on the characteristics of the specific metal used. The resulting biochar possesses improved precipitation, electrostatic attraction, and anion-exchange capabilities, thus enhancing its overall adsorption capacity for ECs (Wu *et al.* 2019a). Wu *et al.* revealed that biochar-supported Ag/Fe nanoparticles (Ag/Fe/MB) effectively eliminate cephalexin from water. This study revealed that within 90 min, the cephalexin removal rate exceeded 86% through a combination of adsorption and reduction processes (Wu *et al.* 2017). In other experiments, the pH level significantly influenced the adsorption of levofloxacin onto Fe/Mn-biochar. It exhibited consistent adsorption capacity for levofloxacin even after five cycles of use (Xiang *et al.* 2019). Cu-biochar demonstrated a significant removal efficiency of 93.22% for doxycycline hydrochloride in water (twice that of unmodified biochar). Considering factors such as cost-effectiveness, efficiency, and practical applications in water systems, Cu-biochar shows significant capacity for improving the removal efficiency of doxycycline hydrochloride from water (Liu *et al.* 2017). Research on clay mineral-modified biochar (biochar–clay composite (CM-BC)) has expanded in recent years with two primary preparation methods: (1) mixing raw materials with clay minerals for pyrolysis and (2) combining biochar with clay minerals. By undergoing modification, clay minerals enhance the porosity of CM-BC and improve its interfacial compatibility with ECs. This enhances the adsorption capacity of CM-BC for ECs (Fu *et al.* 2020). Fu *et al.* utilized montmorillonite–corncob biochar composites for the co-removal of Pb^{2+} and atenolol, achieving maximum capacities of 139.8 and 86.9 mg/g, respectively (Fu *et al.* 2020). Montmorillonite–biochar composite (MT-BC) was prepared by heating a mixture of wheat straw and montmorillonite at 400 °C, and exhibited a 2.4-fold increase in norfloxacin adsorption capacity, influenced by the presence of Cu^{2+} ions, humic acid, and pH (Zhang *et al.* 2018).

Physical activation of biochar is generally achieved using agents such as steam, CO_2, air, or binary gas mixtures (e.g., CO_2/N_2). During pyrolysis, the surface chemistry of biochar undergoes complex transformations, which facilitate subsequent activation by enhancing porosity and functional group development. Unlike physical activation, chemical activation involves the use of additional reagents, thereby increasing the risk of introducing residual chemical contaminants into the activated product. While steam activation is preferred for certain biomass types, CO_2 activation is generally more effective in generating microporous structures and achieving higher specific surface areas (Colomba *et al.* 2022). In addition, thermal modification methods, such as conventional heating and microwave irradiation, have been used to activate biochar. Recently, advanced modification methods involving ultrasound waves, electrically charged gas (plasma), and electrochemical techniques have been applied for biochar activation (Sajjadi *et al.* 2019).

Ball milling is a method used to crush samples into nanometer sizes. It modifies various physicochemical properties of biochar (Wang *et al.* 2017) including the surface area, surface functional groups, and pore structure. The increase in surface area improves the physical adsorption capacity of biochar for ECs, whereas the introduction of new functional groups during milling enhances its chemical adsorption capacity (Qin *et al.* 2020). Milling parameters such as solvent quality and grinding media-to-biochar ratio determine the properties of the resulting biochar (Peterson *et al.* 2012). Research indicates that ball milling can be customized to modify the structure and surface parameters of biochar by using different media. Magnetic biochar, made into ultrafine particles through ball milling, demonstrates a high efficiency in removing pollutants such as carbamazepine from water with straightforward magnetic separation (Shan *et al.* 2016; Naghdi *et al.* 2017). Moreover, particle size enlargement through pelletization or granulation is also employed to improve handling, stability, and application in fixed-bed systems (Thomas 2021).

2.4. Biochar characterization techniques

The aforementioned thermochemical treatments, and surface modification strategies, determine properties such as the specific surface area, pore development, pore distribution, cation exchange capacity, crystallinity, aromaticity, catalytic activity, and functional group density. In addition, surface charge properties, including pH and pH at the point of zero charge (pH_{PZC}), are important because they influence electrostatic interactions between biochar surfaces and ionic contaminants. These properties directly contribute to adsorption mechanisms (Tomczyk *et al.* 2020) and are commonly evaluated

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using a range of characterization and analytical techniques. The surface area and pore characteristics are typically measured by Brunauer–Emmett–Teller analysis. The surface morphology and pore structure are examined using scanning electron microscopy and transmission electron microscopy. Surface elemental composition and chemical states can be determined using X-ray photoelectron spectroscopy (XPS). Functional groups are identified by Fourier-transform infrared (FTIR) spectroscopy, and crystalline mineral phases are analyzed using X-ray diffraction. The elemental composition is generally measured using CHNS elemental analysis. In addition, zeta potential analysis and pH_{PZC} determination are useful for understanding surface charge behavior, whereas thermogravimetric analysis can provide information on thermal stability and compositional changes (Amin *et al*. 2016; Igalavithana *et al*. 2017).

3. APPLICATION OF BIOCHAR AS AN ADSORBENT FOR POLLUTANT REMOVAL

The presence of various contaminants in wastewater has recently increased. This is primarily because of the release of industrial effluents, municipal wastewater, and urban and agricultural runoff into waterways (Xiang *et al*. 2020a). Wastewater contains various hazardous components including heavy metals and organic pollutants. These pollutants are generally introduced into water through various industrial activities such as smelting, mining, chemical manufacturing, leather processing, and battery production (Xiang *et al*. 2020a). Furthermore, municipal wastewater contains macronutrients including N, P, K, and various other ECs such as PFAS (Figure 4). In addition to their complex composition and environmental impact, these substances pose significant hazards to both environmental integrity and human health. This emphasizes the urgent need for effective wastewater management strategies (Enaime *et al*. 2020).

A variety of advanced and nonconventional adsorbents have demonstrated strong potential for wastewater treatment, including natural adsorbents (clays and zeolites), carbon-based hybrid nanomaterials, activated carbon (AC), metal oxide-based hybrid (magnetic and non-magnetic) nanocomposites, metal–organic frameworks, covalent organic frameworks (COFs), polymers and their nanocomposites (Dutta *et al*. 2021). Recent studies favor biochar for wastewater treatment over other adsorbents due to factors such as high surface area, which enhances pollutant removal efficiency, low cost of regeneration, and environmental benefits such as lower greenhouse gases (GHGs) and global warming potential (Cha *et al*. 2016).

Recently, AI and ML have been increasingly integrated into biochar research to support the selection of optimal feedstocks and to improve the prediction and optimization of contaminant adsorption from wastewater. These data-driven approaches help identify the most influential material and process parameters governing adsorption performance.

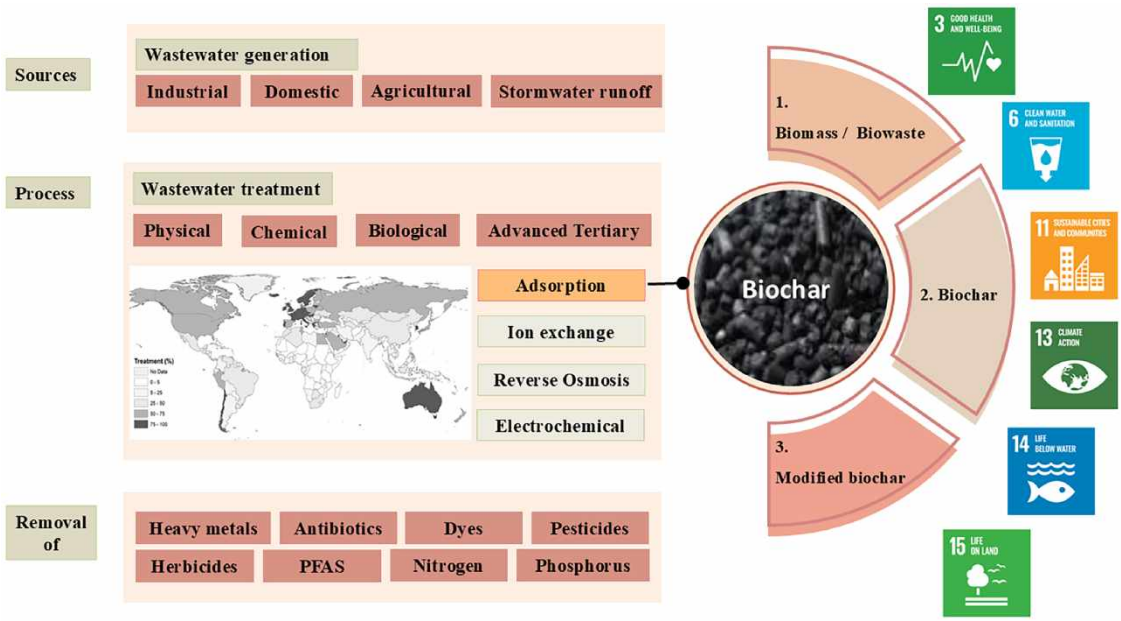


Figure 4 | Application of biochar in wastewater treatment.

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For example, Dashti *et al.* used ML models to evaluate heavy metal removal by biochar and found that the heavy metal-to-biochar concentration ratio and the total carbon content of the biochar were key factors governing removal efficiency (Dashti *et al.* 2023). Similarly, many researchers have conducted ML to identify optimal parameters for different contaminant removal from wastewater (Table 1).

As summarized in Table 1, the most important factors influencing adsorption include solution pH, adsorbent dosage, temperature, contact time, and initial contaminant concentration. For example, as adsorbent dosage increases, it generally improves removal efficiency (percentage removal). Contact time and kinetics determine adsorption rate and the time required to reach equilibrium. Moreover, the initial contaminant concentration affects the driving force for mass transfer.

Regeneration and reusability are important factors to consider when assessing biochar-based adsorbents for wastewater treatment. Even if biochar shows high contaminant removal at the beginning, its real value depends on whether it can be reused in several cycles without a significant drop in performance. This is crucial for both economic feasibility and long-term application (Baaloudj *et al.* 2025). In addition, highly concentrated or diverse wastewater streams may require higher adsorbent dosages, further increasing treatment costs. The presence of toxic pollutants can necessitate regeneration or disposal of spent adsorbents, further adding to the economic burden (Satyam & Patra 2024). Previous studies have reported several regeneration methods, including thermal regeneration, solvent washing, water washing, acid or alkali treatment, and chemical regeneration. In general, regeneration performance is affected by the strength of pollutant binding, pore accessibility, surface functional groups, and possible structural changes during reuse. A gradual reduction in adsorption efficiency is often observed after repeated cycles due to incomplete desorption, pore blockage, surface fouling, and loss of active adsorption sites. Compared with thermal regeneration, chemical regeneration is an efficient approach that can be carried out at lower temperatures, typically between 20 and 35 °C. It helps lower energy consumption and overall treatment costs while making it applicable to a wider range of biochar materials. The choice of chemical regenerant is also not strictly fixed, as both organic and inorganic agents have been successfully used to regenerate different types of biochar with high performance (Dai *et al.* 2019).

