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REVIEW

The Role of Biochar and Earthworms in Pharmaceutical Remediation of Contaminated Soil: A Systematic Review

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Abstract
Pharmaceutical contaminants in soils have become a critical and emerging environmental concern due to their persistence, toxicity, and potential to disrupt both ecosystems and human health. This manuscript provides a comprehensive review of current knowledge on the occurrence, contamination pathways and environmental fate of active pharmaceutical compounds in soil environments globally. The systematic review of recent literature on the topic of pharmaceutical remediation of contaminated soils through biochar and earthworms led to the final inclusion of 116 relevant articles. Based on these findings, the review evaluates sustainable bioremediation strategies that can effectively mitigate pharmaceutical contamination and support global efforts towards environmental sustainability, ecosystem resilience, and public health protection. Biochar, a low-cost, carbon-rich material derived from biomass pyrolysis, has gained significant attention for its high sorption capacity, enabling it to immobilize a broad spectrum of pharmaceutical compounds, thereby reducing their environmental toxicity. Concurrently, vermiremediation using earthworms, offers a promising approach that facilitates the degradation and removal of pharmaceuticals. Moreover, the review highlights the effectiveness of biochar, earthworms and their combined applications in remediating soils contaminated with antibiotics, non-steroidal anti-inflammatory drugs, endocrine disruptors, antihypertensives, anticonvulsants, antidepressants and other pharmaceutical classes. Unlike previous reviews, this study systematically integrates mechanisms and empirical data of biochar-earthworm synergistic remediation, quantifies remediation efficiency differences across drug types, offering new insights for sustainable soil remediation. To advance this promising field, further research is recommended to explore a wider range of pharmaceutical compounds, assess long-term environmental impacts, and optimize the synergistic effects of biochar and earthworms under diverse, real-world conditions.

Highlights

- Biochar makes possible the degradation of pharmaceuticals from contaminated soil.
- Earthworms serve as an alternative for the cleanup of pharmaceutical contaminants.
- The synergy of biochar and earthworm offers an eco-friendly soil decontamination.

Extended author information available on the last page of the article

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1 Introduction

Ensuring access to clean water and soil is a cornerstone of modern sustainable development. This urgency is reflected in the United Nations' 17 Sustainable Development Goals (SDGs), with water and environmental quality directly or indirectly addressed in at least 11 of them (Obaideen et al. 2022). However, escalating anthropogenic pressures including industrial activity, population growth, and intensified pharmaceutical use, have led to the contamination of both water and soil environments with a wide range of emerging pollutants, notably pharmaceutical compounds (Musie and Gonfa 2023; Nishmitha et al. 2025). For this purpose, special treatment should be followed, so that these contaminants are eliminated (Kesari et al. 2021).

Pharmaceuticals in wastewater, surface water, groundwater and soils could reach levels up to mg/L (Alsalihy et al. 2024; Hama Aziz et al. 2024). Although their typically low concentrations, their persistence, bioactivity, and potential for bioaccumulation pose serious ecological and human health risks. Hence, to decrease the environmental and health risk posed by these chemical compounds in the water resources and soil environments, novel methods should be developed to eliminate them (Guo et al. 2017; Samal et al. 2022; Kang et al. 2022; Chauhan et al. 2023; Gkika et al. 2023; Ruziwa et al. 2023; Imreová et al. 2024; Hama Aziz et al. 2024; Gallego-Ramírez et al. 2024). While wastewater treatment plants (WWTPs) are designed to remove organic matter, nutrients, and pathogenic microorganisms, their effectiveness in removing pharmaceutical contaminants in trace levels is often limited (Li et al. 2024). A couple of methods are already in use for the elimination of pharmaceuticals in environmental samples (Guo et al. 2017; Samal et al. 2022; Kang et al. 2022; Chauhan et al. 2023; Gkika et al. 2023; Ruziwa et al. 2023; Imreová et al. 2024; Hama Aziz et al. 2024; Gallego-Ramírez et al. 2024). Each method is characterized by its advantages and drawbacks, which usually depend on the type of pharmaceuticals targeted and the nature of the contaminated medium (Thakur et al. 2023; Khalidi-Idrissi et al. 2023). The methods that are developed in that direction could be divided into physical, chemical and biological processes according to their mechanisms (Guo et al. 2017; Ahmed et al. 2021; Kato and Kansha 2024; Hama Aziz et al. 2024).

Among the physicochemical processes, coagulation (Guo et al. 2017; Kooijman et al. 2020; Alazaiza et al. 2022; Rajagopal et al. 2022), sedimentation (Lee et al. 2015; Guo et al. 2017) and, in some cases, flotation (Ensano et al. 2017; Guo et al. 2017; Kyzas and Matis 2018; Pooja et al. 2025) are employed. Although the technologies are easy in operation, the separation of the organic contaminants is a difficult and complex process (Guo et al. 2017; Kooijman et al. 2020; Alazaiza et al. 2022). Another common physicochemical wastewater treatment technology is membrane separation (Fazal et al. 2015; Shojaee Nasirabadi et al. 2016; Guo et al. 2017; Ma et al. 2024; Shaker et al. 2024; Islam et al. 2025). It is characterized by very high efficiency in the elimination of pharmaceuticals with a minimum environmental impact, but its application is limited by the high cost of the filters and the requirement of frequent replacement of membranes because of fouling of their surfaces.

The application of advanced oxidation processes (AOPs) is a relatively new chemical practice, which offers an environmentally friendly process for the substantial reduction of biological and chemical contaminants and the significant improvement of water quality. The AOPs refer to the formation of chemical oxidants, production of highly reactive oxygen species (ROS) and degradation of harmful organic substances succeeding in wastewater disinfection (Guo et al. 2017; Adeoye et al. 2024; Kanakaraju et al. 2025). They involve a number of redox technologies, like ozonation, photocatalysis activated by semiconductors, UV-based photolysis, electrochemical oxidation, Fenton reaction and combinations thereof. However, the effectiveness of AOPs is affected by multiple factors, whereas more harmful transformation by-products may be produced (Taoufik et al. 2021; Gopalakrishnan et al. 2023).

Activated carbon adsorption is a highly effective and widely applied method combining characteristics from both the physical and chemical approaches. This technology is based on the adsorption of pharmaceuticals onto porous carbon surfaces with large specific surface area (Khanday et al. 2021; Paz et al. 2025). The adsorption capacity is high and the surfaces are chemically stable (Ilavský and Barloková 2023). The technology is applicable for the removal of a wide range of drugs, especially the hydrophobic ones (Shearer et al. 2022). However, this method suffers from the requirement of regeneration or replacement of the carbon surfaces, high costs combined with low efficiency of regeneration and difficult operation (Njewa and Shikuku 2023; Satyam and Patra 2024).

Another physicochemical method involves the use of biochar for the adsorption of organic pollutants from water and soil resources. Biochar, a carbon-rich material produced from biomass via pyrolysis, has emerged as a highly promising solution for the removal of pharmaceuticals from wastewater and soil (Wang et al. 2020; Xiang et al. 2020; Hama Aziz et al. 2024; Dimitriadou et al. 2025). Due to its high surface area, porous structure, and the ability to adsorb a wide range of organic pollutants, biochar has demonstrated significant potential for effectively capturing various pharmaceutical contaminants, while offering the benefits of being relatively inexpensive, reusable, and environmentally friendly. Furthermore, no harmful by-products are produced, potentially improving the overall treatment process and reducing toxicity to a minimum (Solanki and Boyer 2017; Kang et al. 2022; Chauhan et al. 2023; Fu et al. 2024; Satyam and Patra 2024; Hama Aziz et al. 2024; Gallego-Ramírez et al. 2024; Trivedi et al. 2025; Laishram et al. 2025; Fady et al. 2025).

Another sustainable approach for pharmaceutical remediation of contaminated soils is the use of earthworms (Isari et al. 2024). Earthworms offer significant advantages by acting as natural bioturbators and biotransformers. Their burrowing and ingestion processes improve soil structure, aeration, moisture retention, and microbial activity, which increase the bioavailability and breakdown of pharmaceutical residues (Xiao et al. 2022; Gudeta et al. 2023). In this way, the strategic deployment of earthworms offers a safe, cost-effective, and ecologically sustainable approach to pharmaceutical remediation, fostering healthier soil ecosystems without leaving persistent by-products.

This review aims to explore the role of biochar and earthworms in the sustainable remediation of pharmaceutical contaminated soils. Moreover, it provides a comprehensive overview of the sources, types, and environmental behaviors of pharmaceutical pollutants in terrestrial ecosystems. The mechanisms underlying biochar adsorption and earthworm-mediated degradation of pharmaceuticals are also examined, along with the latest advances in engineered biochars and integrated bioremediation strategies. Emphasis is placed on the environmental advantages, technical challenges, and future research directions for optimizing these technologies for large-scale soil cleanup. By integrating biochar and biological agents like earthworms, a more holistic and sustainable approach to pharmaceutical remediation may be achieved.

2 Methodology for Systematic Review Analysis

This systematic review was conducted in accordance with the Preferred Reporting Items for Systematic Reviews and Meta-Analyses (PRISMA) guidelines (Moher et al. 2009; Page et al. 2021). A structured and comprehensive search strategy was developed to identify relevant studies that address the role of earthworms and biochar in pharmaceutical remediation of contaminated soil, which were published the latest 20 years (2005–2025). More specifically, a systematic literature search was conducted in PubMed, Scopus and ScienceDirect and it was limited to articles written in English. The final search was carried out on 15/07/2025. The search strategy combined main keywords, such as “Pharmaceuticals” AND “Biochar” AND “Earthworms”.

The eligible criteria of the study included the involvement of articles in peer-reviewed journals or book chapters published between 2005 and 2025, written in English and the focus of the study was on remediation of pharmaceuticals from soil samples by using biochar or earthworms. Conference abstracts, editorials, perspectives or case studies were excluded from this review. Moreover, studies that did not focus on the primary aims of this review, as well as studies with overlapping datasets or with no significant findings for the systematic review were also excluded from the selection.

Full-text articles were obtained for all studies that met the inclusion criteria. All articles retrieved from the initial search were imported into Mendeley Reference Management, so that they will be easily cited in the manuscript and perform de-duplication. Initially, 591 studies were identified through ScienceDirect, 154 in PubMed and 1907 in Scopus, resulting in a total of 2652 records. Following the automated and manual deduplication process, 735 duplicate records were identified and removed, providing a clear view of the unique dataset retrieved across the three databases.

After removing duplicates ($n=735$), records marked as ineligible by automation tools ($n=1019$) and papers removed for other reasons, like not containing the desired keywords in abstract ($n=536$), 472 studies remained for title/abstract screening. Of these, a significant number of records were excluded by automation tools according to keywords ($n=237$), while 21 out of the remaining 235 records could not be retrieved. Thus, 214 records were selected for full-text review. Of these, the 59 were not found to focus on the primary aims of the systematic review after abstract screening, 7 records were found with overlapping datasets and 32 studies did not have significant findings for the present systematic review. Ultimately 116 studies met the inclusion criteria and were included in the final synthesis. The year distribution of these 116 studies (2005–2025) shows a notable increase in publications after 2015, reflecting the growing scientific interest in biochar-assisted remediation, vermiremediation, and pharmaceutical behavior in soil systems. The selection process is summarized in the PRISMA flow diagram (Fig. 1).

