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Review

Biochar-based fixed filter columns for water treatment: A comprehensive review

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ABSTRACT

Biochar used in fixed filter columns (BFCs) has garnered significant attention for its capabilities in material immobilization and recovery, filtration mechanisms, and potential for scale-up, surpassing the limitations of batch experiments. This review examines the efficacy of biochar in BFCs, either as the primary filtering material or in combination with other media, across various wastewater treatment scenarios. BFCs show high treatment efficiency, with an average COD removal of 80 % ±15.3 % (95 % confidence interval: 72 %, 86 %). Nutrient removal varies, with nitrogen-ammonium and phosphorus-phosphate removal averaging 71 ± 17.1 % (60 %, 80 %) and 57 % ± 25.6 % (41 %, 74 %), respectively. Pathogen reduction is notable, averaging 2.4 ± 1.12 log10 units (1.9, 2.9). Biochemical characteristics, pollutant concentrations, and operational conditions, including hydraulic loading rate and retention time, are critical to treatment efficiency. The pyrolysis temperature (typically 300 to 800 °C) and duration (1.0 to 4.0 h) influence biochar's specific surface area (SSA), with higher temperatures generally increasing SSA. This review supports the biochar application in wastewater treatment and guides the design and operation of BFCs, bridging laboratory research and field applications. Further investigation is needed into biochar reuse as a fertilizer or energy source, along with research on BFC models under real-world conditions to fully assess their efficacy, service life, and costs for practical implementation.

1. Introduction

Many regions face water scarcity, with projections indicating worsening water stress due to climate change, population growth, industrialization, and urbanization. Consequently, there is an urgent need for large-scale interdisciplinary research collaborations aimed at delving into economically viable and sustainable treatment solutions. Within this context, biochar derived from the anaerobic pyrolysis treatment of organic substrates, including agricultural residues, forestry by-products, and municipal solid waste, garners widespread attention owing to its pronounced stability, enhanced surface characteristics, robust pore structure, and abundant functional moieties (La et al., 2019). Moreover, current research indicates the efficacy of biochar in the removal of pollutants such as organic compounds, nutrients, heavy metals, and pathogens from wastewater (Kaetzl et al., 2019; Li et al., 2022; Nguyen et al., 2023).

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Biochar, with its versatile roles as a structural substrate, filtration medium, and catalyst, is crucial in enhancing the efficacy of primary and tertiary treatment modalities in water management (Eshetu et al., 2015; Majumder and Das, 2022). However, when used alone or in batch systems, biochar suffers from limited regeneration and potential contaminant leaching. Moreover, biochar-based batch treatments rely solely on physical-chemical adsorption, lacking biological processes. Due to their limitations in scalability, efficiency, and continuity, batch operations are rarely applied in industrial settings (Mazur et al., 2018). To address this, biochar-based fixed filter columns (BFCs) have been deployed, achieving effective pollutant removal from surface water, agricultural water, and sewage (Dalahmeh et al., 2019b; Deepa et al., 2022; Gao and Wan, 2023; Jayabalakrishnan et al., 2023). Mounting evidence suggests that integrating biochar with sand, gravel, and wood chips brings promising prospects for various water treatment applications, including column filtration systems, bioretention facilities, and constructed wetlands (Boni et al., 2021; Pluer et al., 2016; Xiong et al., 2022). When using biochar as a filtration material, it serves as a carrier for microorganisms to attach to, allowing them to develop and break down pollutants in wastewater (Bolan et al., 2023). Therefore, as a medium in biofiltration systems, biochar combines microbial degradation of pollutants with its adsorbent properties, rendering it an attractive option for biologically mediated water treatment.

Recent reviews have documented the efficacy of various biofiltration methods in removing pathogens, volatile organic compounds, and heavy metals (Maurya et al., 2020; Pachaiappan et al., 2022). Notably, biochar's role as a buffering agent has gained attention for enhancing pollutant removal (Biswal et al., 2022; Quispe et al., 2022; Shahraki and Mao, 2022). While previous studies have explored specific aspects of biochar-based filtration systems, there is a notable lack of comprehensive reviews that provide a detailed synthesis of BFC applications across a wide range of pollutants and wastewater types. For example, while Shahraki and Mao (2022) conducted a focused review on biochar in on-site wastewater treatment systems, highlighting nutrient removal and variability influenced by biochar's source and pyrolysis conditions, their analysis was constrained by a narrow keyword selection that omitted broader biofiltration applications. Similarly, Quispe et al. (2022) investigated biochar's role in column filtration for greywater treatment, providing valuable insights into system configurations and operational mechanisms, albeit relying heavily on data from master's theses and lower-impact journal articles. Biswal et al. (2022) examined biochar in stormwater management through bioretention systems, though their findings, predominantly from batch experiments, might not accurately represent the dynamics of continuous systems (Table S1).

This review seeks to bridge the existing knowledge gaps by providing a comprehensive analysis of the effectiveness of BFCs in removing a wide array of pollutants across different water types and configurations. By synthesizing current research and offering insights into the factors influencing treatment efficacy, this review aims to guide future research and practical applications of BFCs in sustainable water treatment systems. To increase confidence in the highly variable treatment efficacies of BFCs, 95 % confidence intervals (CI)—hereafter referred to as (lower bound, upper bound)—were developed using the bootstrap technique for average values. Additionally, this work explores barriers to adoption and offers future insights for developing BFC in water treatment.

2. Methods

2.1. Methodological overview

This work involves qualitative and quantitative assessments. The selected studies were reviewed to identify and discuss key factors, operational aspects, performance metrics, and gaps. Where applicable, data from the literature were summarized to provide a quantitative overview of BFC performance, including removal efficiencies, operation conditions, and relationships between variables. The primary goal is to provide a comprehensive summary with representative values and confidence ranges.

2.2. Statistical synthesis and analysis

2.2.1. Data synthesis

Operational conditions and biochar characteristics, including residence time, biomass source, production temperature, surface area, hydraulic loading rate (HLR), column diameter (D), column height (H), and removal efficiencies, were derived from the selected studies. Mean values from appropriate experiments were collected as representative samples. When studies reported a range of values, the average of the minimum and maximum values was used to calculate a mean value.

2.2.2. CI estimation

95 % CIs were developed for key performance metrics using the bootstrap technique to understand the variability in treatment efficiencies better. This statistical method was applied to the average values reported in the literature, providing a more nuanced interpretation of the data. Given the skewed nature and high variability of the data, the Bias-Corrected and Accelerated (BCa) bootstrap method ($n = 5000$) was used to construct the CIs.

2.2.3. Software

All statistical analyses and visualizations were performed using the R programming language, utilizing the “boot”, “psych”, “ggplot2”, and “metafor” packages.

3. Overview of biochar and fixed filter columns

3.1. Biochar and configuration of BFCs

Biochar is primarily produced through thermochemical conversion techniques such as pyrolysis, gasification, and torrefaction, with pyrolysis being the most prevalent method (Iwuozor et al., 2023; Yaashikaa et al., 2020). This process involves decomposing biomass in an oxygen-limited environment at temperatures between 300 °C and 850 °C. The properties of the resulting biochar—such as surface area, functional groups, cation exchange capacity, and pH—are influenced by factors including biomass source, temperature, residence time, gas flow rate, and additives (Amalina et al., 2022; Shahraki and Mao, 2022).

Biochar's efficacy in various applications—from soil enhancement to energy production—is well-documented (Lehmann and Joseph, 2015). Its role in wastewater treatment has expanded significantly, leveraging its capacity to adsorb various contaminants, including organic dyes, heavy metals, and pathogens (Jha et al., 2023; Shahraki and Mao, 2022). Further enhancement of biochar's properties can be achieved through physical and chemical modifications both before and after pyrolysis. These modifications can increase attributes such as surface area and pore volume, improving their effectiveness in pollutant removal (Díaz et al., 2024; Huang et al., 2022; Huang et al., 2020; Jayabalakrishnan et al., 2023; Lingamdinne et al., 2022).

BFC systems are utilized to remove organic matter, nutrients, pathogens, and heavy metals from various types of wastewater. Laboratory-scale BFCs are predominantly cylindrical, though rectangular configurations are also noted (Kranner et al., 2019; Singh et al., 2023; Xin et al., 2021). At the field-scale, most filtration systems are predominantly box-shaped (Kholoma et al., 2016; Majumder and Das, 2022; Pluer et al., 2016).

The typical configuration of BFCs includes several main components: influent, top gravel layer, filtration media, bottom gravel layer, and effluent. Water is primarily introduced at the top of these columns and collected at the bottom (Fig. 1). According to the results summa-