3.1. Removal of heavy metals

Heavy metal pollutants primarily encompass metals and their compounds characterized by an atomic weight and density, often defined as having a density exceeding 4 ± 1 g/cm^3. This category encompasses a range of hazardous substances such as Cu, Zn, Hg, Pb, As, Cr, Co, Ni, Cd, Sn, Mn, and Ag, which are widely recognized as the most prevalent and toxic inorganic contaminants in both water and soil ecosystems (Nadeem *et al.* 2006). The ability of biochar to adsorb heavy metals depends on its specific surface area, cation exchange capacity, and surface-active functional groups (Table 2). The adsorption mechanisms of heavy metals to biochar involve physicochemical processes such as ion exchange, surface complexation, electrostatic interactions, and precipitation, while microporosity increases surface area and site availability, potentially enhancing nonspecific interactions such as van der Waals forces (Zhang *et al.* 2020a).

Chemical adsorption relies on the molecular-level reactions occurring between the adsorbent and the metal. The functional groups, including carboxyl, carbonyl, and hydroxyl, present on the surface of biochar will facilitate the adsorption. The dissociation of oxygen-containing functional groups typically imparts a negative charge to biochar, facilitating the efficient removal of toxic heavy metals such as Cu, Cd, Pb, Ni, Hg, and Mn from wastewater (Qambrani *et al.* 2017). In addition to adsorption, heavy metal removal involves a complex interplay of multiple collaborating mechanisms (Zhang *et al.* 2020a). For pollutants such as Pb^{2+}, Cu^{2+}, Cd^{2+}, Ni^{2+}, Zn^{2+}, and Hg^{2+}, adsorption, influenced by specific surface area, is governed by interactions with surface functional groups, such as carboxyl (R-COOH) and hydroxyl (OH) groups. In the case of Cr(VI), it is reduced to Cr(III), which can then be adsorbed by biochar. This adsorption is facilitated by the oxygen-containing functional groups on biochar, particularly at lower pH levels (Enaime *et al.* 2020).

The ability of biochar to adsorb heavy metals is influenced by various factors including the temperature of the aqueous medium, pH, and the presence of ions such as Na^+, K^+, Ca^{2+}, and Mg^{2+} (Ye *et al.* 2022). For example, as the pH of the aqueous solution increases, the functional groups present on the surface of the biochar lose protons, resulting in an augmentation of the negative charge. Consequently, this leads to an enhanced adsorption capacity for heavy metal cations. When the solution contains a high concentration of competing ions including Na^+, K^+, Ca^{2+}, Mg^{2+}, Cl^-, NO_3–, and SO_4^{2-} in the solution, they can effectively outcompete heavy metal ions for adsorption sites, thereby reducing the overall adsorption

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Table 1 | ML studies for different contaminant removal by biochar

Machine learning model	Type of contaminant	Coefficient of determination (R^2)	Mean square error	Key parameters for high adsorption	Reference
Adaptive neuro fuzzy inference system (ANFIS)	Heavy metals	0.942	0.0069	Concentration ratio of heavy metals/biochar, total carbon content in biochar	Dashti *et al.* (2023)
	Textile wastewater (Crystal Violet)	0.8823	~	Solution pH, adsorbent dosage, initial dye concentration, temperature	Kumari *et al.* (2025)
Artificial neural network	Pharmaceuticals (Paracetamol)	1	0.0064	Temperature, initial concentration, contact time	Afolabi *et al.* (2020)
	Pharmaceuticals (tetracycline, sulfamethoxazole)	0.83, 0.93	~	Surface area of biochar, pH of the solution, initial concentration	Zhu *et al.* (2021)
	Textile wastewater (Crystal Violet)	0.9685	~	Solution pH, adsorbent dosage, initial dye concentration, temperature	Kumari *et al.* (2025)
	Heavy metals	0.948	0.0032	Cation exchange capacity (CEC), pH of biochar in water	Zhu *et al.* (2019)
	Pharmaceuticals	>0.99	<1	Surface area of biochar	Mojiri *et al.* (2020)
Coupled simulated annealing-least squares support vector machine	Heavy metals	0.983	0.002	Concentration ratio of heavy metals/biochar, total carbon content in biochar	Dashti *et al.* (2023)
End-to-end machine learning framework	Dye	0.97–0.99	~	Pyrolysis temperature, activator-to-biomass ratio, specific surface area	Gao *et al.* (2026)
Genetic programming (GP)	Heavy metals	0.975	0.003	Concentration ratio of heavy metals/biochar, total carbon content in biochar	Dashti *et al.* (2023)
Gradient boosting trees	Pharmaceuticals (Tetracycline, Sulfamethoxazole)	0.72, 0.89	~	Surface area of biochar, pH of the solution, initial concentration	Zhu *et al.* (2021)
Kernel extreme learning machine	Heavy metals	0.919	0.00008	pH of biochar in water, adsorption environment temperature, pH value of the solution, ionic radius, total carbon content in biochar	Zhao *et al.* (2021b)
k-Nearest neighbor	Textile wastewater (Crystal Violet)	0.9421	~	Solution pH, adsorbent dosage, initial dye concentration, temperature	Kumari *et al.* (2025)
Kriging models	Heavy metals	0.980	0.002	pH of biochar in water, adsorption environment temperature, pH value of the solution, ionic radius, total carbon content in biochar	Zhao *et al.* (2021b)
Particle swarm optimization-ANFIS	Heavy metals	0.971	0.0034	Concentration ratio of heavy metals/biochar, total carbon content in biochar	Dashti *et al.* (2023)
Random forest	Personal care products	0.868, 0.908	~	Surface area of biochar, micropore volume of biochar, (O + N)/C ratio	Zhu *et al.* (2022)
	Pharmaceuticals (tetracycline, sulfamethoxazole)	0.82, 0.9	~	Surface area of biochar, pH of the solution, initial concentration	Zhu *et al.* (2021)

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Table 1 | Continued

Machine learning model	Type of contaminant	Coefficient of determination (R^2)	Mean square error	Key parameters for high adsorption	Reference
	Phosphorus	0.89	~	Solution pH, P concentration, adsorbent dosage, metal composition	Lu *et al.* (2025)
	Heavy metals	0.973	0.0062	Cation exchange capacity (CEC), pH of biochar in water	Zhu *et al.* (2019)
Response surface methodology	Textile wastewater (Crystal Violet)	0.9618	~	Solution pH, adsorbent dosage, initial dye concentration, temperature	Kumari *et al.* (2025)
	Pharmaceuticals	>0.97	<1	Surface area of biochar	Mojiri *et al.* (2020)
XGBoost	Dye (Congo Red)	0.997	~	Solution pH, initial dye concentration, adsorbent dosage	Bibi *et al.* (2023)

capacity of the biochar for heavy metals (Qiu *et al.* 2021). Interactions between different heavy metals in polluted water can also affect the adsorption process (Ye *et al.* 2022). It is important to consider variables such as the application rate, dosage, and recovery method, as well as the regeneration and treatment of biochar adsorbed with metals. These variables are crucial as they directly affect the effectiveness of metal removal using biochar. The application rate and dosage determine the amount of biochar applied and the concentration of pollutants that can be adsorbed effectively. The selection of the recovery method influences the efficiency of metal recovery from spent biochar, which allows for its potential reuse or safe disposal. In addition, effective regeneration and treatment of metal-adsorbed biochar are essential for maintaining its adsorption capacity over multiple cycles and ensuring its long-term effectiveness in treating polluted water (Li *et al.* 2017a). These considerations are crucial for advancing biochar from laboratory validation to scalable, field-ready solutions for water purification.

3.2. Removal of organic contaminants and ECs

Organic pollutants, including antibiotics, agricultural pesticides, herbicides, industrial dyes, oils, phenolic compounds, and persistent organic pollutants, are frequently detected in wastewater and are often regarded as hazardous contaminants (Dai *et al.* 2019). The removal of organic pollutants by biochar is governed by multiple mechanisms, including π–π interactions, hydrophobic interactions, hydrogen bonding, electron donor–acceptor interactions, and pore-filling effects (Panahi *et al.* 2020).

ECs represent a diverse group of chemical pollutants with potential hazards to both human health and ecological balance. These substances are commonly found in water sources, and originate from pharmaceuticals, personal care products, industrial chemicals, and persistent organic pollutants (Liu *et al.* 2019). PFAS (commonly known as 'forever chemicals') are a large class of synthetic organic fluorinated compounds representing thousands of chemicals that have been used extensively in industrial and consumer products since the mid-20th century for their capability to resist water, oil, and stains (Glüge *et al.* 2020; Wang *et al.* 2021), and which have been detected across various water samples (Tran *et al.* 2019; Baluyot *et al.* 2021). ECs undergo constant translocation and transformation in the environment. Although their levels in water may be low, these can accumulate in organisms and pose potential risks to both the environment and human health as they transfer through the food chain (Gomes *et al.* 2017). Several methods have been used to remove ECs, including microbial degradation (Ferreira *et al.* 2016), chemical oxidation, membrane processes, adsorption, and electrochemical methods (Acero *et al.* 2015; Barrios *et al.* 2016). Among the methods used to remove ECs from wastewater, adsorption is widely accepted due to its numerous advantages, including low cost, high efficiency, and versatility in treating a wide range of pollutants. Some of the most commonly utilized adsorbents for this purpose are AC, alumina, adsorbent resin, silica gel, and polyacrylamide (Zhao *et al.* 2016; Qu *et al.* 2018; Wong *et al.* 2018; Peng *et al.* 2019; Wang *et al.* 2019a;

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Table 2 | Heavy metal removal by biochar