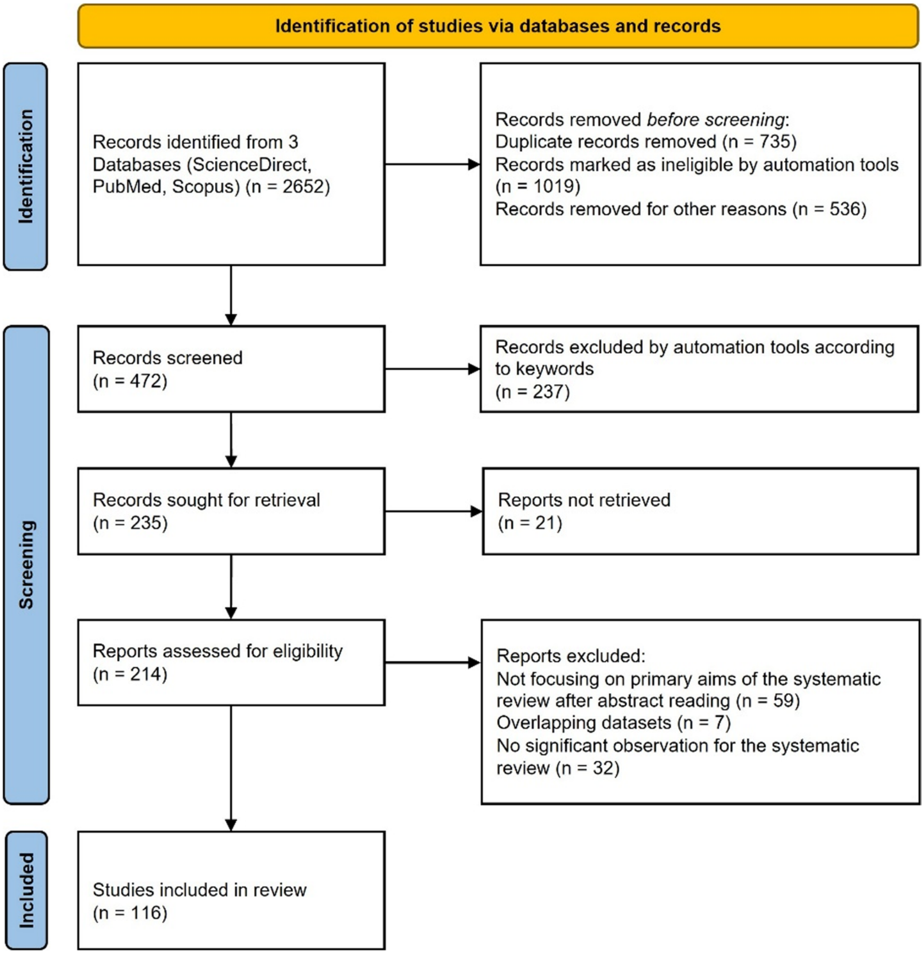

Fig. 1 PRISMA 2020 flow diagram for the systematic review (Page et al. 2021)

3 Pharmaceutical Contamination Pathways of Soils and Environmental Risks

Although unmetabolized pharmaceutical residues are increasingly detected in surface water, groundwater, soils and essentially in all environmental compartments affected by human activity, the precise sources of specific pollutants are often difficult to trace (Całus-Makowska et al. 2023). One of the primary sources is human excretion of active pharmaceutical ingredients (APIs) in urine and feces, discharged into municipal sewage systems from households and hospitals. Similarly, veterinary pharmaceuticals used in livestock and companion animals are excreted into the environment through manure and urine, contributing to the contamination of nearby soils and water bodies. Sludge applications of manure from livestock as fertilizers are widely identified as key sources of veterinary pharmaceutical contamination. Another significant route of contamination is the improper disposal of unused or expired drugs from households, hospitals, and pharmaceutical industries.

Industrial waste streams from pharmaceutical manufacturing facilities may contain high concentrations of under-development or active compounds that can reach WWTPs, while improperly discarded pharmaceuticals may accumulate in landfills, leaching into surrounding soil and groundwater (Fig. 2). These direct inputs are considered point sources of contamination because they originate from identifiable locations.

On the other hand, diffuse sources of pharmaceutical contamination are more challenging to identify and control. These include leaching and runoff from agricultural fields treated with pharmaceutical-laden manure or sewage sludge, as well as leakage from WWTP infrastructure and effluent discharge into natural water bodies (Lapworth et al. 2012; Całus-Makowska et al. 2023). Figure 2 clearly distinguishes between point sources (e.g., WWTP discharge, landfill leachate) and diffuse sources (e.g., agricultural runoff from sludge-amended fields), highlighting their combined contribution to environmental contamination. These diffuse inputs pose unique monitoring and regulatory challenges, as their spatial and temporal dynamics vary with environmental conditions and land-use practices.

Although the magnitude of each route varies across regions and depends on factors such as livestock density, pharmaceutical consumption patterns, and wastewater treatment efficiency, several global assessments provide indicative ranges that help illustrate their comparative influence. Livestock manure and slurry applications are frequently identified as dominant contributors, accounting for approximately 40–60% of pharmaceutical inputs to agricultural soils in areas with intensive animal production (Lapworth et al. 2012). In contrast, municipal wastewater effluent and land-applied biosolids typically contribute around 20–40%, reflecting differences in treatment technologies and disposal practices. Diffuse routes, such as field runoff, septic leakage, and leaching from sludge-amended soils, generally represent lower but still environmentally significant inputs, often estimated at < 10–20% (Lapworth et al. 2012; Całus-Makowska et al. 2023). These values are presented

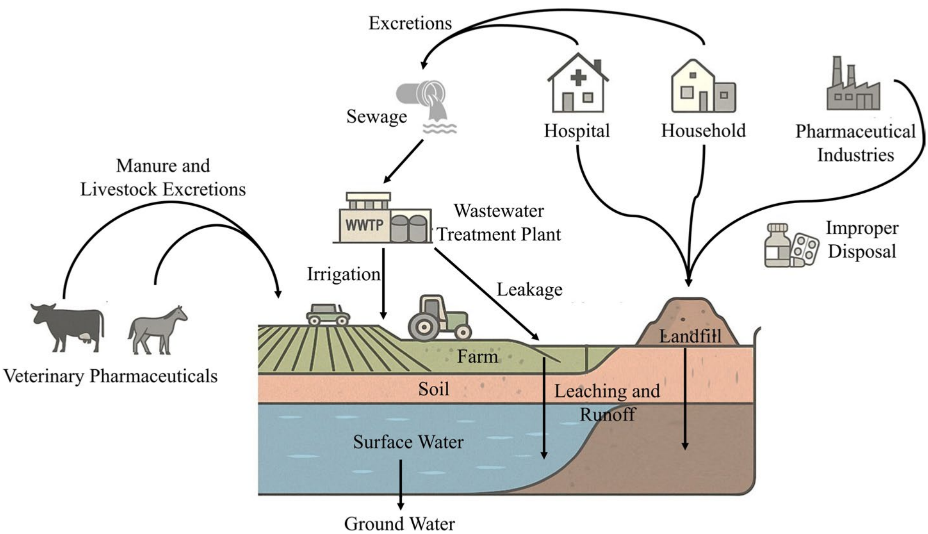

Fig. 2 Pathways of pharmaceutical contamination of soil and water resources. The figure summarizes point and diffuse inputs, illustrating how human, veterinary, and industrial sources introduce pharmaceuticals into soil and aquatic systems

as approximate ranges rather than universal proportions, acknowledging substantial spatial and temporal variability.

4 Pharmaceutical Contaminants in Soils Globally

Pharmaceutical contamination in soils is characterized by the presence of a diverse range of APIs, each exhibiting distinct environmental behaviors influenced by their source, chemical properties, and soil characteristics. Common pharmaceutical categories detected as soil pollutants include antibiotics, analgesics and non-steroidal anti-inflammatory drugs (NSAIDs), antiepileptics and central nervous system agents, beta-blockers, psychotropics, hormonal compounds, and veterinary medicines (Table 1). Beyond documenting contaminant occurrence, the global distribution patterns summarized in Table 1 provide important insights into the regional suitability and adaptability of biochar-earthworm remediation strategies, which are strongly regulated by soil physicochemical properties and climatic conditions.

In a recent study conducted in France (Wakim et al. 2024), nine (9) APIs were identified in soil samples. Among these were the antibiotics penicillin G and roxithromycin, the analgesic paracetamol, and various endocrine disruptors such as progesterone and estrogens (estriol, estrone, 17α-estradiol, 17β-estradiol, and 17α-ethinylestradiol). Paracetamol was present in relatively high concentrations (7.36–43.55 ng/g), as was bisphenol A, a plasticizer linked to endocrine disorders (19.88–36.40 ng/g) (Konieczna et al. 2015). In contrast, roxithromycin (1.36–11.60 ng/g) and penicillin G (0.23–5.96 ng/g) were found in lower concentrations. Progesterone and estrogens were detected at trace levels, typically below 1 ng/g, and in some cases below 0.5 ng/g (Wakim et al. 2024).

A study in central Mexico, revealed multiple pharmaceutical contaminants in soils irrigated with wastewater. Human antibiotics such as sulfamethoxazole (0.98–5.96 ng/g), trimethoprim (0.13–2.44 ng/g), ciprofloxacin (0.35–2.62 ng/g), and clarithromycin (up to 5.43 ng/g) were detected, along with the veterinary antibiotic enrofloxacin ($\leq$1.21 ng/g). Except for antibiotics, NSAIDs including diclofenac (0.10–0.54 ng/g) and naproxen (0.51–3.06 ng/g), the anticonvulsant carbamazepine (0.98–5.96 ng/g), and the antilipemic agent bezafibrate (up to 1.07 ng/g) were also found (Dalkmann et al. 2012).

In Arizona, USA, soils irrigated with reclaimed wastewater were analyzed (Williams and McLain 2012). The antibiotic lincomycin was primarily concentrated in the surface layer (0.009–0.042 ng/g), while carbamazepine showed increased concentrations at deeper layers (0.21–0.28 ng/g at 10–50 cm). Caffeine was also most abundant near the surface (0.89–1.7 ng/g), whereas ibuprofen was barely detectable.

In Ghana, soil samples and leachates from municipal landfills were examined. Leachates contained antibiotics such as chloramphenicol (74.53 ng/mL), doxycycline (3.67 ng/mL), and amoxicillin (1.77 ng/mL), along with NSAIDs like diclofenac (1.20 ng/mL) and analgesics, such as paracetamol (0.29 ng/mL). In the soil samples of the dumpsites, amoxicillin (305.06 ng/g) was the most prevalent, followed by chloramphenicol (253.46 ng/g), doxycycline (53.04 ng/g), diclofenac (25.27 ng/g), and paracetamol (11.70 ng/g) (Dankwa et al. 2024). A separate study, reported the exclusive detection of tramadol in hospital-proximal dumpsite soils, with a mean concentration of 10.50 ng/g (Ishmael et al. 2025).

Soil samples from agricultural fields (cereal, potatoes, cabbage, rice, and citrus) in Spain were examined, revealing widespread contamination with NSAIDs. Salicylic acid was

Table 1 Summary of pharmaceutical contaminants detected in different kinds of soils, their determined concentrations and their geographical distribution

Category of Pharmaceuticals	API	Concentration (ng/g)	Country	Type of Sample	Reference
Antibiotics (β-lactams)	Penicillin G	0.23–5.96	France	Loamy soil – public garden	(Wakim et al. 2024)
	Amoxicillin	1.77 ng/mL	Ghana (Ejisu-Juaben)	Leachates from municipal landfills	(Dankwa et al. 2024)
		305.06	Ghana (Ejisu-Juaben)	Soils from municipal landfills	(Dankwa et al. 2024)
		0.014–0.265 0.374	India	Agricultural field Vegetables uptake	(Akhter et al. 2023)
Antibiotics (Macrolides)	Roxithromycin	1.36–11.60	France	Loamy soil – public garden	(Wakim et al. 2024)
		614	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
	Clarithromycin	≤5.43	Central Mexico	Irrigated soil with untreated wastewater	(Dalkmann et al. 2012)
	Tylosin	< 10	Northern Germany	Soils amended with swine manure	(Tasho and Cho 2016)
		1453	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
	Erythromycin	0.000101–0.018 0.320	India	Agricultural field Vegetables uptake	(Akhter et al. 2023)
		202	USA, Colorado	Irrigated soil with reclaimed water	(Kinney et al. 2006)
Antibiotics (Tetracyclines)	Doxycycline	3.67 ng/mL	Ghana (Ejisu-Juaben)	Leachates from municipal landfills	(Dankwa et al. 2024)
		53.04	Ghana (Ejisu-Juaben)	Soils from municipal landfills	(Dankwa et al. 2024)
		About 2.5	South Africa	Golf course field irrigated by WWTP	(Ademoyegun et al. 2020)
		Not detected	South Africa	Sewage sludge from 3 WWTPs	(Ademoyegun et al. 2020)

Table 1 (continued)

Category of Pharmaceuticals	API	Concentration (ng/g)	Country	Type of Sample	Reference
	Oxytetracycline	6.7	-	Soils amended with manure	(Tasho and Cho 2016)
	Tetracycline	86,000–199,000	Northern Germany	Soils amended with swine manure	(Tasho and Cho 2016)
Antibiotics (Fluoroquinolones)	Ciprofloxacin	0.35–2.62	Central Mexico	Irrigated soil with untreated wastewater	(Dalkmann et al. 2012)
	Enrofloxacin	≤1.21	Central Mexico	Irrigated soil with untreated wastewater	(Dalkmann et al. 2012)
	Ofloxacin	0.014–0.265 5.586	India	Agricultural field Vegetables uptake	(Akhter et al. 2023)
Antibiotics (Sulfonamides)	Sulfamethoxazole	0.98–5.96	Central Mexico	Irrigated soil with untreated wastewater	(Dalkmann et al. 2012)
		9.13	USA, Colorado	Irrigated soil with reclaimed water	(Kinney et al. 2006)
		112	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
	Sulfathiazole	37	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
	Sulfapyridine	38	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
	Sulfamethazine	15	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
Antibiotics (Trimethoprim)	Trimethoprim	0.13–2.44	Central Mexico	Irrigated soil with untreated wastewater	(Dalkmann et al. 2012)
		1.22	USA, Colorado	Irrigated soil with reclaimed water	(Kinney et al. 2006)
		About 4–50	South Africa	Sewage sludge from 3 WWTPs	(Ademoyegun et al. 2020)
		17	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)