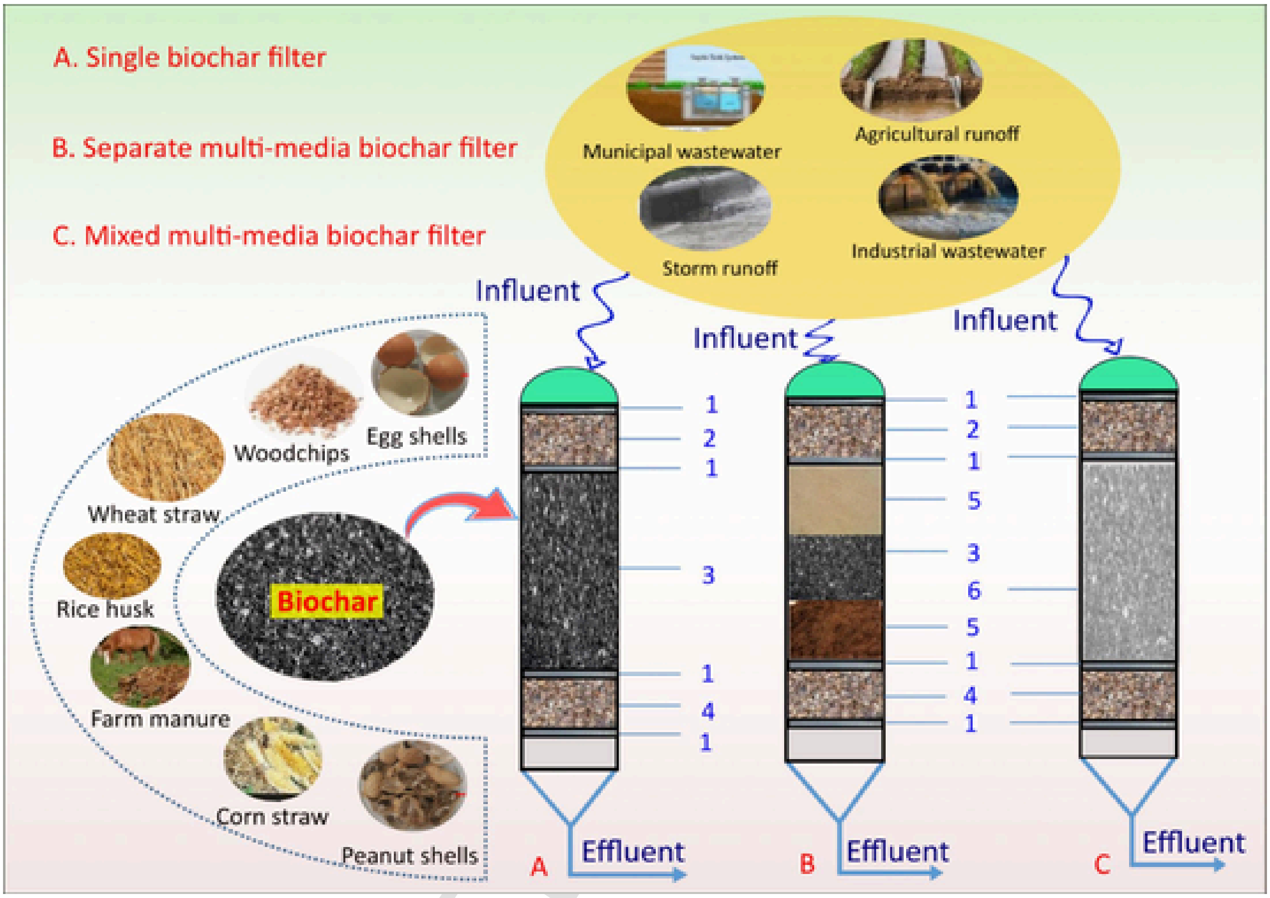

Fig. 1. Configuration and setup of the biochar filtration systems: (1) mesh, (2) top layer, (3) biochar, (4) bottom layer, (5) other filler materials, (6) mixture of biochar and other filler materials.

rized in Table 1, a minority of experiments utilize an upward flow regime (21.1 %), while others employ box-shaped filter tanks that primarily use a horizontal flow regime (15.8 %).

In standard configurations, BFCs feature a top gravel layer and a bottom drainage gravel layer. The top gravel layer promotes the uniform distribution of the influent, prevents biochar particle flotation, and reduces water evaporation. The bottom gravel layer facilitates the flow of effluent. To avoid the loss of finer media grains, mosquito nets and wire mesh are often installed between the gravel and biochar layers. Nevertheless, some studies have opted out of using two gravel layers (36.8 %); in these cases, a mesh layer at both ends of the column supports the filter media and prevents the washing of porous materials.

The specific configurations of BFCs in various studies are detailed in Table 1. Biochar is the primary filter material in the single biochar filter (SBF) system (50.0 %). In the multi-media biochar filter (MMBF), biochar is combined with other materials such as sand, soil, gravel, coconut coir, and wood chips. These materials can be layered in separate MMBF (52.6 %) or mixed MMBF (47.4 %). The mixtures use various ratios; for instance, biochar is mixed with sand at a weight ratio of 8.0 % (Hanandeh et al., 2017), and in the study by Boni et al. (2021), different volume ratios of biochar (7.0 %) and sand (3.0 %) were utilized. In a wood chips column, 10.0 % and 30.0 % volume of biochar were added to enhance the removal of Nitrogen-Nitrate (NO_3^--N) and Phosphorus-Phosphate (PO_4^{3-}-P) (Bock et al., 2015).

BFCs, primarily tested in controlled environments, have shown promise in water and wastewater treatment. Filter column sizes in laboratory setups typically range from 0.8 to 2.5 cm in diameter and 5.0 to 30.0 cm in height. More extensive testing in meso-scale laboratories involves filters ranging from 4.5 to 15.0 cm in diameter and 50.0 to 180.0 cm in height. However, only a tiny fraction of studies (10.5 %) have explored biochar applications in field-scale filter bed systems. This limited field-scale research underscores the need for further investigation into the viability of BFCs on a larger scale. Analysis of 26 samples revealed significant variability in column dimensions. The average height was 71.1 ± 52.3 cm (53.8, 93.2), ranging from 5.0 to 180.0 cm, resulting in a high coefficient of variation (CV) of 73.6 %. Similarly, the average diameter was 9.6 ± 11.7 cm (6.6, 17.7), with diameters ranging from 1.6 to 60.0 cm. The diameters exhibited high skewness (3.2) and kurtosis (10.6), indicating the presence of predominantly small columns with some significant outliers and a high CV of 122.4 %, suggesting substantial inconsistency in diameter sizes (Fig. 2 and Table S2).

3.2. *Removal mechanism of pollutants by BFCs*

In BFCs, pollutant removal occurs through various mechanisms detailed in Fig. 3, including:

- *Non-microbiological processes:* The effectiveness of biochar in pollutant removal is primarily attributed to various non-microbiological mechanisms, including adsorption, ion exchange, chemical reactions, and physical filtration. Adsorption processes on biochar surfaces involve multiple interactions: precipitation with minerals, complexation with oxygen-containing functional groups, coordination with π electrons, ligand and ion exchange, as well as van der Waals forces and hydrogen bonds (Gao et al.,

Table 1
Design parameters of BFCs for wastewater treatment.

Biochar filter configuration	Study scale	Dimension		Description			Flow regime	Reference
		h (cm)	d (cm)	Top	Main filter layers	Bottom		
SBF	Lab	55.0	5.0	Gravel (2.5 cm)	Biochar (50.0 cm)	Gravel (2.5 cm)	VDF	(Perez-Mercado et al., 2018)
	Lab	50.0	7.0	NA	Biochar (50.0 cm)	NA	VDF	(Li et al., 2016)
	Lab	180.0	5.0	Gravel (10.0 cm)	Biochar (50.0 cm)	Gravel (10 0.0 cm)	VUF	(Kaetzl et al., 2018)
	Lab	180.0	5.0	Quartz sand (5.0 cm)	Biochar (55.0 cm)	Quartz gravel (6.0. cm)	VDF	(Kaetzl et al., 2019)
	Lab	91.0	10.0	Gravel (3.0 cm)	Biochar (65.0 cm)	Gravel (5.0 cm, two layers)	VDF	(Chan et al., 2020)
	Lab	60.0	5.0	NA	Biochar (55.0 cm)	NA	VF and HF	(Dalahmeh et al., 2019a)
	Lab	180.0	5.0	Mosquito net	Biochar (54.0 cm)	Quartz-gravel (6.0 cm)	VDF	(Kaetzl et al., 2020)
	Meso-Scale Lab	150.0	15.0	PVC caps	Biochar (5.3 kg)	PVC caps	HF	(Lafdani et al., 2020)
	Lab	61.0	7.0	Pea gravel (7.6 cm)	Biochar (23.0 cm)	Pea gravel (7.6 cm)	VDF	(Reddy et al., 2014)
	Lab	0.0015[a]		NA	Biochar	NA	HF	(Xin et al., 2021)
	Lab	40.0	5.1	Plastic mesh	Biochar	Plastic mesh	VUF	(Paul and Hall, 2021)
	Lab	60.0	10.2	NA	Biochar	Wire mesh + PVC end cap	VDF	(Hunter and Deshusses, 2020)
	Lab	100.0	10.0	NA	Biochar	Plastic mesh	VDF	(Forbis-Stokes et al., 2018)
	Lab	70.0	7.5	Gravel (2 cm)	Biochar (60.0 cm)	Gravel (2.0 cm)	VDF	(Perez-Mercado et al., 2019)
	Lab	100.0	14.0	Gravel (2.5 cm)	Biochar (60.0 cm)	Gravel (2.5 cm)	VUF	(Eshetu et al., 2015)
	Pilot scale	180.0[b]60.0[d]	240.0[c]	NA	Biochar	Plastic liner + gravel (15 cm)	VDF	(Dalahmeh et al., 2016)
	Lab	5.0	1.6	Stainless steel meshes + sand	Biochar (1.0 g)	Stainless steel meshes + sand	VUF	(Gao and Wan, 2023)
	Lab	75.0	60.0	NA	Biochar	NA	VDF	(Jayabalakrishnan et al., 2023)
	Lab	5.1	1.6	Stainless steel meshes + sand	Biochar	Stainless steel meshes + sand	VUF	(Huang et al., 2022)
SMMBF	Field-scale	0.187[a]		Coarse soil (20.0 cm)	Biochar (15.0 cm)	Fine soil (30.0 cm)	VDF	(Majumder and Das, 2022)
	Field-scale	0.187[a]		Coarse soil (20.0 cm)	Biochar (15.0 cm)	Fine soil (30.0 cm)	VDF	(Blum et al., 2019)
	Lab	60.0	6.0	Sand (15.0 cm)	Biochar (30.0 cm)	Gravel stones (15.0 cm)	VDF	(Deepa et al., 2022)
	Lab	0.96[b] 1.42[d]	0.96[c]	Soil (20.0 cm) + Canna plants	Foam (10.0 cm) + Biochar (50.0 cm)	Coconut coir (20.0 cm)	HF	(Singh et al., 2023)
	Lab	14.5	4.5	Gravel (20.0 cm)	Biochar (20.0 cm) + sand (40.0 cm)	Gravel (25.0 cm)	VDF	(Naeem et al., 2019)
	Lab	18.0	1.0	NA	Agricultural soil + biochar	Quartz sand	VDF	(Boni et al., 2020)
	Lab	70.0	4.5	NA	Sand/gas concrete (30.0 cm) + biochar (20.0 cm)	Pebbles (8 cm)	VDF	(Kholoma et al., 2020)
	Lab	NA		Coarse gravel + fine gravel	Lime ($Ca(OH)_2$), sand (SiO_2), biochar	Sponge/ mesh	VDF	(Shabir et al., 2022)
	Field-scale	150.0[b] 80.0[d]	150.0[c]	Gravel (15.0 cm)	Sand/gas-concrete (30 cm) + biochar (20 cm)	A plastic garden mesh + gravel (15.0 cm)	HF	(Kholoma et al., 2016)
	Lab	50.0[b] 30.0[d]	40.0[c]	NA	Coarse sand + biochar + sand (7.6 cm for each).	Gravel (2.5 cm)	VDF	(Kranner et al., 2019)
MMMBF	Lab	21.0	11.0	Crushed stone + a perforated disk + plastic mesh fabric	The mix of sand and biochar (3 layers)	Plastic mesh fabric + a perforated disk + crushed stone	VDF	(Hanandeh et al., 2017)
	Lab	60.0	8.0	Glass beads (1.0 cm) and quartz sand (3.0 cm)	42.0 cm of biochar and quartz sand (7:100 and 3:100)	Quartz sand (5.0 cm) and glass beads (7.0 cm)	VDF	(Boni et al., 2021)
	Bench-scale and field-scale	70.0 [b] 150.0[d]	350.0[c]	Plastics 5-mil polyethylene	Woodchips/ biochar: 9/1	Plastics 5-mil polyethylene	HF	(Pluer et al., 2016)
	Lab	15.0	2.5	Glass wool	Sand/biochar: 7/3	Glass wool	VUF	(Afrooz et al., 2018)
	Lab	30.0	2.5	NA	Sand/biochar: 7/3	NA	VUF	(Valenca et al., 2021)
	Lab	61.0	10.0	NA	Woodchips/ biochar: 9/1	Wire mesh +50-μm filter paper	VDF	(Bock et al., 2015)