Biochar type	Production conditions	Heavy metal	Sorption capacity based on Langmuir isotherm (mg/g)	pH	Removal mechanism	Reference
Brazilian pepper	300 °C, 2 h	Hg(II)	24.2	6	By phenolic hydroxyl and carboxylic functional groups and due to graphite like aromatic structure	Dong *et al.* (2013)
	450 °C, 2 h		18.8			
	600 °C, 2 h		15.1			
Sesame straw	700 °C, 4 h	Pb(II)	102	7		Park *et al.* (2016)
		Cd(II)	86			
		Cr(III)	65			
		Cu(II)	55			
		Zn(II)	34			
Sugar beet residue	600 °C, 2 h	Pb(II)	40.8	5		Inyang *et al.* (2012)
Bamboo	700 °C, 1 h	Cd(II)	77.08	5	Precipitation and cation exchange	Wang *et al.* (2018)
Corn straw	700 °C, 1 h	Cd(II)	76.18			
Pig manure	700 °C, 1 h	Cd(II)	92.68			
Canna indica	300 °C, 2 h	Cd(II)	63.32	5	Precipitation with the minerals and metal ion exchange	Cui *et al.* (2016)
	400 °C, 2 h		105.78			
	500 °C, 2 h		188.79			
	600 °C, 2 h		140.01			
Ramie residues	300 °C, 2 h	Cr(VI)	82.23	2	Chemical interaction between Cr(VI) and carboxyl and hydroxyl groups	Zhou *et al.* (2016)
	450 °C, 2 h		72.32		Electrostatic interactions through film and intraparticle diffusions and chemical binding by polar functional groups	
	600 °C, 2 h		61.18			
Rice straw	100 °C, 6 h	Al(III)	10.7	4.3	The complexation of Al with organic hydroxyl and carboxyl groups, surface adsorption and co-precipitation of Al with silicate particles (as $KAlSi_3O_8$)	Qian & Chen (2013)
	400 °C, 6 h		3.5			
	700 °C, 6 h		9.2			
Spartina alterniflora	400 °C, 2 h	Cu(II)	48.49	6	Surface complexation, ion interaction, and precipitation	Li *et al.* (2013)
Sugar beet pulp	300 °C, 2 h	Cu(II)	12.24	5.5	Ion exchange	Ozer & Tumen (2005)
	400 °C, 2 h		13.44			
	500 °C, 2 h		14.81			
Hickory wood (immersed in $KMnO_4$)	600 °C, 1 h	Pb(II)	153.1	6	Surface adsorption including the surface MnOx particles and oxygen-containing groups	Wang *et al.* (2015a)
		Cu(II)	34.2			
		Cd(II)	28.1			
Pinewood	300 °C, 20 min	Pb(II)	4.25	5	Physical endothermic process	Liu & Zhang (2009)

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Table 2 | Continued

Biochar type	Production conditions	Heavy metal	Sorption capacity based on Langmuir isotherm (mg/g)	pH	Removal mechanism	Reference
Rice husk	300 °C, 20 min	Pb(II)	2.4	5	Physical endothermic process	Liu & Zhang (2009)
Anaerobically digested sludge	600 °C, 2 h	Pb(II)	126.3	7	Physical adsorption, ion exchange adsorption, chemical adsorption	Ni *et al.* (2019)
Dairy manure	200 °C, 4 h	Cu(II)	48.4	7.63	Precipitation with PO_4^{3-} or CO_3^{2-} originating in biochar, and surface complexation through hydroxyl groups or delocalized π electrons	Xu *et al.* (2013)
		Zn(II)	31.6			
		Cd(II)	31.9			
	350 °C, 4 h	Cu(II)	54.4	9.98		
		Zn(II)	32.8			
		Cd(II)	51.4			
Municipal sewage sludge	900 °C	Cd(II)	42.8	6	Surface precipitation by forming Cd compounds in alkaline condition induced by buffer capacity of biochar, and ion exchange	Chen *et al.* (2015)
Anaerobically digested sludge	600 °C, 2 h	Cd(II)	7	7		Ni *et al.* (2019)
Oak bark	450 °C	Pb(II)	30.2	5	As the temperature rose within the range of 25 – 45 °C, the removal efficiencies of Pb^{2+} and Cd^{2+} in the aqueous solution increased correspondingly	Mohan *et al.* (2014)
		Cd(II)	7.4	5		
Oak wood	400 °C	Pb(II)	10.13	5		
		Cd(II)	2.87	5		
Switchgrass	300 °C, 30 min (KOH activated)	Cd(II)	34	5	Adsorption by oxygen-containing functional groups	Regmi *et al.* (2012)
		Cu(II)	31	5		
Cicer arietinum husk	450 °C, 4 h ($MgCl_2$ loaded)	Cu(II)	16.31	6	Adsorption via electrostatic, hydrogen bonding, and π–π bonding	Isaac & Siddiqui (2022)

Wu *et al.* 2019b). Adsorption involves the adhesion of pollutants to the surface of the adsorbent material, effectively removing them from wastewater.

3.2.1. Removal of antibiotics

The use of antibiotics in medicine has led to scientific breakthroughs in combating diseases that historically caused mortality (Kumar *et al.* 2005). However, the extensive use of antibiotics in agriculture, fisheries, and human medicine has increased environmental exposure to pharmaceuticals worldwide (Kumar *et al.* 2005). Detectable quantities enter the natural environment through pathways such as pharmaceutical manufacturing and hospital effluent, agricultural runoff, and urban wastewater (Huang *et al.* 2011; Ferrando-Climent *et al.* 2014). Consequently, antibiotic-resistant pathogens that can complicate important medical procedures, such as organ transplants and cancer care, have become common worldwide. In 2021, 1.14 million deaths were directly attributed to antibiotic-resistant bacteria globally, with nearly 5 million deaths associated with antibiotic-resistant bacteria (GBD 2021 Antimicrobial Resistance Collaborators 2024). Thus, antibiotic contamination must be reduced to limit the development and spread of antibiotic-resistant pathogens.

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Quinolone and macrolide antibiotics bind more strongly to soil particles and have been identified downstream from wastewater treatment facilities (WWTFs) in surface water and river sediments (Ferrando-Climent *et al*. 2014; Marti *et al*. 2014). Because of their recalcitrance in the environment, quinolones and macrolides accumulate to a higher degree than tetracyclines and beta-lactams, which are more susceptible to degradation via photodegradation and hydrolysis, respectively (Marti *et al*. 2014). The accumulation of quinolones in river environments promotes the selection and proliferation of qnr proteins (related to quinolone resistance), facilitating the growth of aquatic antibiotic-resistant bacteria at low antibiotic concentrations (Marti *et al*. 2014). Biochar is an emerging antibiotic remediation method that may offer high efficiency sorption in aqueous settings to limit environmental contamination that can lead to antibiotic-resistant bacteria (Cheng *et al*. 2021).

Municipal wastewater typically contains the following antibiotic classes: tetracyclines, sulfonamides, macrolides, beta-lactams, and quinolones (Du *et al*. 2023). Current spectroscopic techniques for analyzing antibiotic-biochar adsorption mechanisms that identify functional groups of both materials include XPS, FTIR spectroscopy, and Raman spectroscopy (Xiang *et al*. 2020b; Zhuo *et al*. 2023). The variations in peak intensities (or the addition of new peaks) in XPS, FTIR, and Raman spectra can be used to infer the functional group interactions as the antibiotic compound sorbs onto the biochar. Antibiotic adsorption mechanisms are controlled by the structural properties and functional groups on the biochar surface and within the antibiotic structure (Xiang *et al*. 2020b). As the solution pH varies, the overall charge of the biochar will change owing to changes in the protonation state of the surface functional groups, whereas the antibiotic molecule charge depends on the pK_a values for proton-active moieties on specific pharmaceutical compounds (Conte *et al*. 2021).

Antibiotic adsorption onto biochar is classified as physisorption (attraction between sorbent and sorbate) or chemisorption (in the form of chemical bonds). Both types contribute to adsorption (Hu *et al*. 2019; Zhang *et al*. 2020c). Physisorption in the form of electrostatic forces between sorbent and sorbate can make adsorption favorable or unfavorable due to attraction or repulsion between the antibiotic and biochar, which creates pH-dependent adsorption (Xiang *et al*. 2020b; Stylianou *et al*. 2021). Mechanisms of antibiotics adsorption to biochar include π–π interactions and electron donor–acceptor interactions (via electron rich or deficient aromatic rings in the biochar or antibiotic based on functional groups), electrostatic interactions, surface complexation, hydrophobic site interactions, and hydrogen bonding (Xiang *et al*. 2020b; Conte *et al*. 2021; Stylianou *et al*. 2021; Zhuo *et al*. 2023) (Figure 5). Favorable adsorption mechanisms for non-sterically hindered antibiotics (tetracyclines, sulfonamides, beta-lactams, and quinolones) include π-π stacking, hydrogen bonding, and hydrophobic

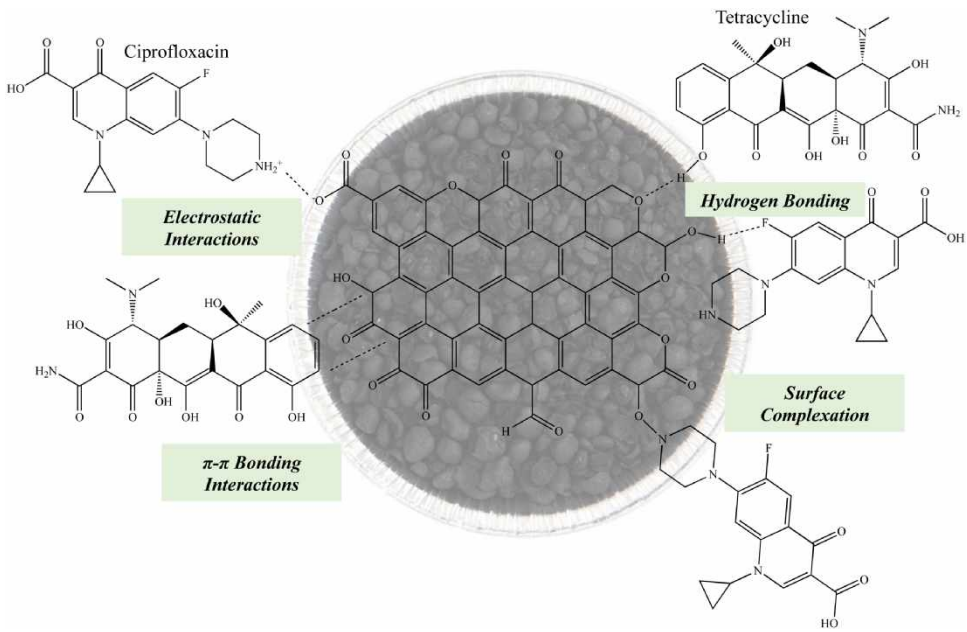


Figure 5 | Antibiotic adsorption mechanisms for biochar (Zhong *et al*. 2020; Xiang *et al*. 2020b; Conte *et al*. 2021; Stylianou *et al*. 2021; Zhuo *et al*. 2023).

