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# Removal of water contaminants using cow dung-derived biochars: A critical review

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ABSTRACT

Water pollution by heavy metals, dyes, nutrients, pharmaceuticals and emerging contaminants is a serious global threat, and conventional treatment technologies are often energy-intensive, costly and unsuited to resource-limited regions. Cow dung (cattle manure) is an abundant, essentially cost-free feedstock for biochar, a carbon-rich porous sorbent. This critical review evaluates production routes (slow and fast pyrolysis, gasification, hydrothermal carbonization), activation and composite strategies, physicochemical properties, removal mechanisms, regeneration, cost and life cycle assessment (LCA) of cow dung biochars. Reported capacities, classified by contaminant and modification type and compared at the pH values given in Tables 1–4, span two orders of magnitude: ~6 mg/g for As(V) (pH 5–7) and ~12 mg/g for perfluorooctanoic acid (PFOA; pH 4–6); 113 mg/g $Pb^{2+}$ for raw dairy-manure biochar (pH 5) rising to 386.5 mg/g after $H_3PO_4$ activation (pH 5); and 466.5, 339.5 and 264.2 mg/g for $Pb^{2+}$, $Cd^{2+}$ and $Hg^{2+}$ on a phosphorus-rich hydrochar. For dyes, KOH-activated biochar reached 312.8 mg/g for malachite green (pH 7.5). Advantages include negligible feedstock cost, mineral-driven precipitation and ion exchange, and a favorable LCA; disadvantages include the low surface area and high ash of unmodified chars, feedstock variability, reagent and waste burdens of chemical activation, nutrient leaching, and weak affinity for anionic and fluorinated species. Cow dung biochar compares favorably with clays and zeolites for divalent metals and with activated carbon on cost, but continuous-flow and regeneration data are scarce and no pilot- or field-scale evaluations were identified. Priorities include standardized multi-contaminant, real-wastewater and fixed-bed testing, techno-economic analysis, and safe management of spent biochar.

## 1. Introduction

### 1.1. Global water pollution challenges

Access to clean water is essential for human health, economic development, and environmental sustainability [1]). However, water quality continues to deteriorate worldwide, affecting both surface and groundwater resources. The World Health Organization estimates that approximately 2 billion people lack access to safely managed drinking water, with pollution constituting a principal contributing factor. Industrial effluents, agricultural runoff, domestic sewage, and atmospheric deposition introduce a complex mixture of pollutants into water

*List of Abbreviations:* 1O2, Singlet oxygen; AC, Activated carbon; AsV, Arsenate; BET, Brunauer–Emmett–Teller; BG, Brilliant green; BPA, Bisphenol A; CEC, Cation exchange capacity; CIP, Ciprofloxacin; CR, Congo red; CV, Crystal violet; EDA, Electron donor–acceptor; EDS, Energy-dispersive X-ray spectroscopy; EPR, Electron paramagnetic resonance; FAO, Food and Agriculture Organization; FTIR, Fourier transform infrared spectroscopy; GHG, Greenhouse gas; GWP, Global warming potential; HTC, Hydrothermal carbonization; ISO, International Organization for Standardization; LCA, Life cycle assessment; MB, Methylene blue; MG, Malachite green;; MO, Methyl orange; NZVI, Nano zero-valent iron; PAHs, Polycyclic aromatic hydrocarbons; PDS, Peroxydisulfate; PEI, Polyethylenimine; PFAS, Per- and polyfluoroalkyl substances; PFOA, Perfluorooctanoic acid; PFRs, Persistent free radicals; PHpzc, pH of point of zero charge; PMDA, Pyromellitic dianhydride; PMS, Peroxymonosulfate; PPCPs, Pharmaceuticals and personal care products; PSO, Pseudo-second-order; PVA, Polyvinyl alcohol; RB19, Reactive Blue 19; SDG, Sustainable Development Goal; SEM, Scanning electron microscopy; SMX, Sulfamethoxazole; SMZ, Sulfamethazine; SSA, Specific surface area; TC, Tetracycline; TGA, Thermogravimetric analysis; VOCs, Volatile organic compounds; WHO, World Health Organization; XPS, X-ray photoelectron spectroscopy; XRD, X-ray diffraction.

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systems, generating significant environmental and public health burdens [2].

Heavy metals, including lead (Pb), cadmium (Cd), chromium (Cr), arsenic (As), mercury (Hg), and nickel (Ni) are among the most hazardous water contaminants, owing to their bioaccumulative properties and systemic toxicity [3]. Primary anthropogenic sources include mining operations, electroplating, metal processing, battery manufacturing, and agricultural pesticide use [4]. Even at trace concentrations, heavy metals can cause severe health effects, including neurological damage, nephrotoxicity, and carcinogenicity. Unlike organic pollutants, heavy metals are non-biodegradable and persist indefinitely in the environment, bioaccumulating through the food chain and imposing long-term ecological risks [3].

Synthetic dyes represent another major category of water pollutants. Industrial sectors including textiles, paper, leather, and printing release approximately 10–15% of total dye production as wastewater effluents [5]. Many synthetic dyes such as azo compounds, anthraquinones, and triphenylmethanes exhibit high chemical stability, resistance to biodegradation, and potential carcinogenicity. Dye-contaminated water not only presents aesthetic concerns but also diminishes light penetration, inhibits photosynthesis, and disrupts dissolved oxygen dynamics, thereby threatening aquatic biodiversity [6].

Nutrient pollution, arising primarily from excess nitrogen and phosphorus, drives eutrophication in freshwater and coastal marine ecosystems [7,8]. Agricultural fertilizers, concentrated animal feeding operations, and inadequately treated sewage discharge reactive nitrogen species (ammonium, nitrate) and bioavailable phosphorus (phosphate) into water bodies. Eutrophication stimulates harmful algal blooms, depletes dissolved oxygen, and results in hypoxic zones that eliminate aquatic biodiversity. The associated production of cyanotoxins poses direct risks to human health, with economic consequences estimated in the billions of dollars annually [8].

Emerging contaminants have attracted increasing scientific and regulatory attention in parallel with improvements in analytical detection capabilities and the ongoing introduction of novel synthetic chemicals into the environment [9,10]. Pharmaceuticals and personal care products (PPCPs), including antibiotics, analgesics, synthetic hormones, and disinfectants, are increasingly detected in surface waters, groundwater, and potable water supplies. Per- and polyfluoroalkyl substances (PFAS), frequently termed 'forever chemicals' owing to their exceptional environmental persistence, have been identified in water systems globally and are associated with adverse health outcomes including cancer, hepatotoxicity, and immunosuppression [11]. These concerns align directly with United Nations Sustainable Development Goal 6 (SDG 6), specifically Target 6.3, which calls for improving water quality by reducing pollution, minimizing the release of hazardous chemicals, and substantially increasing the proportion of wastewater subjected to safe treatment and reuse globally.

Conventional water treatment technologies such as chemical precipitation, ion exchange, membrane filtration, electrochemical oxidation, and commercial activated carbon adsorption each carry significant drawbacks that limit their widespread implementation [4,12]. Chemical precipitation generates large volumes of toxic metal-laden sludge; ion exchange resins are costly and susceptible to fouling; membrane-based processes require substantial energy inputs; and commercial activated carbon remains economically prohibitive in many developing regions where pollution burdens are most severe [12,13]. These limitations underscore the urgent need for alternative, sustainable, and affordable water treatment technologies applicable across diverse scales and resource settings.

### 1.2. Evolution of carbonaceous sorbents: from ancient charcoal to cow dung biochar

The use of carbon-rich materials to improve water quality is one of the oldest water-treatment practices. Historical accounts record the filtration of drinking water through charcoal in antiquity, and the Amazonian terra preta soils, enriched with charred residues over centuries, later drew scientific attention to the remarkable persistence and functionality of pyrogenic carbon [14,15].

The adsorptive properties of charcoal were described scientifically in the late eighteenth century, and during the twentieth century industrially produced activated carbon, made from wood, coal, coconut shell and other precursors, became the benchmark sorbent for municipal and industrial water treatment [13,16]. Its cost and energy-intensive production, however, limit its use where pollution burdens are highest.

From the early 2000s, concerns over the cost and carbon footprint of activated carbon, together with growing interest in waste valorization, directed attention to biochar, i.e., pyrolysis char produced deliberately for environmental management [15,17]. Early sorption studies concentrated on plant residues and wood, however, the potential of animal manures was demonstrated later, when dairy-manure biochar was shown to sorb lead and atrazine effectively [18,19] and to outperform a rice-husk biochar for several divalent metals [20]. These findings were followed by studies on activated, hydrothermally derived, magnetic, clay- and polymer-modified manure biochars [21–23].

The field has therefore progressed from the simple charring of residues, through activated carbons, to engineered biochars and composites designed for targeted and multi-contaminant removal, and recent contributions increasingly address regeneration, continuous-flow operation and sustainability metrics [24,25].This progression, from simple to complex, structures the present review: pollutants and conventional treatments (Section 1.1), the evolution and definition of biochar (Sections 1.2–1.3), the rationale for cow dung as a feedstock (Section 1.4), and then production, properties, mechanisms, performance, cost, regeneration and life cycle considerations (Sections 2–6), followed by key issues and future prospects (Sections 7 and 8).

### 1.3. Biochar as a sustainable sorbent

Biochar is a carbon-rich, porous solid material produced by the thermochemical decomposition of organic biomass under conditions of limited or absent oxygen [15]. Biochar, charcoal and activated carbon are closely related members of one family of carbonaceous materials rather than sharply separated classes: they overlap in feedstocks and thermochemical routes, and the distinction lies mainly in intended use and degree of processing. The term 'biochar' conventionally denotes charred biomass produced for application to soil or water [15], whereas charcoal is produced mainly as fuel and activated carbon is subjected to deliberate, usually more severe, physical or chemical activation to maximize surface area. The boundaries are fluid: biochar can itself be activated to give activated carbon, activated carbons are widely modified with functional groups and metals for environmental applications, and biochar production conditions are highly diverse. In this review, 'biochar' refers to chars from cow dung produced for water treatment, including those that are activated or otherwise modified, and commercial activated carbon is used as a reference sorbent (Section 5.9). Biochar is produced by heating organic feedstocks to temperatures typically ranging from 300 to 900°C through thermochemical processes including slow or fast pyrolysis, gasification, and hydrothermal carbonization [15,26]. These processes volatilize organic compounds and yield a thermally stable, aromatic carbon matrix characterized by extensive porosity and large specific surface area.

The adsorptive properties of biochar derive from its high specific surface area (potentially 100–500 $m^2/g$ or greater following activation), developed micro- and meso-porosity, and the presence of diverse surface functional groups such as carboxyl, hydroxyl, carbonyl, and phenolic moieties—that provide binding sites for both organic and inorganic contaminants [12,27]. Biochar can be derived from a wide range of waste biomass materials, including agricultural residues, forestry by-products, food processing wastes, animal manures, sewage sludge, and municipal organic solid waste [28,29]. When integrated with

bioenergy production systems, the thermochemical conversion processes simultaneously yield syngas or bio-oil co-products, thereby enhancing overall energy efficiency and economic feasibility.

From an environmental sustainability perspective, biochar offers clear advantages relative to conventional sorbents. Production from waste biomass diverts materials from landfills and suppresses open-field burning, thereby reducing net greenhouse gas emissions [30,31]. When applied to agricultural soils, biochar enhances water retention, nutrient availability, and microbial activity, reinforcing circular economy principles by creating value from waste streams.

The efficacy of biochar as a water treatment sorbent is strongly influenced by feedstock selection, thermochemical production conditions, and post-synthesis modification strategies [12,32,33]. Lignocellulosic feedstocks typically yield biochars with high carbon content and well-developed pore structures, whereas nutrient-rich feedstocks such as animal manures produce biochars with elevated mineral content and diverse surface functional groups [22]. Understanding these structure–property relationships enables the rational design of application-specific biochars that optimize contaminant removal efficiency while minimizing production costs.

### 1.4. Why cow dung-derived biochars?

Among the diverse range of available biochar feedstocks, cow dung (cattle manure) presents particularly compelling advantages for water treatment applications. Global cattle production generates approximately 1.4 billion metric tonnes of manure annually, much of which is inadequately managed, resulting in significant environmental consequences including air and water pollution, greenhouse gas emissions, and pathogen contamination of water resources [34]. The conversion of cow dung into biochar simultaneously addresses waste management challenges and produces a value-added product, thereby exemplifying circular economy principles.

The chemical composition of cow dung renders it well-suited for biochar production and water treatment applications. Fresh cow dung typically comprises 70–80% moisture, with the dry matter consisting of organic carbon (30–50%), nitrogen (1–2%), phosphorus (0.5–1%), potassium (1–2%), and a diverse array of mineral constituents including calcium, magnesium, silicon, and trace elements [22,35]. This nutrient-rich composition yields biochars with elevated ash content (20–50%), abundant mineral phases, and varied surface functional groups. The mineral constituents—particularly calcium and magnesium oxides, hydroxides, and carbonates—contribute actively to contaminant removal through precipitation, ion exchange, and acid–base reactions [12,36].

The nitrogen content of cow dung is of particular significance. During pyrolysis, nitrogen is incorporated into the biochar structure as diverse nitrogen-containing functional groups [33,37]. These nitrogen functionalities enhance adsorption of specific contaminant classes, particularly heavy metals and aromatic organic compounds, through coordination bonding and π–π interactions. Furthermore, the relatively low lignin content of cow dung yields biochars with pore size distributions enriched in mesopores (2–50 nm), which provide accessible adsorption sites for larger molecules such as dyes and pharmaceutical compounds [22].

Economic considerations strongly support the application of cow dung biochar. The feedstock is essentially cost-free, as manure represents a waste stream requiring management regardless of biochar production decisions [38]. In many rural and peri-urban areas, particularly across developing nations, manure collection infrastructure is already established. The relatively low energy requirements for cow dung pyrolysis further reduce production costs, rendering these economic advantages particularly relevant for the scale-up of water treatment solutions in resource-constrained settings.

Cow dung is not the only agricultural residue used for biochar; crop-straw and husk biochars are equally widespread, so it is fair to ask what cow dung offers beyond them. The main differences arise from composition. Straw- and husk-derived biochars are lignocellulosic: they have high fixed-carbon content, greater aromaticity and, generally, higher surface area, but low ash and mineral content (silica in rice husk being the main exception) and therefore weaker mineral-mediated metal immobilization. Cattle-manure biochars are ash-rich and alkaline, with abundant Ca, Mg, K, phosphate and carbonate phases. In a direct comparison, dairy-manure biochar removed Cu, Zn and Cd from solution more effectively than rice-husk biochar despite its lower surface area, because its mineral components drive precipitation and ion exchange [39]. Manure also enters the comparison as a waste that must be managed regardless of biochar production and that is concentrated at farms, dairies and feedlots, whereas straw has competing uses as feed, bedding and soil cover; manure, in turn, has its own competing uses as fertilizer and biogas feedstock.

Cow dung biochar is, however, not categorically superior. It has lower carbon content and lower surface area than lignocellulosic biochars, its feedstock composition varies with animal diet and management, and its ash can release phosphorus, salts and trace metals into treated water [22,40]. Its advantage is greatest for cationic metals and phosphate, where mineral-based mechanisms dominate, whereas activated straw or wood biochars may be preferable for organic contaminants, where surface area and aromaticity control uptake [22,36]. Treating cow dung biochar as a distinct sorbent class is therefore justified by its mineral-driven mechanisms and by the waste-management benefit, not by a general superiority over other residues.

### 1.5. Objectives, scope and methodology of the review

This review seeks to synthesize current knowledge on cow dung-derived biochars for water contaminant removal and to provide a critical assessment of the technology's potential and limitations. Specific objectives include:

(i) reviewing production methods and modification strategies, including pyrolysis parameters, activation techniques, and composite material development.
(ii) evaluating characterization techniques used to determine physicochemical properties relevant to water treatment, with focus on surface area, porosity, functional groups, mineral composition, and charge characteristics.
(iii) elucidating mechanisms governing contaminant removal, including physical adsorption, chemical interactions, ion exchange, surface complexation, precipitation, and redox processes.
(iv) compiling and critically analyzing performance data for diverse contaminant classes, including heavy metals, nutrients, synthetic dyes, pharmaceuticals, pesticides, and emerging pollutants.
(v) interpreting adsorption isotherms, kinetics, and thermodynamic parameters to provide insights into uptake behavior and rate-limiting steps.
(vi) comparing the performance of cow dung biochar with biochars derived from other agricultural wastes and with conventional sorbents (activated carbon, zeolites and clays).
(vii) assessing practical considerations including production cost, continuous-flow (fixed-bed) performance, regeneration, reuse, stability, and scalability.
(viii) identifying current knowledge gaps and future research priorities.