(*continued on next page*)

Table 1 (*continued*)

Biochar filter configuration	Study scale	Dimension		Description			Flow regime	Reference
		h (cm)	d (cm)	Top	Main filter layers	Bottom		
	Pilot-scale	50.0	10.0	NA	Woodchips/ biochar: 6.7/3.3	NA	VUF	(Ashoori et al., 2019)
	Lab	125.0	30.0	Plant + gravel	Sand/clay/ biochar: 8.8/0.8/0.4	Gravel	VDF	(Xiong et al., 2022)
	Lab	50.0	7.2	NA	Sand/biochar: 7/3	Coarse gravel (7.2 cm)	VDF	(Rahman et al., 2021)

SMMBF: Separate multi-media biochar filter, MMMBF: Mixed multi-media biochar filter, h: Height, d: Diameter, VDF: Vertical Downflow, VUF: Vertical Upflow, HF: Horizontal Flow, VF: Vertical Flow, NA: Not available.

[a] Volume (m^3).

[b] Length (cm).

[c] Width (cm).

[d] Depth (cm).

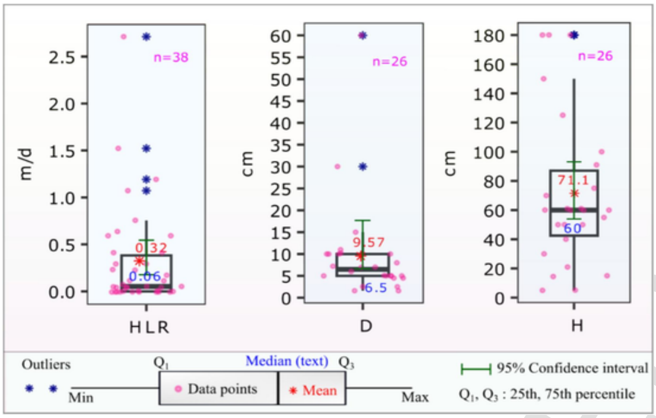

Fig. 2. Boxplot diagram illustrating the design and operation parameters of biochar-based columns. The boxplots display data variability, including the mean (red asterisk and text), median (blue asterisk and text), interquartile range (Q1-Q3), outliers, and the 95 % confidence interval range (green line).

2019; Jiang et al., 2018; Li et al., 2019). Besides, it has been reported that functional groups like alcohols, carbonyls, and carboxylates impart a negative charge to biochar, enhancing its cation exchange capacity (Li et al., 2024; Uras et al., 2012). This feature, combined with biochar's microporous structure and a diversity of organic functional groups, significantly augments its adsorption capacity and ion exchange abilities, thus facilitating the effective removal of pollutants (Yue et al., 2017). Additionally, certain pollutants can undergo chemical reactions with functional groups on the biochar surface, leading to immobilization or transforming into less harmful compounds (Xu et al., 2019). The physical structure of biochar also plays a critical role, as its pore spaces provide sites for the physical trapping of larger particles or contaminants, effectively preventing their migration through the filter media matrix (Perez-Mercado et al., 2019).

- *Microbiological processes:* BFCs utilize not only physical and chemical removal mechanisms but also biological processes such as biodegradation and microbial transformation (Spahr et al., 2019; Ulrich et al., 2017). Biochar's large surface area and porous structure make it an effective carrier for microorganisms, enhancing microbial growth and diversity (Bolan et al., 2023). Pollutants adsorbed on the surface of biochar are degraded by microorganisms residing in the biofilm. Adding biochar to sand increased biofilm formation, leading to greater microbial diversity and enhanced organic matter degradation in biochar filters compared to sand filters (Afrooz and Boehm, 2016; Dalahmeh et al., 2018). Frankel et al. (2016) reported that biotic biochar removed naphthenic acids more effectively (72 %) than sterile biochar (22–28 %) or microbes alone. Microorganisms on biochar utilize organic compounds in wastewater as nutrients, converting them into water, biomass, and CO_2 (Pachaiappan et al., 2022). Biochar also acts as an electron shuttle, enhancing the biodegradation of organic contaminants (Mukherjee et al., 2022). Combining biochar with biofilm improves metal sorption, with microbial processes like biosorption and biotransformation converting heavy metals into inactive forms (Verma et al., 2021). Additionally, biofilm formation on biochar enhances pathogen removal through electrostatic attraction and reduces pore size, facilitating filtration and adsorption (Enaime et al., 2020).

4. Applications of BFCs

4.1. *Wastewater types*

Table 2 illustrates how BFCs have been applied across different water types at various scales. The scope of the study includes a broad spectrum of water categories, each with unique characteristics and treatment requirements. A significant portion of the research—over one-third—focuses on municipal wastewater. Municipal natural and synthetic sewage constitutes about 15 % of the studies, underscoring the versatility of biochar systems in handling typical household waste. An additional 20 % of the research further investigates municipal wastewater. This indicates the potential of biochar in treating wastewater in diverse urban waste conditions.

The use of BFCs extends to treating industrial wastewater, including challenging effluents from processes like textile dyeing and olive milling, which account for another 20 % of the studies. Approximately 15 % of the research explores the potential of biochar in agricultural and livestock wastewaters, such as runoff from dairy and vegetable research farms, emphasizing its ability to manage nutrient and organic loads found in agricultural runoff. Another 15 % of the studies investigate stormwater treatment in urban and natural environments. These studies assess the efficacy of biochar columns in managing runoff from sources like urban areas and forest clear-cuts, highlighting their role in mitigating pollution from non-point sources.

Special categories of wastewater, such as synthetic solutions containing arsenic and tannery wastewater, demonstrate the capability of biochar to treat hazardous and specialized contaminants. Additionally, drinking water treatment using BFCs is a significant area of focus, with studies testing their effectiveness in removing impurities and enhancing water quality to meet health standards. For example, Chan et al. (2020) found that biochar made from a mix of spruce, pine, and fir woodchips effectively removes organic carbon from creek water, achieving removal rates of 100 % for phenanthrene and anthracene, 37–97 % for atrazine, and 49–93 % for naphthalene. This removal efficiency was slightly lower than the fluoride removal efficiency for groundwater

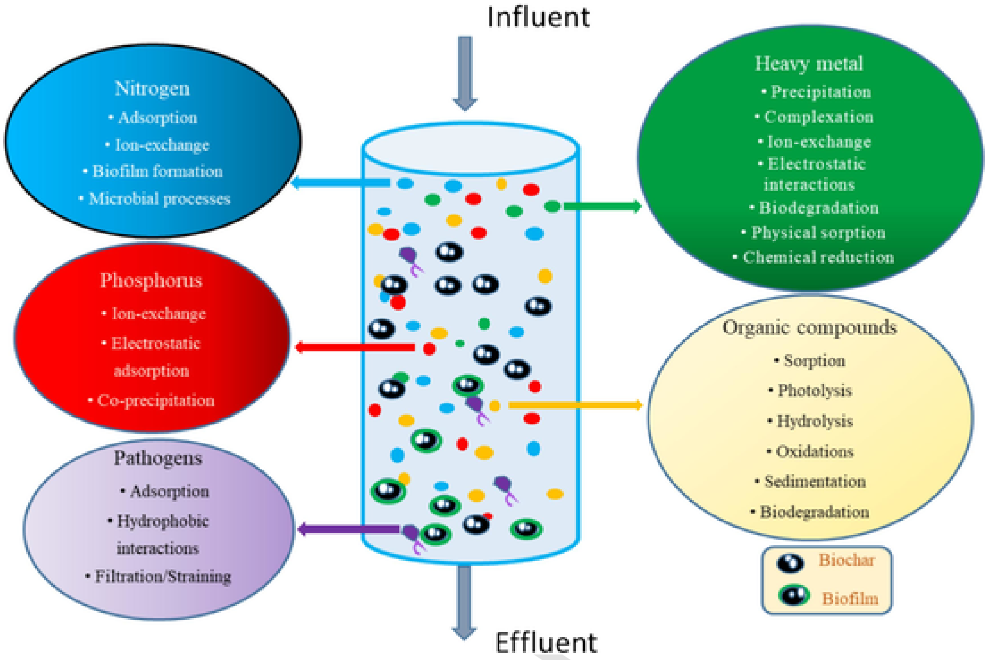

Fig. 3. Diagram illustrating the main removal mechanisms for different pollutants in biochar-based filter columns.

samples (95 ± 0.79 %) with a filter material layer of clay, sawdust, and iron oxide-biochar (Mandoreba et al., 2021).