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interactions (Hu *et al*. 2019; Zhang *et al*. 2020c; Zhuo *et al*. 2023). Hydrogen bonding can occur with oxygen containing and amino functional groups in both the biochar and antibiotic (Xiang *et al*. 2020b; Zhuo *et al*. 2023). Fluoroquinolones have an additional hydrogen bond receptor via the fluorine group that can improve adsorption capacities (Zhuo *et al*. 2023). Biochar that contains a higher fraction of aromatic structures and additional oxygen-containing functional groups favor π–π bond stacking due to their electron-donating capabilities (Zhang *et al*. 2020c).

Variations in FTIR and XPS spectra, and the H:C ratio aid in identifying variations in these electron rich functional groups as compounds adsorb to biochar. Hydrophobic interactions are ideal for antibiotics that have a higher octanol–water partition coefficient (K_{ow}), which leads to increased adsorption due to partitioning into organic matrix of the biochar. The sorption of hydrophobic compounds onto biochar can often be described by linear isotherm or Freundlich isotherms (with a factor of $n > 1$) (Wu *et al*. 2022). Adsorption mechanisms for bulky sterically hindered antibiotics (macrolides) include electrostatic forces, pore filling, and hydrogen bonding (Ndoun *et al*. 2021; Stylianou *et al*. 2021). Owing to the bulky nature of macrolides, such as erythromycin, the pore-filling adsorption mechanism dominates as steric hindrance disrupts other possible mechanisms (Xiang *et al*. 2020b; Stylianou *et al*. 2021; Zhuo *et al*. 2023). Biochars with higher porosity and larger pore sizes have higher degrees of adsorption via pore-filling mechanisms. As the solution pH varies, such that the macrolide and biochar become electrostatically attracted, hydrogen bonding and Van der Waals forces contribute to antibiotic adsorption (Ndoun *et al*. 2021; Stylianou *et al*. 2021). Hydrophilic biochars, which have a higher fraction of oxygen-containing functional groups, can enhance adsorption of macrolides due to an increased availability of sorption sites via electron donor functional groups and hydrogen bond receptors.

In aqueous environments, antibiotic compounds can be present in various charge states due to the presence of proton-active functional groups with multiple pK_a values, which vary as a function of solution pH. The adsorption capacity of biochar varies based on the use of pristine or modified biochar, with the latter having altered physicochemical properties. Biochar modification methods, such as metal doping and acid–base pretreatment, can alter functional groups and increase antibiotic sorption. Du *et al.* reported that modified biochar had a higher average adsorption capacity than unmodified biochar (>20 and 0.1–20 mg/g, respectively) (Du *et al*. 2023). A similar trend is highlighted below (Table 3), where modified biochar has superior maximum adsorption capacity compared with pristine biochar for a range of antibiotic classes at an environmentally relevant temperature (~25 °C) and across a pH range of 3–9, in some cases greater than 100 mg/g. However, modifying biochar will be more time and cost intensive compared with using a simple pristine biochar, which should be considered in large-scale projects.

The degradation of antibiotics in aqueous environments is a key factor determining their capability to persist in natural environments. Quinolones and macrolides persist more than other antibiotic classes because these are more resistant to photodegradation and hydrolytic degradation (Marti *et al*. 2014). *In situ* degradation provided by biochar's modifier(s) regarded as catalysts (highlighted in Table 4) promotes contaminant removal during adsorption, including hydrogen peroxide which is a hydroxyl radical source, metal doping (iron and copper), humic acid pretreatment, peroxydisulfate, and molybdenum disulfide (Shao *et al*. 2018; Luo *et al*. 2019; Yang *et al*. 2020; Zhong *et al*. 2020; Mansoori *et al*. 2021; Fang *et al*. 2022). Moreover, as compounds degrade, the toxicity level and residence time of the degradation products may differ from those of the parent compound. Chen *et al.* studied the photocatalytic degradation of quinolone CIP using artificial solar light irradiation, representing a potential degradation pathway that CIP may undergo before, during, and after wastewater treatment. Their results showed that the CIP degradation products formed after 2.5 h retained 39.4% of the antibacterial activity of the parent compound, based on *Escherichia coli* assays (Chen *et al*. 2018a). These results emphasize the importance of monitoring aqueous antibiotic contaminant degradation products throughout the wastewater treatment process.

3.2.2. Removal of dyes

Dyes (colorants) are characterized by their water or oil solubility and color-imparting characteristics. These are distinct from insoluble pigments. Toxic compounds are widely utilized in industrial processes and generally contribute to wastewater (Srivatsav *et al*. 2020). These compounds include azo dyes, heavy metals, cyanide, and phenols (Yaseen & Scholz 2019).

Recent studies have highlighted the potential of biochar as an effective sorbent for dye removal. For instance, Zhao *et al.* utilized pulp sludge-derived biochar, modified with $ZnCl_2$, to adsorb MB from aqueous media. Under optimal conditions (pH 8, 24 h contact time), the maximum adsorption capacity reached 590.2 mg/g, primarily through electrostatic

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Table 3 | Antibiotic maximum adsorption capacities for different pristine and modified biochars

Biochar biomass	Pyrolysis temp. (°C)	Biochar modification	Antibiotic	Initial conc. (mg/L)	Temperature (°C)	Max ads. pH	Equilibration time (h)	Max adsorption capacity (mg/g)	Ref.
Camphor leaves	650	ZnO	Ciprofloxacin	100	25	4	2	170	Hu *et al.* (2019)
Herbal residue	800	n/a	Ciprofloxacin	100	25	7	12	40	Shang *et al.* (2016)
Bamboo	700	Silica	Tetracycline	20	25	7	48	18	Zhao *et al.* (2019)
Bamboo	700	n/a	Tetracycline	20	25	7	48	2.45	Zhao *et al.* (2019)
Bagasse (sugarcane by-product)	800	Fe	Sulfamethoxazole	50	25	5	4	205	Zhang *et al.* (2020c)
Bagasse (Sugar cane by-product)	800	n/a	Sulfamethoxazole	50	25	5	4	35.4	Zhang *et al.* (2020c)
coffee grounds	700	Phosphoric acid	Sulfadiazine	10	25	5	24	98.5	Zeng *et al.* (2022)
Cotton stalks	600	Fe	Oxytetracycline	50	25	3	24	20	Gu *et al.* (2021)
Cotton stalk	600	n/a	Oxytetracycline	50	25	3	24	7.5	Gu *et al.* (2021)
Sewage sludge	550	Co & Fe	Ciprofloxacin	2–20	25–35	5	48	8.25	Zheng *et al.* (2022)
Sewage sludge	550	n/a	Ciprofloxacin	2–20	25–35	5	48	5	Zheng *et al.* (2022)
Corn stalk	600	Ball milled and N-doped	Norfloxacin	50	25	5	48	13	Wu *et al.* (2022)
Corn stalk	600	n/a	Norfloxacin	50	25	3	48	3.5	Wu *et al.* (2022)
Sewage sludge	500	MnO_2	Ciprofloxacin	10	25	5	6	16.69	Luo *et al.* (2022)
Sewage sludge	500	n/a	Ciprofloxacin	10	25	5	6	10	Luo *et al.* (2022)
Sewage sludge	500	Fe/Cu	Tetracycline	10	25	9	6	192	Liu *et al.* (2020)
Sewage sludge	500	n/a	Tetracycline	10	25	9	6	50	Liu *et al.* (2020)

interactions, cation exchange, and π–electron interactions, followed by physical adsorption (Zhao *et al.* 2021a). In another study, a magnetic biochar catalyst derived from Fe-tanned collagen fibers was used as a persulfate activator to successfully remove MB (Guo *et al.* 2022). Mahmoud *et al.* investigated the efficient adsorption capabilities of biochar derived from acid-

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Table 4 | Antibiotic degradation by different biochars with and without modifier catalysts

Biochar biomass	Pyrolysis temperature (°C)	Degradation catalyst	Antibiotic	Degradation % of parent compound	Degradation product toxicity (relative to parent compound)	Reference
Corn stalk	600	Ball milled and N-doped	Norfloxacin	93%	Less toxic	Wu *et al.* (2022)
Corn stalk	600	n/a	Norfloxacin	89%	Less toxic	Wu *et al.* (2022)
Sewage sludge	500	MnO_2	Ciprofloxacin	85%	Not Considered	Luo *et al.* (2022)
Sewage sludge	500	n/a	Ciprofloxacin	52%	Not Considered	Luo *et al.* (2022)
Sewage sludge	500	Fe/Cu	Tetracycline	92.5%	Not Considered	Liu *et al.* (2020)
Corn stalk	700	Fe_3O_4 and electrodes	Sulfadiazine	85%	Not Considered	Wei *et al.* (2022)

treated Kenaf *Hibiscus cannabinus* prepared via slow pyrolysis. They observed that it achieved 95 wt% removal of MB dye at a concentration of 50 mg/L. Optimal dye sorption occurred at a pH of 8.5 (Mahmoud *et al.* 2012).

Several studies have been conducted on pollutants from dye-based industries using different types of biochar. Studies have evaluated the effectiveness of Gliricidia biochar, biochar derived from agricultural crop residues, and a composite of stalk biochar synthesized with nano-zero-valent iron (nZVI). Gliricidia biochar exhibited enhanced adsorption capabilities because of its physicochemical properties including pH, specific surface area, and pore volume. Meanwhile, the stalk biochar composite displayed a synergistic effect in improving the persulfate catalytic activity for the removal of dyes (e.g., new coccine, methyl orange, Congo red, and direct blue 71) (Liu *et al.* 2022a). Moreover, the electrostatic attractions increased the adsorption of cationic dyes such as methyl violet, crystal violet, MB, basic blue, and rhodamine by peanut hull biochar (Wathukarage *et al.* 2019; Saravanan *et al.* 2020; Jegan *et al.* 2020a, 2020b; Liu *et al.* 2022a). Other studies have revealed that the reactions between dye molecules and sorbent surfaces are strongly influenced by pH, with basic conditions enhancing electrostatic reactions and acidic conditions promoting hydrogen bonding. Moreover, multiple mechanisms contribute to dye sorption, including electrostatic attraction, hydrogen affinity, pore filling, hydrophobic interactions, and π–π interactions (Li *et al.* 2016; Praveen *et al.* 2022).