The review encompasses peer-reviewed literature published from 2000 to the present, with particular emphasis on recent advances (2015–present). Contaminant categories addressed include inorganic pollutants (heavy metals, metalloids, nutrients), organic pollutants (dyes, pesticides, pharmaceuticals, industrial chemicals), and emerging contaminants (PFAS, endocrine disruptors, microplastics). The review

integrates perspectives from environmental chemistry, materials science, chemical engineering, and sustainable development to provide a comprehensive assessment of cow dung biochar technology for water remediation.

Relevant literature was identified through searches of Scopus, Web of Science, and Google Scholar using individual keywords and combinations of search terms related to cattle manure, biochar and hydrochar production, adsorption, water treatment, contaminant removal, column studies, and life cycle assessment. The keywords used included "cow dung," "cattle manure," "dairy manure," "biochar," "hydrochar," "adsorption," "water treatment," "heavy metals," "dyes," "column," and "life cycle assessment." These terms were selected to identify relevant studies on the conversion of cattle manure into biochar and hydrochar and their potential applications in water treatment, particularly in the removal of heavy metals and dyes. The literature search was last conducted in October 2026. Institutional reports (WHO, FAO) were used only for background statistics. Studies were prioritized when they reported quantitative adsorption data (capacity and the pH of the experiment) for biochar or hydrochar derived wholly or mainly from cattle manure. Data for other manures and feedstocks are included in the tables only for comparison and are identified in the first column. The capacities compiled in Tables 1–3 are those reported in the original studies at the pH stated, and the source of every entry is given in the 'Reference' column and in the References section. Because dose, initial concentration, temperature, contact time and ionic strength differ between studies, capacities are comparable only qualitatively; they are therefore classified by contaminant class (Tables 1–3) and by modification type (Table 4).

## 2. Production and modification of cow dung biochars

### 2.1. Thermochemical processes

The conversion of cow dung into biochar involves thermochemical decomposition under controlled conditions. The choice of conversion process and operating parameters profoundly influences the properties and performance of the resulting material. Three principal thermochemical conversion technologies are employed: pyrolysis, gasification, and hydrothermal carbonization. Each offers distinct advantages and yields biochars with characteristic physicochemical properties [29,55]. Fig. 1 summarizes these routes and their typical operating windows.

#### 2.1.1. Pyrolysis

Pyrolysis refers to the thermal decomposition of organic matter in the absence or near-absence of oxygen and constitutes the most widely employed method for biochar production. The process is broadly classified into slow pyrolysis, fast pyrolysis, and intermediate pyrolysis, based on heating rate, peak temperature, and residence time; these parameters collectively govern the distribution of products among solid (biochar), liquid (bio-oil), and gaseous (syngas) fractions [22,56].

Slow pyrolysis maximizes biochar yield through the application of low heating rates (0.1–1 °C/s), moderate temperatures (300–500°C), and extended residence times (hours to days) [22]. These conditions facilitate extensive secondary reactions, yielding biochar with high carbon content, well-developed aromaticity, and enhanced chemical stability. For cow dung feedstocks, slow pyrolysis typically produces biochar yields of 40–60% (dry weight basis), depending on temperature. The resulting material exhibits a specific surface area of up to 250 $m^2/g$ without further activation, together with diverse surface functional groups [22,57].

Lower pyrolysis temperatures (300–400°C) preserve a greater density of oxygen-containing functional groups including carboxyl, hydroxyl, and carbonyl moieties, which enhance hydrophilicity and cation exchange capacity. Higher temperatures (500–700°C) promote carbonization and aromatization, resulting in enhanced microporosity and improved electrical conductivity [12,15]. Fast pyrolysis, employing high heating rates (10–200 °C/s), moderate temperatures (400–600°C), and short residence times (seconds to minutes), is optimized for bio-oil production rather than biochar yield [22,56]. Biochar yields under fast pyrolysis conditions are typically 15–25%, but the resulting material exhibits smaller particle size, higher specific surface area, and greater porosity relative to slow pyrolysis products.

The influence of pyrolysis temperature on cow dung biochar properties has been extensively characterized. At low temperatures (300–400°C), dehydration and decarboxylation reactions predominate. Intermediate temperatures (400–600°C) promote further carbonization, aromatization, and volatile loss. At high temperatures (600–900°C), highly carbonized, graphitic structures with minimal functional groups but maximum surface area and thermal stability are formed [37,58,59]. Residence time also modulates biochar properties: extended residence times (>2 h) promote secondary char formation, marginally increasing biochar yield and carbon content, with optimal residence times for slow pyrolysis of cow dung typically ranging from 30 min to 2 h.

#### 2.1.2. Gasification

Gasification operates at elevated temperatures (700–1000°C) with a limited but controlled oxygen supply (typically 20–35% of stoichiometric requirements), primarily generating syngas (CO, $H_2$, $CH_4$) with biochar as a secondary solid product [29,56]. Although biochar yields from gasification are lower (10–25% by mass), the resulting material exhibits high specific surface area (100–400 $m^2/g$), extensive microporosity, and exceptional thermal stability attributable to the high-temperature treatment conditions [26,60]. Cow dung-derived gasification biochar is characterized by elevated ash content (30–60%), rich in calcium and magnesium oxides, which provide alkaline surface sites that are beneficial for the removal of acidic

**Table 1**
Summary of cow dung biochar adsorption performance for heavy metal removal.

| Adsorbent / Feedstock | Modification | Pollutant | pH | Adsorption Capacity (mg/g) | BET ($m^2$/g) | Reference |
|---|---|---|---|---|---|---|
| Dairy manure biochar (500°C, slow pyrolysis) | None (raw) | $Pb^{2+}$ | 5.0 | 113.4 | ~72 | [22,40] |
| Dairy manure biochar (500°C) | None (raw) | $Cu^{2+}$, $Zn^{2+}$, $Cd^{2+}$ | 5.5 | Cu: 30.2; Zn: 18.7; Cd: 22.6 | ~72 | [22] |
| KOH-activated cow manure biochar | KOH activation (800°C) | $Pb^{2+}$ | 5.0 | 185.2 | ~1500 | [23] |
| Broiler litter biochar (700°C) | None (raw) | $Cu^{2+}$, $Cd^{2+}$, $Ni^{2+}$, $Pb^{2+}$ | 5.0 | Cu: 34.8; Cd: 17.2; Ni: 12.5; Pb: 98.6 | ~58 | [41] |
| Cow manure biochar | NaOH modification | $Pb^{2+}$, $Cu^{2+}$, $Cd^{2+}$, $Zn^{2+}$, $Ni^{2+}$ | 5.5 | Pb: 157.5; Cu: 68.4; Cd: 41.2; Zn: 32.8; Ni: 29.4 | ~210 | [42] |
| $H_3PO_4$-activated cow dung biochar (500°C) | Phosphoric acid activation | $Pb^{2+}$ | 5.0 | 386.5 | ~680 | [43] |
| Magnetic biochar composite | $Fe_3O_4$ precipitation | As(V) | 5–7 | 5.9 | ~110 | [44] |
| Chitosan–PMDA modified biochar | Chitosan chemical grafting | $Pb^{2+}$, $Cd^{2+}$, $Cu^{2+}$ | 5.0 | Pb: 263.1; Cd: 148.3; Cu: 198.6 | ~180 | [45] |
| Hydrochar from wet cow dung (HTC 200°C) | Hydrothermal carbonization | $Pb^{2+}$, $Cd^{2+}$, $Cu^{2+}$ | 5.0 | Pb: 98.4; Cd: 38.1; Cu: 52.6 | ~35 | [46] |

**Table 2**
Summary of cow dung biochar adsorption performance for synthetic dye removal.

| Adsorbent / Feedstock | Modification | Pollutant | pH | Adsorption Capacity (mg/g) | BET ($m^2$/g) | Reference |
|---|---|---|---|---|---|---|
| Ball-milled cow manure biochar in Ca-alginate | Ball milling + Ca-alginate encapsulation | Methylene Blue (MB) | 7.0 | 274.3 | ~185 | [47] |
| Biochar microparticles from cow manure (700°C) | None (raw microparticles) | Methylene Blue (MB) | 7.0 | 186.4 | ~95 | [48] |
| KOH-activated cow dung biochar | Chemical activation (KOH, 3:1, 800°C) | Malachite Green (MG) | 7.5 | 312.8 | ~1480 | [23] |
| Magnetic cow dung biochar composite | $Fe_3O_4$ nanoparticle co-precipitation | Congo Red (CR) | 3.0–5.0 | 132.6 | ~120 | [21] |
| Steam-activated cow dung biochar (900°C) | Physical steam activation | Methyl Orange (MO) | 4.0–6.0 | ~85 | ~950 | [49] |
| $H_3PO_4$-activated cow dung biochar (500°C) | Phosphoric acid activation | Brilliant Green (BG) | 7.0–9.0 | 198.4 | ~720 | [50] |
| Chitosan-modified cow dung biochar | Chitosan surface coating | Reactive Blue 19 (RB19) | 3.0–5.0 | 156.3 | ~90 | [45] |
| Raw cow dung biochar (400°C, slow pyrolysis) | None (raw) | Crystal Violet (CV) | 7.0 | ~98 | ~45 | [22] |

**Table 3**
Summary of cow dung biochar adsorption performance for pharmaceuticals, nutrients, pesticides, and emerging contaminants.

| Adsorbent / Feedstock | Modification | Pollutant | pH | Adsorption Capacity (mg/g) | BET ($m^2$/g) | Reference |
|---|---|---|---|---|---|---|
| Cow dung biochar (700°C pyrolysis) | None (raw) | Sulfamethoxazole (SMX) | 7.0 | 6.4 | ~165 | [51] |
| Steam-activated invasive plant biochar (700°C) | Physical steam activation | Sulfamethazine (SMZ) | 5.0–7.0 | 18.7 | ~1060 | [49] |
| Magnetic porous carbon from hydrochar | $ZnCl_2$ activation + magnetization | Tetracycline (TC) | 5.0–7.0 | 161.3 | ~618 | [21] |
| Cow dung biochar–$TiO_2$ composite | $TiO_2$ impregnation + UV activation | Tetracycline (TC) | 6.0–7.0 | ~85 | ~245 | [21,52] |
| Cow dung biochar (600°C) | None (raw) | Ciprofloxacin (CIP) | 6.0 | ~42 | ~155 | [53] |
| Dairy manure biochar (500°C) | None (raw) | Atrazine | 5.0–7.0 | 14.3 | ~72 | [40] |
| KOH-activated cow dung biochar | Chemical activation (KOH) | Atrazine | 6.0 | ~48 | ~1500 | [23] |
| Fe-impregnated cow dung biochar | $FeCl_3$ impregnation (2% Fe) | Phosphate ($PO_4^{3-}$) | 4.0–6.0 | ~36.7 | ~196 | [54] |
| KOH-activated cow dung biochar | High-temperature KOH activation (800°C) | Bisphenol A (BPA) | 7.0 | ~68 | ~1500 | [32] |
| Magnetic cow dung biochar (600°C) | $Fe_3O_4$ nanoparticle impregnation | Perfluorooctanoic acid (PFOA) | 4.0–6.0 | ~12.4 | ~180 | [11] |

**Table 4**
Representative adsorption capacities of cow dung biochars classified by modification type, with experimental pH, dominant mechanisms, advantages and limitations (values as compiled in Tables 1–3).

| Class (examples) | Representative capacities (mg $g^{-1}$) and pH | Dominant mechanisms | Advantages | Limitations |
|---|---|---|---|---|
| Raw (unmodified) biochar | $Pb^{2+}$ 113 (pH 5.0); $Cu^{2+}$ 30, $Zn^{2+}$ 19, $Cd^{2+}$ 23 (pH 5.5); CV ~98 and MB 186 (pH 7); CIP ~42 (pH 6); atrazine 14 (pH 5–7); SMX 6 (pH 7) | Precipitation, ion exchange, complexation; π–π and pore filling for organics | No reagents; lowest cost; mineral-driven metal removal | SSA ~20–150 $m^2$/g; moderate capacity; ash-derived leaching; pH rise can mimic adsorption |
| Chemically activated (KOH, $H_3PO_4$, NaOH) | KOH: $Pb^{2+}$ 185, MG 313 (pH 7.5), BPA ~68 (pH 7); $H_3PO_4$: $Pb^{2+}$ 386.5 (pH 5), BG 198 (pH 7–9); NaOH: $Pb^{2+}$ 158 (pH 5.5) | Pore filling, π–π (KOH); pyromorphite precipitation ($H_3PO_4$); complexation | Highest SSA (>1500 $m^2$/g, KOH); highest Pb capacity ($H_3PO_4$) | Corrosive reagents, washing and effluent; energy; P release; Pb bound irreversibly after $H_3PO_4$ activation |
| Physically activated (steam, $CO_2$) | MO ~85 (pH 4–6) for steam-activated biochar (900°C) | Pore filling; hydrophobic interactions | No chemical reagents; clean surface | High temperature (700–900°C); mass loss; few cow dung data |
| Hydrochar (HTC) | $Pb^{2+}$ 98, $Cd^{2+}$ 38, $Cu^{2+}$ 53 (pH 5); P-rich hydrochar: $Pb^{2+}$ 466.5, $Cd^{2+}$ 339.5, $Hg^{2+}$ 264.2 (pH not specified in source) | Precipitation, complexation, ion exchange | No pre-drying; mild temperature; abundant O-groups | Low SSA (20–100 $m^2$/g); process water; limited stability and regeneration data |
| Magnetic / metal-oxide composites | As(V) 5.9 (pH 5–7); CR 133 (pH 3–5); PFOA 12 (pH 4–6); TC 161 (pH 5–7); phosphate 37 (Fe-impregnated, pH 4–6) | Complexation, electrostatic, ligand exchange; photocatalysis ($TiO_2$) | Magnetic separation; reuse > 5 cycles reported; catalytic functions | Low capacity for As(V) and PFOA; Fe leaching at low pH; nanoparticle cost and safety |
| Polymer, clay and alginate composites | $Pb^{2+}$ 263, $Cd^{2+}$ 148, $Cu^{2+}$ 199 (pH 5.0, chitosan); RB19 156 (pH 3–5); MB 274 (pH 7, alginate) | Amine complexation, electrostatic | Mechanical stability; suited to columns; selectivity | Higher cost; polymer degradation; diffusion limits; few cow dung data |

contaminants and the precipitation of heavy metals [22].

### 2.1.3. Hydrothermal carbonization (HTC)

Hydrothermal carbonization (HTC) converts wet biomass (typically 15–25% solids content) into a carbonaceous product, termed hydrochar, through treatment with subcritical water (180–260°C) under autogenous pressure (2–10 MPa) for several hours [55,61]. In contrast to pyrolysis or gasification, HTC does not require pre-drying of the feedstock, conferring a significant energy advantage for high-moisture feedstocks such as cow dung (70–80% moisture content).