The investigations also extend to wastewater from anaerobic processes, such as digests from human waste and other anaerobic digestion effluents, demonstrating the capability of biochar in polishing these streams. Different water types, including grey water and surface water, are also explored, showcasing the extensive applicability of biochar technology.

4.2. *Organic matter*

The study involved 18 observations and revealed a mean COD removal efficiency of 80 % (72, 86 %) and CV of 19 % (Fig. 4 and Table S2). This interval suggests high reliability and consistency in the column's performance across various trials. Efficiency rates for COD removal ranged from 52 % (Kaetzl et al., 2019) to 99 % (Kaetzl et al., 2019), demonstrating BFC's high potential under optimal conditions. The data showed a negative skewness (−0.57) and kurtosis (−1.17), indicating a distribution skewed toward higher efficiencies and more variation than a standard curve, reflecting diverse efficacy across setups or biochar types (Table S2). In evaluating the organic treatment efficiency of BFCs, comparisons were made with other media biofilters. Results indicate that most BFCs were more effective in organic treatment than the control media ($p < 0.05$), as presented in Table 3 A. Specifically, Kaetzl et al. (2019) demonstrated that the BFC with biochar as the primary material (single BFC) achieved a 52 % COD removal efficiency, significantly surpassing the rice husk filter that increased the COD concentration compared to the influent, resulting in a processing efficiency of −19 %.

Similarly, biochar showed significantly higher effectiveness ($p < 0.001$) compared to other filtrations in a study aimed at polishing effluents from a greywater treatment plant (Eshetu et al., 2015). The high COD removal rate of BFCs is attributed to biochar's significantly larger surface area, which facilitates biofilm formation and supports microbial attachment, biological degradation, and mineralization of organic matter (Kaetzl et al., 2019). A larger specific area of biochar with numerous functional groups (such as –COOH and –COH) enhanced the capacity for adsorption and the precipitation of various pollutants (Xin et al., 2021). Furthermore, given its lower particle density and bulk density compared to other materials of similar particle size, biochar is more convenient for transportation and handling (Perez-Mercado et al., 2018). The thermal treatment of biomass during pyrolysis results in the formation of hydrophobic functional groups on the surface of biochar, consequently enhancing its water retention capacity (Adhikari et al., 2023). The water retention capacity and high porosity increase the HRT in BFCs. For example, in a study by Perez-Mercado et al. (2018), the HRT in BFCs ranged from 2.9 to 4.5 days, while the HRT in sand filters was only 0.5 h. Prolonged hydraulic residence time prolongs the interaction between wastewater and biofilm, elevating the potential for organic matter degradation.

4.3. *Heavy metals*

The scale of heavy metal treatment in the studies reviewed predominantly involves small-scale column setups, with operational times ranging from as little as 3 h to a maximum of 50 h. The majority of these studies design BFC for treatment targeting individual metals such as chromium (Cr) (Deepa et al., 2022; Imran et al., 2020), arsenic (As) (Boni et al., 2021), lead (Pb) (Boni et al., 2020), cadmium (Cd) (Naeem et al., 2019), and uranium (U) (Lingamdinne et al., 2022). However, some research, such as the study by Singh et al. (2023), explores the concurrent removal of multiple metals, including zinc (Zn), Pb, and nickel (Ni), from wastewater. These studies report removal efficiencies ranging from 6 % to 91 %. Notably, heavy metal removal efficiency in stormwater runoff using single BFCs was relatively low, ranging from 18 % to 24 % (Reddy et al., 2014), potentially due to competition among metals for binding sites on the biochar surface.

Similar to its performance in organic matter processing, the BFC also shows a higher capability of removing heavy metals than other biofilters. For example, Deepa et al. (2022) found that biochar co-immobilized with sodium alginate microbial beads (Co-im/B.M.B) had a higher Cr treatment capacity (99 %) than immobilized sodium alginate microbial beads alone (Im/S.AM.B) at 93 %. This increased capacity was attributed to additional functional groups (carboxylic, hydroxylic, and carbonyl) provided by the biochar and microbial biomass, enhancing the binding sites for Cr ions. Another study by Boni et al. (2020) observed that the treatment capacity was 231.0 mg/g in

Table 2
Case studies and research findings on the use of BFCs.

Source of biomass	Production methods	Type of ww	HLR (m/h)	OLR (gCOD/m^3/d)	Efficiency (%)	References
Pine-spruce Willow	Commercial	Domestic ww	0.0014	20 ± 5*	COD: 94–99 TN: 50–52 PO_4^{3-}-P: 89 ± 7, TP: 86 ± 9 (willow biochar) PO_4^{3-}-P and TP: 32–60 (pine-spruce biochar)	(Perez-Mercado et al., 2018)
Porous palm residues	Pyrolysis for 2 h at 700 °C	Livestock and poultry ww	36.00–48.00[a]	NA	NH_4^+-N: 80 TP: 68	(Li et al., 2016)
Rice husk	Commercial	Municipal ww	0.05	63.0 ± 16.0	*E. coli*: 2.3 ± 0.58[c] Enterococci: 2.4 ± 0.72[c] Bacteriophages: 1.9 ± 0.36[c] COD: 52	(Kaetzl et al., 2019)
Solid waste from olive oil processing	Pyrolysis for 1,5 h at 550 °C	Real domestic ww	6.6×10^{-3}	0.5**	TP: 83 ± 3.2	(Hanandeh et al., 2017)
Virgin coniferous wood	Commercial biochar	Synthetic arsenic solution	0.06 0.12 0.18	NA	2.6–4.0 mg/g As	(Boni et al., 2021)
Softwoods	Pyrolysis 700 °C	Municipal ww	0.05	252.0	COD: 87 ± 2.6, TOC: 77 ± 3.6 TN: 14 ± 8.1, TP: 13 ± 9.0 FIB: up to 1.7[c]	(Kaetzl et al., 2018)
A mix of spruce, pine, and firwood chips	Pyrolysis >450 °C	Drinking water	0.60	NA	Phenanthrene and anthracene: 100 Atrazine: 37–97 Naphthalene: 49–93	(Chan et al., 2020)
Mango peels	Pyrolysis 600 °C	Real domestic ww	1×10^{-2} - 1.33×10^{-2}	NA	COD: 97, BOD: 88 TN: 74, NH_4^+-N: 78 NO_3^--N: 54, *E.coli*: 100	(Majumder and Das, 2022)
Hardwood	Pyrolysis 500 °C	Real domestic ww	NA		74 organic compounds: 44–86	(Blum et al., 2019)
Eggshells	Pyrolysis 800 °C for 4 h	Synthetic tannery ww	NA	NA	Cr: 99, COD: 96 Sulphate: 94	(Deepa et al., 2022)
Rice husk	Pyrolysis 600 °C	Real domestic ww	NA	NA	Potentially toxic metals: 6–91	(Singh et al., 2023)
Eucalyptus wood	Pyrolysis 550–850 °C for 2 h	Laundry ww	0.30[b], 0.40[b], and 0.50[b]	NA	COD, BOD: 79 to ≥83	(Yaseen et al., 2019)
Pine-spruce wood	Pyrolysis 800 °C	Municipal ww	2.1×10^{-3}	NA	Polyfluoroalkyl: 20–99	(Dalahmeh et al., 2019a)
Wheat straw	Pyrolysis 350 °C for 30 min	Synthetic ww	0.08	NA	Cd: 89–93 (in the first 90 min)	(Naeem et al., 2019)
Virgin coniferous wood	Pyrolysis at 600 °C	Synthetic ww	0.76	NA	163.9 mg/g for Pb	(Boni et al., 2020)
Miscanthus	Pyrolysis 850 ± 20 °C for 30 min	Real municipal ww	0.05	509.0 ± 173.0	COD: 74 ± 18 *E.coli:* 1.35 ± 0.27[c]	(Kaetzl et al., 2020)
Sewage sludge	Pyrolysis 700 °C for 60 min	Industrial ww	0.05	NA	42.3 mg/g for dye	(Al-Mahbashi et al., 2022)
Quinoa biomass	Pyrolisis 400 °C for 1 h	Synthetic ww	0.06	NA	Cr: <90 in the first 180 min	(Imran et al., 2020)
Spruce woodchips	Pyrolysis 600 °C	Forest Clear-Cut Runoff Water	1.53–2.72	NA	TN: 58	(Lafdani et al., 2020)
Birch, aspen, and alder woodchips	Pyrolysis 500 °C	Ww from the septic tank	1.7×10^{-3}	NA	Turbidity: 99 PO_4^{3-}-P: 95, DOC: >60	(Kholoma et al., 2020)
Various agriculture wastes	Commercial biochar	Olive mill ww	NA	NA	COD: 70, BOD: 40 Total organiccarbon: 82 Phenolic: 90	(Shabir et al., 2022)
Coconut shell	Pyrolysis	Real textile dyeing ww	0.42	NA	Colour: 75 BOD: 58, COD: 87	(Jayabalakrishnan et al., 2023)
Waste wood pellets	Gasification 520 °C	Synthetic stormwater runoff	NA	NA	NO_3^--N: 86, PO_4^{3-}-P: 47 Heavy metals: 18–24 Phenanthrene: 100 Naphthalene: 76 *E.Coli*: 27	(Reddy et al., 2014)
Bamboo biomass	Pyrolysis 500 °C for 2 h	Digested swine ww	NA	NA	NH_4^+-N: 63, TN: 51 TP: 53, COD: 80	(Xin et al., 2021)
Porous palm residues	Pyrolysis 2 h at 700 °C	Synthetic ww	12.0–72.0[a]	NA	NH_4^+-N: 80 TP: 68	(Li et al., 2016)
A mixture of pine and spruce wood biomass	Commercial biochar	Real domestic ww	9.6×10^{-4} 1.3×10^{-3} 1.6×10^{-3}	NA	COD: 90–93 TN: 71 ± 12, NH_4^+-N: 93 ± 2 *E. coli:* 1.1–2.1[c] (vertical flow filter) 2.5–3.4[c] (horizontal flow filter)	(Dalahmeh et al., 2019b)
Hardwood	Pyrolysis 500 °C	Domestic ww	1.7×10^{-3}	NA	PO_4^{3-}-P: 40, TP: 30 (gas concrete-biochar) PO_4^{3-}-P: 26, TP: 24 (sand-biochar) PO_4^{3-}-P: 20, TP: 23 (sand)	(Kholoma et al., 2016)