3.2.3. Removal of pesticides and herbicides

Among the different adsorbents, biochar demonstrates a higher adsorption capacity for removing pesticides and herbicides from wastewater. This provides a cost-effective solution with an effective removal efficiency (Saleh *et al.* 2020). Atrazine is an herbicide commonly used in maize and sugarcane cultivation to control broadleaf weeds. It was removed using biochar derived from corn straw. The adsorption process involves various mechanisms including π–electron bonding, hydrogen bonding, and van der Waals force attraction. It results in the removal of approximately 97% of atrazine from agricultural wastewater (Suo *et al.* 2019). In another study, hardwood biochar produced from beech, birch, and maple pyrolyzed at 350 °C, improved the removal efficiencies to ≥97% for atrazine, ≥99% for chlorantraniliprole, and ≥88% for glyphosate (Jarry *et al.* 2026). Acid-modified hydrochar derived from passion fruit residues had a higher adsorption capacity, with values of 18.05, 10.83, and 5.05 μg/g for atrazine, ametryn, and metribuzin, respectively, corresponding to removal efficiencies of approximately 62, 72, and 52%, respectively (Almeida *et al.* 2026). In the case of triazine herbicides, biochar-based systems have also shown excellent performance, with maximum adsorption capacities reported to reach as high as 260.5 mg/g (Sun & Liang 2026). Another study reports *Bacopa monnieri* leaf–derived biochar for the removal of pendimethalin from aqueous solution, achieving an optimum removal efficiency of 97.56% at an equilibrium time of 120 min, pH of 8, an initial concentration of 40 mg/L, and an adsorbent dosage of 5 mg (Thakor *et al.* 2026). A study by Baharum revealed an improved efficiency in removing diazinon (insecticide) using biochar derived from carbonized coconut shells prepared via a pyrolysis process at 700 °C and modified with H_3PO_4 and acid–alkali (NaOH) (Baharum *et al.* 2020). Devi *et al.* revealed that pentachlorophenol, a prevalent pesticide in agricultural wastewater, was effectively removed

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using zero-valent iron magnetic sludge biochar derived from paper mill waste (Devi & Saroha 2014). Klasson *et al.* demonstrated that dibromochloropropane pesticide was removed effectively by biochar derived from almond shells pyrolyzed at 800 °C for 45 min (Klasson *et al.* 2013). Several studies have demonstrated the effectiveness of switchgrass biochar and magnetic switchgrass biochar in removing herbicides from wastewater under conditions characterized by low solubility pH and fluctuating temperatures (Essandoh *et al.* 2017a, 2017b; Xiang *et al.* 2020a).

3.2.4. Removal of per and polyfluoroalkyl substances

PFAS are extensively used in various industrial and consumer applications, including food packaging, textiles, cosmetics, and firefighting foams. This is owing to their desirable properties such as waterproofing, stain resistance, and chemical stability (Prevedouros *et al.* 2006; Gagliano *et al.* 2020; Ji *et al.* 2020). PFAS have raised significant national and global health concerns owing to their persistence in the environment, potential toxicity, and widespread human exposure to contaminated water and food (Figure 6). Many PFAS are highly resistant to degradation. This enables these to accumulate in ecosystems and living organisms. Over time, they pose severe health risks (Glüge *et al.* 2020). Moreover, their extensive use has led to increasing concerns about their widespread distribution in the environment (Tavasoli *et al.* 2021). In general, PFAS adsorption onto biochar depends on the physicochemical characteristics of both the PFAS molecules and the biochar adsorbent. This process is determined by several interaction mechanisms, including electrostatic attraction, hydrophobic interactions, and π–π interactions. In addition, the structural properties of biochar, such as its surface area and pore structure, are key factors in determining adsorption performance. Biochar with a high surface area and well-developed micropores and mesopores can enhance PFAS uptake through pore filling and van der Waals interactions. Therefore, controlling and optimizing the porosity of biochar via pyrolysis and post-modification mechanisms is an effective strategy for improving PFAS removal (Ihsanullah *et al.* 2025).

Among the most widely used and studied PFAS are long-chain perfluoroalkyl acids such as perfluorooctanoic acid (PFOA) and perfluorooctane sulfonate (PFOS) (Tokranov *et al.* 2021). PFAS are observed worldwide in surface and groundwater sources, aquatic matrices, coastal waters, and public drinking water supplies before being distributed to consumers (Wei *et al.* 2018; Nakayama *et al.* 2019; Evich *et al.* 2022). Long-chain PFAS tend to accumulate in wastewater sludge solids, potentially contaminating groundwater when these materials are applied to agricultural land or disposed of in landfills (Lindstrom *et al.* 2011; Sepulvado *et al.* 2011). Long-chain PFAS were observed to be more prevalent in sludge than

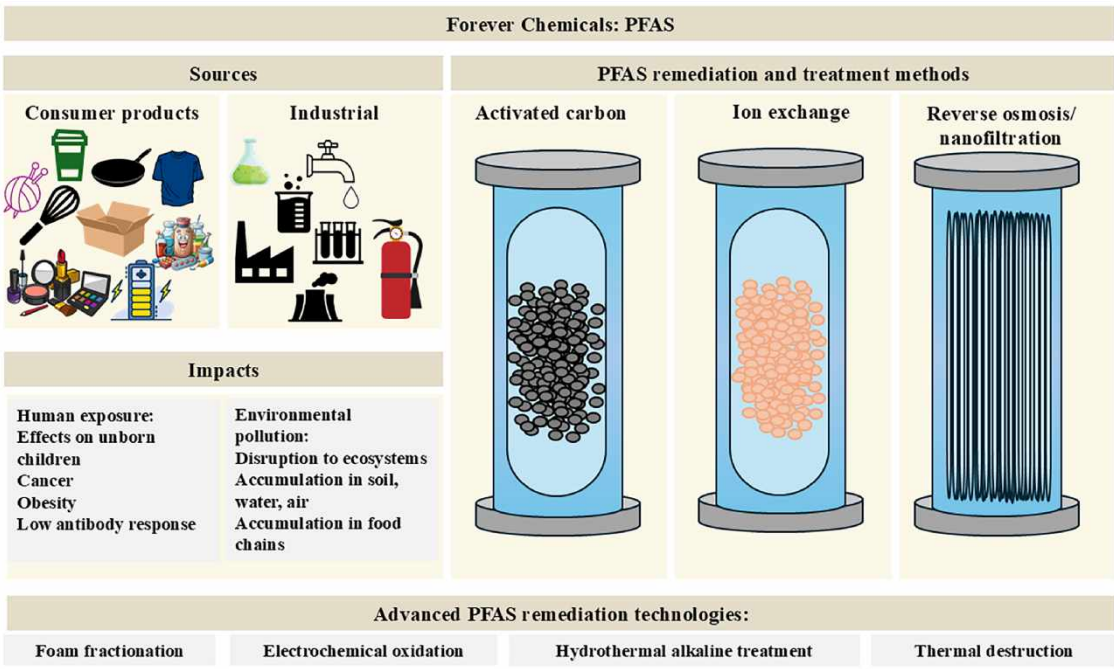


Figure 6 | PFAS remediation technologies for wastewater.

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short-chain PFAS because of their high tendency to adsorb onto solids. This aligns with studies indicating that the partition coefficient (*Kd*) increases with the number of CF2 groups, which causes a higher adsorption of long-chain PFAS (Adams *et al.* 2021). Various studies have identified high concentrations of conventional PFAS in wastewater treatment plant effluents, often exceeding 100 ng/L. Alternative PFAS have also been found in significant amounts, with levels reaching several hundred nanograms per liter (Houtz *et al.* 2018; Jiawei *et al.* 2019; Vo *et al.* 2020). Conventional water treatment techniques such as coagulation, flocculation, sedimentation, and filtration are largely ineffective in removing PFAS from contaminated water (Wanninayake 2021). Although techniques such as sonolysis and electrochemical treatment have proven effective in degrading PFAS in laboratory experiments, their high energy requirements limit their practicality to small, concentrated waste streams (Campbell *et al.* 2009; Schaefer *et al.* 2015; Horst *et al.* 2018). Granular activated carbon is widely regarded as a practical solution for PFAS removal, although it struggles with limitations such as reduced efficiency for short-chain PFAS and interference from substances such as anions and organic matter. Furthermore, its slow adsorption rate can lead to early breakthrough. Despite some potential, large-scale PFAS treatment remains challenging (Du *et al.* 2014; Rahman *et al.* 2014).

Militao *et al.* developed biochar-alginate beads derived from albumin and rice straw to remove PFAS. The beads exhibited impressive and rapid adsorption of PFOS, achieving removal efficiencies between 87 and 99% in under 16 h. It was determined that hydrophobic sorption was the dominant mechanism over electrostatic sorption, highlighting the significant potential of biochar-alginate composites for PFAS removal from water (Militao *et al.* 2023). In a different study on softwood-derived biochar, the time required to achieve sorption equilibrium for PFAS ranged from 12 to 48 h. The removal efficiency for PFAS in mixed solutions was reduced by 48.7–76.7%, compared with the higher removal rates observed when treating single compounds like PFOA, PFOS, perfluorobutanoic acid (PFBA), and perfluorobutanesulfonic acid (PFBS) (Zhang *et al.* 2021). The biochar–polymer composite material produced using wood chip biochar through slow pyrolysis had maximum adsorption capacities for PFOS, PFOA, PFBS, and PFBA 634, 536, 301, and 264 mg/g, respectively (Deng *et al.* 2023). With regard to PFAS in soil, biochar derived from waste timber (activated with steam and CO_2) has shown varying performance depending on soil properties. Specifically, sorption was significantly weaker in soils with high total organic carbon, and short-chain PFAS (CF3–CF4) exhibited weaker sorption than longer-chain PFAS (>CF5). Furthermore, perfluoroalkyl sulfonic acids (PFSAs) demonstrated stronger sorption than perfluoroalkyl carboxylic acids (PFCAs). This highlights the influence of both PFAS structure and soil composition on sorbent effectiveness (Sørmo *et al.* 2021).

Based on the studies mentioned above, it is evident that modified biochar, including functionalized biochar such as amine/cationic-modified and metal-modified biochar, shows improved PFAS removal compared with pristine biochar. These modifications enhance adsorption by introducing stronger electrostatic interactions, increased surface area, additional active sites, and, in some cases, improved catalytic functionality. Consequently, the removal efficiencies of both long- and short-chain PFAS can be improved. However, the removal of short-chain PFAS remains more challenging than that of long-chain PFAS (Imran *et al.* 2025; Teng *et al.* 2025). Therefore, ongoing monitoring and assessment of wastewater treatment systems are essential to understand their role in the fate and movement of emerging PFAS.