HTC hydrochar typically exhibits lower carbon content (50–70%) but higher cation exchange capacity than pyrolysis biochar, owing to the preservation of carboxyl and hydroxyl functional groups. Specific surface areas are generally lower (20–100 $m^2$/g) unless subsequent activation is applied [46,62]. Recent investigations have demonstrated that HTC-derived cow dung hydrochar can achieve comparable or superior heavy metal removal capacities relative to pyrolysis biochar, particularly for lead, cadmium, and copper, notwithstanding its lower surface

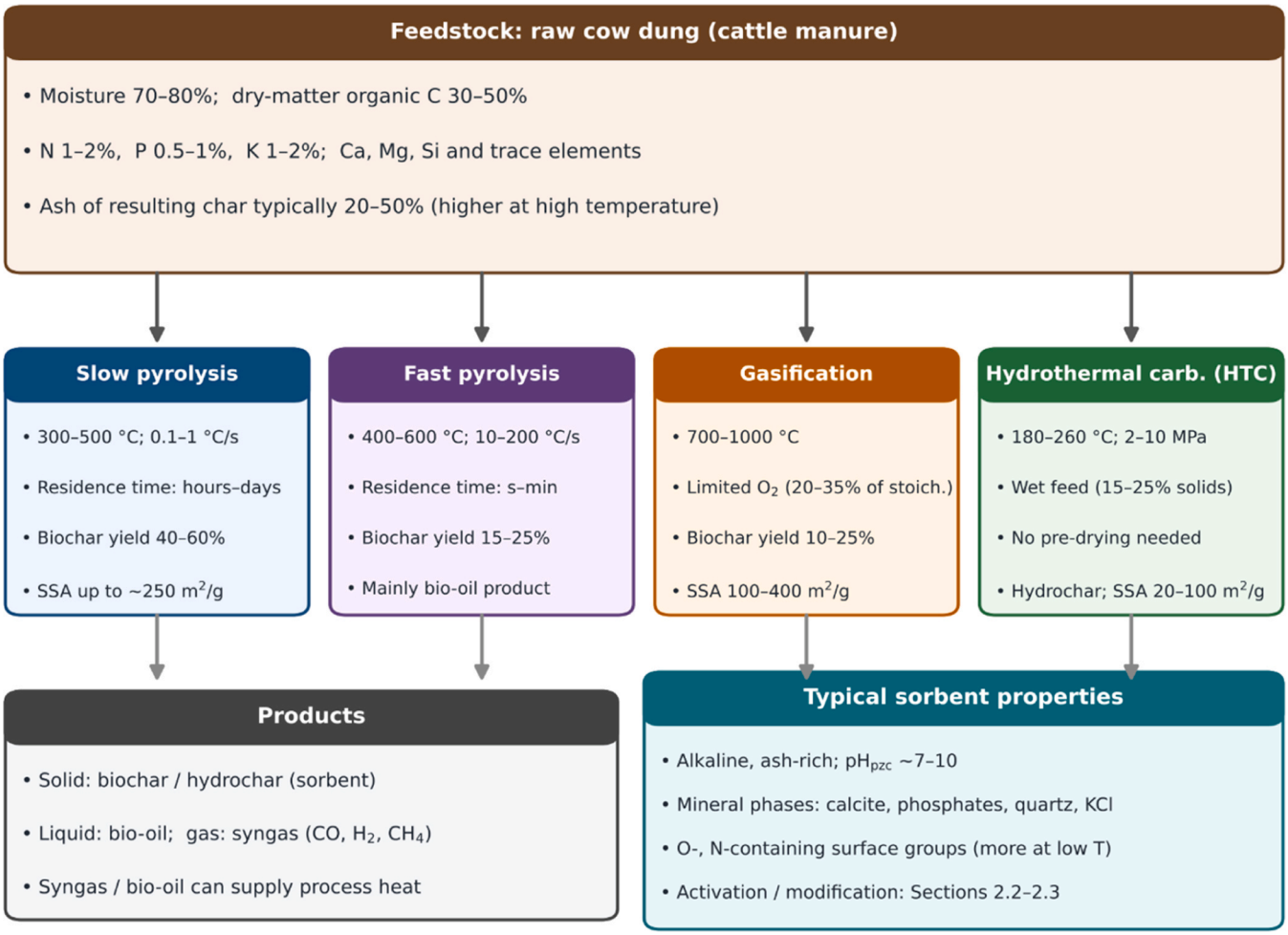



**Fig. 1.** Overview of biochar production from cow dung showing the principal thermochemical conversion pathways and typical operating windows (SSA, specific surface area). Schematic compiled by the authors from Lehmann & Joseph [15], Cha et al., [29], Libra et al. [55] and the values given in Section 2.1.

area [63-65]. This observation suggests that surface chemistry—including oxygen-containing functional groups, mineral phases, and phosphorus-bearing sites—may exert greater influence on removal performance than physical pore structure for certain contaminants. For example, a phosphorus-rich cow manure hydrochar prepared by hydrothermal carbonization exhibited exceptionally high adsorption capacities of 466.45 mg $g^{-1}$ for Pb(II), 339.48 mg $g^{-1}$ for Cd(II), and 264.19 mg $g^{-1}$ for Hg(II), with removal driven principally by precipitation, surface complexation, and ion exchange mechanisms [63].

### 2.2. *Physical and chemical activation*

Although raw cow dung biochar possesses inherent adsorption properties, activation treatments substantially enhance performance by increasing specific surface area, developing porosity, and modifying surface functional group chemistry. Activation may be accomplished through physical or chemical means, each offering distinct advantages and mechanistic pathways. The principal activation and modification strategies are summarized inFig. 2.

#### 2.2.1. *Physical activation*

Physical activation involves the controlled treatment of biochar with oxidizing gases—most commonly steam, carbon dioxide ($CO_2$), or air—at elevated temperatures (700–1000°C). This process induces controlled gasification of amorphous carbon and residual tars, thereby creating and enlarging pores and increasing accessible surface area [66, 67]. The procedure typically encompasses two stages: initial carbonization at 400–600°C followed by activation at higher temperatures using the selected oxidizing agent [49,67]. Steam activation at 700–900°C is the most aggressive physical activation method, yielding biochars with exceptionally high surface areas (>1000 $m^2/g$) and well-developed microporosity. $CO_2$ activation at 700–900°C provides more controlled pore development, predominantly generating microporosity while minimizing mass loss, and typically achieves surface areas of 300–800 $m^2/g$ [50,67]. Air oxidation at lower temperatures (200–400°C) constitutes a mild treatment that primarily introduces surface oxygen functional groups rather than generating substantial additional porosity [66].

#### 2.2.2. *Chemical activation*

Chemical activation involves impregnating the biochar precursor—or pre-carbonized biochar—with a chemical activating agent, followed by thermal treatment at temperatures typically ranging from 400 to 800°C. This approach operates at lower temperatures than physical activation while yielding biochars with substantially higher surface areas (in some cases exceeding 3000 $m^2/g$) and the capacity for precise modulation of surface chemistry [23,50].

Potassium hydroxide (KOH) is among the most widely employed chemical activating agents, generating biochars with exceptionally high surface areas and abundant micropore volume [23,50]. KOH-activated cow dung biochar routinely achieves specific surface areas exceeding 1500 $m^2/g$ with a predominantly microporous architecture, rendering it particularly effective for the removal of low-molecular-weight organic contaminants. Optimal activation typically requires KOH-to-biochar

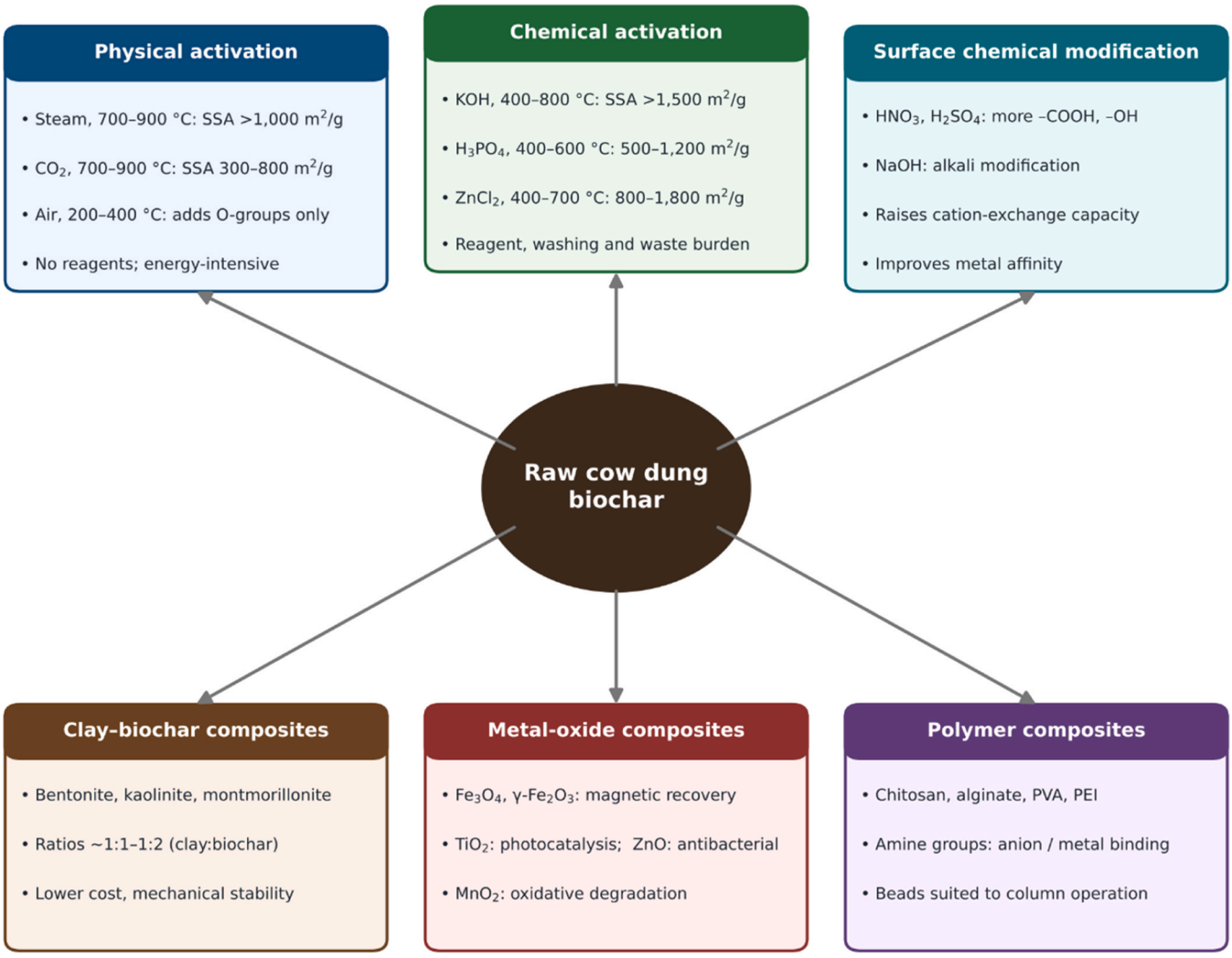



**Fig. 2.** Overview of activation and modification strategies for enhancing the physicochemical performance of raw cow dung biochar in water treatment applications. Schematic compiled by the authors from Qiu et al. [12], Han et al. [21], Inyang et al. [36] and Sections 2.2–2.3.

mass ratios of 1:1–4:1.

Phosphoric acid ($H_3PO_4$) activation proceeds at slightly lower temperatures (400–600°C) and benefits from the simultaneous occurrence of carbonization and activation in a single processing step. Residual phosphorus incorporated into the biochar structure contributes additional reactive sites for contaminant binding. $H_3PO_4$-activated cow dung biochar typically achieves surface areas of 500–1200 $m^2/g$ with a hierarchical pore structure encompassing both micro- and meso-pores [50]. Zinc chloride ($ZnCl_2$) activation at 400–700°C yields surface areas of 800–1800 $m^2/g$ with pronounced mesoporosity, which is advantageous for the adsorption of larger molecular contaminants. However, the toxicity and cost of $ZnCl_2$, combined with the necessity for thorough post-activation washing to remove residual zinc, reduce the practical appeal of this approach [50,67]. Additional chemical activating agents, including nitric acid ($HNO_3$), sulfuric acid ($H_2SO_4$), and sodium hydroxide (NaOH), have also been investigated. Nitric acid treatment is particularly notable for its capacity to introduce abundant carboxyl and hydroxyl surface groups, substantially increasing cation exchange capacity and improving affinity for heavy metals [42,68].

## 2.3. Composite and multi-material biochars

Composite and multi-material biochars represent advanced sorbent architectures developed to address the inherent limitations of raw or singly activated biochar. While activation improves surface area, porosity, and functional group density, the integration of cow dung biochar with complementary materials introduces synergistic physicochemical properties that further enhance adsorption performance, selectivity, and post-treatment recovery. The natural mineral content of cow dung biochar—including calcium, magnesium, and potassium phases—renders it particularly amenable to composite formation with metal oxides, clay minerals, and polymeric materials. The resulting hybrid materials exhibit improved selectivity and higher removal efficiencies for target contaminants through multiple cooperative mechanisms, including ion exchange, surface complexation, and electrostatic interactions [69,70].

### 2.3.1. Clay–biochar composites

The integration of biochar with clay minerals—including bentonite, kaolinite, montmorillonite, and sepiolite—combines the adsorptive properties of both components with enhanced mechanical stability and reduced material cost [69,70]. Bentonite–cow dung biochar composites have demonstrated particularly strong performance for heavy metal removal. Optimized bentonite-to-biochar ratios (1:1–1:2) have been shown to achieve 20–50% higher heavy metal removal efficiencies relative to either component in isolation [70].

#### 2.3.2. Metal oxide biochar composites

The incorporation of metal oxides into cow dung biochar imparts enhanced chemical reactivity, magnetic properties that facilitate post-treatment separation, and in some instances catalytic activity. Commonly employed metal oxides include magnetite ($Fe_3O_4$), maghemite ($\gamma$-$Fe_2O_3$), zinc oxide (ZnO), titanium dioxide ($TiO_2$), and manganese oxides ($MnO_2$, $Mn_3O_4$) [21,44,71]. Magnetic biochar composites incorporating $Fe_3O_4$ or $\gamma$-$Fe_2O_3$ nanoparticles can be rapidly recovered from treated water through the application of an external magnetic field. Studies have reported successful reuse of magnetic cow dung biochars for more than five treatment cycles with minimal loss of adsorption capacity; saturation magnetization values typically range from 10 to 40 emu/g, sufficient for efficient magnetic separation while maintaining high adsorption performance [21,44,72]. $TiO_2$ biochar composites enable the treatment of recalcitrant organic contaminants through the combination of adsorption and photocatalytic degradation pathways [21,52]. ZnO–biochar composites provide both antibacterial and adsorption functionality, while $MnO_2$–biochar composites demonstrate excellent performance in the oxidative degradation of persistent organic pollutants [21,52].

#### 2.3.3. Polymer–biochar composites

Polymeric modification of biochar improves mechanical integrity, reduces dust generation during handling, and introduces new functional groups suited to specific target contaminants. Commonly employed polymers include chitosan, alginate, polyvinyl alcohol (PVA), and polyethylenimine (PEI) [45,47,70]. Chitosan–biochar composites leverage the amine ($–NH_2$) and hydroxyl (–OH) groups of chitosan, which provide effective binding sites for cationic contaminants through combined coordination and electrostatic mechanisms. Alginate–biochar beads retain structural integrity under flow conditions and are particularly well-suited for column-based treatment systems. PEI modification introduces primary, secondary, and tertiary amine groups onto the biochar surface, substantially improving the capacity for anionic contaminant removal, including phosphate, chromate, and arsenate.

### 2.4. Characterization techniques

Comprehensive characterization of cow dung biochar is essential for interpreting adsorption performance, elucidating removal mechanisms, and guiding the optimization of material properties. Different analytical techniques provide complementary information on surface morphology, pore structure, elemental composition, crystalline phases, and surface chemistry.

#### 2.4.1. Scanning electron microscopy (SEM)

Scanning electron microscopy (SEM) is widely applied to characterize the surface morphology and pore development of biochar materials. Raw cow dung typically exhibits compact fibrous structures associated with undigested lignocellulosic biomass. Pyrolysis progressively transforms these structures through thermal decomposition; at lower temperatures (300–400°C), partial preservation of plant-derived cellular architecture and limited pore formation are commonly observed, while at elevated temperatures (500–700°C), volatilization and carbonization promote structural fragmentation, yielding a more porous surface with well-developed cavities and channels. Mineral particles derived from ash constituents become increasingly visible at higher pyrolysis temperatures. When coupled with energy-dispersive X-ray spectroscopy (EDS), SEM provides elemental mapping and semi-quantitative compositional analysis, enabling the identification of mineral phases and the confirmation of contaminant deposition following adsorption experiments [12,21,36].

#### 2.4.2. Surface area and pore size analysis (BET)

Brunauer–Emmett–Teller (BET) analysis quantifies the specific surface area, pore volume, and pore size distribution of biochar through nitrogen adsorption–desorption isotherms measured at cryogenic temperature (77 K). These parameters strongly govern adsorption capacity and mass transfer behavior during water treatment. Unmodified cow dung biochar generally exhibits moderate specific surface areas, typically 20–150 $m^2/g$ depending on feedstock composition and pyrolysis conditions. Chemical activation substantially enhances porosity and surface area; KOH activation has been reported to increase the surface area of manure-derived biochars to values exceeding 2000 $m^2/g$ under optimized conditions, while $H_3PO_4$ activation typically yields surface areas of 400–1000 $m^2/g$ with hierarchical micro–mesoporous architectures [12,21,73].