(*continued on next page*)

Table 2 (*continued*)

Source of biomass	Production methods	Type of ww	HLR (m/h)	OLR ($gCOD/m^3/d$)	Efficiency (%)	References
A blended mix of wood	Pyrolysis180 to 395 °C for 6 h	Synthetic stormwater	0.12	NA	Pathogenic bacteria: 3.9^c *E. coli*: 1.9^c MS2: 1.8^c	(Afrooz et al., 2018)
Unknown components	Commercial biochar	Synthetic stormwater	0.23	NA	*E.coli:* 3.6 ± 0.7^c (Rogue Biochar) 3.7 ± 0.72^c (Agricultural Carbons) 2.0 ± 0.38^c (Terra Char) 1.9 ± 0.5^c (NAKED Char)	(Valenca et al., 2021)
Mixed biomass	Pyrolysis 394 °C	Natural stormwater	NA	NA	*E.coli:* 0.8 to 3.0^c	(Kranner et al., 2019)
Pine	Pyrolysis 350 °C for 4 h	Synthetic aquaculture ww	0.24	NA	NO_3^--N: 84 ± 0.06 (50.0 mg NO_3^--N/L) 93 ± 0.01 (125.0 mg NO_3^--N /L)	(Paul and Hall, 2021)
Pine origin (*Pinus sp.*)	Commercial biochar	Vegetable farm ww	NA	NA	4.0 g $N/m^3/d$	(Pluer et al., 2016)
Pine pellets	Commercial biochar	Human waste anaerobic digestate ww	4×10^{-4}	NA	TN: 24, NH_4^+-N: 50 COD: 56, PO_4^{3-}-P: 68	(Hunter and Deshusses, 2020)
Pine pellets	Pyrolysis 900 °C for 1 h	Anaerobic digestion effluent	7×10^{-3}	380.0	COD: 56 NH_4^+-N: 81	(Forbis-Stokes et al., 2018)
Pine Hardwood	Commercial biochar	Synthetic ww	NA	NA	PO_4^{3-}-P: 65 after 18.0 h NO_3^--N: 86 after 18.0 h and 97 after 72.0 h	(Bock et al., 2015)
Pine wood	Commercial biochar	Urban stormwater runoff	1.53×10^{-2}	NA	NO_3^--N: >99, Zn: 50 Cd, Cu, Ni, Pb: >80 Trace organic contaminant: 100	(Ashoori et al., 2019)
Rice straw husk	Pyrolysis 500 °C	Rainwater runoff	NA	NA	Cu, Zn, Pd, Cd: >98	(Xiong et al., 2022)
Wood	Commercial biochar	Dairy runoff	1.08	NA	NH_4^+-N: 77 (Biochar Supreme) 49 (Biochar Now)	(Rahman et al., 2021)
Hardwood	Commercial biochar	Municipal ww	1.4×10^{-3} 8.3×10^{-3} 1.67×10^{-3}	19.4 ± 2.2 3.9 ± 1.6 4.9 ± 1.3 14.6 ± 3.1*	Bacteria: 1.6–4.5^c Virus: 1.0–2.3^c (at HLR 1.4×10^{-3})	(Perez-Mercado et al., 2019)
Unknown components	Commercial biochar	Grey water	1.17×10^{-2}	NA	COD: 80, PO_4^{3-}-P: 22, TP: 58, NH_4^+-N: 43, NO_3^--N: 70, TN: 66, *E.coli*: 45	(Eshetu et al., 2015)
Unknown components	Commercial biochar	Grey water	4.8×10^{-3}	40.0*	$BOD_{5:}$ 93, TP: 42 NH_4^+-N: 89 *E.coli*: 13.7	(Dalahmeh et al., 2016)
Pinewood	Pyrolysis 600 °C for 1 h Modified with $FeCl_3$	Municipal ww	0.30	NA	P: 175 mg/kg (GIB) 99 after 49.1 min (PIB)	(Gao and Wan, 2023)
Moso bamboo	Pyrolysis 400–600 °C for 1 h Modified with $MgCl_2$	Synthetic ww	0.65	NA	PO_4^{3-}-P: 60.7, 61.2 and 62.2 mg/g	(Jiang et al., 2018)
Coconut shell	Pyrolysis 550 °C for 1 h Modified with $ZnCl_2$	Real textile dyeing ww	0.42	NA	Colour: 99, BOD: 80.6 COD:90.5	(Jayabalakrishnan et al., 2023)
Quinoa biomass	Pyrolysis 400 °C for 1 h Modified with HNO_3	Synthetic ww	$1.5 \times 10^{-3\ b}$	NA	Cr: 90 in first 180.0 min 75 after 300.0 min 33 after 600.0 min	(Imran et al., 2020)
Wheat straw	Pyrolysis 350 °C for 30 min Modified with H_3PO_4	Synthetic ww	$2.1 \times 10^{-3\ b}$	NA	Cd: 92–96 in first 90.0 min 78–86 after 240.0 min	(Naeem et al., 2019)
Bamboo	Pyrolysis 600 °C for 1.5 h Modified with $AlCl_3$	Synthetic ww	0.30, 0.60 &1.20	NA	Sulfonamides: 650.0 and 1400.0 mg/kg	(Huang et al., 2022)
Watermelonrinds	Pyrolysis 500 °C for 1 h Modified with $FeCl_3$	Surface water	NA	NA	U: 99 for the initial 3 cycles	(Lingamdinne et al., 2022)

Lab: Laboratory, Ww: Wastewater, [a]: HRT in h, Q: Flow rate ([b]: Q in L/min), Log_{10} unit: Reduction efficiency ([c]: Log_{10} MPN/100 mL), *: g $BOD_5/m^2/d$, **: g $P/m^2/d$, DOC: Dissolved Organic Carbon, BOD: Biochemical Oxygen Demand, GIB: Granular iron biochar, PIB: Powder iron biochar, OLR: organic loading rate, NA: Not available.

columns filled with separate layers of soil and biochar, significantly higher than the 67.1 mg/g in soil-only columns. Xiong et al. (2022) also reported that a mixture of biochar, sand, and clay amendment in bioretention systems resulted in better heavy metal removal in rainwater runoff than traditional sand bioretention, with average reductions of Cu, Zn, Pb, and Cd by 55 %, 61 %, 20 %, and 36 %, respectively. Factors such as low ash content, higher cation exchange capacity, and a higher content of hydroxyl groups contribute to biochar's superior metal-removal effectiveness.

However, Ashoori et al. (2019) reported that the filter column with a mixture of woodchips and biochar did not significantly improve metal removal compared to columns without biochar (woodchips and woodchips-straw), exhibiting similar efficiencies in removing Cd, Cu, Ni, Zn, and Pb from synthetic stormwater ($p > 0.2$). While BFCs show potential in treating heavy metals, their effectiveness varies and can be low, especially in systems treating multiple metals simultaneously. The efficacy of metal removal depends on the specific characteristics of the biochar and the treated metals. More extensive and longer-term studies are needed to fully assess the factors influencing the metal treatment capabilities of BFCs, which will aid in selecting the most suitable biochar types for efficient metal removal.

4.4. *Pathogens*

Various conventional techniques are currently employed to eradicate pathogens from wastewater, including coagulation, filtration, chlorination, activated sludge treatment, and anaerobic digestion (Nasir et al., 2022). BFCs have been used to treat pathogens such as *Escherichia*

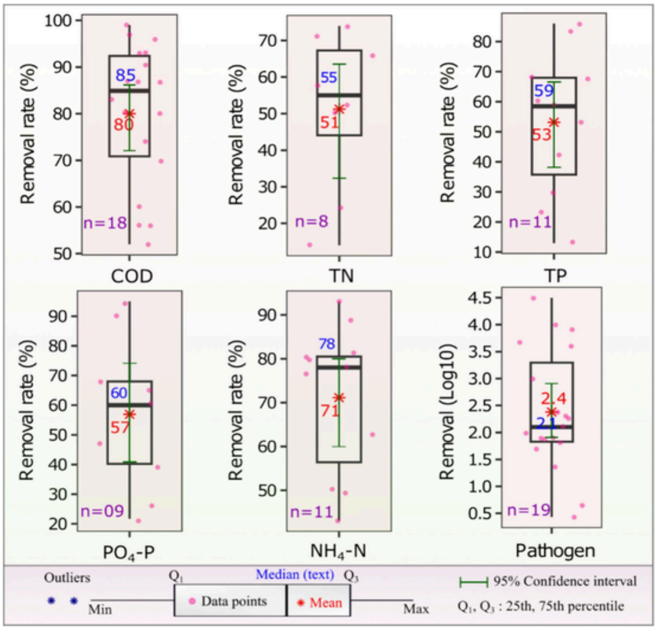

Fig. 4. Plots presenting removal efficiencies of biochar-based fixed filter columns. The boxplots display variability in data, including the mean (red asterisk and text), median (blue asterisk and text), interquartile range (Q1-Q3), outliers, and the 95 % confidence interval range (green line).

coli (E. coli), bacteriophages, and enterococci in municipal wastewater and stormwater (Table 2). The effectiveness of microbe removal by BFCs is mainly due to the filter material's adsorption capacity, the biofilm's properties that develop on filter surfaces, and the physical entrapment or straining occurring within the small pore spaces of the filter.