3.3. Removal of nitrogen (N) and phosphorus (P)

The presence of N and P in wastewater contributes to eutrophication. In the aqueous phase, these nutrients (primarily in the form of NH_4^+, NO_3^-, and PO_4^{3-}) are the main drivers of eutrophication and can be removed effectively by biochar during the tertiary treatment of wastewater (Yin *et al.* 2018; Xiang *et al.* 2020a). Excessive N and P originating from industrial, agricultural, and domestic sources are released into aquatic environments through agricultural runoff, and industrial and municipal wastewater systems (Zhang *et al.* 2020b). Various adsorbents have been assessed for their efficacy in removing N and P from water systems. These include bentonite, zeolites, nanoparticles, polymeric ion exchangers, and aluminum oxides (Zhang & Gao 2013; Angar *et al.* 2017; Li *et al.* 2017b; Alshameri *et al.* 2018; Xu *et al.* 2018). Many studies have demonstrated the utility of biochar in the removal of N and P from water systems. In this context, the effectiveness of biochar for sorption mainly relies on the characteristics of the feedstock utilized in its preparation, thermochemical procedures involved during its production, and subsequent chemical modifications to the biochar (Hassan *et al.* 2020).

Many researchers have experimented with the capability of NH_4–N adsorption by biochars derived from pine sawdust, wheat straw, peanut shell, corncob, cotton stalk, and giant reed (Gao *et al.* 2015; Hou *et al.* 2016; Liu *et al.* 2016; Yang *et al.* 2018; Lee *et al.* 2021). Overall, the majority of the reported NH_4–N adsorption capacities of biochar were found to

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be below 20 mg N/g (Zhang *et al*. 2020b). Yao *et al.* observed a minor nitrate adsorption capacity in biochars derived from peanut hulls, sugarcane bagasse, pepperwood, and bamboo as the pyrolysis temperature increased up to 600 °C (Yao *et al.* 2012). Fan *et al.* proposed that the presence of Mg^{2+} significantly enhances the adsorption of NH_4^+ by phosphate-rich biochar through ($MgNH_4PO_4$) precipitation (Fan *et al*. 2019). Furthermore, it has been demonstrated that treating biochar with concentrated HCl effectively enhances the nitrate adsorption capacity of biochars (Chintala *et al*. 2013). Wang *et al.* hypothesized that the augmentation of basic functional groups in La-modified biochar-enhanced NH_4^+ and NO_3^- adsorption by 1.9 and 11.2 folds respectively, compared with unmodified oak sawdust biochar (maximum adsorption for La-modified biochar 10.1 mg/g and 100mg/g respectively) (Wang *et al*. 2015b). Tang *et al.* used biochar derived from sewage sludge pyrolyzed at 450 °C. It exhibited exceptional efficacy in removing ammonium from wastewater owing to its high specific surface area and advantageous functional groups (Tang *et al.* 2019). For municipal wastewater treatment, a combination of biochar and a biofilter has been used to extract N and P (Cole *et al*. 2017).

In water, the inorganic form of phosphorus (PO_4^-P) exists in various protonation states, including $H_2PO_4^-$, HPO_4^{2-}, and PO_4^{3-}, depending on the pH. Phosphate anions typically encounter repulsion from the negatively charged surfaces of biochar. This results in generally low adsorption capacities of unmodified biochars for inorganic phosphorus (Zhang *et al*. 2020b). Fang *et al.* observed higher removal efficiencies of PO_4^-P achieved by corn biochar pyrolyzed at 300, 450, and 600 °C (Fang *et al*. 2014). Engineered biochar achieved by modifying the physicochemical properties of biochar combined with $Al(OH)_3$ exhibits enhanced adsorption capacity for phosphorus removal. This provides the potential for reuse in fertilizer industries (Zheng *et al*. 2019). A study on phosphate adsorption by incorporating metals and metal oxides into biochar feedstocks was conducted. Among the modifiers, Mg has received considerable attention, given its ability to notably enhance the PO_4^-P adsorption capacity of biochar through robust divalent cation bridging between Mg and P (Wang *et al*. 2020).

Yao *et al.* discovered that biochar derived from magnesium-enriched tomato leaves could adsorb substantial amounts of PO_4^-P, surpassing 100 mg PO_4^-P/g (Yao *et al*. 2013). According to Zeng *et al.*, biochar obtained from *Thalia dealbata*, a phytoremediation plant, demonstrated notably greater adsorption capacity for PO_4^-P (approximately 2.54–4.96 mg P/g) than biochars derived from three other phytoremediation plants, which exhibited minimal P removal efficiencies. They suggested that the elevated magnesium content in *T. dealbata* relative to that in the other plants could be the primary factor behind this disparity (Zeng *et al*. 2013). Another study by Novais *et al.* evaluated doped Al to enhance the capability of biochar to remove P. For example, Al-doped biochars from poultry manure and sugarcane straw showed 701.65 and 758.96 mg P/g respectively, compared to unmodified biochar (356.04 and 468.84 mg P/g respectively) (Novais *et al*. 2018).

3.4. Enhancement of anaerobic digestion of sewage sludge

In addition to being used as an adsorbent for N and P, biochar can also be used as an additive in the AD of sewage sludge originating from municipal wastewater treatment. AD is an inexpensive and environmentally friendly method for reducing chemical and biochemical oxygen demand. It can be utilized as part of the municipal wastewater treatment process or to manage the concentrated waste-activated sludge fraction after conventional aerobic treatment. This technology degrades organic matter to produce methane-enriched biogas through complex biological processes mediated by microorganisms. Microbial processes, inhibitory substances and byproducts, recalcitrant substrates, and low growth rates can limit AD. This can be mitigated by adding biochar to the process to increase the degradation rate or methane production via direct interspecies electron transfer, microbial immobilization and growth support, adsorption of inhibitory compounds, and retention of microbial biomass.

Biochar used for AD augmentation is primarily derived from agricultural residues; for example, Shen *et al.* produced biochar from pinewood and white oak, and reported a 7.6–10.3% increase in methane yield for mesophilic AD. Under thermophilic conditions, pinewood biochar delivered a 2.68% improvement over the control (Shen *et al*. 2016). Zhang *et al.* evaluated nine types of biochar (3 feedstocks × 3 pyrolysis temperatures), corn straw, coconut shell, and sewage sludge pyrolyzed at 400, 500 and 600 °C for 120 min on AD of sewage sludge. Corn straw biochar at 600 °C performed best with 86.1% increase of methane yield above the control. It performed 13 and 35% better than sewage sludge and coconut shell derived biochar, respectively (Zhang *et al*. 2019c). Likewise, Shen *et al.* reported a 37% increase in methane yield during a thermophilic AD with biochar produced from corn stover (Shen *et al*. 2017). Shen *et al.* pyrolyzed corn stover at 300, 500, and 700 °C for 2 h and reported that methane production was increased by 46.9, 28.9, and 11.2%, respectively, although the lag phase was prolonged (Shen *et al*. 2021).

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Biochar can also be produced from manure or sludge. For example, Jang *et al.* produced biochar from dairy manure, and with a 10 g/L addition to AD, reported a 24.9% increase in methane production (Jang *et al.* 2018). Liu *et al.* pyrolyzed sludge at 400 and 600 °C and found that methane production was improved by up to 54.5% with the lower temperature biochar and 39.7% for the latter. This beneficial effect was only reported at a higher dose of biochar addition, with lower concentrations being similar in yield to the control, or below it (Liu *et al.* 2022b).

To improve the efficacy of biochar, surface chemical modifications can be performed to incorporate different properties. Zhang *et al.* synthesized corn stover biochar pyrolyzed at 550 °C for 120 min and modified with nZVI to improve the AD of sewage sludge. An increase in methane yield by 115.4% with 2.5 g of modified biochar added was observed (39.1% increase with biochar alone (0.75g), 63.8% increase with nZVI biochar (0.75g)) (Zhang *et al.* 2019b). Zhang and Wang prepared iron–manganese–modified biochar from corn straw pyrolyzed at 600 °C and reported a 55.8% higher methane yield at a dose of 1.5 g per 300 ml of wastewater, with a volatile fatty acid degradation rate increase of 35.4% compared with the control. At a dose of 5 g per 300 ml of wastewater, the methane production was reduced by 26.4%, revealing an inhibitory effect of overdosing (Zhang & Wang 2020). Li *et al.* utilized manganese oxide to modify corn straw biochar pyrolyzed at 600 °C for 2 h. The modified biochar increased 121.1 L/kg volatile solids of methane with an optimum dosing of 2.36 g. The use of this modified biochar resulted in a cumulative increase in methane yield of up to 122% compared with the control (Li *et al.* 2019a).

4. LIFE CYCLE ASSESSMENT OF BIOCHAR USE IN WASTEWATER TREATMENT

The environmental and economic outcomes of biochar production and use are context dependent and vary significantly across regions due to differences in feedstock availability, production technologies, and end-use strategies. LCA can be used as a key tool to evaluate the environmental trade-offs associated with biochar systems (Roberts *et al.* 2010) (Figure 7).

Past LCA studies on biochar systems have identified the feedstock stage as the major contributor to overall environmental impacts. Thermal conversion processes (such as pyrolysis and gasification) and transportation are among the next main contributors. For example, in the production of Miscanthus-derived biochar, emissions were quantified at 93.0 kg CO_2 eq/t for feedstock cultivation, 23.3 kg CO_2 eq/t for pyrolysis, and 4.8 kg CO_2 eq/t for transportation (Tadele *et al.* 2019). Willow-derived biochar is often favored over that produced from pig manure, owing to its significantly lower energy requirements for feedstock drying. The high moisture content of pig manure requires substantial energy input during pretreatment, thereby