Table 1 (continued)

Category of Pharmaceuticals	API	Concentration (ng/g)	Country	Type of Sample	Reference
Antibiotics (Lincosamides)	Lincomycin	0.009–0.042 (surface layer)	USA, Arizona	Irrigated soil with reclaimed wastewater	(Williams and McLain 2012)
Antibiotics (Chloramphenicol)	Chloramphenicol	74.53 ng/mL	Ghana (Ejisu-Juaben)	Leachates from municipal landfills	(Dankwa et al. 2024)
		253.46	Ghana (Ejisu-Juaben)	Soils from municipal landfills	(Dankwa et al. 2024)
		About 2	South Africa	Golf course field irrigated by WWTP	(Ademoyegun et al. 2020)
		About 4–99	South Africa	Sewage sludge from 3 WWTPs	(Ademoyegun et al. 2020)
Antifungal	Miconazole	1.41	USA, Colorado	Irrigated soil with reclaimed water	(Kinney et al. 2006)
Paracetamol	Paracetamol	7.36–43.55	France	Loamy soil – public garden	(Wakim et al. 2024)
		0.29 ng/mL	Ghana (Ejisu-Juaben)	Leachates from municipal landfills	(Dankwa et al. 2024)
		11.70	Ghana (Ejisu-Juaben)	Soils from municipal landfills	(Dankwa et al. 2024)
		0.4	Spain	Soil from agricultural fields	(Aznar et al. 2014)
		33.2	USA, Colorado	Irrigated soil with reclaimed water	(Kinney et al. 2006)
		About 4	South Africa	Golf course field irrigated by WWTP	(Ademoyegun et al. 2020)
		About 6–99	South Africa	Sewage sludge from 3 WWTPs	(Ademoyegun et al. 2020)
		419	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
NSAIDs	Diclofenac	0.10–0.54	Central Mexico	Irrigated soil with untreated wastewater	(Dalkmann et al. 2012)

Table 1 (continued)

Category of Pharmaceuticals	API	Concentration (ng/g)	Country	Type of Sample	Reference
		1.20 ng/mL	Ghana (Ejisu-Juaben)	Leachates from municipal landfills	(Dankwa et al. 2024)
		25.27	Ghana (Ejisu-Juaben)	Soils from municipal landfills	(Dankwa et al. 2024)
		About 10–50	South Africa	Sewage sludge from 3 WWTPs	(Ademoyegun et al. 2020)
		83	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
	Naproxen	0.51–3.06	Central Mexico	Irrigated soil with untreated wastewater	(Dalkmann et al. 2012)
		0.7	Spain	Soil from agricultural fields	(Aznar et al. 2014)
		0.55	Mexico City	Irrigated soil from wastewater channels	(Durán-Alvarez et al. 2009)
		4	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
	Ibuprofen	Barely detected	USA, Arizona	Irrigated soil with reclaimed wastewater	(Williams and McLain 2012)
		0.5	Spain	Soil from agricultural fields	(Aznar et al. 2014)
		0.25	Mexico City	Irrigated soil from wastewater channels	(Durán-Alvarez et al. 2009)
		8.65	South Africa	Golf course field irrigated by WWTP	(Ademoyegun et al. 2020)
		About 13–97	South Africa	Sewage sludge from 3 WWTPs	(Ademoyegun et al. 2020)
		114	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
	Salicylic Acid	4.4	Spain	Soil from agricultural fields	(Aznar et al. 2014)
	Mefenamic Acid	1.5	Spain	Soil from agricultural fields	(Aznar et al. 2014)

Table 1 (continued)

Category of Pharmaceuticals	API	Concentration (ng/g)	Country	Type of Sample	Reference
	Fenoprofen	0.8	Spain	Soil from agricultural fields	(Aznar et al. 2014)
	Aspirin	1.16	South Africa	Golf course field irrigated by WWTP	(Ademoyegun et al. 2020)
		About 20–60	South Africa	Sewage sludge from 3 WWTPs	(Ademoyegun et al. 2020)
Opioid Medications	Tramadol	10.50	Ghana (Sefwi Wiawso)	Dumpsite soil near hospital	(Ishmael et al. 2025)
	Codeine	About 20–45	South Africa	Sewage sludge from 3 WWTPs	(Ademoyegun et al. 2020)
Hormonal Medications	Progesterone	0.32–0.95	France	Loamy soil – public garden	(Wakim et al. 2024)
	Estriol	0.04–0.12	France	Loamy soil – public garden	(Wakim et al. 2024)
		113	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
	Estrone	0.35–0.43	France	Loamy soil – public garden	(Wakim et al. 2024)
		137	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
	17α-estradiol	0.03–0.22	France	Loamy soil – public garden	(Wakim et al. 2024)
	17β-estradiol	0.07–0.11	France	Loamy soil – public garden	(Wakim et al. 2024)
	17α-ethinylestradiol	0.18–79.6	France	Loamy soil – public garden	(Wakim et al. 2024)
		313	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
	Diethylstilbestrol	184	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
	Estradiol-3-sulfate	28	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
	Estrone-3-sulfate	28	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)

Table 1 (continued)

Category of Pharmaceuticals	API	Concentration (ng/g)	Country	Type of Sample	Reference
Anticonvulsants	Carbamazepine	0.98–5.96	Central Mexico	Irrigated soil with untreated wastewater	(Dalkmann et al. 2012)
		0.21–0.28 (deeper layer) 0.14–0.25 (surface layer)	USA, Arizona	Irrigated soil with reclaimed wastewater	(Williams and McLain 2012)
		1.2	Spain	Soil from agricultural fields	(Aznar et al. 2014)
		6.48	Mexico City	Irrigated soil from wastewater channels	(Durán-Alvarez et al. 2009)
		17.6	USA, Colorado	Irrigated soil with reclaimed water	(Kinney et al. 2006)
		About 3.5	South Africa	Golf course field irrigated by WWTP	(Ademoyegun et al. 2020)
		About 10–60	South Africa	Sewage sludge from 3 WWTPs	(Ademoyegun et al. 2020)
		42	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
Antilipemic Agent	Bezafibrate	≤1.07	Central Mexico	Irrigated soil with untreated wastewater	(Dalkmann et al. 2012)
		9	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
	Gemfibrozil	3.6	Spain	Soil from agricultural fields	(Aznar et al. 2014)
	Fenofibrate	7.0	Spain	Soil from agricultural fields	(Aznar et al. 2014)
	Clofibric Acid	0.7	Spain	Soil from agricultural fields	(Aznar et al. 2014)
		10	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
Antihyperuricemin Agents	Allopurinol	1.3	Spain	Soil from agricultural fields	(Aznar et al. 2014)

Table 1 (continued)

Category of Pharmaceuticals	API	Concentration (ng/g)	Country	Type of Sample	Reference
Antihypertensives (Calcium Channel Blockers)	Diltiazem	0.13	USA, Colorado	Irrigated soil with reclaimed water	(Kinney et al. 2006)
	Dehydronifedipine	1.39	USA, Colorado	Irrigated soil with reclaimed water	(Kinney et al. 2006)
Antihypertensives (Beta Blockers)	Propranolol	26	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
	Metoprolol	21	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
Anticoagulants	Warfarin	23.9	USA, Colorado	Irrigated soil with reclaimed water	(Kinney et al. 2006)
Antidepressants	Fluoxetine	8.8	USA, Colorado	Irrigated soil with reclaimed water	(Kinney et al. 2006)
Benzodiazepines	Diazepam	About 10–36	South Africa	Sewage sludge from 3 WWTPs	(Ademoyegun et al. 2020)
Antihistamines	Diphenhydramine	1.56	USA, Colorado	Irrigated soil with reclaimed water	(Kinney et al. 2006)
Gastrointestinal Medications	Ranitidine	3	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
	Omeprazole	1	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)
Psychostimulants – Psychoactive Drugs	Caffeine	0.89–1.7	USA, Arizona	Irrigated soil with reclaimed wastewater	(Williams and McLain 2012)
		6.81	USA, Colorado	Irrigated soil with reclaimed water	(Kinney et al. 2006)
		About 6	South Africa	Golf course field irrigated by WWTP	(Ademoyegun et al. 2020)
		About 25–85	South Africa	Sewage sludge from 3 WWTPs	(Ademoyegun et al. 2020)
		74	Spain	Sewage sludge from 2 STPs	(Nieto et al. 2010)

Table 1 (continued)

Category of Pharmaceuticals	API	Concentration (ng/g)	Country	Type of Sample	Reference
	Cotinine	3.4	USA, Colorado	Irrigated soil with reclaimed water	(Kinney et al. 2006)
Antiseptics	Triclosan	4.4	Mexico City	Irrigated soil from wastewater channels	(Durán-Alvarez et al. 2009)
		About 3	South Africa	Golf course field irrigated by WWTP	(Ademoyegun et al. 2020)
		About 12–89	South Africa	Sewage sludge from 3 WWTPs	(Ademoyegun et al. 2020)
Plasticizers in Pharmaceutical Industry	Bisphenol A	19.88–36.40	France	Loamy soil – public garden	(Wakim et al. 2024)
	Butylbenzylphthalate	131.0	Mexico City	Irrigated soil from wastewater channels	(Durán-Alvarez et al. 2009)
	Bis-2-ethylhexyphthalate	820.0	Mexico City	Irrigated soil from wastewater channels	(Durán-Alvarez et al. 2009)
	Di-n-butylphthalate	244.0	Mexico City	Irrigated soil from wastewater channels	(Durán-Alvarez et al. 2009)
Other Chemicals in Pharmaceutical Industry	4-nonylphenol	41.0	Mexico City	Irrigated soil from wastewater channels	(Durán-Alvarez et al. 2009)

found in all samples (up to 4.4 ng/g), along with mefenamic acid (1.5 ng/g), ibuprofen (0.5 ng/g), fenoprofen (0.8 ng/g), and naproxen (0.7 ng/g). Paracetamol (0.4 ng/g) was also frequently detected in the soil samples. Although gemfibrozil (3.6 ng/g) and fenofibrate (7.0 ng/g) were detected in relative high concentrations, they were present in limited samples, while clofibric acid (0.7 ng/g) was found only in one field in relative lower concentrations. Additionally, carbamazepine (1.2 ng/g) and allopurinol (1.3 ng/g) were found in nearly half of the samples (Aznar et al. 2014).

In another study, soils irrigated by wastewater channels in Mexico City were analyzed (Durán-Alvarez et al. 2009). Carbamazepine was detected at elevated levels (6.48 ng/g), suggesting its environmental persistence, while naproxen (0.55 ng/g) and ibuprofen (0.25 ng/g) were barely measurable, and estrogens were below detection limits.

Veterinary antibiotics are also major contributors to soil contamination. In a systematic review, oxytetracycline concentrations of 6.7 ng/g after the use of manure were reported, while tetracycline levels ranging from 86 to 199 µg/g were found after application of swine manure. Sulfamethazine and tylosin were also present in lower concentrations. The existence of these veterinary antibiotics in soil samples raises the concern of plant uptake and their potential bioaccumulation (Tasho and Cho 2016).

The antibiotics persistence in soil and in leaves of vegetables was also investigated in a study conducted in India (Akhter et al. 2023). Ofloxacin at a maximum concentration of 0.265 ng/g of soil, as well as amoxicillin and erythromycin from an agricultural field irrigated with water from a nearby river were observed. These antibiotics were also present in green vegetables from the same fields, sometimes at higher concentrations, namely ofloxacin (5.586 ng/g), amoxicillin (0.374 ng/g), and erythromycin (0.320 ng/g).

A study conducted across three sites in Colorado, USA, detected 13 pharmaceutical compounds in soils. Erythromycin had the highest concentration (202 ng/g), followed by sulfamethoxazole (9.13 ng/g) and trimethoprim (1.22 ng/g). Other pharmaceuticals identified in the soil samples included paracetamol (33.2 ng/g), warfarin (23.9 ng/g), carbamazepine (17.6 ng/g), fluoxetine (8.8 ng/g), miconazole (1.41 ng/g), diphenhydramine (1.56 ng/g), diltiazem (0.13 ng/g), dehydronifedipine (1.39 ng/g), caffeine (6.81 ng/g) and cotinine (3.4 ng/g) (Kinney et al. 2006). Reclaimed water use was identified as the primary pharmaceutical contamination and accumulation source in these soil samples.