This study evaluates the efficiency of BFCs in pathogen removal by a logarithmic reduction scale. Across 19 samples, the average log reduction in pathogen concentration was 2.4 ± 1.1, with a 95 % CI ranging from 1.9 to 2.9 (Fig. 4 and Table S2). Boehm et al. (2020) report that conventional biofilters, which use materials like rocks, gravel, sand, granulated activated carbon (GAC), and synthetic plastics, typically achieved an average reduction of 0.4 $\log_{10}$ units for fecal indicator bacteria (FIB) in stormwater. In contrast, biochar-enhanced filter columns demonstrated a significantly higher capability, with potential reductions reaching up to 3.5 $\log_{10}$ units.

Further studies, such as those by Kaetzl et al. (2020) and Kaetzl et al. (2019), have shown that single BFCs can remove FIB better than other materials filters like wood chips, gravel, rice husk, and sand filters ($p < 0.05$). Some comparative results are presented in Table 3B, where the average efficiency of BFCs in removing *E. coli* and enterococci was higher than that of woodchip and gravel filters. In addition, the mean removal rates of *E. coli*, enterococci, and bacteriophages by biochar were statistically significantly higher than those by sand columns. This increased effectiveness is attributed to biochar's high organic carbon content, which fosters hydrophobic attachments of bacteria on its surface and provides increased surface area for viral and bacterial attachment. Furthermore, the adsorption capacity of the filter material, the characteristics of the biofilm, and the physical entrapment within small pore spaces are critical factors in microbe removal (Dalahmeh et al., 2019b; Kranner et al., 2019).

4.5. Nutrients

The study analyzed nutrient removal across multiple samples, presenting average efficiencies and variations within defined CIs, as de-

Table 3
Compare the effectiveness of pollutant treatment between BFCs and other media filters.

Parameters	Removal efficiency (%)	OLR (gCOD/m^3/d)	*P*-value	Reference
A: Organic matter				
COD	Biochar: 87 ± 2.6 Woodchips: 70 ± 4.7 Gravel: 74 ± 2.6	252.0	*<0.05*	(Kaetzl et al., 2018)
	Biochar: 74 ± 18 Sand: 61 ± 12	509.0 ± 173.0	*<0.05*	(Kaetzl et al., 2020)
	Biochar: 52 Rice husk: −19	63.0 ± 16.0	*<0.05*	(Kaetzl et al., 2019)
	Biochar: 94–99 Sand: 90–97	20.0 ± 5.0^b 5.0^b	*<0.05*	(Perez-Mercado et al., 2018)
	Biochar: 56 GAC: 62 Zeolite: 22 Gravel: 25	380.0	NA	(Forbis-Stokes et al., 2018)
	Biochar: 80 Filtralite: 63	NA	*<0.001*	(Eshetu et al., 2015)
	Biochar: 80 Hydrophobic polypropylene resin: 72	NA	NA	(Xin et al., 2021)
B: Pathogens				
E. coli	Biochar: 0.99^a Wordchips: 0.78^a Gravel: 0.84^a	252.0	*<0.05*	(Kaetzl et al., 2018)
	Biochar: 2.3 ± 0.58^a Sand: 2.0 ± 0.63^a Rice husk: 2.3 ± 0.63^a	63.0 ± 16.0	*≤0 .05*	(Kaetzl et al., 2019)
	Biochar: 1.4 ± 0.27^a Sand: 1.2 ± 0.31^a	509.0 ± 173.0	*<0.01*	(Kaetzl et al., 2020)
Enterococci	Biochar: 1.0^a Wordchips: 0.8^a Gravel: 0.8^a	252.0	*<0.05*	(Kaetzl et al., 2018)
	Biochar: 2.4 ± 0.7^a Sand: 2.2 ± 0.7^a Rice husk: 2.4 ± 0.6^a	63.0 ± 16.0	*<0.05*	(Kaetzl et al., 2019)
Bacteriophages	Biochar: 1.9 ± 0.4^a Sand: 1.7 ± 0.5^a Rice husk: 1.3 ± 0.4^a	63.0 ± 16.0	*<0.05*	(Kaetzl et al., 2019)
C: Nutrients				
TP	Biochar: 53 Hydrophobic polypropylene resin: 37	NA	NA	(Xin et al., 2021)
	Gas concrete-biochar: 30 Sand-biochar: 24 Sand: 23	NA	*0.04*	(Kholoma et al., 2016)
	Biochar: 175.0 mg/kg Quartz sand: 10.3 mg/kg	NA	*<0.05*	(Gao and Wan, 2023)
PO$_4^{3-}$-P	Gas concrete-biochar: 40 Sand-biochar: 26 Sand: 20	NA	*0.04*	(Kholoma et al., 2016)
	Gas concrete-biochar: 35 Sand-biochar: 45 Sand: 35	NA	*0.02*	(Kholoma et al., 2020)

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Table 3 (*continued*)

Parameters	Removal efficiency (%)	OLR (gCOD/m^3/d)	*P*-value	Reference
NH_4^+-N	Biochar: 81 GAC: 83 Zeolite: 83 Gravel: 76	380.0	NA	(Forbis-Stokes et al., 2018)
	Biochar: 50 GAC: 25 Zeolite: 69	740.0	NA	(Hunter and Deshusses, 2020)
TN	Biochar: 50–52 Sand: <5	20.0 ± 5.0^b 5.0^b	<0.05	(Perez-Mercado et al., 2018)
	Biochar: 24 GAC: 20 Zeolite: 39	740.0	NA	(Hunter and Deshusses, 2020)

OLR: organic loading rate, NA: Not available.

[a] : Log_{10} MPN/100 mL.

[b] : $gBOD_5/m^2/d$.

picted in Fig. 4. NH_4^+-N removal demonstrated an average efficiency of 71 ± 17.1 % (60, 80 %), indicating moderate sample variability (CV = 24 %). Total nitrogen (TN) removal averaged 51 ± 21.7 % (32, 64 %), and exhibited higher variability (CV = 42 %), reflecting efficiencies ranging from 14 % to 74 %. PO_4^{3-}-P and total phosphorus (TP) removals reported average efficiencies of 57 ± 25.6 % (41 %, 74 %) and 53 ± 23.8 % (38, 67 %), respectively. Both nutrients showed high variability (CVs ~45 %), with data distributions indicating a broad range of outcomes dependent on specific sample conditions.

Significant variability in nutrient removal by BFCs was observed, with some studies achieving up to 93 % for NH_4^+-N, 74 % for TN, 96 % for PO_4^{3-}-P, and 86 % for TP. Most experiments suggest that BFCs' nutrient removal efficiency was higher than other media filters (Table 3 C). According to Kholoma et al. (2020), the filter column consisting of layers of biochar and sand or gas concrete exhibited higher treatment efficiency than those without. Notably, the removal of TN in different BFCs was 12-fold higher than in sand filters despite all filters operating under similar hydraulic loading rates (HLR) of 1.33×10^{-3} - 1.42×10^{-3} m/h. Columns amended with biochar showed significantly higher TN removal compared to sand ($p < 0.05$), suggesting that the addition of biochar enhanced NH_4^+-N adsorption and nitrification (Rahman et al., 2021). In contrast, other media filters showed relatively low nutrient removal efficiencies; for instance, the TN reduction by woodchips and gravel was only 7 ± 8.8 % and 12 ± 4 %, respectively (Kaetzl et al., 2018). Further, the TN removal rate only achieved 30 % for sand-based filters (Martikainen et al., 2023) and was slightly higher at 34 % for stone (Rasool et al., 2018).

The higher efficiency in nutrient removal in BFCs can be explained by the large surface area of biochar, which enhances the adsorption of nutrients from wastewater and provides a conducive environment for bacterial growth, thereby increasing treatment efficiency. Also, the high porosity and abundance of micropores and nanopores in biochar create numerous anaerobic zones, enhancing the activity of denitrifying bacteria (Zainudin et al., 2020). Moreover, the efficiency of TP and PO_4^{3-}-P removal is also related to the mineral content on the surface of the filter material (such as calcium, iron, and aluminum), which increases the adsorption and precipitation of PO_4^{3-}-P from wastewater (Perez-Mercado et al., 2018).

However, Perez-Mercado et al. (2018) demonstrated that the efficiency of P treatment with various biochar buffering materials could be either higher or lower than that of the sand medium. For example, the P removal rate in single BFCs using willow biochar was higher (89 ± 7 % and 86 ± 9 % for PO_4^{3-}-P and TP, respectively) than sand media (75–83 % for TP and PO_4^{3-}-P), while the removal rates for pine-spruce biochar ranged from 32 % to 60 %, which was lower than that of sand medium. Similarly, according to Kholoma et al. (2020), the efficiency of PO_4^{3-}-P removal using biochar (42 %) was found to be lower gas concrete (96 %). The low efficiency in removing PO_4^{3-}-P by biochar could be attributed to the negative overall surface charge of biochar, which leads to a limited capacity for the adsorption of anionic pollutants (Shahraki and Mao, 2022). Similarly, the NH_4^+-N removal efficiency of biochar (50 %) was lower than that of zeolite (69 %), as demonstrated in the report by Hunter and Deshusses (2020) when using a single BFC. Ashoori et al. (2019) also reported that the effectiveness of treating NO_3^--N by a mixture of woodchip and biochar filter supplementation did not show significant differences compared to woodchip and woodchips-straw filter columns ($p = 0.31$ and $p = 0.53$).