#### 2.4.3. Fourier transform infrared spectroscopy (FTIR)

Fourier transform infrared spectroscopy (FTIR) identifies the surface functional groups present on biochar by measuring infrared absorption at characteristic vibrational frequencies. Published studies on cow dung and manure-derived biochars report characteristic absorption bands associated with hydroxyl groups (O–H, ~3200–3600 $cm^{-1}$), aliphatic C–H stretching (~2850–2950 $cm^{-1}$), carbonyl groups (C═O, ~1700 $cm^{-1}$), aromatic C═C bonds (~1580–1620 $cm^{-1}$), and C–O stretching vibrations from alcohols, phenols, or ethers (~1000–1300 $cm^{-1}$). The relative intensities of these bands depend strongly on pyrolysis temperature and activation treatment. Post-adsorption FTIR spectra frequently reveal shifts in peak position, changes in band intensity, or the appearance and disappearance of specific absorption features, thereby providing mechanistic evidence for complexation, electrostatic attraction, hydrogen bonding, or surface precipitation processes [12,21].

#### 2.4.4. X-ray diffraction (XRD)

X-ray diffraction (XRD) identifies crystalline mineral phases in biochar and evaluates structural changes following modification or contaminant adsorption. Cow dung biochar commonly contains mineral phases such as calcite ($CaCO_3$), quartz ($SiO_2$), calcium phosphates, sylvite (KCl), and various metal oxides, the abundance and crystallinity of which generally increase with pyrolysis temperature as organic matter decomposes and ash-forming elements become concentrated. Post-adsorption XRD analysis has been employed to detect newly formed crystalline precipitates, providing evidence for contaminant immobilization through surface precipitation mechanisms. The identification of lead phosphate, lead carbonate, cadmium carbonate, or metal hydroxide phases in heavy metal-loaded biochars confirms the contribution of mineral-mediated reactions to overall removal performance [12,21,36].

#### 2.4.5. X-ray photoelectron spectroscopy (XPS)

X-ray photoelectron spectroscopy (XPS) is a surface-sensitive analytical technique used to determine the elemental composition and chemical bonding states of atoms within the outermost few nanometers of a material surface. Survey spectra of cow dung biochar commonly detect carbon (C 1 s), oxygen (O 1 s), nitrogen (N 1 s), and inorganic elements including calcium, potassium, phosphorus, magnesium, and silicon derived from ash constituents. High-resolution spectra provide detailed bonding environment information; the C 1 s spectrum is routinely deconvoluted into contributions from C–C/C═C (~284.6 eV), C–O (~286.0 eV), C═O (~287.0 eV), and O–C═O (~288.5 eV) species. Following contaminant adsorption, XPS studies frequently report the appearance of new peaks associated with immobilized metals (e.g., Pb 4 f, Cd 3d, Cu 2p, Cr 2p, As 3d), confirming successful surface retention. Shifts in binding energy or changes in peak intensity provide direct mechanistic evidence for electron transfer, surface complexation, or precipitation reactions [12,36].

#### 2.4.6. Additional characterization methods

Raman spectroscopy is employed to assess the degree of graphitization and structural order within biochar by characterizing the D band (disordered carbon, ~1350 $cm^{-1}$) and G band (graphitic carbon,

~1580 $cm^{-1}$). The D/G band intensity ratio provides a quantitative measure of structural disorder and aromaticity. Thermogravimetric analysis (TGA) evaluates thermal stability, fixed carbon content, volatile matter, and ash yield by monitoring mass changes as a function of temperature under controlled atmospheric conditions. Proximate and ultimate analyses quantify moisture, volatile matter, fixed carbon, and ash content, as well as elemental ratios (H/C, O/C) that serve as indicators of biochar aromaticity and polarity. The point of zero charge (pHpzc) represents the solution pH at which the net surface charge of biochar is zero; below this value, the surface carries a net positive charge favorable for anion adsorption, while above it, the negatively charged surface preferentially attracts cations. Cow dung biochar typically exhibits pHpzc values between 7 and 10, reflecting the alkaline nature of its mineral-rich composition.

### 2.5. Production cost of cow dung biochars

Whether cow dung biochar can compete with commercial sorbents depends largely on its production cost, which is governed by feedstock cost, pre-treatment (drying), reactor scale and energy integration, and post-treatment (activation, washing and modification). The feedstock itself is essentially free, but its high moisture content (70–80%) makes drying a major energy burden unless solar drying or hydrothermal carbonization is used. A techno-economic and environmental assessment of on-site pyrolysis of cattle manure in a portable unit in Idaho, USA, estimated a production cost of about USD 237 per tonne of biochar, i.e., approximately USD 0.24 $kg^{-1}$ [74].

Costs rise substantially once activation or modification is needed, because of reagents (KOH, $H_3PO_4$, $ZnCl_2$, iron salts, polymers), higher process temperatures, washing water and effluent treatment. Cost estimates for adsorbents should also be made at both laboratory and industrial scale and expressed per unit of pollutant removed rather than per kilogram of sorbent [75]. A meta-analysis of life cycle studies reported an average energy demand of 6.1 MJ $kg^{-1}$ for biochar compared with 97 MJ $kg^{-1}$ for activated carbon, and found the cost of heavy-metal adsorption on biochar to be lower than on activated carbon for Cr and Zn and comparable for Pb and Cu [76].

A simple normalization illustrates the point. At USD 0.24 $kg^{-1}$ and the $Pb^{2+}$ capacity of 113.4 mg $g^{-1}$ reported for raw dairy-manure biochar (Table 1), the sorbent cost is about USD 2.1 per kg of $Pb^{2+}$ removed, assuming single use and saturation. $H_3PO_4$-activated biochar (386.5 mg $g^{-1}$ $Pb^{2+}$) would be more cost-effective per kilogram of $Pb^{2+}$ removed than the raw material only if activation raised the price by less than about 3.4-fold (386.5/113.4), and regeneration would lower the effective cost further. This is an order-of-magnitude estimate built from different studies, not a measured cost; cost data specific to activated and composite cow dung biochars, ideally reported in USD per kg of sorbent and per kg of contaminant removed, remain scarce and are a priority for future work (Section 7).

## 3. Properties of cow dung-derived biochars relevant to water treatment

### 3.1. Surface area and porosity

Surface area and porosity constitute fundamental determinants of biochar adsorptive performance toward environmental contaminants. Pore architecture, size distribution, and pore network connectivity strongly influence both the adsorption capacity and selectivity of biochar for different pollutant classes by regulating contaminant diffusion and accessibility to active sorption sites. This relationship is complex, depending on the molecular dimensions of the target contaminant, the physicochemical structure of the biochar matrix, and the governing adsorption mechanisms, including pore filling, electrostatic attraction, hydrogen bonding, and π–π stacking interactions. Furthermore, pore connectivity and tortuosity significantly affect intraparticle mass transfer rates, thereby modulating adsorption kinetics. Micropores (diameter <2 nm) are particularly significant for the adsorption of low-molecular-weight contaminants and small organic molecules, as their dimensions are comparable to adsorbate molecular sizes, enhancing confinement effects and adsorption potential [12,21,36].

Mesopores (2–50 nm) serve as the primary sorption sites for medium-molecular-weight contaminants, including many dyes and pharmaceutical compounds, while simultaneously functioning as diffusion conduits providing access to micropore networks. Macropores (>50 nm) contribute relatively little to total adsorption capacity but serve as primary access channels facilitating the transport of adsorbates from the bulk solution to the internal pore structure. Understanding pore size distribution and its relationship to target contaminant dimensions is therefore essential for the rational optimization of cow dung biochar for specific water treatment applications.

### 3.2. Surface Chemistry and Functional Groups

Surface chemistry governs the affinity of cow dung biochar for ionic and polar contaminants. Carboxyl, hydroxyl, carbonyl and phenolic groups, formed from the cellulose, hemicellulose and lignin remaining in the manure, provide cation-exchange and complexation sites, while the relatively protein-rich manure contributes nitrogen-containing groups (amine, amide and pyridinic N) that can coordinate metals and participate in π–π and hydrogen-bonding interactions [12,33,37]. Pyrolysis temperature is the main control: chars made at 300–400°C retain abundant oxygen-containing groups and have higher cation exchange capacity, whereas higher temperatures remove H and O, lowering H/C and O/C ratios and increasing aromaticity, hydrophobicity and conductivity [15,58]. A trade-off therefore exists between the polar surface that favors metal binding and the condensed aromatic surface that favors organic contaminants.

### 3.3. Mineral composition and ash content

Ash typically accounts for 20–50% of cow dung biochar and often exceeds that of lignocellulosic chars several-fold. It is dominated by Ca, Mg, K, P, Si and Na present as calcite, phosphates, quartz, sylvite and oxides (Section 2.4.4). These phases enable mechanisms that carbon surfaces alone cannot provide: precipitation of metals as carbonates, hydroxides and phosphates (including pyromorphite for Pb), ion exchange with $Ca^{2+}$, $Mg^{2+}$ and $K^{+}$, and ligand exchange with phosphate [20,36]. The same minerals have drawbacks. Ash can block pores and dilute the carbon fraction, and soluble salts, phosphate and, depending on animal feed, trace metals can leach into the water being treated [22,40]. The balance between inherent phosphorus release and pollutant immobilization is therefore a design variable, and uptake should be assessed together with the release of Ca, Mg, P and other ions.

### 3.4. Solution pH, pHpzc and surface charge

Carbonates, oxides and hydroxides make cow dung biochar alkaline, and its pH and pHpzc generally increase with pyrolysis temperature (pHpzc typically 7–10; Section 2.4.6). Because of this alkalinity and buffering capacity, contact with the biochar can raise the pH of acidic metal solutions and promote hydroxide and carbonate precipitation. Part of the 'adsorption' reported in batch tests at fixed initial pH may therefore be precipitation induced by the sorbent itself, and the pH should be measured at equilibrium and the removed species verified (e.g., by XRD or XPS) before capacities are compared. Surface charge is positive below the pHpzc and negative above it, which explains the preference for cations and cationic dyes at near-neutral pH and the need for acidic conditions or cationic modification for anionic dyes and arsenate (Sections 4.2 and 5.2).

### 3.5. Chemical stability and leaching

The carbon in cow dung biochar is less stable than that of woody biochar because of its lower fixed-carbon and higher ash content, but it is still resistant to biological degradation over years to centuries [15,18]. For water treatment the more immediate concern is chemical stability in contact with water: dissolution of carbonates at low pH, release of P, K and Na, release of dissolved organic carbon from low-temperature chars, and mobilization of trace metals originating from feed additives. These factors argue for washing and conditioning of biochar before use, leaching tests (batch and column) on every new material, and reporting of effluent quality alongside removal efficiency.

## 4. Mechanisms of contaminant removal

### 4.1. Physical adsorption and pore filling

Physical adsorption, or physisorption, constitutes one of the principal pathways through which cow dung biochar removes contaminants from aqueous systems. This process is driven by non-covalent van der Waals forces between the adsorbate and the biochar surface, and typically does not involve the formation or breaking of chemical bonds. Pore filling is a particularly important physical adsorption mechanism, whereby contaminant molecules migrate into and become physically entrapped within the pore structure of biochar, driven by concentration gradients and diffusive transport processes. This mechanism is most effective when the molecular dimensions of the target contaminant are compatible with the prevailing pore size distribution of the biochar [12,21].

For nonpolar and weakly polar organic contaminants, hydrophobic interactions with the carbonaceous matrix of biochar also contribute to physical adsorption. Biochars produced at higher pyrolysis temperatures, which exhibit greater graphitization and reduced surface oxygen content, typically display stronger hydrophobic character and correspondingly higher affinity for nonpolar organic molecules through this mechanism.

### 4.2. Electrostatic interactions

Electrostatic interactions between charged surface sites on biochar and ionic contaminants in solution represent an important mechanism governing removal efficiency. The surface charge of cow dung biochar is primarily determined by its pHpzc and the prevailing solution pH. At pH values above the pHpzc (typically 7–10 for cow dung biochar), the biochar surface carries a net negative charge, promoting electrostatic attraction of cationic contaminants such as heavy metal ions ($Pb^{2+}$, $Cd^{2+}$, $Cu^{2+}$, $Ni^{2+}$, $Zn^{2+}$) and cationic dyes (e.g., methylene blue, malachite green) [12]. Conversely, at pH values below the pHpzc, the positively charged surface favors the adsorption of anionic contaminants, including phosphate, arsenate, and anionic dyes. Understanding the pH-dependent surface charge behavior of cow dung biochar is therefore essential for optimizing treatment conditions and predicting removal performance under varying solution chemistry.

### 4.3. Ion exchange

Ion exchange is a particularly significant removal mechanism for heavy metals and nutrients in cow dung biochar systems. The mineral-rich nature of cow dung biochar results in the presence of readily exchangeable base cations—principally $Ca^{2+}$, $Mg^{2+}$, $K^{+}$, and $Na^{+}$—associated with mineral phases or loosely bound to surface functional groups [22]. During adsorption, these indigenous cations can be displaced by target heavy metal ions from solution, constituting a classic cation exchange process. The degree to which ion exchange contributes to overall removal can be assessed by quantifying the release of exchangeable base cations concurrent with target metal uptake. Studies on cow dung-derived biochars have consistently reported significant contributions from ion exchange, particularly for divalent metals such as $Pb^{2+}$, $Cu^{2+}$, and $Cd^{2+}$ [22,42,77].

### 4.4. Surface complexation

Surface complexation involves the formation of coordinate bonds between contaminant ions or molecules and surface functional groups on cow dung biochar. This mechanism is particularly important for heavy metal ions, which can form inner-sphere or outer-sphere complexes with oxygen-containing surface groups, including carboxyl (–COOH), hydroxyl (–OH), and phenolic ($–C_6H_4OH$) functionalities [12]. Inner-sphere complexation, involving direct coordination of metal ions to surface oxygen donor atoms, is generally stronger and more specific than outer-sphere complexation, which involves electrostatic association with minimal direct bonding. For organic contaminants, surface complexation may involve hydrogen bonding between functional groups of the adsorbate and surface moieties of biochar, π–π electron donor–acceptor interactions between aromatic structures, and n–π interactions between electron-donating nitrogen or oxygen atoms and π-acceptor aromatic systems [12,21].

### 4.5. Redox reactions

Redox reactions constitute an important mechanism governing the removal of redox-active contaminants by biochar, involving electron transfer processes between redox-active surface moieties and contaminants exhibiting variable oxidation states. This mechanism is particularly relevant for the immobilization or transformation of inorganic pollutants such as chromium(VI), mercury species, and certain redox-sensitive organic compounds. Among these, the reduction of highly toxic and mobile Cr(VI) to the less toxic, less soluble Cr(III) has been extensively investigated and reported. Following reduction, Cr(III) commonly precipitates as $Cr(OH)_3$ under mildly acidic to alkaline conditions ($pH > 5.5$), thereby decreasing its mobility and bioavailability [78,79].

X-ray photoelectron spectroscopy analyses have confirmed the presence of Cr(III) species on biochar surfaces following exposure to Cr(VI) solutions, providing direct evidence of reductive immobilization. Concurrent oxidation of oxygen-containing functional groups on the biochar surface suggests that these groups function as electron donors during the reduction process. The characteristic color change of chromium solutions from yellow (Cr(VI)) to green (Cr(III)) provides further qualitative confirmation of the redox transformation [78,79].

The redox functionality of biochar can be further enhanced through the incorporation of transition metals such as Fe, Mn, and Ti, yielding engineered composite biochars with catalytic properties. These materials can activate chemical oxidants such as persulfate (peroxydisulfate, PDS, or peroxymonosulfate, PMS) or hydrogen peroxide to generate reactive oxygen species that degrade recalcitrant organic pollutants resistant to conventional adsorption [21,47,52].