However, pharmaceuticals were prevalent not only in soil samples, but also in sewage sludge, as reported in South Africa. Chloramphenicol, paracetamol and ibuprofen concentrations in sludge approached 100 ng/g, with caffeine and the antibacterial agent triclosan detected at approximately 80 ng/g. Other drugs found in the sludge included carbamazepine, aspirin, diclofenac, diazepam, codeine and trimethoprim. These compounds predominantly accumulated in surface soil layers, diminishing with depth (Ademoyegun et al. 2020).

In Spain, sludge from two sewage treatment plants (STPs) was also collected and analyzed for pharmaceutical contaminants. This study revealed high levels of sulfonamides, such as sulfamethoxazole (112 ng/g), sulfatiazole (37 ng/g), sulfapyridine (38 ng/g) and sulfamethazine (15 ng/g), as well as trimethoprim (17 ng/g), the macrolide roxithromycin (614 ng/g) and the veterinary antibiotic tylosin (1453 ng/g). Moreover, paracetamol (419 ng/g), naproxen (4 ng/g), diclofenac (83 ng/g) and ibuprofen (114 ng/g) were also detected in the sludge samples. Additional contaminants in the sludge from both treatment plants included caffeine (74 ng/g), carbamazepine (42 ng/g), propranolol (26 ng/g), bezafibrate (9 ng/g) and clofibric acid (10 ng/g), as well as multiple hormonal medications in significant amounts, such as 17α-ethinylestradiol (313 ng/g), diethylstilbestrol (184 ng/g), estrone (137 ng/g), estriol (113 ng/g), estradiol-3-sulfate (28 ng/g) and estrone-3-sulfate (28 ng/g). Metoprolol (21 ng/g), ranitidine (3 ng/g) and omeprazole (1 ng/g) were barely identified at least in one of the two sludge treatment plants (Nieto et al. 2010). Hence, these findings raise significant public health concerns mainly due to the potentially developed antibiotic resistance bacteria and the endocrine-disrupting effects.

5 Biochar Technology for Pharmaceutical Remediation of Contaminated Soils

5.1 Biochar and its Properties

Biochar is a stable, carbonaceous material produced through the pyrolysis of organic biomass, like wood and plants, at temperatures between 300 °C and 700 °C under limited or no oxygen conditions. This process excludes fossil fuel-derived feedstocks (Bhattacharya et al. 2024). Specifically, biochar is recognized as a means of carbon sequestration, and a soil amendment due to its ability to reduce ammonia, greenhouse emissions and to adsorb various environmental pollutants (Sanchez-Hernandez et al. 2019).

Several thermochemical methods have been developed for biochar production, including pyrolysis, hydrothermal carbonization, gasification, and torrefaction (Yaashikaa et al. 2020). The choice of method depends on the desired biochar characteristics, biomass type, pyrolysis temperature, and residence time. Parameters such as feedstock type (e.g., wood, agricultural residues, or sewage sludge), heating rate, and particle size significantly influence the resulting biochar's physical and chemical properties, especially its adsorption capacity (Gai et al. 2014; Tomczyk et al. 2020; Yaashikaa et al. 2020).

5.1.1 Biochar's Physical Properties

The efficiency of biochar in remediating pharmaceutical-contaminated soils is closely linked to its physical characteristics, particularly specific surface area (SSA), pore size, and total pore volume (V) (Dayoub et al. 2024). Studies have shown that higher pyrolysis temperatures enhance surface area and porosity, improving biochar's adsorption capacity (Tomczyk et al. 2020). Biomass material is also a key factor for succeeding optimal physical properties.

Biochar contains macropores (>50 nm) facilitating microbial colonization, mesopores (2–50 nm) and micropores (<2 nm), which are primarily responsible for contaminant trapping and mass transfer. The V typically ranges from 0.016 to 0.083 mL/g, but it can reach up to 1.8 mL/g (Nguyen et al. 2023). SSA generally fall between 8 and 132 m^2/g, although values as high as approximately 3300 m^2/g have been reported under optimal conditions. Woody feedstocks tend to produce biochar with superior SSA and porosity, in comparison to non-woody materials (Ippolito et al. 2020), affecting the adsorption capacity of biochar (Nguyen et al. 2023; Dayoub et al. 2024).

5.1.2 Biochar's Chemical Properties

The chemical properties are also critical for the adsorption capacity of biochar. Key parameters include electrical conductivity, pH, carbon and ash contents, macronutrients percentage and cation exchange capacity (Nguyen et al. 2023). These characteristics are also largely influenced by feedstock type and pyrolysis temperature (Dayoub et al. 2024).

The cation exchange capacity reflects the biochar's ability to retain positively charged ions (cations) and plays a crucial role in both soil fertility and pollutant adsorption. It generally decreases with increasing pyrolysis temperature (Tomczyk et al. 2020). Biochar from agricultural waste, manure, or sewage sludge typically showing higher values than those from woody biomass (Ippolito et al. 2020).

Biochar yield decreases as pyrolysis temperature increases (Kamarudin et al. 2022). High yields are often associated with non-woody biomass such as sewage sludge and animal manure, which also results in higher ash content, sometimes exceeding 65% at 700 °C. In contrast, wood-based biochars, such eucalyptus wood generally exhibits lower ash content but higher carbon content, which is advantageous for stability and sorption capacity. The carbon and ash contents were found to be increased, generally with increasing temperature (Jafri et al. 2018; Tomczyk et al. 2020; Dayoub et al. 2024).

Biochar pH is typically neutral to weakly alkaline across feedstock types, with higher pyrolysis temperatures resulting in increased pH values (Tomczyk et al. 2020). Similar behavior is also observed for the electrical conductivity. Nitrogen content generally declines with temperature, whereas phosphorus content increases. Potassium levels vary less but are highest in rice straw-derived biochars and lowest in those from eucalyptus wood (Dayoub et al. 2024).

Two additional crucial factors describing the chemical properties of biochar are the O/C and H/C ratios. An O/C ratio below 0.2 signifies high stability, while values above 0.6 indicate a less stable product. Both O/C and H/C ratios decrease with increasing pyrolysis temperature, regardless the feedstock type, due to the loss of volatile compounds and reduced hydrogen and oxygen content (Tomczyk et al. 2020; Nguyen et al. 2023; Dayoub et al. 2024). Thus, a meaningful rise of biochar stability (O/C ratio below 0.2) could be succeeded by increasing the pyrolysis temperature.

Therefore, in selecting biochar for pharmaceutical remediation, a balance between physical and chemical properties and stability of biochar is essential. The choice of biomass feedstock and optimization of pyrolysis conditions are critical for producing biochar with high potential of adsorption and long-term environmental effectiveness (Ippolito et al. 2020).

5.2 Pharmaceutical Remediation through Biochar

Biochar's ability to remediate soil and wastewater contaminated with multiple pollutants is well-established (Dimitriadou et al. 2025). Its high efficiency in adsorbing pharmaceuticals is primarily attributed to its large surface area and high porosity, which facilitate both physical and chemical adsorption of organic contaminants. The surface of biochar contains various functional groups, such as hydroxyl (-OH), carboxyl (-COOH), and carbonyl groups (-C=O). These functional groups can interact with pharmaceutical pollutants through physical adsorption mechanisms, such as hydrogen bonding, hydrophobic or electrostatic interactions, and pore-filling, or through chemical adsorption mechanisms like surface complexation, ion pairing, and π-π interactions (Kang et al. 2022; Qiu et al. 2022). The adsorption process involves steps such as external diffusion, particle diffusion, and surface reaction. Multiple studies show a wide variety of pharmaceuticals in soils and assess biochar's effectiveness in adsorbing them (Table 2).

5.2.1 Antibiotics

Antibiotics, such as tetracycline can be removed through a two-stage adsorption process. The first stage is rapid, characterized by the diffusion of the antibiotic into the micropo-

Table 2 Summary of pharmaceuticals remediation using Biochar; the feedstock types and pyrolysis conditions for the generation of the Biochar, its specific surface area and the adsorption capacity of the API on the Biochar

Category of Pharmaceuticals	API	Biochars				References
		Feedstock	Pyrolysis Parameters	Specific Surface Area (m^2/g)	Adsorption Capacity (mg/g) [1]	
Antibiotics (Tetracyclines)	Tetracycline	Rice straw	700 °C for 2 h	372.2	552.0	(Chen et al. 2018)
		Swine manure	700 °C for 2 h	319.0	365.4	(Chen et al. 2018)
		Municipal Solid Wastes	500 °C for 30 min	8.7	78.0	(Premarathna et al. 2019)
		Agricultural wastes	300 °C for 12 h	Not applicable	9.5	(Hoslett et al. 2021)
		Peanut husk	600 °C for 1 h	Not applicable	71.7	(Egbedina et al. 2023)
Antibiotics (β-lactams)	Cephalexin	Oil palm fiber	550 °C for 30 min	76.1	7.9	(Grisales-Cifuentes et al. 2021)
	Amoxicillin	Banana pseudostem fibers	350 °C for 2 h	100.9	100.0	(Chakhtouna et al. 2021)
			650 °C for 2 h	190.5	99.8	
		Peanut husk	600 °C for 1 h	Not applicable	28.5	(Egbedina et al. 2023)
Antibiotics (Fluoroquinolones)	Ciprofloxacin	Corncob	600 °C for 2 h	306.0	0.4	(Dang et al. 2022)
		Used tea leaves	450 °C for 30 min	8.1	238.1	(Li et al. 2018)
		Municipal Solid Wastes	450 °C for 4 h	4.3	122.2	(Ashiq et al. 2019)
		Municipal Solid Wastes with montmorillonite composite	450 °C for 4 h	6.5	167.4	
	Ofloxacin	Textile milled sludge modified with iron oxide particles	400 °C for 4 h	91.0	19.7	(Singh and Srivastava 2020)

Table 2 (continued)

Category of Pharmaceuticals	API	Biochars				References
		Feedstock	Pyrolysis Parameters	Specific Surface Area (m^2/g)	Adsorption Capacity (mg/g) [1]	
	Levofloxacin	Corn husks with pre-pyrolysis $FeCl_3$ impregnation	300 °C for 1 h	112.5	56.6	(Chen et al. 2019)
		Corn husks with post-pyrolysis $FeCl_3$ impregnation	300 °C for 1 h	94.9	273.7	
Antibiotics (Chloramphenicol)	Chloramphenicol	Wood of *Eucalyptus globulus*	400 °C for 2 h	Not applicable	21.4	(Ahmed et al. 2017)
Antibiotics (Sulfonamides)	Sulfamethoxazole	Wood of *Eucalyptus globulus*	400 °C for 2 h	Not applicable	28.3	(Ahmed et al. 2017)
		Raw bamboo	450 °C for 2 h	299.0	25.7	(Huang et al. 2020)
		Conifer cone	600 °C for 1 h	110.0	3% removal rate	(Imreová et al. 2024)
		Plum kernel	600 °C for 1 h	128.0	43% removal rate	
		"Biochar 4073" and conifer cone (1:2)	600 °C for 1 h	330.0	46% removal rate	
	Sulfathiazole	Wood of *Eucalyptus globulus*	400 °C for 2 h	Not applicable	45.2	(Ahmed et al. 2017)
	Sulfamethazine	Wood of *Eucalyptus globulus*	400 °C for 2 h	Not applicable	20.7	(Ahmed et al. 2017)
		Tea waste	700 °C for 7 h	421.3	33.8	(Rajapaksha et al. 2014)
	Sulfapyridine	Hickory chips	450 °C for 2 h	299.0	58.6	(Huang et al. 2020)
		Conifer cone	600 °C for 1 h	110.0	0% removal rate	(Imreová et al. 2024)
		Plum kernel	600 °C for 1 h	128.0	48% removal rate	
		"Biochar 4073" and conifer cone (1:2)	600 °C for 1 h	330.0	80% removal rate	

Table 2 (continued)