4.6. *Organic contaminants*

Organic compounds in wastewater can be divided into two categories: biodegradable organic compounds (such as acetone, methanol, amino acids, humic substances, and carbohydrates) and persistent organic compounds (such as pesticides, herbicides, antibiotics, phenols, and polycyclic aromatic hydrocarbons) (Xiang et al., 2020). BFCs have demonstrated a high capacity for removing organic compounds, effectively addressing both biodegradable and biologically refractory organic compounds across various wastewater types. Furthermore, BFC systems have been specifically applied to treat persistent organic compounds in wastewater, such as colour, polycyclic aromatic hydrocarbons, and *per*-and polyfluoroalkyl substances. Notably, the removal efficiency for some persistent compounds has been exceptionally high, with phenanthrene removal reaching 100 % (Reddy et al., 2014).

However, some compounds showed negative removal efficiencies, such as perfluorobutanoic acid (−40 % to 60 %), perfluoropentanoic acid (−30 % to 80 %), perfluorohexanoic acid (−10 % to 90 %), and perfluoroheptanoic acid (−515 % to 100 %). These results occurred with biochar lacking biofilm, where removal depended solely on adsorption. The short carbon chains, high hydrophilicity, and water solubility of these polyfluoroalkyl substances led to poor adsorption efficiency (Dalahmeh et al., 2019a). Despite these findings, there remains a scarcity of studies utilizing biochar in medium and large-scale column filter systems to treat challenging, biologically resistant organic compounds. The results from the synthesized studies suggest that biochar is a highly effective filtering material for removing organic compounds from various types of wastewater, including those that are biologically recalcitrant. The treatment efficiencies vary depending on the specific organic compound targeted. However, it is essential to note that BFCs may not be effective for some biologically recalcitrant compounds. In certain instances, they might even increase the concentration of these compounds in the treated wastewater relative to the influent.

5. Influencing factors on removal efficiency

5.1. *Biochar characteristics*

Biochar characteristics such as chemical composition, surface area, porosity, and particle size directly influence its cation exchange capacity, water retention, and adsorption capacity, thereby affecting pollutant removal efficiency (Amalina et al., 2022; Tan et al., 2021; Weber and Quicker, 2018). These properties are shaped by feedstock, production temperature, residence time, heating rate, and pre- and post-production treatments (Ahmad et al., 2014; Daful and R Chandraratne, 2020). Different feedstocks result in varied inorganic mineral content and chemical structures (Leng et al., 2021). The impact of biochar particle size on pollutant removal efficiency varies. For example, sand mixed with coarse biochar (0.49 mm) achieved 77 % to 88 % efficiency, outperforming sand with fine biochar (0.31 mm), which ranged from 56 % to 84 % ($p = 0.05$) (Hanandeh et al., 2017). Conversely, Perez-Mercado et al. (2018) found smaller particles of biochar more effective in treating COD, with removal rates of 94 % for particles sized d10 = 2.8 mm and 99 % for d10 = 1.4 mm and 0.7 mm, though particle size showed no significant impact on nutrient removal efficiency.

Biochar feedstock is generally categorized into woody biomass (forest and tree waste) and non-woody biomass (sludge, livestock waste, crops, and residues). Woody biomass is distinguished by its low voidage, high bulk density, high calorific value, and low ash and moisture content (Sivaranjanee et al., 2024). Biochar production temperatures typically range from 300 °C to 800 °C, with pyrolysis durations from 1 to 4 h (Figs. 5 a & b). The production conditions of biochar, particularly the production temperature, notably influenced its properties (Yaashikaa et al., 2020). Although Fig. 5 does not correlate production temperature with BET surface area, some biochars exhibit large BET surface areas (>700 m^2/g) at pyrolysis temperatures above 900 °C. Studies show that increasing production temperature generally increases SSA. For instance, Elnour et al. (2019) reported that the SSA of date palm biochar rose from 2.04 to 249.13 m^2/g as temperature increased from 300 °C to 700 °C. Similarly, Chowdhury et al. (2016) found that wood sawdust biochar's SSA increased from 2.567 to 220.989 m^2/g as the temperature rose from 350 °C to 550 °C. Guo et al. (2020) observed that maize-straw biochar exhibited an SSA range of 0.47 to 635.24 m^2/g as temperature increased from 200 °C to 900 °C.

Modifying biochar further enhances its surface area; for example, HCl modification increased the SSA of coconut shell biochar from 52.35 to 590.8 m^2/g (Jayabalakrishnan et al., 2023). Similarly, $FeCl_3$ treatment increased the SSA of watermelon rind biochar from 52.1 to 86.35 m^2/g (Lingamdinne et al., 2022). Increased surface area typically improves adsorption efficiency (Tan et al., 2021). In Jiang et al. (2018), Moso bamboo biochar modified with $MgCl_2$ achieved an SSA of 311 m^2/g at 400 °C and 399 m^2/g at 600 °C, with corresponding PO_4^{3-}-P adsorption efficiencies of 60.7 to 62.2 mg/g. Furthermore, Jayabalakrishnan et al. (2023) reported BOD and COD removal efficiencies of 58 % and 87 % for pristine biochar and 80.6 % and 90.5 % for modified biochar.

Biochar's diverse characteristics, such as surface area, particle size, and feedstock origin, simultaneously influence the pollutant removal efficiency of BFCs. The variability in experimental conditions across studies complicates the quantitative ranking of these factors affecting treatment efficiency. Therefore, more targeted research and customized experiments are needed to address current uncertainties.

5.2. Concentration of pollutants

The contaminant concentration significantly influences the movement of contaminant molecules toward active sites on the adsorbent surface (Imran et al., 2020). The removal efficiency of Cd increased from 56 % to 90 % as the initial Cd concentration decreased from 100 mg/L to 5 mg/L in the filter column with two layers of biochar and sand (Naeem et al., 2019). Similarly, Deepa et al. (2022) reported a removal efficiency of 98 % at low Cr concentrations, whereas at higher Cr concentrations, it decreased to 92 %. The filtering materials in this study consist of three layers: biochar, sand, and gravel stones. This decline might be attributed to the saturation of all active sites at lower concentrations, where fewer metal ions effectively compete for binding. The increased number of competing ions at higher concentrations reduces the adsorption rate.

On the contrary, Lafdani et al. (2020) found that the outlet nitrogen concentration significantly decreased when the initial concentration was highest. In contrast, lower initial concentrations diminished removal efficiency, mainly when the TN concentration in water was below 0.4 mg/L, suggesting minimal TN adsorption under these conditions. Similar trends were observed in the studies by Reddy et al. (2014) and Majumder and Das (2022). For example, the removal efficiency of *E. coli* was only 27 % at an initial concentration of 7400 MPN/100 mL, while it reached 100 % when the *E. coli* concentration was approximately $3 \times 10^7 \pm 2.4 \times 10^7$ MPN/100 mL.

These observations highlight a general trend: as the input concentration of pollutants increases, treatment efficiency typically decreases due to heightened competition among ions for available binding sites on the adsorbent, leading to reduced adsorption rates. This underscores the importance of optimizing biochar for maximum efficiency without excessive resource expenditure. Conversely, deficient initial concentrations can also impair treatment efficiency, potentially due to inadequate contaminant presence needed for effective degradation. Understanding these dynamics is crucial for designing adaptable and cost-effective water treatment systems that can handle varying pollutant loads while maintaining regulatory compliance and ensuring the safety of treated water.

5.3. Hydraulic conditions

The HLR values from 38 experiments averaged 0.32 ± 0.60 m/h (0.19, 0.55) (Fig. 4 and Table S2). The median HLR was significantly lower at 0.06 m/h, indicating a skewed distribution with a high concentration of samples at meager rates. The skewness and kurtosis values were 2.7 and 7.9, respectively, suggesting that most samples clustered at lower rates with fewer high rates. The CV for HLR was exceptionally high at 171.9 %, highlighting it as the parameter with the most significant variation among those measured.

In field-scale studies, the flow rate is critical for evaluating the continuous treatment of metal effluents using sorbents. HRT and HLR are

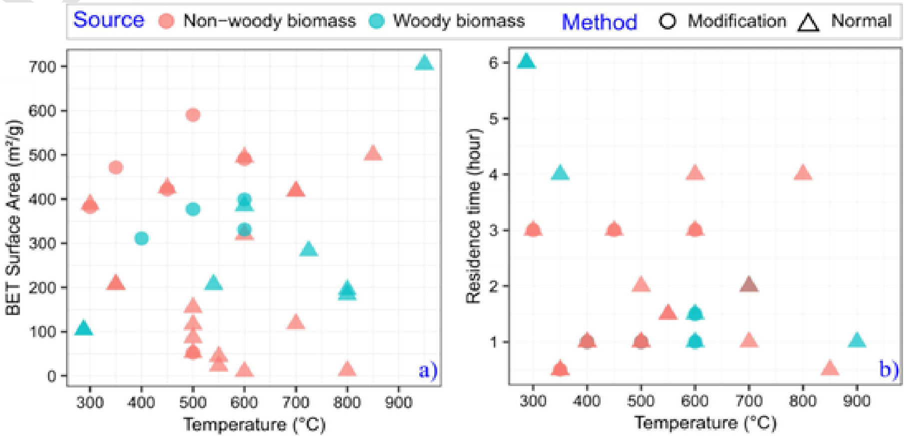

Fig. 5. Relationship between pyrolysis temperature of biochar and BET surface area (a) and residence time (b) across different biomass types and production methods. "Normal" refers to conventional pyrolysis, while "Modification" indicates chemically modified production methods.

inversely related; as HLR increases, HRT decreases, affecting pollutant removal efficiency. For example, Dalahmeh et al. (2019b) found that increasing the HLR in wood biochar filter from 8.33×10^{-4} to 1.25×10^{-3} and then to 1.67×10^{-3} m/h led to a progressive decrease in TN removal from 81 % to 71 % and then to 59 % ($p < 0.05$). Similarly, (Li et al., 2016) observed that extending the HRT from 12 to 48 h enhanced the removal rates of NH_4^+-N and TP. Additionally, nitrate and nitrite removal rates improved as HRT was extended from 12 to 60 h ($p = 0.05$). Extended HRT increased contact between wastewater and packing materials, enhancing degradation, while high HLR shortened contact times, reducing adsorption and interaction with pollutants. However, extending HRT further from 36 to 72 h did not significantly impact NH_4^+-N and TP removal, and increasing HRT from 60 to 72 h showed no significant effect on nitrate and nitrite removal. Perez-Mercado et al. (2019) also showed a similar influence on pathogens when using filters with hardwood biochar as the primary materials; the reduction rates of microbes at HLR of 1.67×10^{-2} m/h were significantly lower ($p < 0.01$) than at HLRs of 8.33×10^{-3} and 1.25×10^{-2} m/h, with no significant differences between the latter two.