Persulfate activation by carbon materials, including biochar, proceeds through both radical and non-radical pathways, and the contribution of each depends on defect density, persistent free radicals (PFRs), oxygen functional groups (notably C═O groups), the $sp^2$ carbon network and any incorporated metals [80–82]. The radical pathway generates sulfate ($SO_4^{\bullet-}$) and hydroxyl (•OH) radicals by single-electron transfer from PFRs and surface donor groups to PMS or PDS, and superoxide ($O_2^{\bullet-}$) through reactions involving dissolved oxygen and the oxidant. The non-radical pathways include singlet oxygen ($^1O_2$), formed from PMS on carbonyl and defect sites, and direct electron transfer, in which the carbon surface mediates electron flow from the adsorbed pollutant to the surface-bound oxidant; high-valent iron-oxo species may add to these in Fe-containing composites. Non-radical routes are more selective toward electron-rich pollutants and are less affected by chloride, bicarbonate and natural organic matter than $SO_4^{\bullet-}$ and •OH,

which is important for real wastewater [80,82].

The relative importance of these species is usually assigned from quenching experiments (e.g., methanol, tert-butanol, p-benzoquinone, furfuryl alcohol or L-histidine) combined with EPR spin trapping (DMPO, TEMP). Scavengers are not fully selective, so EPR and electrochemical evidence should accompany them. Mechanistic studies of persulfate activation specifically on cow dung biochar remain limited; the inherent Fe, Mn and nitrogen species of manure may contribute, but this has yet to be demonstrated and most evidence derives from other biochars.

### 4.6. *Synergistic mechanisms in complex systems*

In practical water and wastewater treatment systems, contaminant removal by biochar rarely proceeds through a single dominant pathway. Instead, multiple physicochemical mechanisms operate simultaneously or sequentially, often interacting synergistically to enhance overall remediation efficiency. This multi-mechanistic behavior is most evident in the removal of heavy metals, where electrostatic attraction, ion exchange, surface complexation, precipitation, and intraparticle diffusion collectively contribute to adsorption, immobilization, and long-term stabilization.

For heavy metal contaminants, the initial removal stage is commonly dominated by electrostatic attraction, whereby positively charged metal ions are rapidly drawn to negatively charged biochar surfaces. Ion exchange with indigenous base cations follows, with subsequent surface complexation through the formation of inner-sphere coordination bonds between metal ions and oxygen-containing surface functional groups. Under favorable pH and solution chemistry conditions, precipitation as metal hydroxides, carbonates, or phosphates further reduces contaminant mobility. Slow intraparticle diffusion enables metal ions to migrate into internal pore structures over extended periods, where they may become physically entrapped or strongly retained [36,41,83,84].

A comprehensive understanding of these interacting mechanisms is essential for the rational design and optimization of biochar-based sorbents. For heavy metal remediation, biochars with elevated cation exchange capacity, abundant oxygen-containing functional groups, and alkaline mineral constituents are generally preferred. In contrast, organic pollutant removal is favored by biochars with high specific surface area, well-developed porosity, condensed aromatic domains, and hydrophobic surface characteristics. For nutrient removal, surface charge optimization and specific functional group chemistry are particularly important, as adsorption behavior depends strongly on the ionic speciation of the nutrient in solution [12,36,69]. The principal mechanisms are summarized schematically in Fig. 3.

## 5. Adsorption performance of cow dung biochars for different contaminant classes

The adsorption performance of cow dung-derived biochar varies substantially depending on the chemical properties, charge, molecular dimensions, and solubility of the target contaminant, as well as the surface characteristics of the biochar [21,36]. For heavy metals, adsorption primarily proceeds through ion exchange, electrostatic attraction, surface complexation, and precipitation with mineral components present in the biochar [12]. For synthetic dyes, removal is governed principally by pore filling, π–π interactions, and charge-based attraction between dye molecules and the biochar surface [12]. Pharmaceuticals and personal care products are typically adsorbed through hydrophobic interactions, hydrogen bonding, and donor–acceptor mechanisms depending on molecular structure [21], while nutrient removal is influenced by ion exchange capacity and mineral precipitation reactions [85]. Biochar production conditions modulate these mechanisms: elevated pyrolysis temperatures generally increase surface area and promote organic pollutant adsorption, while lower temperatures preserve functional groups that enhance metal binding [15].

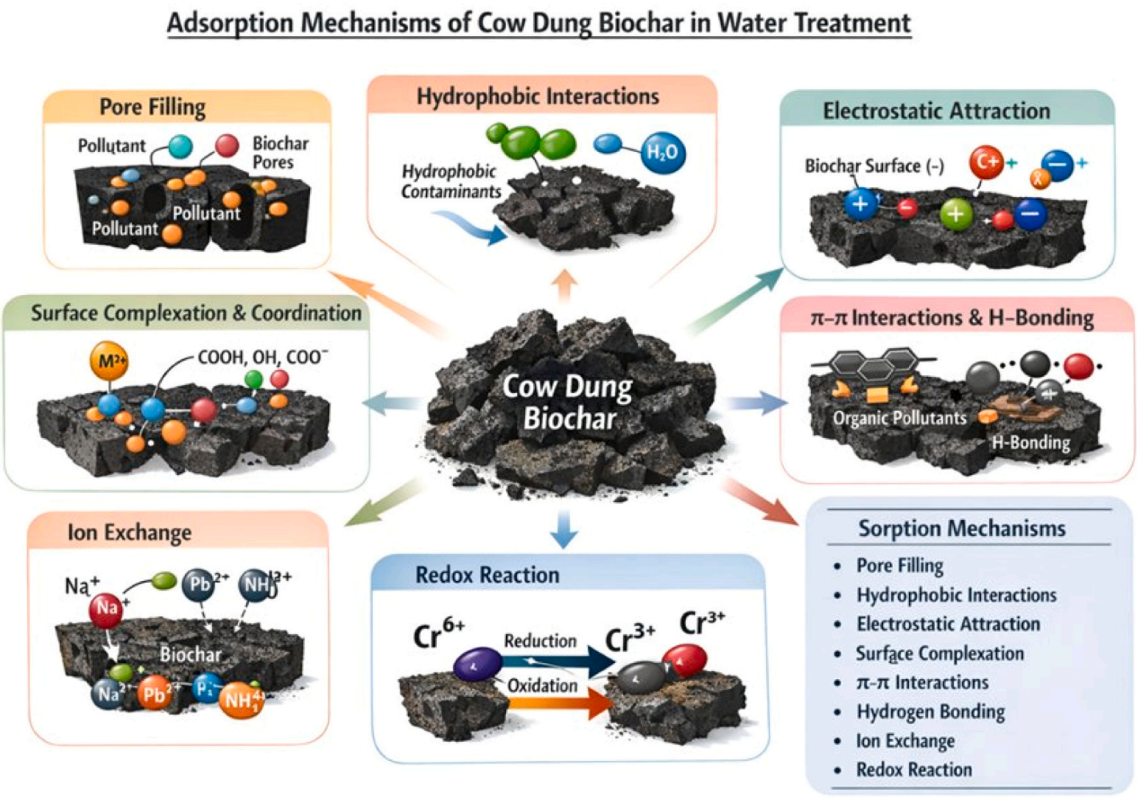



**Fig. 3.** Schematic illustration of the principal adsorption mechanisms operating at the cow dung biochar–water interface during contaminant removal. (Adapted from [21].).

### 5.1. Heavy metals

Heavy metals—including Pb, Cd, Cu, Cr, As, Zn, and Ni—represent the most extensively studied contaminant class in the context of cow dung-derived biochar systems. At near-neutral pH, the majority of these metals exist as cationic species and are attracted to the negatively charged surfaces of cow dung biochar through electrostatic attraction. Multiple additional mechanisms contribute to overall removal, including surface complexation with oxygen-containing functional groups (–COOH and –OH), ion exchange with inherent base cations ($Ca^{2+}$, $Mg^{2+}$), and precipitation as insoluble hydroxide, carbonate, and phosphate phases [22,36]. Experimental evidence consistently demonstrates that these mechanisms operate simultaneously, collectively enhancing sorption efficiency.

Modification of cow dung biochar substantially improves heavy metal removal performance. Phosphoric acid ($H_3PO_4$) activation greatly enhances $Pb^{2+}$ immobilization through the formation of stable lead phosphate minerals, notably pyromorphite [40,43]. Incorporation of additives such as magnetic $Fe_3O_4$, chitosan, or nano zero-valent iron increases specific surface area, introduces complementary functional groups, and promotes redox-assisted adsorption [45,47]. These modifications are reflected in reported adsorption capacities that increase from approximately 113 mg $g^{-1}$ in raw dairy manure biochar (pH 5.0) to over 380 mg $g^{-1}$ following $H_3PO_4$ activation (pH 5.0; Table 1), highlighting the critical contribution of phosphate-induced precipitation to high removal efficiencies.

### 5.2. Synthetic dyes

Synthetic dyes are a major class of organic pollutants for which manure-derived biochars have been extensively evaluated as adsorbents. They are broadly categorized as cationic dyes (e.g., methylene blue, malachite green, crystal violet) and anionic dyes (e.g., Congo red, methyl orange, reactive blue). Cow dung biochars generally exhibit stronger affinity for cationic dyes at neutral to slightly alkaline pH, where the biochar surface carries a net negative charge and promotes electrostatic attraction with positively charged dye molecules. Removal of anionic dyes is less favorable and typically requires acidic conditions to protonate surface sites or chemical modification with cationic agents such as chitosan or polyethyleneimine (PEI) [45,69].

Beyond electrostatic interactions, multiple physicochemical mechanisms contribute to dye adsorption, including pore filling, hydrophobic interactions, and π–π electron donor–acceptor interactions between the aromatic structures of biochar and dye molecules [21,23]. Activation treatments—particularly KOH or steam activation—substantially enhance these processes by increasing surface area and pore volume. Activated biochars can achieve BET surface areas exceeding 900 $m^2/g$, resulting in substantially improved dye uptake capacities. These combined mechanisms explain the superior performance of engineered cow dung biochars for the removal of a broad spectrum of synthetic dyes.

### 5.3. Pharmaceuticals and personal care products (PPCPs)

Pharmaceuticals—including antibiotics (sulfonamides, tetracyclines, fluoroquinolones), analgesics, and synthetic hormones—are increasingly recognized as persistent environmental contaminants of concern. Cow dung biochars can effectively remove these compounds through multiple interacting mechanisms, including π–π interactions between the aromatic rings of pharmaceutical molecules and the graphitic domains of biochar, hydrogen bonding involving surface functional groups such as –OH and –COOH, hydrophobic partitioning of nonpolar drug molecules into the carbon matrix, and electrostatic interactions that depend on solution pH and the speciation of both the pharmaceutical and the biochar surface [12,21,49].

Modification and activation substantially enhance PPCP removal performance. Activated biochars—particularly those treated with $ZnCl_2$ or engineered into magnetic porous composite materials—exhibit substantially higher adsorption capacities due to increased surface area, pore volume, and functional group density. $ZnCl_2$-activated magnetic porous carbon has been reported to achieve tetracycline adsorption capacities of approximately 161 mg $g^{-1}$, substantially exceeding the performance of unmodified biochars [21]. These findings underscore the critical importance of pore structure, surface chemistry, and functionalization for the efficient removal of pharmaceutical contaminants using cow dung-derived biochars.

### 5.4. Nutrients (Phosphate, Ammonium, and Nitrate)

The removal of nutrient species, including phosphate ($PO_4^{3-}$), ammonium ($NH_4^+$), and nitrate ($NO_3^-$), is critical for mitigating eutrophication in receiving water bodies. Cow dung biochar removes phosphate primarily through precipitation and surface complexation mechanisms. Calcium-rich mineral phases naturally present in manure biochar—including hydroxyapatite and $CaCO_3$—facilitate phosphate immobilization through ligand exchange and precipitation reactions, while iron or aluminum modification introduces additional high-affinity binding sites. Iron-modified biochars have been shown to significantly enhance phosphate uptake through the formation of strong Fe–O–P bonds, with adsorption capacities increasing from approximately 18 mg $g^{-1}$ in raw biochar to around 37 mg $g^{-1}$ following iron incorporation [54].

Ammonium removal is governed primarily by cation exchange and electrostatic attraction, as $NH_4^+$ ions interact with negatively charged functional groups on the biochar surface [21,86]. Nitrate removal presents a greater challenge, as $NO_3^-$ is anionic; effective adsorption typically requires biochar modification to introduce positive surface charges or cationic functionalization. Although nutrient adsorption capacities are generally lower than those reported for heavy metals or dyes, engineered cow dung biochars offer practical value for treating agricultural wastewater and anaerobic digestate, where simultaneous nutrient recovery and pollution control are required.

### 5.5. Pesticides

Pesticides such as atrazine, chlorpyrifos, and glyphosate are persistent agricultural pollutants for which removal using manure-derived biochars has been extensively investigated. Cow dung biochar adsorbs pesticides primarily through hydrophobic partitioning and π–π interactions, mechanisms particularly effective for nonpolar or weakly polar pesticide molecules that preferentially associate with the aromatic, carbon-rich domains of the biochar matrix [12,40]. The degree of adsorption is strongly influenced by pyrolysis temperature; biochars produced at temperatures $\geq 500°C$ exhibit higher aromaticity and hydrophobicity, enhancing affinity for organic contaminants.

Chemical activation markedly improves pesticide removal performance. KOH activation significantly increases surface area and pore volume, creating additional adsorption sites. For example, dairy manure biochar produced at 500°C adsorbs approximately 14.3 mg $g^{-1}$ of atrazine, while KOH-activated variants can achieve capacities of approximately 48 mg $g^{-1}$ [23]. This enhancement is attributable to improved pore structure and increased accessible surface area, which facilitate the diffusion and retention of pesticide molecules. These results emphasize the importance of both pyrolysis conditions and post-treatment modifications for optimizing cow dung biochar in pesticide remediation applications.

### 5.6. Emerging contaminants (PFAS and Endocrine Disruptors)

Emerging contaminants, including per- and polyfluoroalkyl substances (PFAS) and endocrine-disrupting compounds such as bisphenol A (BPA), represent among the most challenging targets for biochar-based treatment systems. PFAS are particularly resistant to adsorption

owing to the exceptional strength of C–F bonds and their amphiphilic molecular structure, which limits effective interaction with conventional sorbents [11]. In cow dung biochars, removal of PFAS such as perfluorooctanoic acid (PFOA) is generally modest, typically requiring high-surface-area materials or surface modification (e.g., magnetic or functionalized biochars) to achieve appreciable uptake. Reported adsorption capacities for PFOA (~12 mg $g^{-1}$) remain substantially lower than those observed for dyes or pharmaceuticals, reflecting fundamental limitations in affinity and binding interactions.

In contrast, endocrine disruptors such as BPA are more amenable to biochar adsorption. Hydrophobic interactions and π–π electron donor–acceptor interactions dominate their removal, particularly in highly aromatic and porous biochars [12,32]. KOH-activated biochars significantly enhance BPA removal performance by increasing surface area and exposing additional graphitic domains, with adsorption capacities reported to reach approximately 68 mg $g^{-1}$. Despite these advances, further research is needed to engineer biochar surfaces with tailored functionalities capable of improving PFAS affinity, particularly through strategies that enhance electrostatic attraction and fluorophilic interactions [11].

### 5.7. Comparative performance across pollutant classes

Across different pollutant categories, the adsorption behavior of cow dung biochar is commonly described using classical equilibrium and kinetic models. For heavy metals and many dye systems, the Langmuir isotherm provides the best fit to experimental data, supporting the concept of monolayer adsorption onto a finite number of uniform active sites [22,40]. In contrast, pesticides and many pharmaceutical compounds are more accurately described by the Freundlich isotherm, reflecting adsorption on heterogeneous surfaces with multiple energy sites and the possibility of multilayer interactions [23]. Across most pollutant classes, adsorption kinetics are governed by the pseudo-second-order (PSO) model, indicating that chemisorption is the rate-limiting step, involving electron sharing or transfer between adsorbent and adsorbate [87].

Comparative studies consistently demonstrate that engineered biochars substantially outperform unmodified materials. Activation using KOH or $H_3PO_4$ dramatically increases surface area, pore volume, and functional group density, yielding significantly higher adsorption capacities across all pollutant types. These activated biochars can improve removal efficiencies by approximately 3–8-fold for organic contaminants such as dyes and pharmaceuticals, with more moderate improvements (1.5–3-fold) observed for heavy metals, where mineral precipitation and surface complexation also contribute substantially to removal [36]. Modified biochars including magnetic and polymer-functionalized composites not only improve adsorption selectivity, particularly for anionic contaminants, but also enable easier post-treatment separation from aqueous systems [47]. Overall, these comparisons reveal a consistent trend: specific surface area dominates organic contaminant removal, while both surface chemistry and mineral composition jointly govern heavy metal adsorption efficiency.