Category of Pharmaceuticals	API	Biochars				References
		Feedstock	Pyrolysis Parameters	Specific Surface Area (m^2/g)	Adsorption Capacity (mg/g) [1]	
Antibiotics (Trimethoprim)	Trimethoprim	Conifer cone	600 °C for 1 h	110.0	6% removal rate	(Imreová et al. 2024)
		Plum kernel	600 °C for 1 h	128.0	31% removal rate	
		"Biochar 4073" and conifer cone (1:2)	600 °C for 1 h	330.0	95% removal rate	
Antibiotics (Macrolides)	Clarithromycin	Conifer cone	600 °C for 1 h	110.0	0% removal rate	(Imreová et al. 2024)
		Plum kernel	600 °C for 1 h	128.0	0% removal rate	
		"Biochar 4073" and conifer cone (1:2)	600 °C for 1 h	330.0	91% removal rate	
Antibiotics (Lincosamides)	Clindamycin	Conifer cone	600 °C for 1 h	110.0	0% removal rate	(Imreová et al. 2024)
		Plum kernel	600 °C for 1 h	128.0	24% removal rate	
		"Biochar 4073" and conifer cone (1:2)	600 °C for 1 h	330.0	84% removal rate	
NSAIDs	Naproxen	Walnut shells	600 °C for 2 h	786.0	533.0	(Anfar et al. 2020)
	Ketoprofen	Walnut shells	600 °C for 2 h	786.0	494.0	(Anfar et al. 2020)
	Salicylic Acid	Walnut shells	600 °C for 2 h	786.0	683.0	(Anfar et al. 2020)
	Diclofenac Sodium	Rice hull	Torrefaction at 350 °C for 1 h	Not applicable	3.3	(Filipinas et al. 2021)
		Moringa seeds powder	450 °C for 2 h	Not applicable	100.9	(Bagheri et al. 2020)

Table 2 (continued)

Category of Pharmaceuticals	API	Biochars				References
		Feedstock	Pyrolysis Parameters	Specific Surface Area (m^2/g)	Adsorption Capacity (mg/g) [1]	
		Conifer cone	600 °C for 1 h	110.0	0% removal rate	(Imreová et al. 2024)
		Plum kernel	600 °C for 1 h	128.0	26% removal rate	
		"Biochar 4073" and conifer cone (1:2)	600 °C for 1 h	330.0	70% removal rate	
	Ibuprofen	Coconut shells activated by steam	450 °C for 1 h	726.0	9.7	(Chakraborty et al. 2019)
		Coconut shells activated chemically by H_3PO_4	450 °C for 1 h	805.0	12.2	
		Date stone seeds activated by steam	700 °C for 1 h	513.0	9.7	(Chakraborty et al. 2020)
		Date stone seeds activated chemically by H_3PO_4	700 °C for 1 h	342.0	12.2	
		Wood apple fruit shell without modification	650 °C for 1 h	4.4	5.0	(Chakraborty et al. 2018)
		Wood apple fruit shell activated by steam	650 °C for 1 h	308.0	12.7	
		Wood chips	800 °C for 7 h	841.0	132.0	(Luo et al. 2020)
		Mung bean husk activated by steam	550 °C for 1 h	Not applicable	59.8	(Mondal et al. 2016)
Non-opioid analgesic	Paracetamol	Oil palm fibers	550 °C for 30 min	76.1	7.3	(Grisales-Cifuentes et al. 2021)
		Wood chips	800 °C for 7 h	841.0	196.0	(Luo et al. 2020)

Table 2 (continued)

Category of Pharmaceuticals	API	Biochars				References
		Feedstock	Pyrolysis Parameters	Specific Surface Area (m^2/g)	Adsorption Capacity (mg/g) [1]	
		Pomelo peels	700 °C for 3 h	1033.0	147.0	(Tran et al. 2020)
		Pure glucose	900 °C for 3 h	1292.0	286.0	
Antidepressants	Fluoxetine	Com-mercial biochar	Not available	270.0	4.1	(Escudero-Curiel et al. 2021)
	Citalopram	Conifer cone	600 °C for 1 h	110.0	36% removal rate	(Imreová et al. 2024)
		Plum kernel	600 °C for 1 h	128.0	28% removal rate	
		"Biochar 4073" and conifer cone (1:2)	600 °C for 1 h	330.0	97% removal rate	
	Venlafaxine	Conifer cone	600 °C for 1 h	110.0	0% removal rate	(Imreová et al. 2024)
		Plum kernel	600 °C for 1 h	128.0	28% removal rate	
		"Biochar 4073" and conifer cone (1:2)	600 °C for 1 h	330.0	77% removal rate	
Anticonvulsants	Carbamazepine	Conifer cone	600 °C for 1 h	110.0	0% removal rate	(Imreová et al. 2024)
		Plum kernel	600 °C for 1 h	128.0	30% removal rate	
		"Biochar 4073" and conifer cone (1:2)	600 °C for 1 h	330.0	84% removal rate	
	Lamotrigine	Conifer cone	600 °C for 1 h	110.0	4% removal rate	(Imreová et al. 2024)
		Plum kernel	600 °C for 1 h	128.0	35% removal rate	
		"Biochar 4073" and conifer cone (1:2)	600 °C for 1 h	330.0	78% removal rate	

Table 2 (continued)

Category of Pharmaceuticals	API	Biochars				References
		Feedstock	Pyrolysis Parameters	Specific Surface Area (m^2/g)	Adsorption Capacity (mg/g) [1]	
Psychostimulants – Psychoactive Drugs	Caffeine	Pine needles oxidized with boiling HNO_3	650 °C	Not applicable	6.5	(Anastopoulos et al. 2020)
		Bovine bone and $AlCl_3$ and $MgCl_2$ loading	650 °C	46.3	26.2	(dos Santos Lins et al. 2019)
Opioid Drugs	Tramadol	Conifer cone	600 °C for 1 h	110.0	0% removal rate	(Imreová et al. 2024)
		Plum kernel	600 °C for 1 h	128.0	30% removal rate	
		"Biochar 4073" and conifer cone (1:2)	600 °C for 1 h	330.0	77% removal rate	
Antihypertensives	Valsartan	Oil palm fibers	550 °C for 30 min	76.1	23.9	(Grisales-Cifuentes et al. 2021)
	Telmisartan	Conifer cone	600 °C for 1 h	110.0	31% removal rate	(Imreová et al. 2024)
		Plum kernel	600 °C for 1 h	128.0	25% removal rate	
		"Biochar 4073" and conifer cone (1:2)	600 °C for 1 h	330.0	96% removal rate	
	Metoprolol	Conifer cone	600 °C for 1 h	110.0	7% removal rate	(Imreová et al. 2024)
		Plum kernel	600 °C for 1 h	128.0	25% removal rate	
		"Biochar 4073" and conifer cone (1:2)	600 °C for 1 h	330.0	96% removal rate	

Table 2 (continued)

Category of Pharmaceuticals	API	Biochars				References
		Feedstock	Pyrolysis Parameters	Specific Surface Area (m^2/g)	Adsorption Capacity (mg/g) [1]	
	Sotalol	Conifer cone	600 °C for 1 h	110.0	0% removal rate	(Imreová et al. 2024)
		Plum kernel	600 °C for 1 h	128.0	31% removal rate	
		"Biochar 4073" and conifer cone (1:2)	600 °C for 1 h	330.0	69% removal rate	
	Bisoprolol	Conifer cone	600 °C for 1 h	110.0	11% removal rate	(Imreová et al. 2024)
		Plum kernel	600 °C for 1 h	128.0	11% removal rate	
		"Biochar 4073" and conifer cone (1:2)	600 °C for 1 h	330.0	97% removal rate	
Antihistamines	Cetirizine	Conifer cone	600 °C for 1 h	110.0	0% removal rate	(Imreová et al. 2024)
		Plum kernel	600 °C for 1 h	128.0	20% removal rate	
		"Biochar 4073" and conifer cone (1:2)	600 °C for 1 h	330.0	85% removal rate	
	Fexofenadine	Conifer cone	600 °C for 1 h	110.0	0% removal rate	(Imreová et al. 2024)
		Plum kernel	600 °C for 1 h	128.0	27% removal rate	
		"Biochar 4073" and conifer cone (1:2)	600 °C for 1 h	330.0	87% removal rate	

Table 2 (continued)

Category of Pharmaceuticals	API	Biochars				References
		Feedstock	Pyrolysis Parameters	Specific Surface Area (m^2/g)	Adsorption Capacity (mg/g) [1]	
Hormonal Medications	17α-ethinylestradiol	Grain and straw residues	650 °C for 1 h	Not applicable	0.13–0.37	(Caban et al. 2020)
		Sugarcane harvest residues	550 °C	58.9	0.055	(Wei et al. 2019)
		Corn stalks	500 °C for 1.5 h	Not applicable	Not applicable	(Xu et al. 2015)
	Estrone	Forest pine wood	650 °C for 30 min	544	Not applicable	(Li et al. 2020)
		Softwood	450 °C	Not applicable	Not applicable	(Mann et al. 2016)
	17β-estadiol	Softwood	450 °C	Not applicable	Not applicable	(Mann et al. 2016)
		Wheat straw	400 °C for 6 h	43.4	3.1	(Zhang et al. 2017)
		Rice straw		28.0	2.8	
		Corn straw		52.6	2.7	
	Progesterone	Spruce trees	450 °C (for softwood) for 2.5 h 750 °C (for hardwood) for a few min	Not applicable	Not applicable	(Alizadeh et al. 2016)

[1] The adsorption capacities were extracted from the respective publications under their reported experimental conditions

rous structure of biochar surface. This stage also involves an increase in π-π interactions and hydrogen bonding. These hydrogen bonds occur primarily between the hydroxyl and carboxyl groups on biochar and the amine ($-NH_2$), phenolic (-OH), and carbonyl groups of tetracycline, while the aromatic rings of both molecules promote π–π stacking interactions. The second stage corresponds to the equilibrium phase, during which biochar becomes saturated, making it the slower phase of adsorption (Kang et al. 2022). Various feedstocks have been employed for the production of biochars aimed at eliminating tetracycline (Chauhan et al. 2023). These include rice straw or swine manure heated at 700 °C for 2 h (Chen et al. 2018), solid wastes heated at 500 °C for 30 min (Premarathna et al. 2019) and agricultural waste pyrolyzed at 300 °C for 12 h (Hoslett et al. 2021). The highest SSA and adsorption capacity (q_t) were observed for biochar produced from rice straw (SSA = 372.2 m^2/g, V = 0.23 mL/g, q_t = 552.0 mg/g), followed by biochar derived from swine manure, which had a larger V (SSA = 319.0 m^2/g, V = 0.25 mL/g, q_t = 365.4 mg/g) (Chen et al. 2018). A peanut husk biochar prepared by heating at 600 °C for 1 h resulted in a relatively low adsorption capacity of 71.7 mg/g (Egbedina et al. 2023).

Cephalexin is adsorbed by biochar through hydrogen bonding between the polar groups of biochar and the amine (-NH) functional groups of cephalexin, electrostatic interactions, and π-π stacking (Kang et al. 2022). A biochar generated by pyrolysis of oil palm fiber at 550 °C for 30 min was tested, exhibiting a satisfactory SSA (76.1 m^2/g) and V (0.12 mL/g), but its adsorption capacity was relatively low (q_t = 7.9 mg/g) (Grisales-Cifuentes et al. 2021). pH was found to be crucial for the adsorption capacity of cephalexin, with optimal adsorption occurring at an acidic pH (pH = 3) due to stronger hydrogen bonds and π-π interactions between the API and the surface of the adsorbent (Grisales-Cifuentes et al. 2021; Chauhan et al. 2023).

The same adsorption mechanisms have also been reported for amoxicillin (Chauhan et al. 2023). Banana pseudostem fibers pyrolyzed at 350 °C to 650 °C for 2 h with a rate of 10 °C/min, were used as feedstock for biochar production aimed at amoxicillin removal. This biochar exhibited a high SSA (100.9–190.5 m^2/g) and an adsorption capacity of approximately 100 mg/g at neutral pH (Chakhtouna et al. 2021). Another study tested the application of peanut husk biochar generated after pyrolysis at 600 °C for 1 h for the remediation of amoxicillin. This biochar resulted in a maximum adsorption capacity of 28.5 mg/g (Egbedina et al. 2023).

Similarly, ciprofloxacin is adsorbed by biochar through the same mechanisms (Kang et al. 2022). Biochar derived from corncob heated at 600 °C for 2 hours resulted in the highest SSA (306 m^2/g), but the lowest adsorption capacity (q_t = 0.4 mg/g) at acidic pH (pH = 4) (Dang et al.). The highest adsorption capacity (q_t = 238.1 mg/g) was achieved by biochar prepared from used tea leaves heated at 450 °C for 30 min, which had a low SSA (8.06 m^2/g) and V (0.01 mL/g) (Li et al. 2018). Another feedstock, municipal solid waste pyrolyzed at 450 °C for 4 h, also achieved high adsorption capacity (q_t = 122.2 mg/g) but had a low SSA (4.33 m^2/g) (Ashiq et al. 2019). Further modification of this biochar with a montmorillonite composite resulted in a 50% increase in the SSA and a 37% increase in the adsorption capacity (Ashiq et al. 2019).

Ofloxacin was successfully removed using biochar produced by the thermal treatment of textile milled sludge at 400 °C for 4 h and modified with iron oxide particles incorporated into the mesopores. The biochar had a SSA of 91.0 m^2/g at pH 6 and an adsorption capacity of 19.7 mg/g for ofloxacin removal (Singh and Srivastava 2020).