Despite these findings, Dalahmeh et al. (2019b) noted that in pine and spucre wood biochar filter, variations in HLR did not significantly impact the removal of *E. coli* and Salmonella, nor did they affect COD removal, which remained stable at 90–93 % across different rates. Similarly, (Perez-Mercado et al., 2018) found consistent performance in single BFCs at HLR of 1.25×10^{-2} and 8.33×10^{-3} m/h for COD and PO_4^{3-}-P removal. These results indicate that while increasing HLR reduces the effectiveness of removing TN, NH_4^+-N, and TP due to shorter contact times, it does not significantly affect the removal of *E. coli*, Salmonella, COD, and PO_4^{3-}-P. Thus, the effectiveness of treatment adjustments, such as operational parameter changes, depends on factors like biochar type, filter configuration, wastewater characteristics, and the operational phase of the system.

6. Challenges and future perspectives

6.1. Challenges

6.1.1. Reduction efficiency over time

BFCs exhibit a decline in treatment efficiency over time, similar to other biofiltration technologies (Mohamed et al., 2023; Shahraki and Mao, 2022). This decline is due to alterations in the functional groups within biochar's pores and changes in the microbial communities' activity on its surface. These factors critically influence pollutant removal mechanisms. Moreover, the performance of BFCs is susceptible to operational parameters and environmental condition fluctuations, compounding the challenge. This challenge can be mitigated by adopting strategies such as reactivating or replacing biochar, enhancing microbial communities, optimizing operational conditions, managing environmental fluctuations, and engaging in ongoing research and development.

6.1.2. Clogging issues

Clogging is a significant operational challenge for BFCs. Over time, sediment and solid particles accumulate, obstructing the filter media and forming a biological membrane by bacteria. This increases pressure drops, channeling, dead zones, and uneven microbial growth, severely diminishing the system's contaminant removal efficiency (Dobslaw et al., 2018). Some studies report the onset of clogging within a range of 70 to 260 days (Forbis-Stokes et al., 2018; Kaetzl et al., 2020). Remedial actions like backwashing or replacing filter media are necessary but introduce additional operational challenges and potential health risks for maintenance personnel (Majumder and Das, 2022). To address clogging issues, selecting the appropriate size of biochar and other filtering materials is essential. Smaller biochar particles are more prone to causing clogging (Le et al., 2020; Ramezanzadeh et al., 2023). Additionally, pre-treatment of wastewater before it enters the filtration process is also essential. Biological filters can become clogged due to accumulating suspended solids, especially those smaller than six μm (Le et al., 2020). When clogging occurs, a backwashing process or replacement of a significant portion of the filter material is required to extend the filter's operational time.

6.1.3. Scale from lab to the field

Discrepancies between laboratory-scale studies and real-world applications significantly hinder understanding pollutant removal mechanisms. Laboratory conditions often fail to replicate complex environmental, hydrological, and operational realities, leading to inconsistent field performance (Boehm et al., 2020; Kholoma et al., 2016; Pluer et al., 2016). This gap underscores the need for more robust predictive models and scaled-up research. To overcome the discrepancies between laboratory-scale studies and real-world applications, it is crucial to enhance field research, develop predictive models that more accurately reflect real-world conditions, and design large-scale (bench-scale, pilot-scale) and long-term studies to evaluate results more effectively.

6.1.4. The influence of factors on treatment efficiency

Multiple factors, including influent loading rates, biochar properties, HRT, and HLR, significantly influence BFC treatment efficiency. Studies examining these influences have not always provided unbiased findings due to the interconnected nature of these effects and the limitations of existing models to capture these interactions fully. The variability in biochar characteristics, pollutant concentrations, and hydraulic conditions complicates the accurate prediction of treatment outcomes. To address these challenges, it is essential to develop comprehensive models, conduct controlled experiments, enhance data collection and analysis, optimize operational conditions, and improve biochar selection and treatment processes.

6.2. Future perspectives

6.2.1. Desorption and reuse of media

While restoring biochar's adsorption capacity is critical for its sustainable use, forming a biofilm on biochar surfaces after extended operation can facilitate the biological degradation of pollutants. This could diminish the need to reuse biochar media. Further, the desorption of contaminants can be prohibitively expensive, accounting for up to 50 % of the cost of new biochar production (Alsawy et al., 2022; Wang et al., 2020). Thus, repurposing saturated biochar as a fertilizer presents a viable alternative, as nutrient-enriched biochar has been shown to significantly enhance plant growth in agricultural settings (Shahraki and Mao, 2022). Although nutrient-enriched biochar can effectively enhance soil quality, there are concerns that other contaminants may be released from biochar during its application to soil. For instance, heavy metals may leach from biochar during the phosphorus recovery (Shahraki and Mao, 2022). A controlled release of organic and inorganic adsorbates from biochar pores can occur through diffusion. Conversely, chemical interactions and competition for adsorption sites between biochar functional groups and target contaminants can influence the kinetics of contaminant release from biochar, depending on the system's pH (Mukherjee et al., 2022).

Therefore, it is crucial to carefully consider any harmful components emitted from waste biochar, as they could be absorbed by crops and enter the food chain, potentially impacting the environment and human health. Biochar can also be used as an energy source, in fuel cells, as a catalyst, and for other applications (Ganesapillai et al., 2023; Setyawan et al., 2024). However, since biochar adsorbs pollutants from wastewater, burning it can release these pollutants into the surrounding environment, such as volatile organic compounds, NO_x, and metal oxides. Therefore, it is essential to treat the biochar before use, control the combustion process, and conduct a thorough environmental impact assessment before reusing biochar.

6.2.2. Economic issues and operational duration

The cost of biochar varies between countries, depending on factors such as local availability of feedstock, collection and transportation, processing requirements, technology scale, and production conditions (Mohan et al., 2014; Murtaza et al., 2024). However, biochar is less expensive than materials like activated carbon, zeolite, and metal-organic frameworks (Satyam and Patra, 2024). Table S4 compares the costs of biochar with other materials, highlighting the significant variability in biochar prices. Despite extensive research, several economic and technical challenges remain unresolved. It is challenging to obtain comprehensive information on the prices and production costs of biochar. A deeper exploration of these aspects is necessary before biochar can be effectively and widely implemented. Preparing raw materials for biochar, such as collection, cleaning, and grinding, requires significant effort before pyrolysis, and any modifications to enhance biochar's efficacy increase production costs. Optimizing the production process to reduce costs is crucial for future applications.

The durability of biochar in biofilters is also poorly understood due to a lack of long-term field-scale studies assessing its lifespan. Such studies are essential to evaluate biochar's performance over extended periods and under varying conditions. More comprehensive research is needed to estimate the operational duration of BFC systems and to develop long-term maintenance strategies. Effective maintenance strategies for biofilter systems using biochar should include continuous performance monitoring, biochar management, and regeneration, optimization of operating conditions, clogging management, process evaluation and improvement, as well as staff training and guidance.

6.2.3. Field-scale research

While most BFC research is conducted in laboratories and often only considers single pollutants, future studies should investigate the adsorption capabilities of BFC in the presence of multiple analytes. This is essential for practical application, as real wastewater contains numerous competing pollutants that can affect treatment efficiency. The real-world aging of BFCs and their efficiency decline over time remain underexplored. Long-term field-scale studies are crucial for addressing these gaps. Empirical data from such studies should be used to refine numerical models to evaluate biochar-augmented biofilters under various development scenarios. These models should simulate unit processes within biofilters to enable detailed scenario testing and performance assessment. Economic feasibility studies are also necessary to assess the cost-effectiveness of deploying large-scale biochar treatment systems.

7. Conclusion

This review assesses the effectiveness of biochar in filtration systems for wastewater treatment. BFCs are widely studied, with 50 % using SBF systems and the rest employing multi-layer systems, showing significant variability in column dimensions. The systems demonstrate high reliability and consistency, with BFCs generally outperforming control media in organic treatment, achieving an average COD removal of 80 ± 15.3 % (72, 86 %). Nutrient removal shows variability, with NH_4^+-N removal averaging 71 ± 17.1 % (60, 80 %), TN at 51 ± 21.7 % (32, 64 %), PO_4^{3-}-P at 57 ± 25.6 % (41, 74 %), and TP at 53 ± 23.8 % (38, 67 %). Pathogen concentrations are reduced by an average log of 2.4 ± 1.1 (1.9, 2.9). Biochar characteristics, pollutant concentration, and HLR influence pollutant reduction efficiency. Future research should focus on field-scale applications to evaluate economic viability, lifespan, environmental impacts, and optimized designs (optimal HLR, biochar particle size, and column configurations) tailored to specific operational conditions.

CRediT authorship contribution statement

Vu Khac Hoang Bui: Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Resources, Methodology, Data curation, Conceptualization. **T. Phuong Nguyen:** Writing – review & editing, Writing – original draft, Validation, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **T.C. Phuong Tran:** Writing – original draft, Validation, Investigation, Formal analysis. **T.T. Nguyen Nguyen:** Writing – original draft, Validation, Investigation, Formal analysis. **T. Nghi Duong:** Writing – original draft, Visualization, Investigation, Formal analysis, Data curation. **V.-Truc Nguyen:** Writing – original draft, Validation, Investigation, Formal analysis, Data curation. **Chong Liu:** Writing – original draft, Validation, Formal analysis, Data curation. **D. Duc Nguyen:** Writing – original draft, Visualization, Validation, Conceptualization. **Xuan Cuong Nguyen:** Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Project administration, Methodology, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Data availability

Data will be made available on request.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2024.176199.

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