### 5.8. Continuous-flow (Fixed-Bed) performance

Most capacities in Tables 1–3 come from batch experiments, in which a fixed mass of sorbent is equilibrated with a fixed volume of solution. Real treatment units, such as household filters, community fixed beds and permeable reactive barriers, operate under continuous flow, where performance is set by the shape and timing of the breakthrough curve rather than by the equilibrium capacity, and depends on bed height, flow rate (empty-bed contact time), influent concentration, particle size and competing ions [38]. Column capacities are usually lower than batch capacities because contact time is short and equilibrium is not reached, but they are the data needed for design. Breakthrough data are commonly analysed with the Bohart–Adams, Thomas and Yoon–Nelson models [88–90], which yield the bed capacity, rate constant and time to 50% breakthrough and allow scale-up, for example by the bed-depth service-time approach.

Fixed-bed studies have been reported for biochars from several feedstocks. Alkali-modified biochar was evaluated for the removal of Pb, Cu, Cd, Zn and Ni in combined batch and column tests [42], and $H_2O_2$-modified hydrochar from peanut hull showed improved heavy-metal removal in columns relative to the unmodified material [91]. Reviews of adsorption under continuous-flow conditions have compared adsorbents using column performance criteria such as saturation capacity, service time and the number of regeneration cycles [92], although that review addresses dye adsorbents rather than metals on biochar. For cow dung and cattle-manure biochars specifically, column studies are far fewer than batch studies, and few report full breakthrough curves, bed parameters and regeneration; continuous-flow evidence for these materials should therefore be regarded as scarce rather than absent. The earlier statement of an 'absence' of such studies in the literature is accordingly restricted to complete breakthrough analyses for cow dung biochar.

Practical issues specific to manure biochar include fines and dust that clog beds, low bulk density that causes channelling or flotation, and ash dissolution that raises effluent pH and releases phosphate and salts during the first bed volumes. These can be mitigated by pelletizing or granulating the biochar, using alginate, clay or polymer binders (Section 2.3), mixing with sand to maintain permeability, and pre-washing. Future column studies should report bed height, flow rate or empty-bed contact time, particle size, influent concentration, capacity at 5–10% and 50% breakthrough, effluent pH and leachates, performance in real water matrices, and performance over repeated loading–regeneration cycles.

### 5.9. Comparison with other adsorbents

The value of cow dung biochar is best judged against the sorbents it would replace: other biochars, commercial activated carbon, and low-cost mineral sorbents such as natural zeolites and clays (Table 5). Commercial activated carbon has far higher surface area than any unmodified biochar and remains the benchmark for organic contaminants and dyes, where surface area and pore structure dominate uptake [16,93]. For divalent metals, however, cow dung biochar benefits from mineral-driven precipitation and ion exchange, and the $Pb^{2+}$ capacities compiled in Table 1 (113 mg $g^{-1}$ raw to 386.5 mg $g^{-1}$ $H_3PO_4$-activated) lie at or above the range usually reported for natural zeolites, clays and activated carbon, which for metals is generally of the order of tens of mg $g^{-1}$ [94–96]. Because test conditions differ, such comparisons are indicative only; the most direct evidence is the meta-analysis of Alhashimi & Aktas [76], which found lower energy demand for biochar than for activated carbon and lower or comparable cost per unit of heavy metal adsorbed.

Zeolites and clays have the advantages of low cost, availability, selectivity for particular cations (e.g., $NH_4^+$ and $Pb^{2+}$ on clinoptilolite) and cation-exchange capacity, but they have low surface area, little affinity for organic compounds and anions without modification, and limited regeneration options [95,96]. Cow dung biochar combines some of these mineral functions with an organic carbon matrix, and clay–biochar composites (Section 2.3.1) combine both. Its weaknesses relative to activated carbon are lower surface area, variable quality and ash-related leaching, and relative to zeolites, a less defined and less reproducible mineral phase.

### 5.10. Regeneration and reusability

Reusability determines the real cost of a sorbent and the amount of waste it generates. Four approaches are used. Acid or chelant washing (dilute HCl or $HNO_3$, EDTA) desorbs metals held by ion exchange or complexation; solvent or alkaline washing (ethanol, methanol, NaOH) is

**Table 5**
Qualitative comparison of cow dung biochar with other adsorbents for water treatment (compiled from [22,76,94–96] and Tables 1–3).

| Sorbent | Typical properties | Strengths | Limitations | Relative cost and energy |
|---|---|---|---|---|
| Cow dung biochar (this review) | SSA ~20–150 $m^2$/g raw, > 1500 $m^2$/g after KOH activation; ash 20–50%; alkaline | Mineral-driven removal of divalent metals and phosphate; dyes and drugs after activation; waste valorization | Variable composition; leaching of P and salts; weak for As (V), nitrate, PFAS | Low (≈USD 0.24 $kg^{-1}$ for raw biochar; higher if activated) |
| Lignocellulosic (straw, wood) biochar | Higher C and SSA, low ash | High affinity for organics after activation | Weaker mineral-mediated metal uptake | Low to moderate |
| Commercial activated carbon | SSA typically several hundred to > 1000 $m^2$/g | Benchmark for organics, dyes, taste and odor; well-established | Cost and energy-intensive production; metals uptake often modest unless modified | Higher (energy demand ~97 MJ/kg vs ~6 MJ/kg for biochar; [76]) |
| Natural zeolite (e.g., clinoptilolite) | Low SSA; framework cation exchange | Selective for $NH_4^+$, $Pb^{2+}$, $Cd^{2+}$; low cost; stable | Poor for organics and anions; limited capacity | Low |
| Clay minerals (bentonite, kaolinite) | Low to moderate SSA; permanent charge (bentonite) | Low cost; cation exchange; good for cationic dyes and metals | Poor for anions and organics unless modified; separation difficulties | Low |

used for dyes and organic contaminants; thermal treatment (300–600°C) can restore capacity for organics; and oxidative regeneration (persulfate, Fenton, photocatalysis) can destroy adsorbed organics, particularly on composites [12,32]. Magnetic cow dung biochar composites have been reused for more than five cycles with minimal loss of capacity, and thermal regeneration is reported to recover 60–90% of the initial capacity over five cycles (Section 6.4).

Regeneration is more difficult for cow dung biochar than for lignocellulosic biochar, because a large part of its capacity comes from mineral phases rather than from surface sites. Acid washing dissolves carbonate and phosphate phases, so capacity is lost with each cycle; metals precipitated as very insoluble minerals such as pyromorphite ($Pb_5(PO_4)_3Cl$) are effectively irreversibly bound, so that sorbents used for Pb removal by phosphate-driven precipitation are better managed as stabilized wastes than regenerated; and the ash may sinter during thermal regeneration. Capacity losses over repeated cycles also arise from incomplete desorption, pore blocking, attrition and loss of the sorbent during handling. Studies reporting more than five cycles, regeneration in column mode, or regeneration in real wastewater remain few.

Reusability studies should report the regeneration agent and conditions, desorption efficiency, capacity retention per cycle, mass loss, and the leaching of metals and nutrients in each cycle, and should be accompanied by an economic and environmental comparison of regeneration with single use followed by safe disposal or valorization (Section 6.4). For phosphate and ammonium, in particular, single use followed by application of the nutrient-laden biochar as a slow-release fertilizer may be preferable to regeneration.

## 6. Life cycle assessment of cow dung-derived biochars

While the adsorption performance of cow dung-derived biochars is well documented, understanding their environmental sustainability requires more than just laboratory efficiency measures. Life Cycle Assessment (LCA) is a standardized methodology (ISO 14040/14044) that quantifies environmental impacts throughout a product's full life cycle, from raw material extraction and production through use and end-of-life management. For cow dung biochars, LCA reveals environmental dimensions that adsorption studies alone cannot capture. It accounts for energy consumed during pyrolysis, greenhouse gas emissions arising from thermochemical conversion, avoided emissions from agricultural waste diversion, and long-term carbon storage in soils or other receiving environments. As the application of biochar in water treatment advances from laboratory scale toward real-world implementation, LCA provides the evidentiary basis necessary to support investment decisions, guide process optimization, and substantiate environmental claims [97,98].

### 6.1. LCA methodology and system boundaries for biochar systems

A comprehensive LCA of cow dung biochar systems encompasses four principal life cycle phases: (i) feedstock collection and pretreatment, including drying and size reduction; (ii) thermochemical conversion, such as pyrolysis, gasification, or hydrothermal carbonization; (iii) biochar transport and application in water treatment; and (iv) end-of-life management of spent biochar. The functional unit is typically defined as one tonne of biochar produced, one tonne of $CO_2$ equivalents sequestered, or one kilogram of contaminant removed, depending on the analytical context. A central methodological challenge lies in assigning environmental credits for avoided burdens: when cow dung is diverted from open-field decomposition or conventional manure lagoon storage, the associated methane and nitrous oxide emissions are avoided, generating an environmental credit that substantially improves the net environmental balance of the biochar system [99,100].

Defining system boundaries is particularly important for manure-derived biochar systems. If the boundary is set at the farm gate, avoided emissions from conventional manure management practices are excluded, presenting a less favorable LCA profile. When a broader, cradle-to-grave boundary is adopted, these avoided emissions substantially enhance net environmental performance. A comprehensive evaluation framework proposed by Azzi et al. [101] identifies six categories of biochar-specific effects frequently overlooked by standard LCA approaches. These include carbon storage in biochar, changes in soil greenhouse gas fluxes, substitution of conventional soil amendments, effects on crop yields, modifications of soil organic carbon stocks, and improvements in downstream water quality—all of which are relevant when spent cow dung biochar from water treatment is subsequently applied to agricultural land. The life cycle stages, burdens and benefits considered are summarized in Fig. 4.

### 6.2. Environmental impacts during cow dung biochar synthesis

The synthesis stage constitutes the most energy-intensive phase of the biochar life cycle. Slow pyrolysis of agricultural residues and animal manures at 400–700°C requires substantial thermal energy, which is typically supplied through combustion of co-produced syngas or bio-oil, or from external fossil fuel sources. When pyrolysis is configured as an energy-integrated system, heat recovery from syngas and bio-oil combustion can substantially reduce the demand for external energy inputs; under such conditions, the carbon footprint of biochar production decreases markedly and may become net-negative. A 2022 life cycle assessment of pyrolysis processes across multiple agricultural feedstocks demonstrated a net carbon sequestration potential when accounting for syngas combustion credits, positioning biochar as a promising alternative to commercial activated carbon [102].

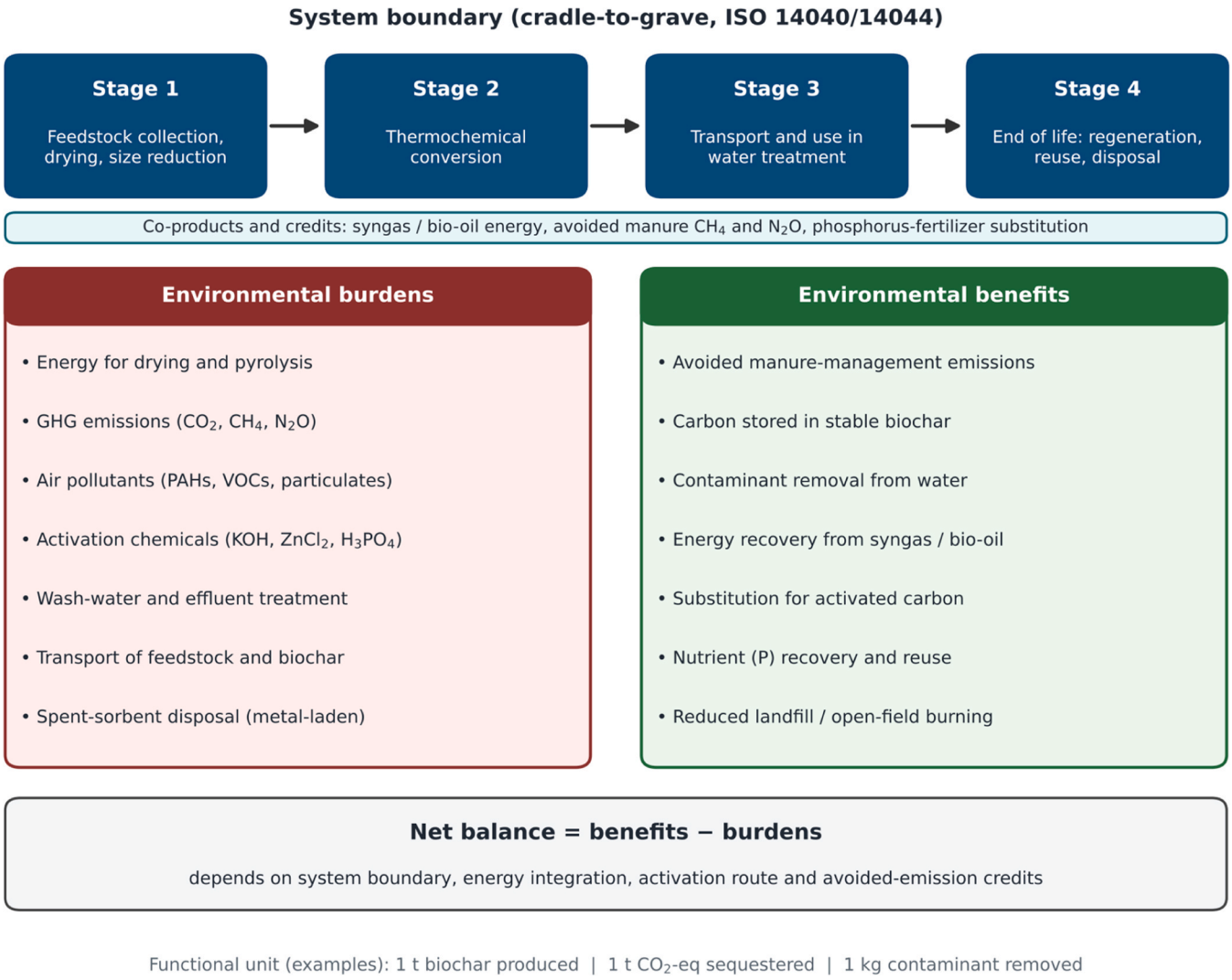



**Fig. 4.** Life cycle assessment (LCA) framework for cow dung-derived biochar in water treatment (ISO 14040/14044): life cycle stages, environmental burdens and benefits.

For cattle manure specifically, a techno-economic and environmental assessment conducted in Idaho, USA found that on-site pyrolysis using a portable refinery unit produced biochar at approximately $237 per metric tonne, with a carbon footprint of 951 kg $CO_2$ equivalents per metric tonne of biochar produced [74]. While this figure appears substantial in isolation, it must be evaluated against the avoided emissions from conventional manure management practices, which significantly offset the production burden. Comparable findings emerged from a comparative LCA of cattle manure anaerobic digestion versus direct land application, which reported a lower cumulative energy demand and reduced climate change impact for the thermochemical pathway when co-products including syngas-derived electricity and nutrient-rich ash were accounted for [103].

The choice of pyrolysis temperature exerts considerable influence on the LCA profile. Higher pyrolysis temperatures (600–900°C) demand greater energy input but yield biochar with superior chemical stability and elevated fixed carbon content, thereby securing a greater proportion of carbon per unit of feedstock in long-term storage. Lower temperatures (300–500°C) are more energy-efficient but produce biochars with higher O:C ratios and reduced long-term stability, diminishing the effective carbon sequestration credit. Comparative LCA analyses have demonstrated that carbon sequestration potential varies substantially with both feedstock type and pyrolysis temperature. Animal manure-derived biochars generally exhibit higher ash content and lower fixed carbon per unit mass relative to lignocellulosic biochars, which can constrain their net climate benefits per unit mass while still delivering favorable environmental performance per unit of contaminant removed [104].