Biochar has potential as an adsorbent for levofloxacin remediation. A more complex adsorption mechanism involving physical and chemical pathways, including electrostatic interactions, hydrogen bonding between the proton acceptors (-COOH, -C = O) of levofloxacin and the donor groups of the biochars (-COOH and -OH), π-π stacking, electron donor-acceptor interactions, and functional group surface complexation, has been described (Kang et al. 2022). Corn husks were used as feedstock, which were pyrolyzed at 300 °C for 1 h, with two modification schemes tested. In the first, $FeCl_3$ was impregnated prior to pyrolysis, resulting in a biochar with elevated SSA (112.45 m^2/g) but low adsorption capacity (q_t = 56.6 mg/g). In the second, $FeCl_3$ impregnation occurred after pyrolysis, leading to a slight change in SSA (94.9 m^2/g) but a significant increase in adsorption capacity (q_t = 273.7 mg/g) (Chen et al. 2019).

Sulfamethoxazole is adsorbed by biochar through π-π interactions, hydrogen bonding between electronegative nitrogen and oxygen atoms of the API with dissociated hydrogen atoms of biochar -OH and -COOH groups, and hydrophobic interactions, whereas only π-π interactions are involved in the adsorption of sulfadiazine (Kang et al. 2022). Remediation of sulfamethoxazole was achieved using biochar produced from the pyrolysis of *Eucalyptus globulus* wood at 400 °C for 2 h (Ahmed et al. 2017) and raw bamboo at 450 °C for 2 h (Huang et al. 2020). Both biochars exhibited sulfamethoxazole adsorption capacities between 25 mg/g and 30 mg/g, with the bamboo-derived biochar having a high SSA (299.0 m²/g). The biochar from *Eucalyptus globulus* was also effective for removing chloramphenicol, sulfathiazole, and sulfamethazine, with adsorption capacities of 21.4 mg/g, 45.2 mg/g, and 20.7 mg/g, respectively (Ahmed et al. 2017). Additionally, tea waste biochar heated at 700 °C for 2 h achieved an increased adsorption capacity for sulfamethazine (q_t = 33.8 mg/g) and a very high SSA (421.3 m²/g) (Rajapaksha et al. 2014). For sulfapyridine removal, biochar derived from hickory chips pyrolyzed at 450 °C for 2 h exhibited a large SSA (299 m²/g) and a high adsorption capacity (q_t = 58.6 mg/g) (Huang et al. 2020).

5.2.2 Non-Steroidal Anti-Inflammatory Drugs (NSAIDs)

Naproxen has been effectively remediated via adsorption onto biochar produced from the pyrolysis of walnut shells at 600 °C for 2 h. This biochar exhibited a high SSA (786.0 m²/g) and an excellent adsorption capacity for naproxen (q_t = 533.0 mg/g) (Anfar et al. 2020). The adsorption mechanism involves only π-π interactions (Kang et al. 2022). The same walnut shell-derived biochar was also used for the removal of ketoprofen providing an adsorption capacity of 494 mg/g. Additionally, this biochar demonstrated a very high adsorption capacity for salicylic acid (q_t = 683.0 mg/g) (Anfar et al. 2020).

Several biochars have been tested for the remediation of sodium diclofenac, achieving satisfactory removal rates. Biochar produced through torrefaction of rice hulls at 350 °C for 1 h resulted in a modest adsorption capacity (q_t = 3.3 mg/g) (Filipinas et al. 2021). A more effective approach involved the pyrolysis of moringa seed powder at 450 °C for 2 h, which yielded a significantly higher adsorption capacity (q_t = 100.9 mg/g) (Bagheri et al. 2020). The presence of carbonyl and carboxyl groups on the surface of this biochar played a key role in the adsorption of diclofenac sodium (Chauhan et al. 2023). Furthermore, three different biochars were tested for diclofenac removal. Among them, only the mixed biochar (a blend of commercial biochar and conifer cone biochar) demonstrated a high removal efficiency, achieving a 70% reduction in diclofenac concentration (Imreová et al. 2024).

Ibuprofen, a widely used over-the-counter analgesic and antipyretic, is adsorbed onto biochar via π-π interactions, hydrogen bonding among the carboxylic group of the API and the hydroxyl (-OH), carbonyl (-C=O) and carboxylic groups of biochars, and pore-filling mechanisms (Kang et al. 2022). Various biomass sources have been evaluated for biochar production to remove ibuprofen, yielding similar adsorption capacities (q_t = 9–13 mg/g) in most cases (Chauhan et al. 2023). Among the studies presented in Table 2, the highest adsorption capacity was recorded for the biochar derived from wood chips (q_t = 132.0 mg/g), which also had the highest SSA (841.0 m²/g) (Luo et al. 2020). The second-highest adsorption capacity was observed with mung bean husk-derived biochar (q_t = 59.8 mg/g) (Mondal et al. 2016). In nearly all cases, the SSA exceeded 300 m²/g (Mondal et al. 2016; Chakraborty et al. 2019, 2020; Luo et al. 2020), except for the wood apple fruit shell biochar without modification, which had a much lower SSA (4.4 m²/g) and the lowest adsorption capacity (q_t = 5.0 mg/g) (Chakraborty et al. 2018).

5.2.3 Acetaminophen (Paracetamol)

The adsorption of acetaminophen onto various biochars has been reported in several studies (Chauhan et al. 2023). The adsorption mechanisms primarily involve π-π interactions, hydrogen bonding among the -OH, -C=O and -NH groups of paracetamol, which serve as proton acceptors, and the proton donor groups (-OH, -COOH) of biochars, as well as electrostatic interactions between the pharmaceutical and the biochar surface (Kang et al. 2022). For instance, pyrolysis of oil palm fibers at 550 °C for 30 min produced a biochar with a relatively low SSA (76.1 m^2/g) and modest adsorption capacity for paracetamol (q_t = 7.3 mg/g) (Grisales-Cifuentes et al. 2021). In contrast, other feedstocks yielded significantly better results. Biochar produced from wood chips at 800 °C for 7 h (Luo et al. 2020), pomelo peels at 700 °C for 3 h, and pure glucose at 900 °C for 3 h (Tran et al. 2020) demonstrated significantly higher adsorption capacities and SSA (Table 2) (Tran et al. 2020; Luo et al. 2020).

5.2.4 Antidepressants

Fluoxetine can be removed from environmental samples through adsorption onto biochar, primarily via π-π interactions (Kang et al. 2022). A commercially available biochar with a SSA of 270 m^2/g was employed for fluoxetine remediation. Its adsorption capacity, determined using the Freundlich isotherm, was calculated to be 4.1 mg/g (Escudero-Curiel et al. 2021).

In another study, a blended biochar was significantly more effective than the individual biochar in reducing the concentration of citalopram, achieving up to 97% removal. A similar trend was observed for venlafaxine, with the mixed biochar achieving a 77% removal efficiency (Imreová et al. 2024).

5.2.5 Anticonvulsants

Carbamazepine and lamotrigine were effectively removed from contaminated samples using the blended biochar developed by Imreová et al. This biochar achieved 84% removal of carbamazepine and 78% removal of lamotrigine (Imreová et al. 2024).

5.2.6 Psychostimulants – Psychoactive Drugs

Caffeine, a common psychostimulant found in various pharmaceuticals (e.g., analgesics, cold/flu medications, antihistamines), has also been investigated for removal using biochar. The adsorption mechanisms include outer-sphere complexation, electrostatic interactions, and other physical adsorption processes (Chauhan et al. 2023). Two different biochars have been studied for caffeine removal. One was produced from pine needles pyrolyzed at 650 °C and subsequently oxidized with boiling HNO_3, achieving an adsorption capacity of 6.54 mg/g (Anastopoulos et al. 2020). The other was derived from bovine bone, treated at 650 °C and modified by $AlCl_3$ and $MgCl_2$ loading, which showed a significantly higher adsorption capacity of 26.2 mg/g (SSA=46.3 m^2/g) (dos Santos Lins et al. 2019).

5.2.7 Opioid Drugs

For the removal of tramadol, the three biochars developed by Imreova et al. were evaluated. The blended biochar achieved a significantly higher removal efficiency of 77% (Imreová et al. 2024).

5.2.8 Antihypertensives

Valsartan was removed using biochar produced from oil palm fibers pyrolyzed at 550 °C for 30 min. This biochar exhibited a SSA of 76.1 m^2/g and an adsorption capacity of 23.9 mg/g (Grisales-Cifuentes et al. 2021). The primary adsorption mechanism involved hydrogen bonding between the carboxylic and carbonyl groups of valsartan and the phenolic groups on the biochar surface. Additional interactions, such as π-π stacking, hydrophobic effects, electrostatic interactions, and pore-filling also contributed to the overall adsorption process, though to a lesser extent (Grisales-Cifuentes et al. 2021).

Another angiotensin II receptor antagonist, telmisartan, as well as three beta-blockers, namely bisoprolol, metoprolol, and sotalol, were also successfully removed using the blended biochar described by Imreova et al., achieving removal rates of up to 97% for these APIs (Imreová et al. 2024).

5.2.9 Antihistamines

The same three (3) biochars were also tested for the remediation of two (2) common antihistamines, fexofenadine and cetirizine. Once again, the blended biochar proved to be the most effective. It achieved removal rates of 87% for fexofenadine and 85% for cetirizine, highlighting its potential as a reliable biochar blend for antihistamine remediation (Imreová et al. 2024).

5.2.10 Hormonal Medications

The synthetic estrogen 17α-ethinylestradiol has been successfully removed from clay and sandy soils using biochar produced from grain and straw residues pyrolyzed at 650 °C for 1 h. Adsorption capacities ranged from 0.13 mg/g to 0.37 mg/g, with π-π stacking and electrostatic interactions identified as the dominant adsorption mechanisms (Caban et al. 2020). Similarly, a biochar derived from sugarcane harvest residues and pyrolyzed at 550 °C was applied to sandy loam and clay soils, achieving an adsorption capacity of up to 0.055 mg/g. This biochar had a SSA of 58.9 m^2/g (Wei et al. 2019). In another study, the removal of 17α-ethinylestradiol from agricultural soils was assessed using biochar derived from corn stalks pyrolyzed at 500 °C for 1.5 h. The key adsorption mechanisms involved pore filling, hydrogen bonding, and π-π stacking (Xu et al. 2015).

Estrone was removed using a biochar produced from forest pine wood pyrolyzed at 650 °C for 30 min. This biochar exhibited a SSA of 544 m^2/g and a V of 0.25 mL/g. The primary adsorption mechanisms were π-π interactions and electrostatic forces (Li et al. 2020). Another biochar, produced from softwood pyrolyzed at 450 °C, was also tested for estrone remediation in a sandy-loam-clay soil mixture (Mann et al. 2016).

In the same study, Mann et al. evaluated the performance of softwood-derived biochar for the removal of a third estrogen, 17β-estradiol (Mann et al. 2016). Additionally, biochars made from wheat, rice, and corn stalks pyrolyzed at 400 °C for 6 h were tested. The wheat straw biochar had a SSA of 43.4 m^2/g, the rice straw 28.0 m^2/g, and the corn straw 52.6 m^2/g. These biochars were applied to greenhouse soils, achieving similar adsorption capacities of approximately 3 mg/g. In this case, pore-filling was identified as the dominant adsorption mechanism for 17β-estradiol (Zhang et al. 2017).

Finally, progesterone was remediated from mixed soil samples (sand, silt, and clay) using biochar produced from spruce trees. The biomass was pyrolyzed at 450 °C for softwood for 2.5 h and 750 °C for hardwood for a few minutes. The main mechanisms driving progesterone adsorption were partitioning and hydrophobic interactions (Alizadeh et al. 2016).

6 The Role of Earthworms in Pharmaceutical Remediation of Contaminated Soils

6.1 Earthworms

Another eco-sustainable method for soil remediation and cleanup of pharmaceutical contaminants is the use of earthworms, a natural component of soil fauna (Tagliabue et al. 2023). Earthworms are commonly found in terrestrial environments and play a crucial role in maintaining soil fertility and structure. They contribute to nutrient cycling by transporting nutrients deeper into the soil profile, making them more accessible to plant roots. Although earthworms can bioaccumulate organic pollutants such as pharmaceutical compounds, raising concerns due to their position within the food web and potential public health risks (Carter et al. 2014), some studies have demonstrated notable biodegradation of APIs by earthworms (Rodriguez-Campos et al. 2014; Carter et al. 2016; Shi et al. 2020; Xiao et al. 2022; Gudeta et al. 2023; Tagliabue et al. 2023; Fučík et al. 2024).