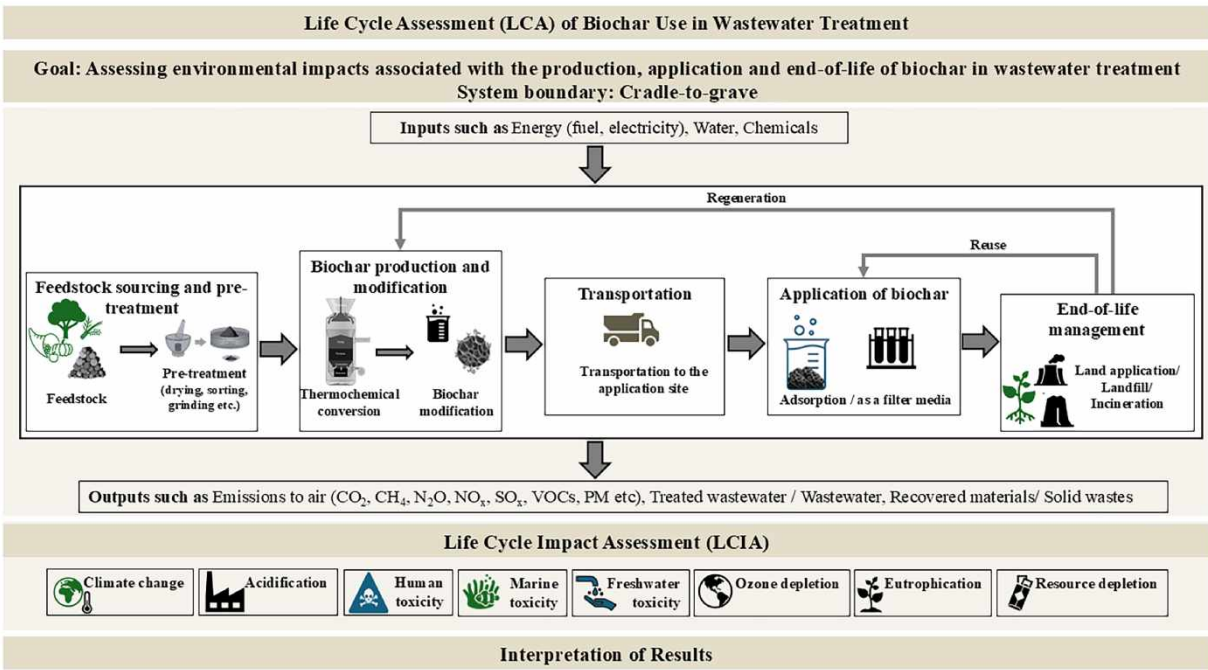


Figure 7 | LCA of biochar use in wastewater treatment.

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increasing the environmental impacts and reducing the overall sustainability of the biochar system (Rajabi Hamedani *et al.* 2019). The production of biochar is associated with the release of various gaseous pollutants, including CO_2, nitrogen oxides (NO_x), and particulate matter, contributing to its overall environmental footprint. A study assessing a fixed-bed pyrolysis system in China reported a greenhouse gas emission intensity of 1.55×10^{-2} kg CO_2eq/MJ (Yang *et al.* 2016). Recent research findings suggest that iron in biochar feedstocks may facilitate nitrogen fixation during pyrolysis, thereby reducing N_2O generation and release into the atmosphere (Cayuela *et al.* 2014; Lin *et al.* 2017). In addition, the biochar production process can generate a range of contaminants, including polycyclic aromatic hydrocarbons (PAHs), volatile organic compounds, heavy metals, and carbon-based nanoparticles, which may persist within the biochar matrix (Lian & Xing 2017). These elements can be readily leached from the biochar matrix into surrounding water bodies, for example, potassium (K, up to 64%), calcium (Ca, up to 40%), and phosphorus (P, up to 75%). Given that relatively large quantities of biochar are often applied in water and wastewater treatment systems, there is a risk of releasing high concentrations of these and potentially toxic co-occurring constituents into the environment (Limwikran *et al.* 2018). Hence, there is a clear need to develop application-specific regulatory standards for potentially toxic elements in biochar, especially for use in contaminated water and wastewater treatment systems. Despite increasing interest in biochar-based remediation, LCA studies targeting wastewater applications remain limited, highlighting a critical research gap in understanding the broader environmental implications and ensuring the safe, sustainable deployment of biochar technologies at scale.

Thompson *et al.* (2016) reported that wood-derived biochar effectively removes micropollutants in WWTFs and improves overall environmental performance. Their LCA revealed that biochar serves not only as an effective adsorbent but also as a net carbon sink, sequestering approximately 6.5 Gg of CO_2eq/yr. This amount exceeds the 5.0 Gg of CO_2 emissions usually linked to 12.5 million gallons per day (mgd) WWTF that uses organic and nutrient removal (Thompson *et al.* 2016). Alhashimi & Aktas (2017) conducted an LCA to evaluate the comparative environmental and economic performance of biochar and AC for removing heavy metals from water. Their analysis found that biochar is a cost-effective option for adsorbing chromium and zinc, with results significant at the 95% confidence level. For lead and copper, the adsorption costs for both materials were similar although the outcomes were influenced by biochar type and market variability. Importantly, when tailored for specific contaminants, biochar can match the adsorption performance of AC, while achieving lower environmental impacts (Alhashimi & Aktas 2017).

Huang *et al.* reported that sewage sludge-derived biochar, synthesized using 300 W microwave pyrolysis, removed 83% of trichloroethylene at pH 3.1 through heterogeneous Fenton oxidation. The catalytic reactivity of the biochar was maintained over multiple cycles, demonstrating its potential for long-term solution as a permeable reactive barrier in groundwater remediation. Although trace metals including chromium, copper, nickel, lead, and zinc were detected in the biochar leachate, their concentrations remained below regulatory thresholds for nonpotable water (Huang *et al.* 2020). Góngora *et al.* evaluated the environmental performance of biochar-amended constructed wetlands compared with aerated treatment units in their LCA of onsite wastewater systems. The study concluded that adding biochar to constructed wetlands improved environmental results, while changes in biochar biodegradation had only a minor effect on the overall impact scores. These findings suggest that biochar-enhanced wetlands provide strong and sustainable environmental benefits (Gongora *et al.* 2021).

Choudhary and Philip conducted LCA of empty palm bunch (PEB) biochar to evaluate its efficacy in removing pharmaceutical and personal care products from secondary wastewater effluents. The biochar achieved over 75% removal efficiency in both controlled multipollutant setups and real wastewater systems. Furthermore, the total environmental impact of PEB biochar was approximately 20-fold lower than that of commercial AC (Choudhary & Philip 2022). A combined LCA and TEA conducted for a tertiary wastewater treatment system using a biochar-integrated reactive filtration (RF) strategy was evaluated in two configurations: biochar-only filtration (BC-RF) and iron-ozone catalytic oxidation coupled with biochar filtration (Fe–CatOx–BC–RF). Both systems reached high treatment efficiency, removing 90–99% of total phosphorus and allowing for potential resource recovery through phosphorus adsorption onto biochar. Additionally, the system achieved over 90% removal of micropollutants (Taslakyan *et al.* 2023).

An LCA by Zakrisson *et al.* compared the environmental performance of biochar filters to conventional sand filters for onsite household wastewater treatment. Biochar filters had similar eutrophication impacts and showed slight variations in acidification effects based on site sensitivity. Climate impacts ranged from higher to lower than sand filters, influenced by energy source, feedstock, and end-of-life use. Notably, willow chip biochar and syngas-heated pyrolysis systems exhibited

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the lowest climate impacts. Scenarios using renewable energy and biochar reuse in landscaping significantly reduced environmental impacts (Zakrisson *et al*. 2024).

Biochar can effectively adsorb pollutants in wastewater treatment; however, it often reaches its maximum capacity over time, requiring periodic replacement or regeneration to sustain removal performance and comply with treatment standards. Therefore, it is essential to evaluate biochar's effectiveness not only for short-term applications but also under long-term operational conditions, where saturation, aging, operational conditions (pH, temperature, and competing ions), and fouling may reduce its functionality. Once saturated, biochar requires regeneration (e.g., chemical desorption, thermal reactivation) or replacement to maintain compliance with discharge standards. A comprehensive understanding of biochar's removal dynamics across its service lifespan, along with insights from LCA, is key to assessing its environmental sustainability, economic viability, and practical deployment in full-scale wastewater treatment systems.

5. TECHNO-ECONOMIC ASSESSMENT OF BIOCHAR USE IN WASTEWATER TREATMENT

TEA is important for determining the feasibility of adsorbents in wastewater treatment. Adsorbent cost can be evaluated using many parameters, including capital cost, operating cost, and financial and regulatory costs (Figure 8). Adsorbents costing less than USD 1/mol (1 g of adsorbent for the removal of 1 mole of a contaminant in the aqueous phase) are generally considered inexpensive, whereas those above USD 200/mol are considered costly (Ighalo *et al*. 2022). Biochar production is inherently influenced by regional feedstock availability and geographic energy economics, both of which are key factors in overall production viability and cost. As highlighted by Pratt and Moran, electricity prices in North America are comparatively higher than those in Europe, whereas some Middle Eastern countries benefit from exceptionally low electricity costs, offering a more favorable economic environment for large-scale biochar production and use (Pratt & Moran 2010). Biochar is a cost-effective alternative to AC. While AC typically costs between USD 1,100–1,700/ton, biochar prices range significantly lower, between USD 350 and 1,200/ton (Ganguly & Wright 2022).

To address these economic challenges, novel production technologies have recently been developed to enhance energy efficiency and reduce the economic barriers associated with conventional biochar synthesis (Pratt & Moran 2010). Recent studies have revealed that pristine biochar is not sufficiently reactive and does not possess sufficient functional

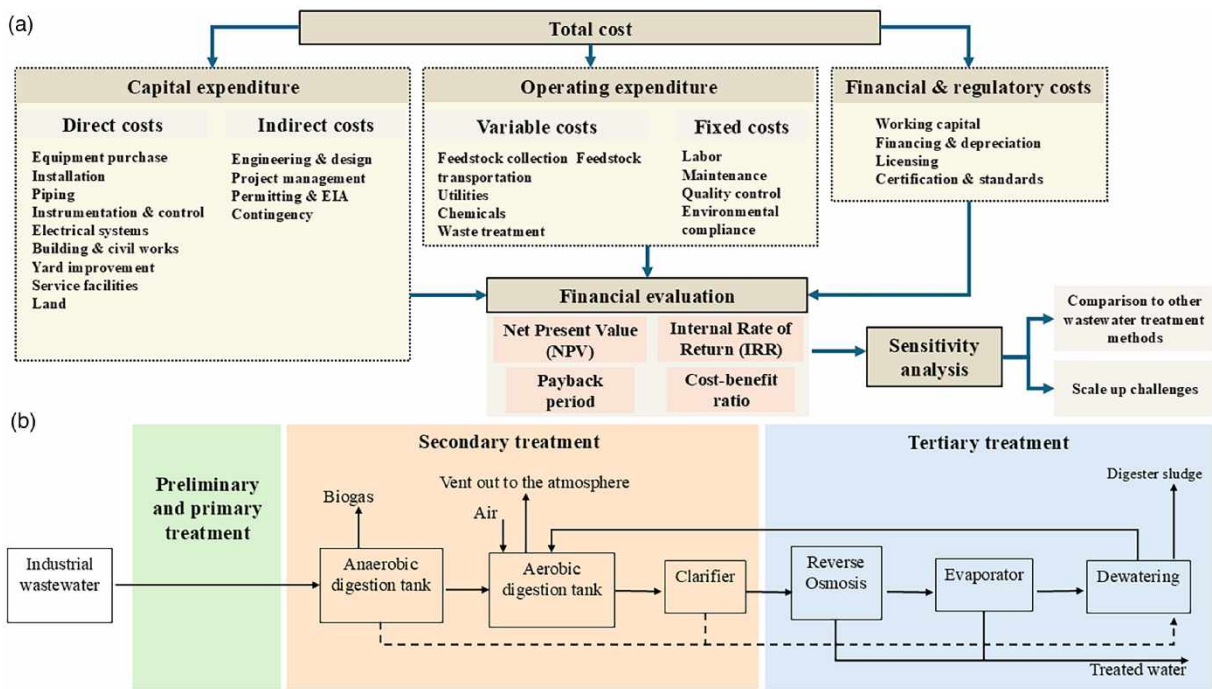


Figure 8 | (a) Techno-economic assessment cost categories and (b) industrial wastewater treatment process for identifying possible cost venues (modified from the study by Ganguly & Wright (2022)).