Beyond climate change, the synthesis stage introduces additional environmental burdens recognized by LCA. Pyrolysis off-gases may contain polycyclic aromatic hydrocarbons (PAHs), volatile organic compounds (VOCs), and fine particulate matter if gas treatment systems are inadequate, contributing to photochemical oxidant formation and human toxicity impacts. Chemical activation treatments—particularly those employing KOH and $ZnCl_2$—introduce supplementary environmental burdens associated with reagent production and the necessity for wastewater treatment to recover activation chemicals and wash effluents. A comparative LCA examining willow-derived and pig manure biochar in Belgium found willow biochar to be environmentally superior across multiple impact categories, underscoring the significance of feedstock selection beyond process design alone in determining overall LCA outcomes [105]. For cow dung biochar, the higher mineral content and lower calorific value of this feedstock relative to woody biomass indicate that greater external energy inputs may be required unless the pyrolysis kiln is specifically designed for high-moisture, high-ash manure substrates.

### 6.3. Carbon sequestration potential and climate change mitigation

Carbon sequestration represents the most frequently cited environmental benefit of biochar production and is central to LCA-based comparisons with conventional water treatment sorbents. When biomass carbon that would otherwise return to the atmosphere through biological decomposition becomes trapped within the stable aromatic matrix of biochar, a durable carbon sink is established. For cow dung feedstocks, approximately 30–50% of the dry-weight organic carbon is retained in the biochar following slow pyrolysis, with the remainder released as $CO_2$, CO, and hydrocarbon gases during the conversion process. The carbon retained in biochar exhibits considerable resistance to decomposition, with estimated mean residence times in soil spanning hundreds to thousands of years—substantially exceeding the weeks-to-months residence time characteristic of unprocessed manure.

A systematic review of biochar-to-soil LCA studies [106] found that the majority of assessed systems demonstrated net greenhouse gas benefits (Table 6), with carbon sequestration and avoided emissions from feedstock decomposition consistently outweighing production-phase emissions, particularly when renewable energy sources were employed in the thermochemical process. A gasification-based biochar system evaluated in Italy demonstrated a net greenhouse gas reduction of approximately 8300 kg $CO_2$ equivalents per tonne of biochar, incorporating credits for electricity generation from syngas combustion [107]. These results highlight the critical importance of integrated biorefinery designs that recover energy from thermochemical co-products; the practice of venting or flaring syngas without energy recovery substantially degrades the LCA profile.

In agricultural system contexts, co-application of biochar with cattle manure compost has been reported to reduce methane emissions from composting operations by up to 79%, contributing a significant reduction in global warming potential [108]. This finding is particularly relevant in circular economy scenarios where cow dung biochar production occurs on a livestock farm. Biochar produced from one fraction of the manure stream can be used to treat farm-generated wastewater while concurrently being incorporated into composting operations to mitigate methane emissions, thereby generating a cascade of environmental co-benefits that are not typically captured within single-process LCA boundaries.

### 6.4. Environmental impacts at end-of-life: disposal and reuse of spent biochar

The environmental fate of spent cow dung biochar following water treatment represents a significant but frequently neglected dimension of LCA. Once biochar has adsorbed heavy metals, synthetic dyes, pharmaceuticals, or nutrients, it becomes contaminated and cannot be applied to agricultural soils without risk of pollutant remobilization to the environment. Three principal end-of-life pathways are available for contaminant-laden spent biochar: (i) regeneration and reuse; (ii) co-combustion or incineration for energy recovery; and (iii) controlled landfill disposal. Each option entails distinct environmental trade-offs.

Regeneration—involving the restoration of adsorption capacity through thermal treatment (300–600°C), chemical washing with acid or base solutions, or solvent extraction—is the preferred option because it extends biochar service life and reduces demand for fresh material production. However, regeneration consumes energy and chemical reagents. Repeated regeneration cycles progressively degrade the biochar's pore structure and surface chemistry, ultimately reducing adsorption efficiency over time. LCA analyses indicate that thermal regeneration recovers 60–90% of original adsorption capacity over five cycles, providing a net environmental benefit relative to single-use disposal when the avoided burden of fresh biochar production is incorporated [12,32]. For cow dung biochar specifically, the high mineral ash content complicates thermal regeneration, as the ash matrix does not regenerate and may undergo sintering at elevated temperatures, necessitating careful temperature management.

When spent biochar carries heavy metal contamination, land application is contraindicated. In such cases, co-incineration with municipal solid waste or industrial waste streams can recover residual energy from the carbon fraction while mineralizing organic contaminants. However, incineration releases sequestered carbon as $CO_2$, negating long-term carbon storage benefits, and may volatilize heavy metals unless adequate flue gas treatment is in place. The choice between incineration (energy recovery with loss of carbon sequestration credit) and landfill disposal (preservation of sequestered carbon but with associated land

**Table 6**
Summary of life cycle assessment studies relevant to cow dung and cattle manure biochar systems.

| Study / Feedstock | System Boundary | Conversion Method | GHG Impact (kg $CO_2$ eq./t biochar) | Key Environmental Benefits | Key Burdens / Limitations | Net Outcome / Reference |
|---|---|---|---|---|---|---|
| Cattle manure pyrolysis (Idaho, USA) | Cradle-to-gate | Slow pyrolysis (portable kiln) | 951 | Carbon sequestration; nutrient-rich biochar; effective waste management | High transportation energy demand; avoided emissions not accounted for | Net positive when avoided emissions included; Patel et al. [74] |
| Agro-residue pyrolysis (multi-feedstock) | Cradle-to-grave | Slow pyrolysis with syngas combustion | Net carbon-negative | Long-term carbon sequestration; potential replacement for activated carbon | Variability in feedstock properties; reliance on syngas energy credits | Promising substitute for activated carbon; Roberts et al. [102] |
| Biochar-to-soil systems (review) | Cradle-to-grave | Multiple (pyrolysis/ gasification) | Predominantly net-negative | Carbon sequestration; enhanced soil fertility; reduced emissions | Data limitations; insufficient characterization of eutrophication impacts | Overall GHG benefits outweigh production impacts; Tisserant & Cherubini [106] |
| Gasification biochar (Italy) | Cradle-to-grave | Gasification with electricity generation | -8300 | Strong carbon sink; renewable energy co-generation | High capital investment; sensitive to electricity grid emission factors | Highly carbon-negative performance; Puzone et al. [107] |
| Dairy manure compost with biochar addition | Gate-to-use | Composting with biochar amendment | 79% $CH_4$ reduction vs. control | Lower global warming potential; improved compost quality; methane mitigation | Limited system boundary (excludes biochar production impacts) | Significant GWP reduction; Sun et al. [108] |
| Biochar as green sorbent for soil remediation (review) | Cradle-to-grave | Pyrolysis (various feedstocks) | Net benefit vs. conventional methods | Lower energy demand than pump-and-treat; effective pollutant immobilization | Uncertain toxicity impacts; unclear spent sorbent disposal | Favorable alternative to conventional remediation; Pennesi et al. [109] |
| Biochar production (circular economy review) | Cradle-to-grave | Multiple thermochemical routes | Carbon-efficient; net positive with co-product credits | Climate change mitigation; improved resource efficiency; supports circular economy | Variability in end-of-life scenarios; sensitivity to allocation methods | Promising pathway for sustainable resource use; Tisserant & Cherubini [110] |

use and potential leachate risks) depends substantially on the carbon intensity of the local electricity grid and the engineering standards of available landfill infrastructure. In regions with highly decarbonized electricity grids, the energy credit attributable to incineration is diminished, making engineered landfill containment a more climatically favorable option.

A particularly promising pathway within a circular economy framework involves the use of phosphate-laden spent biochar as a slow-release fertilizer. Cow dung-derived engineered biochar can effectively adsorb phosphorus from wastewater streams and subsequently release it gradually upon land application, improving crop nutrition while simultaneously recovering a nutrient of finite global availability from water treatment effluent [62]. In this scenario, spent biochar is transformed from a waste requiring management into an agricultural resource, eliminating disposal requirements and substituting for the production of synthetic phosphate fertilizer—an energy-intensive process dependent on non-renewable mineral extraction. This approach embodies LCA principles of circular economy design and represents an environmentally superior outcome relative to conventional spent biochar management strategies.

### 6.5. Comparative LCA: cow dung biochar versus conventional water treatment sorbents

Situating cow dung biochar within the broader water treatment landscape requires systematic comparison with established sorbent technologies. Commercial activated carbon, produced primarily from coal, coconut shells, or wood through energy-intensive thermochemical activation processes, remains the benchmark sorbent for water and wastewater treatment. Its production generates substantial greenhouse gas emissions and imposes significant energy demands. By contrast, cow dung biochar can be manufactured with considerably lower energy requirements and reduced carbon dioxide emissions, advantages that are amplified when evaluated on the basis of contaminant removal per kilogram of sorbent. The elevated mineral content of cow dung biochar facilitates the removal of heavy metals and phosphate through precipitation and ion exchange mechanisms without additional energy expenditure, contributing to favorable performance metrics in functional unit comparisons.

Several LCA studies have documented positive environmental outcomes for cow dung and manure-derived biochars. Analyses incorporating soil application scenarios have identified benefits across climate change, acidification, and eutrophication impact categories. Comparative assessments of biochar-based soil remediation relative to conventional pump-and-treat technologies consistently demonstrate the environmental advantages of biochar-based approaches in terms of both energy consumption and greenhouse gas emissions. However, the LCA evidence base for cow dung biochar specifically in water treatment applications remains limited, and not all studies report uniformly favorable outcomes. Chemical activation treatments, particularly those employing KOH or $ZnCl_2$, can introduce substantial additional energy and chemical burdens that offset benefits derived from carbon sequestration and waste diversion. Furthermore, the compositional variability of cow dung—influenced by animal diet, seasonal patterns, and farm management practices—introduces uncertainty into LCA results and complicates the generalization of findings across different production contexts.

### 6.6. Gaps and future directions in LCA of cow dung biochar for water treatment

Despite growing interest in biochar-based water treatment, the LCA evidence base for cow dung biochar in this specific application context remains sparse. The preponderance of existing LCA studies addresses biochar applied to agricultural soils rather than water treatment systems, leaving significant knowledge gaps regarding the environmental performance of cow dung biochar across the full water treatment life cycle. Critical areas requiring investigation include the environmental implications of advanced modification strategies—such as magnetic nanoparticle incorporation, polymer coating, and metal oxide compositing—which are rarely evaluated within a complete LCA framework. Additionally, most available studies assess only a limited subset of environmental impact categories, frequently omitting human toxicity, ecotoxicity, land use, and water consumption impacts that may be material to a comprehensive environmental evaluation.

Future research priorities include the development of LCA studies conducted under conditions representative of scaled deployment, with particular attention to the environmental and economic performance of cow dung biochar production and water treatment in low- and middle-income countries where both water quality deficits and agricultural waste management challenges are most acute. Dynamic LCA approaches that model temporal changes in carbon storage stability, sorbent degradation, and end-of-life management outcomes over extended time horizons are needed to complement static inventory analyses. Integration of techno-economic assessment with environmental LCA will be essential to evaluate the commercial viability of cow dung biochar water treatment systems and to identify cost–environment trade-offs that inform both technology selection and policy development. Finally, the development of standardized LCA methodologies specifically adapted to biochar water treatment systems—addressing issues of co-product allocation, functional unit definition, and system boundary selection—would substantially enhance the comparability of results across studies and strengthen the evidence base for regulatory and investment decision-making.

## 7. Current challenges and key issues to be resolved

The preceding sections show that cow dung biochar is a promising, low-cost sorbent, but the evidence base is dominated by short-term batch experiments with synthetic solutions. The following issues must be resolved before the technology can be recommended for practical water treatment.

### 7.1. Standardization and data Quality

Reported capacities are hard to compare because feedstock origin, diet, manure age, pyrolysis conditions, particle size, dose, initial concentration, temperature, contact time and ionic strength are not standardized and are often incompletely reported. Precipitation caused by the alkalinity and phosphate of the sorbent is frequently counted as adsorption, mineral phases are seldom verified after sorption, and isotherm parameters are often obtained by linearized fits. Minimum reporting standards, including equilibrium pH, dose, effluent leachates and mechanism verification, and the use of non-linear model fitting would greatly improve comparability.

### 7.2. Real wastewater and multi-contaminant systems

Nearly all capacities were measured in single-contaminant, deionized-water systems. Real waters contain competing cations ($Ca^{2+}$, $Mg^{2+}$), anions (phosphate, carbonate, sulfate), natural organic matter and mixtures of metals, dyes and drugs, which can strongly reduce uptake or alter the mechanism. Tests with real surface water, groundwater and industrial or agricultural effluents, and with competitive multi-solute systems, are largely missing.

### 7.3. Scale-up, continuous flow and field validation

No pilot- or field-scale evaluation of cow dung biochar for water treatment was identified, and column data are limited (Section 5.8). Issues of hydraulics, clogging, particle size and binders, bed lifetime and effluent quality at scale have not been addressed. Demonstration units at

household or community scale in regions with high pollution burdens, with defined performance and cost targets, are the next logical step.

### 7.4. Regeneration and management of spent biochar

Regeneration data for cow dung biochar are scarce, and the mineral-based capacity of the material makes conventional acid or thermal regeneration problematic (Section 5.10). The environmental fate of metal-laden, dye-laden and drug-laden spent biochar, including the stability of the immobilized contaminants and their toxicity characteristic leaching, needs systematic study to avoid shifting pollution from water to soil or landfill (Section 6.4).

### 7.5. Safety, leaching and feedstock quality

Cattle manure may contain trace metals from feed additives, veterinary residues and pathogens. Pyrolysis destroys pathogens and many organic residues, but metals are concentrated in the char, and PAHs can form under poor combustion control. Release of phosphate, salts and metals from the biochar into treated water, and the resulting eutrophication or toxicity risk, must be assessed for each feedstock and production condition, and quality standards for biochar intended for drinking-water or effluent treatment are needed.

### 7.6. Techno-economics and life cycle evidence

The cost of cow dung biochar has been estimated for basic pyrolysis (Section 2.5), but not for activated or composite materials, and few LCAs address water treatment rather than soil application (Section 6.6). Integrated techno-economic and life cycle assessments using a functional unit based on contaminant removed, and including regeneration or disposal, are needed to show whether the performance gain from activation justifies the added cost and burden.

### 7.7. Emerging contaminants and mechanistic understanding

Performance toward PFAS, nitrate, arsenate and other anionic or highly polar species is poor (Sections 5.4 and 5.6), and microplastics and antibiotic resistance genes have received little attention. Mechanistic claims often rest on a limited set of ex situ spectra; combining in situ spectroscopy, quantitative phase analysis, surface complexation and reactive-transport modelling with controlled experiments would separate the contributions of adsorption, precipitation, ion exchange and redox processes.

## 8. Future prospects and research directions

(i) Design of low-cost, low-burden activation: steam or $CO_2$ activation, co-pyrolysis of manure with lignocellulosic residues to raise surface area, and mineral engineering (Ca, Mg, Fe or clay addition) can tune the balance between surface area and mineral reactivity while avoiding corrosive reagents and wash effluents.

(ii) Application-specific materials: matching the biochar to the target, for example mineral-rich chars for Pb, Cd and phosphate, high-surface-area chars for dyes and pharmaceuticals, and functionalized or composite materials for anions, rather than seeking a single universal sorbent.

(iii) Engineered configurations: granulated or pelletized biochar for fixed beds, biochar-amended sand filters, constructed wetlands and permeable reactive barriers, and magnetic composites for batch treatment, with performance, hydraulics and lifetime demonstrated at pilot scale.

(iv) Circular use of spent biochar: phosphate- and ammonium-loaded biochar as a slow-release fertilizer, and stabilization or safe disposal of metal-laden material, assessed through LCA and leaching tests.

(v) Data-driven design: systematic, standardized datasets and machine-learning models linking feedstock composition, production conditions and sorption capacity could reduce trial-and-error experimentation, provided that the underlying data meet the reporting standards discussed in Section 7.1.

(vi) Integration with decentralized systems in low- and middle-income countries: where manure is abundant and conventional infrastructure is lacking, community-scale pyrolysis units coupled with household or community filters could provide an affordable route to safe water, provided that quality control, cost and local acceptance are addressed.

(vii) Policy and standards: biochar quality standards and guidance for the use of manure-derived biochars in water treatment would give regulators and users confidence in the technology.