The use of earthworms for the removal of organic pollutants from soils remains an emerging area of study but is gaining gradually increasing attention among researchers (Carter et al. 2016; Xiao et al. 2022; Gudeta et al. 2023; Tagliabue et al. 2023; Fučík et al. 2024). This process, termed vermiremediation, refers to the transformation, degradation, and removal of pollutants from soils and other environmental media through the activity of earthworms (Rodriguez-Campos et al. 2014). Earthworms contribute significantly to altering the physical, chemical, and biological properties of soil through their burrowing and ingestion activities. These behaviors modify soil porosity, organic matter composition, and the spatial distribution of elements within the soil. Due to these ecological functions, earthworms are often referred to as "ecosystem engineers" (Tagliabue et al. 2023).

Several earthworm species are commonly utilized in vermiremediation. These include *Eisenia fetida*, commonly known as red worms (Carter et al. 2014, 2016; Lin et al. 2021; Xiao et al. 2022; Gudeta et al. 2023; Tagliabue et al. 2023; Fučík et al. 2024), *Amynthas robustus* (Lin et al. 2021), *Metaphire guillelmi* (Yin et al. 2021), *Pheretima guillelmi* (Zhang et al. 2022), *Lumbricus terrestris* (Almutairi 2019; Singh et al. 2023) and species of *Perionyx* (Singh et al. 2023). Among these, *E. fetida* is the most widely adopted species due to several advantages, among which it efficiently decomposes organic matter, adapts well to a variety of environmental conditions, reproduces rapidly, and requires minimal maintenance

(Carter et al. 2014, 2016; Xiao et al. 2022; Gudeta et al. 2023; Tagliabue et al. 2023; Fučík et al. 2024). The predominance of *E. fetida* is also linked to its ease of laboratory breeding, short generation times, and extensive experimental background in ecotoxicology and vermiremediation studies.

6.1.1 Mechanisms of Vermiremediation

Earthworms contribute to contaminant removal from soils through both direct and indirect mechanisms. Direct mechanisms include the absorption and digestion of pharmaceuticals through the intestinal epithelium, followed by biotransformation during gut transit. Some pharmaceutical molecules are bioaccumulated within earthworm tissues and subsequently excreted in modified or partially degraded forms (Carter et al. 2014). These transformations are increasingly attributed to gut-associated microbial communities, which play a more active role than previously assumed.

Recent molecular studies using 16 S rRNA gene sequencing, metagenomics, and metatranscriptomics have revealed that the earthworm gut hosts diverse microbial taxa with metabolic capabilities relevant to pharmaceutical degradation. Dominant groups such as *Proteobacteria*, *Actinobacteriota*, *Firmicutes*, and *Bacteroidota* harbor genes encoding enzymes involved in redox, deamination, dehydroxylation, hydrolysis, and conjugation reactions, all of which can modify pharmaceutical compounds (Pass et al. 2015; Sapkota et al. 2020). For example, bacteria from the families *Enterobacteriaceae*, *Pseudomonadaceae*, *Streptomycetaceae*, and *Bacillaceae* are frequently enriched during pharmaceutical exposure and are known to metabolize different kind of pharmaceuticals. In the case of tetracycline, gut microbial sequencing in *E. fetida* has identified increased abundance of genera capable of encoding monooxygenases and esterases, known to catalyze oxidative degradation or deactivation of tetracycline molecules (Gasparrini et al. 2020). Similar pathways have been proposed for β-lactam antibiotics, where gut microbiota expressing β-lactamases or esterase-like hydrolases facilitate structural breakdown during gut passage. In addition, several studies report transcriptional upregulation of microbial genes associated with xenobiotic metabolism when earthworms are exposed to pharmaceuticals, suggesting an adaptive enhancement of degradative potential (Zhu et al. 2021; Song et al. 2022).

Indirect mechanisms involve earthworm-mediated stimulation of soil microbial communities. Earthworm casts and mucus contain easily decomposable carbon, nutrients, and bioactive compounds that promote microbial proliferation and enzymatic activity in the surrounding soil (Sizmur et al. 2017). This stimulates microbial processes such as co-metabolism, enzyme secretion, and biofilm formation, all of which enhance the degradation or immobilization of pharmaceuticals. These indirect effects complement the biochemical transformations occurring within the gut, creating a multifaceted remediation system supported by both earthworm physiology and their associated microbiota (Tagliabue et al. 2023).

6.2 Earthworms for Pharmaceutical Remediation

Studies investigating the use of earthworms for the remediation of pharmaceutical-contaminated soils are still limited but growing. Red worms, in particular, are frequently applied to soils contaminated with various classes of pharmaceuticals, including antibiotics, beta-blockers, anticonvulsants, and NSAIDs. A summary of the pharmaceuticals targeted through vermiremediation and the corresponding earthworm species employed is provided in Table 3. In general, bioaccumulation factors (BAFs) for these compounds are less than 1, indicating that significant accumulation within earthworm tissues is unlikely. Substantial degradation of many pharmaceutical compounds by earthworms has been observed in both biochar-amended and unamended soils, although certain classes, such as macrolides, appear more resistant to degradation (Fučík et al. 2024) (Table 3).

6.2.1 Antibiotics

Earthworms of the species *E. fetida* and *A. robustus* have been evaluated for their ability to remediate sterile and natural soils contaminated with tetracyclines. No significant differences were observed between the two soil types. However, both earthworm species demonstrated significantly greater remediation capacity in sterile soils compared to bacterial treatments. The reduction in tetracycline concentrations after 40 days differed only slightly between *E. fetida* (57.0%) and *A. robustus* (61.9%) (Lin et al. 2021). Compared to *E. fetida*, *M. guillelmi* exhibited greater sensitivity to certain contaminants and achieved an approximately 20% faster removal of tetracyclines (Yin et al. 2021; Tagliabue et al. 2023). In another study, low BAFs were observed in non-amended soils for tetracycline (BAF = 0.0467), chlortetracycline (BAF = 0.0568), and oxytetracycline (BAF = 0.0253) in earthworms of the species *E. fetida*. Notably, degradation of these compounds was more pronounced in non-amended soils than in biochar-amended soils. After 21 days, *E. fetida* facilitated the removal of 82% of tetracycline, 76% of chlortetracycline, and 59% of oxytetracycline in non-amended soils (Fučík et al. 2024).

For macrolide antibiotics, BAFs remained below 1 for all four (4) compounds tested. Erythromycin exhibited the highest bioaccumulation (BAF = 0.317), followed by roxithromycin (BAF = 0.111), while clarithromycin (BAF = 0.057) and azithromycin (BAF = 0.027) showed minimal accumulation. Interestingly, clarithromycin and azithromycin degraded more efficiently in biochar-amended soils, whereas no significant difference was found for roxithromycin between the two (2) soil types. Conversely, erythromycin showed a higher degradation rate in non-amended soil (24%) compared to biochar-amended soil (12%) after 21 days (Fučík et al. 2024).

Regarding fluoroquinolones, *E. fetida* showed negligible bioaccumulation. BAFs were below 0.1 for moxifloxacin, enrofloxacin, ciprofloxacin, and ofloxacin, indicating limited persistence of these compounds in the earthworms' digestive system. Nevertheless, high degradation rates were recorded in non-amended soils after 21 days: 72% for moxifloxacin, 69% for enrofloxacin, 65% for ciprofloxacin, and 56% for ofloxacin. These values were significantly higher than those observed in biochar-amended soils (Fučík et al. 2024).

P. guillelmi was tested for its ability to remediate sulfamethoxazole-contaminated soils, achieving nearly complete removal (99.55%) of the compound. The primary mechanisms identified were microbial activation in the soil and detoxification of the antibiotic within the earthworm gut (Zhang et al. 2022; Tagliabue et al. 2023). Although *E. fetida* showed relatively higher uptake for sulfamethoxazole (BAF = 0.329), sulfacetamide (BAF = 0.163), and sulfamethoxypyridazine (BAF = 0.163), the overall bioaccumulation remained ecologically insignificant (Fučík et al. 2024). All evaluated sulfonamides showed greater degradation in non-amended soils compared to biochar-modified soils, with sulfacetamide (96%), sulfadimethoxine (89%), sulfathiazole (89%), sulfamethoxazole (83%), and sulfapyridine (80%) achieving the highest degradation rates 21 days since the initial application of earthworms. Only sulfamethazine and sulfamethoxypyridazine, demonstrated improved degradation in biochar-amended soils (76% and 74%, respectively) compared to non-amended soils (66% and 63%, respectively). Trimethoprim exhibited low bioaccumulation in *E. fetida* (BAF = 0.0520), along with relatively low degradation rates in both non-amended (52%) and biochar-amended soils (41%) (Fučík et al. 2024).

6.2.2 Antihypertensives

The ability of *E. fetida* to remediate soil contaminated with acebutolol and nadolol was also investigated. Both compounds exhibited low bioaccumulation, with BAFs of 0.193 and 0.154, respectively. A significant degradation rate was recorded after 21 days for nadolol in both non-amended (91%) and biochar-amended soils (85%). However, the degradation of acebutolol was less efficient, with approximately 50% of the compound removed in both soil types (Fučík et al. 2024).

6.2.3 Other Medications

Carter, et al. (Carter et al. 2016), evaluated the influence of different soil types and pH levels on the bioaccumulation of various pharmaceuticals by the red worms. The study focused on diclofenac, carbamazepine, fluoxetine, and orlistat. Diclofenac showed the highest bioaccumulation among the tested compounds, with BAF values exceeding 1 across all soil types (Carter et al. 2014, 2016). Carbamazepine showed low accumulation across all soil types, with BAFs below 1 (Carter et al. 2016). Similarly, fluoxetine showed consistently low accumulation regardless of soil type or pH, with BAFs ranging from 0.20 to 0.37 (Carter et al. 2016). Orlistat bioaccumulation (BAF = 0.49–1.54) varied with soil type and pH.

7 Earthworm-Biochar Synergy in Remediation of Pharmaceutical Contaminated Soils

The combined application of biochar and earthworms has emerged as a promising strategy for enhancing soil remediation, although research remains more limited than studies on each amendment individually (Huang and He 2023; Zhu et al. 2024). In the context of pharmaceutical-contaminated soils, evidence for synergistic effects is still emerging and is supported primarily by indirect and semi-quantitative indicators rather than standardized removal metrics. Their co-application creates physicochemical and biological interactions capable of improving contaminant immobilization, transformation, and overall soil health. The synergistic effects stem from multiple mechanisms, which can be decomposed into physical, biochemical, and microbial pathways, each contributing to enhanced remediation efficiency.

Table 3 Summary of pharmaceuticals vermiremediation using earthworms; the worms species, BAFs in earthworms in non-amended soils and degradation rate in non-amended soils and biochar-amended soils (Carter et al. 2016; Tagliabue et al. 2023; Fučík et al. 2024)

Category of Pharmaceuticals	API	Earthworms Species	BAF[1]	Degradation Rate in Non-amended Soils	Degradation Rate in Bio-char-amended Soils
Antibiotics (Tetracyclines)	Tetracycline	*E. fetida*	-	57.0%	-
		A. robustus	-	61.9%	-
		M. guillelmi	-	83.2%	-
		E. fetida	0.0467	82.0%	60.0%
	Chlortetracycline	*E. fetida*	0.0568	76.0%	67.0%
	Oxytetracycline	*E. fetida*	0.0253	59.0%	43.0%
Antibiotics (Macrolides)	Clarithromycin	*E. fetida*	0.0570	27.0%	35.0%
	Erythromycin	*E. fetida*	0.3170	24.0%	12.0%
	Azithromycin	*E. fetida*	0.0270	30.0%	40.0%
	Roxithromycin	*E. fetida*	0.1110	26.0%	24.0%
Antibiotics (Fluoroquinolones)	Moxifloxacin	*E. fetida*	0.0630	72.0%	47.0%
	Enrofloxacin	*E. fetida*	0.0402	69.0%	45.0%
	Ciprofloxacin	*E. fetida*	0.0216	65.0%	49.0%
	Ofloxacin	*E. fetida*	0.0309	56.0%	42.0%
Antibiotics (Sulfonamides)	Sulfamethoxazole	*P. guillelmi*	-	99.6%	-
		E. fetida	0.3290	83.0%	77.0%
	Sulfacetamide	*E. fetida*	0.1630	96.0%	96.0%
	Sulfadimethoxine	*E. fetida*	0.0616	89.0%	26.0%
	Sulfamethazine	*E. fetida*	0.0562	66.0%	76.0%
	Sulfamethoxypyridazine	*E. fetida*	0.1630	63.0%	74.0%
	Sulfapyridine	*E. fetida*	0.0875	80.0%	67.0%
	Sulfathiazole	*E. fetida*	0.0920	89.0%	84.0%
	Trimethoprim	*E. fetida*	0.0520	52.0%	41.0%
Antihypertensives (Beta-blockers)	Acebutolol	*E. fetida*	0.1930	48.0%	49.0%
	Nadolol	*E. fetida*	0.1540	91.0%	85.0%

Table 3 (continued)

Category of Pharmaceuticals	API	Earthworms Species	BAF[1]	Degradation Rate in Non-amended Soils	Degradation Rate in Bio-char-amended Soils
NSAIDs	Diclofenac	*E. fetida*	1.01–12.36	-	-
Anticonvulsants	Carbamazepin	*E. fetida*	0.36–0.97	-	-
Antidepressants	Fluoxetine	*E. fetida*	0.20–0.37	-	-
Weight Loss Medications	Orlistat	*E. fetida*	0.49–1.54	-	-

[1] BAF is a dimensionless factor defined by the following ratio: $BAF = \frac{C_{earthworm}}{C_{soil}}$

7.1 Decomposition Mechanisms Mediated by Earthworm-Biochar Synergy

As far as the physical decomposition pathways are concerned, earthworm burrowing creates continuous bioturbation that redistributes biochar throughout the soil matrix, increasing contact between sorbent particles and contaminants. During ingestion, biochar is fragmented and mixed with fine soil particles, while passage through the gut results in the formation of organic-rich casts that embed biochar within microaggregates. These processes increase biochar dispersion, surface exposure, and accessibility to contaminants. Furthermore, certain endogenic species, which ingest large amounts of soil organic matter, may further enhance biochar-contaminant interactions as they facilitate deeper integration and greater contact between biochar and soil biota (Sanchez-Hernandez et al. 2019).