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specificity to be effective in neutralizing a broad spectrum of environmental pollutants, particularly in complex wastewater matrices. Such a constraint has generated an increasing need for engineered or modified biochars with enhanced surface properties, catalytic activity, and adsorption capacities tailored to target emerging pollutants (Vikrant *et al*. 2018). For example, magnesium-impregnated cow dung–derived biochar developed by Chen *et al.* exhibited reduced P leaching from the soil by 89.25%, with a relatively low production cost of USD 1.20 per ton (excluding feedstock and labour costs). This highlights the potential of cost-effective surface modification strategies to significantly enhance the functional performance of biochar in wastewater treatment applications (Chen *et al*. 2018b). Several studies have attempted to estimate the overall operating costs associated with the application of biochar for wastewater treatment (Table 5). For example, amine-modified biochar derived from chicken manure was tested for the adsorption of dimethyl sulfide, reporting an adsorption capacity of 1.14 mg/g. The total production cost, including transportation, chemical reagents, and electrical energy, was reported at USD 2.60/kg (Nguyen & Lee 2015). Praveen *et al*. (2021) reported that the adsorption capacities for the removal of Basic Red 09 from wastewater using coconut shell, groundnut shell, and rice husk biochars were 10, 46.3, and 44 mg/g, respectively. Moreover, the economic analysis showed that the approximate costs of removing 1 g of Basic Red 09 were USD 0.01, 0.06, and 0.01 for groundnut shell, coconut shell, and rice husk, respectively. These adsorbents were regarded as economically favorable because they could be subsequently reused for soil applications (Praveen *et al*. 2021).

The sustainability implications of engineered nanomaterials for treating contaminated water and wastewater were reported an estimated production costs ranging from USD 0.03–1.21 per gram for TiO_2 nanoparticles and USD 0.05–0.10 per gram for nZVI (Adeleye *et al*. 2016). A TEA by Taslakyan *et al.* (2023) demonstrated that a biochar-integrated advanced treatment system including phosphorus removal, micropollutant degradation, and disinfection can achieve carbon-neutral wastewater treatment at competitive costs. For a 1130 m^3/day facility, the mean treatment cost was USD $0.18 \pm 0.01/m^3$. At a larger scale of 3780 m^3/day, the cost decreased to USD $0.08 \pm 0.01/m^3$, with biochar contributing only USD $0.03/m^3$ (Taslakyan *et al.*2023).

Most of the current evidence for the performance of biochar in wastewater treatment is based on laboratory-scale experiments, with minimal validation under actual operating conditions. In order for the entire potential of biochar to be utilized

Table 5 | The cost for different biochar types for wastewater treatment

Adsorbent (Type of biochar)	Target contaminant	Removal efficiency (%)	Cost	Reference
Magnesium-impregnated cow dung	Phosphorus	89.25	1.2 USD/ton of biochar	Chen *et al*. (2018b)
Amine-modified chicken manure	Dimethyl sulfide	0.11	2.6 USD/kg of biochar	Nguyen & Lee (2015)
Coconut shell	Basic Red 09	1	~0.06 USD/g of Basic red 09 removal	Praveen *et al*. (2021)
Groundnut shell	Basic Red 09	4.63	~0.01 USD/g of Basic red 09 removal	Praveen *et al*. (2021)
Rice husk	Basic Red 09	4.4	~0.01 USD/g of Basic red 09 removal	Praveen *et al*. (2021)
Gasification waste-derived activated carbon	Malachite green	22.6	0.23 USD/kg of biochar	Ahmad *et al*. (2020)
Distillers grains hydrochar	Methylene blue	34.03	~30 USD/ton of biochar	Xu *et al*. (2021)
Distillers grains hydrochar	Phosphate	9.69	~30 USD/ton of biochar	Xu *et al*. (2021)
H_3PO_4 activated carbon from cherry kernels	Pb (II)	94.48	41.92 USD/kg of biochar	Vukelic *et al*. (2018)
H_3PO_4 activated carbon from cherry kernels	Cd (II)	92.42	41.92 USD/kg of biochar	Vukelic *et al*. (2018)
H_3PO_4 activated carbon from cherry kernels	Ni (II)	66.22	41.92 USD/kg of biochar	Vukelic *et al*. (2018)
Rice straw	Zn (II)	3.57	2.579 USD/kg of biochar	Sakhiya *et al*. (2021)

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in wastewater treatment, there is an urgent need for field-scale or operating system-based studies that account for the variability and complexity of actual treatment conditions. However, existing evidence indicates that engineered biochar is an efficient and sustainable alternative to AC, offering comparable performance but with lower environmental footprints. These features position biochar as a durable and scalable solution for advanced and decentralized wastewater treatment processes.

6. CONCLUSIONS AND FUTURE PERSPECTIVES

Biochar is a potential solution for wastewater treatment and is well known for its efficiency and cost-effectiveness as an adsorbent. Biochar is derived from various biomass sources, such as agricultural residues, forestry residues, organic municipal waste, manure, and sewage sludge. This study summarizes biochar production technologies and analyzes the stages of feedstock pretreatment, thermal conversion, and post-treatment methodologies. Furthermore, the mechanisms underlying the capability of biochar to adsorb contaminants within wastewater matrices were evaluated. The major observations of this review are as follows: (1) Material performance and modification: The adsorption efficacy of biochar is highly dependent on its feedstock origin, pyrolysis conditions, and surface modification strategies. Enhancing the specific surface area, catalytic activity, and functional group density remains key to improving contaminant removal capacities. (2) Mechanistic diversity: Biochar removes pollutants through mechanisms such as ion exchange, surface complexation, electrostatic attraction, and precipitation. These contribute to the effective removal of heavy metals, organic compounds, nutrients, and ECs from wastewater matrices. (3) Scale-up and practical deployment: Although laboratory-scale studies have demonstrated considerable potential, real-world applications require further optimization to account for the complexity of actual treatment systems. Factors such as variable wastewater composition, competing ions, dissolved organic matter, microbial interactions, fluctuating hydraulic conditions, site-specific constraints, and economic feasibility can all influence treatment performance. Therefore, future research should prioritize long-term pilot-scale and field-scale studies under real-world conditions.

Despite its promise, several challenges must be addressed. Biochar's stability decreases with prolonged exposure to aqueous solutions, particularly with respect to the potential leaching of heavy metals from certain biochar variants. The carbonaceous structure of biochar undergoes alterations after multiple use cycles, particularly when it functions as a catalyst (Huang *et al*. 2019). Biochars, particularly those sourced from sewage sludge, may contain elevated levels of heavy metals and pathogens. These metals have the potential to leach during application, leading to further contamination with heavy metals (Wang & Wang 2019). The regeneration and recovery of biochar after use is another significant consideration. Because adsorption facilitates the adherence of contaminants to the biochar surface, it is crucial to convert bound toxic pollutants into nontoxic states for efficient management (Chen *et al*. 2011; Li *et al*. 2018b). For example, it is necessary to incorporate application-specific contaminant release pathways by combining total-content thresholds for contaminants such as heavy metals and PAHs with standardized leaching tests under realistic operating and regeneration conditions. Moreover, these should be linked to toxicity-related impact categories and end-of-life scenarios. Additional challenges arise, particularly in developing countries that lack adequate industrial infrastructure. Considering the various environmental issues associated with the practical use of biochar, extensive research is required to address these concerns. It is imperative to identify solutions to potential environmental problems and provide feasible research directions to expand the application of biochar in such regions.

The challenges surrounding cost implications and scalability emphasize the need for a balance between optimization efforts and practical applicability. Although feedstocks are abundant, they require extensive pretreatment (e.g., drying, grinding, and cleaning), and functionalization processes can increase production costs relative to conventional AC. For example, adsorbents costing less than USD 1/mol are generally considered very cheap, whereas those above USD 200/mol are considered very expensive (Ighalo *et al*. 2022). Therefore, future research should aim to achieve a trade-off between optimizing the production process and enhancing the applicability of biochar to minimize costs.

Most studies on biochar sorption have focused on single contaminants in aqueous solutions. However, most real-world wastewaters contain complex mixtures of pollutants, which can result in competitive adsorption and reduced overall efficiency. Limited empirical data are available on the sorption of co-contaminants. This highlights the need for simultaneous sorption models to better understand these interactions. Reports on biochar sorption should provide detailed information on sorbent properties and conditions to facilitate future research in this area.

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Novel methodologies such as magnetic biochar and biochar–biofilm amalgamation present noteworthy prospects for enhancing wastewater treatment efficiency. Nevertheless, a strong focus on environmental impact assessments with techno-economic evaluations, and policy frameworks is required to guide sustainable implementation. Field-scale pilot projects are needed to validate laboratory findings under realistic conditions. Multistakeholder collaboration among academia, government, and industry will be key to establishing standardized protocols and encouraging regulatory acceptance. Incentive schemes that link biochar use with circular economy strategies, such as sludge valorization and carbon sequestration credits, can further drive uptake.

In conclusion, although the path toward realizing the full potential of biochar in wastewater treatment may be challenging, its inherent versatility and efficacy as an adsorbent can potentially promote sustainable water management practices.

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DECLARATION OF GENERATIVE AI AND AI-ASSISTED TECHNOLOGIES IN THE WRITING PROCESS

During the preparation of this work the authors used ChatGPT (GPT-4.0) in order to correct grammar and check English language. After using this tool, the authors reviewed and edited the content as needed and take full responsibility for the content of the publication.

CONFLICT OF INTEREST

The authors declare no conflicts of interest.

DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

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