Regarding the biochemical mechanisms of pharmaceuticals decomposition due to the earthworm-biochar synergy, they are primarily laid on the ingestion of the APIs and biochar by the earthworms and the digestion mechanisms conducted in the earthworms' gut. As biochar passes through the earthworm gut, it becomes coated with mucus, digestive enzymes, and partially decomposed organic matter, modifying its surface and increasing its sorption reactivity. Moreover, earthworms secrete enzymes such as carboxylesterase, phosphatase, and dehydrogenase, which may bind to biochar surfaces and facilitate contaminant transformation or detoxification. For instance, mucus-derived alkaline compounds can increase local soil pH, promote conditions that enhance biochar sorption capacity and reduce contaminant bioavailability (Huang and He 2023). These processes create a "vermi-biochar" with altered surface properties compared to the common untreated biochars.

Another crucial pharmaceutical remediation mechanism due to the earthworms-biochar synergy involves the microbial pathway. Earthworm activity stimulates microbial biomass and enzymatic activity in surrounding soil, promoting biodegradation processes that complement biochar's sorption function. Biochar provides microhabitats and electron-accepting surfaces that support microbial colonization, while earthworms supply carbon and nutrients that enhance microbial metabolism. The combined effects lead to shifts in community structure and increased activity of C-, N-, P-, and S-cycling enzymes, which may accelerate the transformation or immobilization of organic contaminants, such as the pharmaceutical molecules (Sanchez-Hernandez 2018).

Beyond these physical interactions, bioturbation caused by earthworms can also affect the long-term stability and mobility of pharmaceuticals adsorbed to biochar. As earthworms fragment, ingest, and redistribute biochar particles within the soil profile, the resulting changes in particle size, surface exposure, and microaggregate incorporation may modify sorption equilibria. Depending on biochar characteristics and environmental conditions, such processes can either promote further stabilization of adsorbed contaminants - through enhanced aging and organometallic associations - or, in certain cases, facilitate the release of previously adsorbed compounds. These dynamics highlight that earthworm–biochar interactions extend beyond sorption enhancement and can also shape the long-term environmental fate of pharmaceutical residues. According to Shi et al. (2023), earthworm activity can redistribute and fragment biochar particles in soil, forming organic-rich components that incorporate biochar within microaggregates. This bioturbation can enhance the exposure of biochar surfaces to contaminants and affect the mobility and bioavailability of potentially toxic substances, including PAHs and other organic compounds. These findings support the need for long-term field-scale studies to fully understand the ecological impacts of earthworm–biochar interactions (Shi et al. 2023).

Together, these three (3) mechanisms create a synergistic remediation system wherein biochar enhances earthworm-driven soil processing, and earthworms accelerate the activation, dispersion, and functionalization of biochar.

7.2 Case Verification of Earthworm-Biochar Synergy

Although the combined effects of earthworms and biochar have been widely demonstrated in the remediation of metals (Boughattas et al. 2025) and organic pollutants such as PAHs (Hou et al. 2023), quantitative evidence specific to pharmaceutical-contaminated soils remains scarce and fragmented, often embedded within broader ecotoxicological or bioavailability studies.

One relevant study showed that the co-application of *Aporrectodea caliginosa* and willow-chip biochar resulted in improved remediation efficiency, with the magnitude of response strongly dependent on soil type (Garbuz et al. 2020). Another study demonstrated that the addition of earthworms such as *A. caliginosa* or *L. terrestris* to soils amended with biochar derived from pine needles or spent coffee grounds increased the activities of enzymes associated with C-, P-, and S-cycling, which were retained on biochar surfaces (Sanchez-Hernandez 2018). While these studies did not directly quantify pharmaceutical removal efficiencies, the observed biochemical enhancements are widely recognized as functional proxies for increased contaminant transformation and immobilization capacity.

In addition, several studies report beneficial ecological outcomes that indirectly support synergistic remediation potential, such as reduced pollutant-induced DNA damage in earthworms and enhanced gut microbial diversity when biochar is present (Sanchez-Hernandez et al. 2019). This indicates that biochar may not only improve soil conditions for earthworms but also increase their tolerance and functional performance under pollutant stress.

7.3 Effect Comparison of Earthworm-Biochar Synergy

Direct quantitative comparisons of remediation efficiency for pharmaceuticals in earthworm-biochar systems remain limited. However, available evidence illustrates that combined treatments often outperform single amendments. For example, soils amended with biochar and earthworms exhibit greater enzymatic activity and improvements in microbial resilience, and biochar activation can occur. All of these factors contribute to enhanced pharmaceutical immobilization compared to either amendment alone (Table 3) (Sanchez-Hernandez 2018; Garbuz et al. 2020; Fučík et al. 2024; Zhu et al. 2024). These comparative improvements, such as enhanced enzyme binding to biochar surfaces, increased microbial activity, and greater biochar distribution throughout the soil, highlight the potential gains associated with synergistic application.

Despite these promising results, only a small number of studies explicitly quantify pharmaceutical concentrations, bioavailability reductions, or dissipation rates under combined earthworm-biochar treatments, indicating that this remains an emerging research area. In the study of Fucik et al. (Fučík et al. 2024), in which 21 APIs were investigated, biochar affected the bioavailability of pharmaceuticals to earthworms only on the first day, while a significant persistence of pharmaceuticals in soils was observed in the presence of biochar, in contrast to other recent studies (Table 3) (Fučík et al. 2024). This divergence highlights the sensitivity of synergistic outcomes to experimental design, biochar properties, and exposure duration. Future research should therefore prioritize standardized quantitative endpoints, including removal efficiency, adsorption capacity, and bioavailability reduction, to more robustly substantiate earthworm-biochar synergy in pharmaceutical-contaminated soils.

8 Challenges and Prospects

While the combined application of biochar and earthworms is a promising strategy for remediating pharmaceutical contaminated soils, several practical considerations remain before widespread field implementation. These factors do not undermine the feasibility of the approach but highlight areas where further optimization will enhance its effectiveness and applicability.

a) Practical Considerations for Large-Acale Biochar Application Biochar production and field incorporation require thoughtful planning, particularly regarding feedstock choice, pyrolysis conditions and application rates. Although higher amendment rates are commonly used in laboratory studies, field-scale projects may employ lower, more cost-effective doses tailored to site conditions. Moreover, the growing availability of agricultural and forestry residues as low-cost feedstocks, along with the improvements in pyrolysis technology, are expected to reduce production costs over time. Continued research into optimizing biochar properties for specific soil and contaminant profiles will further support its practical deployment. Importantly, full scale techno economic studies will further support the applicability of biochar.

b) Earthworm Performance in Contaminated Environments Earthworms generally exhibit good tolerance to environmentally relevant concentrations of pharmaceuticals, and several species (e.g., *E. fetida*, *A. caliginosa*) maintain active burrowing and feeding even in moderately contaminated soils. While very high API concentrations may affect earthworm physiology or behavior, biochar addition often helps mitigate these effects by reducing contaminant bioavailability. Identifying species with naturally higher tolerance, and understanding how biochar amendments support earthworm health, will facilitate reliable vermiremediation across a wider range of soil conditions.

c) Influence of Variable Soil Conditions Soil pH, texture, moisture, and co-contaminants can influence pharmaceutical sorption and biological activity, but these factors are common to most soil remediation approaches. Importantly, both biochar and earthworms tend to improve several soil properties, including pH buffering, aeration, and microbial activity, which may offset some of the inhibitory effects observed under complex soil environments. At a broader scale, global contamination patterns (Table 1) indicate that soil type and climatic conditions (e.g., temperature and moisture regimes) regulate the regional adaptability and transferability of biochar-earthworm remediation. Accordingly, site- and region-specific selection of biochar properties and earthworm species is recommended to optimize remediation efficiency under different climatic conditions. Therefore, future studies using more realistic multi-contaminant systems will help refine predictive models and guide site- and climatic-specific application strategies.

d) The Potential Formation of Pharmaceutical Transformation Products The potential formation of pharmaceutical transformation products due to their partial degradation from vermiremediation and adsorption mediated by biochar may jeopardize the safety of the soil ecosystem. Current evidence indicates that transformation products can be generated during earthworm-mediated degradation or microbially driven processes associated with biochar-amended soils. For example, studies on antibiotic degradation have shown that biotransformation can generate intermediate metabolites that may retain antimicrobial activity or exhibit increased toxicity compared to the parent compound, as demonstrated for certain sulfonamide and fluoroquinolone antibiotics in soil systems. Indeed, several fluoroquinolone transformation products have been identified as persistent intermediates, while sulfonamide degradation pathways can produce oxidized compounds with greater ecotoxicological risk than the parent compound (Sören Thiele-Bruhn 2019; Maculewicz et al. 2022). Furthermore, recent work has identified multiple transformation products of sulfamethazine and tetracycline in earthworms and soil, highlighting that antibiotic degradation can produce distinct metabolites in biota (Vergara-Luis et al. 2024). Nevertheless, limited studies have focused on identifying these transformation products, leaving the fate and potential toxicity of these by-products in combined soil-biochar-biota systems essentially unresolved. These findings underscore the need for future research integrating chemical analysis with ecotoxicological assessment to better evaluate the risks associated with pharmaceutical transformation products under biochar-earthworm remediation scenarios.

Therefore, the combined use of earthworms and biochar is increasingly supported by advances in materials engineering, soil ecology and microbial genomics. Tailored biochars with enhanced functional groups, the identification of resilient earthworm species, and -omics analyses of gut microbial pathways offer clear opportunities for improved performance. As field trials expand and methodologies become more standardized, the biochar-earthworm synergy has strong potential to develop into a practical, sustainable solution for long-term remediation of pharmaceutical contaminated soils.

9 Conclusions

The increasing presence of pharmaceutical contaminants in soil poses a significant environmental and public health challenge. While numerous remediation strategies have been explored, eco-friendly and sustainable approaches are gaining traction. Among these, biochar has emerged as a promising material with high sorption capacity, capable of immobilizing a wide range of pharmaceutical compounds. Similarly, earthworms, offer a nature-based solution through the process of vermiremediation, contributing to the degradation and transformation of persistent pollutants. A diverse array of pharmaceuticals, including antibiotics, NSAIDs, endocrine disruptors, antihypertensives, anticonvulsants, antidepressants, antipsychotics, and psychostimulants, have been successfully removed from contaminated soils using either biochar, earthworms, or their combined application. This review provides a comprehensive synthesis of both mechanistic and empirical data on biochar-earthworm remediation, highlighting differences in removal efficiencies among drug classes and offering practical insights for the design of sustainable soil remediation strategies. These findings highlight the potential of integrating biological and bio-based strategies for sustainable soil remediation. To fully realize the potential of these ecological technologies, further studies should be conducted to include a broader range of APIs, commonly found in the environment due to widespread human use, explore synergistic effects, and evaluate long-term efficacy under real-world conditions. Future studies should also consider regulatory frameworks, acceptable residue limits, and human health risk assessment methodologies to ensure that these remediation strategies can be translated into regulatory-compliant solutions. Embracing such nature-based solutions not only supports the cleanup of contaminated soils but also aligns with broader goals of environmental sustainability and ecosystem resilience.

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Data Availability The datasets generated during and/or analysed during the current study are available from the corresponding author on reasonable request.

Declarations

Clinical Trial Number Not applicable.

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