## 9. Conclusions

Cow dung biochar is a low-cost, mineral-rich sorbent whose strongest performance is the removal of divalent metals and phosphate by precipitation and ion exchange, and of cationic dyes and organic micropollutants after activation. Reported capacities range from about 6 mg $g^{-1}$ for As(V) to several hundred mg $g^{-1}$ for $Pb^{2+}$, $Cd^{2+}$ and $Hg^{2+}$ on phosphorus-rich materials, but they depend strongly on pH, modification and test conditions. Compared with activated carbon, cow dung biochar has lower surface area but lower cost and energy demand; compared with zeolites and clays, it offers an organic carbon matrix in addition to mineral functions. Its main limitations are ash-related leaching, feedstock variability, the burden of chemical activation, difficult regeneration and weak affinity for anionic and fluorinated contaminants. Life cycle evidence is favorable but sparse for water treatment. Before the material can be recommended for practical use, standardized testing, fixed-bed and real-wastewater studies, regeneration and spent-sorbent management, and techno-economic assessments are required.

## CRediT authorship contribution statement

**Purity Ngui:** Writing – original draft, Methodology, Investigation. **Victor O. Shikuku:** Writing – review & editing, Supervision, Conceptualization. **Emily C. Ng'eno:** Writing – review & editing. **John O. Otieno:** Writing – review & editing.

## Consent to publish declaration

Not applicable.

## Ethics approval and consent to participate

Not applicable.

## Clinical trial registration

Not applicable.

## Declaration of Generative AI and AI-assisted technologies in the writing process

During the preparation of this work the authors used Google Gemini to produce enhanced Figures. After using this tool, the authors reviewed and edited the content as needed and take full responsibility for the content of the publication.

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

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## Data availability

No data was used for the research described in the article.

## References

[1] World Health Organization (WHO). Progress on Household Drinking Water, Sanitation and Hygiene 2000–2022. Geneva: WHO/UNICEF; 2023.
[2] Lin L, Yang H, Xu X. Effects of water pollution on human health and disease heterogeneity: a review. Front Environ Sci 2022;10:880246.
[3] Järup L. Hazards of heavy metal contamination. Br Med Bull 2003;68(1):167–82.
[4] Qasem NAA, Mohammed RH, Lawal DU. Removal of heavy metal ions from wastewater: a comprehensive and critical review. NPJ Clean Water 2021;4:36.
[5] Al-Tohamy R, Ali SS, Li F, et al. A critical review on the treatment of dye-containing wastewater: ecotoxicological and health concerns of textile dyes and possible remediation approaches for environmental safety. Ecotoxicol Environ Saf 2022;231(113160):113160.
[6] Lellis B, Fávaro-Polonio CZ, Pamphile JA, Polonio JC. Effects of textile dyes on health and the environment and bioremediation potential of living organisms. Biotechnol Res Innov 2019;3(2):275–90.
[7] Carpenter SR, Caraco NF, Correll DL, Howarth RW, Sharpley AN, Smith VH. Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecol Appl 1998;8(3):559–68.
[8] Smith VH, Tilman GD, Nekola JC. Eutrophication: impacts of excess nutrient inputs on freshwater, marine, and terrestrial ecosystems. Environ Pollut 1999;100(1–3):179–96.
[9] Ankley GT, Brooks BW, Huggett DB, Sumpter JP. Repeating history: pharmaceuticals in the environment. Environ Sci Technol 2007;41(24):8211–7.
[10] Kümmerer K. The presence of pharmaceuticals in the environment due to human use. J Environ Manag 2009;90(8):2354–66.
[11] Brusseau ML, Anderson RH, Guo B. PFAS concentrations in soils: background levels versus contaminated sites. Sci Total Environ 2020;740(140017):140017.
[12] Qiu M, Liu L, Ling Q, et al. Biochar for the removal of contaminants from soil and water: a review. Biochar 2022;4:19.
[13] Inyang M, Dickenson E. The potential role of biochar in the removal of organic and microbial contaminants from potable and reuse water: A review. Chemosphere 2015;134:232–40.
[14] Glaser B, Haumaier L, Guggenberger G, Zech W. The 'Terra Preta' phenomenon: a model for sustainable agriculture in the humid tropics. Naturwissenschaften 2001;88(1):37–41.
[15] Lehmann J, Joseph S, editors. Biochar for Environmental Management: Science, Technology and Implementation. 2nd ed. London. Routledge; 2015.
[16] Mohan D, Sarswat A, Ok YS, Pittman CU. Organic and inorganic contaminants removal with biochar, a renewable, low cost and sustainable adsorbent – A critical review. Bioresour Technol 2014;160:191–202.
[17] Ahmad M, Rajapaksha AU, Lim JE, et al. Biochar as a sorbent for contaminant management in soil and water: a review. Chemosphere 2014;99:19–33.
[18] Cao X, Harris W. Properties of dairy-manure-derived biochar pertinent to its potential use in remediation. Bioresour Technol 2010;101(14):5222–8.
[19] Cao X, Ma L, Gao B, Harris W. Dairy-manure derived biochar effectively sorbs lead and atrazine. Environ Sci Technol 2009;43(9):3285–91.
[20] Xu X, Cao X, Zhao L. Comparison of rice husk- and dairy manure-derived biochars for simultaneously removing heavy metals from aqueous solutions. Chemosphere 2013;92(8):955–61.
[21] Han M, Liu Z, Huang S, et al. Application of biochar-based materials for effective pollutant removal in wastewater treatment. Nanomaterials 2024;14(23):1933.
[22] Lee S, Kim M, Park G, Jung S, Kwon EE. Animal manure biochar for the removal of hazardous pollutants from wastewater. Biochar X 2025;1(0):e003.
[23] Wang J, Zhang M. Adsorption characteristics and mechanisms of Pb(II) and Zn(II) by a novel KOH-activated biochar derived from cow manure waste. RSC Adv 2017;7(31):18941–50.
[24] Jiang H, Li X, Dai Y. Phosphoric acid activation of cow dung biochar for adsorbing enrofloxacin in water: Icing on the cake. Environ Pollut 2024;341(122887):122887.
[25] Mao X, Liu Y, Long S. Research progress on adsorption mechanisms and regeneration applications of modified biochar for heavy metals in wastewater. Desalin Water Treat 2025;324(101399):101399.
[26] Manya JJ. Pyrolysis for biochar purposes: a review to establish current knowledge gaps and research needs. Environ Sci Technol 2012;46(15):7939–54.
[27] Wang J, Wang S. Preparation, modification and environmental application of biochar: A review. J Clean Prod 2019;227:1002–22.
[28] Sohi SP, Krull E, Lopez-Capel E, Bol R. A review of biochar and its use and function in soil. Adv Agron 2010;105:47–82.
[29] Cha JS, Park SH, Jung S, Ryu C, Jeon J, Shin M, Park Y. Production and utilization of biochar: A review. J. Ind. Eng. 2016;40:1–15. https://doi.org/10.1016/j.jiec.2016.06.002.
[30] Jeffery S, Verheijen FGA, van der Velde M, Bastos AC. A quantitative review of the effects of biochar application to soils on crop productivity. Agric Ecosyst & Environ 2011;144(1):175–87.
[31] Lehmann J, Gaunt J, Rondon M. Biochar sequestration in terrestrial ecosystems: A review. Mitig Adapt Strateg Glob Change 2006;11:403–27.
[32] Wang J, Wang S. Preparation, modification and environmental application of biochar: A review. J Clean Prod 2019;227:1002–22.
[33] Mukherjee A, Zimmerman AR, Harris W. Surface chemistry variations among a series of laboratory-produced biochars. Geoderma 2011;163(3–4):247–55.
[34] FAO. Livestock's Long Shadow – Environmental Issues and Options. Rome: Food and Agriculture Organization of the United Nations; 2006.
[35] Vandecasteele B, Sinicco T, D'Hose T, Vanden Nest T, Mondini C. Biochar amendment before or after composting affects compost quality and N losses, but not P plant uptake. J Environ Manag 2016;168:200–9.
[36] Inyang MI, Gao B, Yao Y, et al. A review of biochar as a low-cost adsorbent for aqueous heavy metal removal. Crit Rev Environ Sci Technol 2016;46(4):406–33.
[37] Keiluweit M, Nico PS, Johnson MG, Kleber M. Dynamic molecular structure of plant biomass-derived black carbon (biochar). Environ Sci Technol 2010;44(4):1247–53.
[38] Gwenzi W, Chaukura N, Noubactep C, Mukome FND. Biochar-based water treatment systems as a potential low-cost and sustainable technology for clean water provision. J Environ Manag 2017;197:732–49.
[39] Xu X, Cao X, Zhao L, Wang H, Yu H, Gao B. Removal of Cu, Zn, and Cd from aqueous solutions by the dairy manure-derived biochar. Environ Sci Pollut Res 2013;20(1):358–68.
[40] Wang H, Wen Y, Ding Y, et al. Rapid and effective lead elimination using cow manure derived biochar: balance between inherent phosphorus release and pollutants immobilization. Toxics 2023;11(1):1.
[41] Uchimiya M, Lima IM, Klasson KT, Chang SC, Wartelle LH, Rodgers JE. Immobilization of heavy metal ions (CuII, CdII, NiII, and PbII) by broiler litter-derived biochars in water and soil. J Agric Food Chem 2010;58(9):5538–44.
[42] Ding Z, Hu X, Wan Y, Wang S, Gao B. Removal of lead, copper, cadmium, zinc, and nickel from aqueous solutions by alkali-modified biochar: batch and column tests. J Ind Eng Chem 2016;33:239–45.
[43] Yao Y, Gao B, Inyang M, et al. Biochar derived from anaerobically digested sugar beet tailings: characterization and phosphate removal potential. Bioresour Technol 2011;102(10):6273–8.
[44] Zhang M, Gao B, Varnoosfaderani S, Hebard A, Yao Y, Inyang M. Preparation and characterization of a novel magnetic biochar for arsenic removal. Bioresour Technol 2013;130:457–62.
[45] Deng J, Liu Y, Liu S, et al. Competitive adsorption of Pb(II), Cd(II) and Cu(II) onto chitosan-pyromellitic dianhydride modified biochar. J Colloid Interface Sci 2017;506:355–64.
[46] Regmi P, Garcia Moscoso JL, Kumar S, Cao X, Mao J, Schafran G. Removal of copper and cadmium from aqueous solution using switchgrass biochar produced via hydrothermal carbonization. J Environ Manag 2012;109:61–9.
[47] Wang B, Gao B, Wan Y. Entrapment of ball-milled biochar in Ca-alginate beads for the removal of methylene blue dye from aqueous solution. J Ind Eng Chem 2019;76:313–9.
[48] Lonappan L, Rouissi T, Das RK, et al. Adsorption of methylene blue on biochar microparticles derived from different waste materials. Waste Manag 2016;49:537–44.
[49] Rajapaksha AU, Vithanage M, Ahmad M, et al. Enhanced sulfamethazine removal by steam-activated invasive plant-derived biochar. J Hazard Mater 2015;290:43–50.
[50] González JF, Román S, Encinar JM, Martínez G. Pyrolysis of various biomass residues and char utilization for the production of activated carbons. J Anal Appl Pyrolysis 2009;85(1–2):134–41.
[51] Yao Y, Gao B, Chen H, et al. Adsorption of sulfamethoxazole on biochar and its impact on reclaimed water irrigation. J Hazard Mater 2012;209-210(209–210):408–13.
[52] Fang G, Gao J, Liu C, Dionysiou DD, Wang Y, Zhou D. Key role of persistent free radicals in hydrogen peroxide activation by biochar. Environ Sci Technol 2014;48(3):1902–10.
[53] Pan M, Chu LM. Adsorption and degradation of five selected antibiotics in agricultural soil. Sci Total Environ 2016;545-546(545–546):48–56.
[54] Michalekova-Richveisova B, Fristak V, Pipiska M, Duriska L, Moreno-Jimenez E, Soja G. Iron-impregnated biochars as effective phosphate sorption materials. Environ Sci Pollut Res 2017;24(1):463–75.
[55] Libra JA, Ro KS, Kammann C, et al. Hydrothermal carbonization of biomass residuals: a comparative review of the chemistry, processes and applications. Biofuels 2011;2(1):71–106.
[56] Qian K, Kumar A, Zhang H, Bellmer D, Huhnke R. Recent advances in utilization of biochar. Renew Sustain Energy Rev 2015;42:1055–64.
[57] Claoston N, Samsuri AW, Ahmad Husni MH, Mohd Amran MS. Effects of pyrolysis temperature on the physicochemical properties of empty fruit bunch and rice husk biochars. Waste Manag Res 2014;32(4):331–9.
[58] Uchimiya M, Chang SC, Klasson KT. Screening biochars for heavy metal retention in soil: role of oxygen functional groups. J Hazard Mater 2011;190(1–3):432–41.

[59] Zhang J, Liu J, Liu R. Effects of pyrolysis temperature and heating time on biochar obtained from the pyrolysis of straw and lignosulfonate. Bioresour Technol 2014;176:288–91.
[60] Novak JM, Lima I, Xing B, et al. Characterization of designer biochar produced at different temperatures and their effects on a loamy sand. Ann Environ Sci 2009;3:195–206.
[61] Fang J, Zhan L, Ok YS, Gao B. Minireview of potential applications of hydrochar derived from hydrothermal carbonization of biomass. J Ind Eng Chem 2018;57:15–21.
[62] Fang J, Gao B, Zimmerman AR, Ro KS. Removing phosphorus from aqueous solutions using engineered biochar and its potential for nutrient recycling. Chemosphere 2018;201:594–600.
[63] Fu H, Wang B, Wang H, Liu H, Xie H, Han L, Wang N, Sun X, Feng Y, Xue L. Assessment of livestock manure-derived hydrochar as cleaner products: Insights into basic properties, nutrient composition, and heavy metal content. J. Clean. Prod. 2022;330:129820. https://doi.org/10.1016/j.jclepro.2021.129820.
[64] Zhang C, Sun S, Xu S, Johnston C, Wu C. Phosphorus Removal from Dirty Farmyard Water by Activated Anaerobic-Digestion-Derived Biochar. Ind. Eng. Chem. Res. 2022;62(45):19216–24. https://doi.org/10.1021/acs.iecr.2c02668.
[65] Fan M, Li C, Sun Y, Zhang L, Zhang S, Hu X. In situ characterization of functional groups of biochar in pyrolysis of cellulose. Sci. Total Environ. 2021;799:149354. https://doi.org/10.1016/j.scitotenv.2021.149354.
[66] Shen W, Li Z, Liu Y. Surface chemical functional groups modification of porous carbon. Recent Pat Chem Eng 2008;1(1):27–40.
[67] Zhang C, Sun S, Xu S, Wu C. CO2 capture over steam and KOH activated biochar: effect of relative humidity. Biomass– Bioenerg 2022;166(106608):106608.
[68] Boehm HP. Surface oxides on carbon and their analysis: a critical assessment. Carbon 2002;40(2):145–9.
[69] Rajapaksha AU, Chen SS, Tsang DC, et al. Engineered/designer biochar for contaminant removal/immobilization from soil and water. Chemosphere 2016;148:276–91.
[70] Tan XF, Liu YG, Gu YL, et al. Biochar-based nano-composites for the decontamination of wastewater: a review. Bioresour Technol 2016;212:318–33.
[71] Zhang G, Liu H, Liu R, Qu J. Removal of phosphate from water by a Fe-Mn binary oxide adsorbent. J Colloid Interface Sci 2009;335(2):168–74.
[72] Wang S, Zhao M, Zhou M, et al. Biochar-supported nZVI (nano zero-valent iron) composite for contaminant removal. Sci Total Environ 2019;671:1095–109.
[73] Foo KY, Hameed BH. Insights into the modeling of adsorption isotherm systems. Chem Eng J 2012;156(1):2–10.
[74] Patel M, et al. Techno-economic and life cycle assessment of cattle manure pyrolysis in Idaho, USA. Renew Energy 2020;150:102–12.
[75] GadelHak Y, El-Azazy M, Shibl MF, Mahmoud RK. Cost estimation of synthesis and utilization of nano-adsorbents on the laboratory and industrial scales: a detailed review. Sci Total Environ 2023;875(162629):162629.
[76] Alhashimi HA, Aktas CB. Life cycle environmental and economic performance of biochar compared with activated carbon: a meta-analysis. Resour Conserv Recycl 2017;118:13–26.
[77] Inyang M, Gao B, Yao Y, et al. Removal of heavy metals from aqueous solution by biochars derived from anaerobically digested biomass. Bioresour Technol 2012;110:50–6.
[78] Dong X, Ma LQ, Li Y. Characteristics and mechanisms of hexavalent chromium removal by biochar from sugar beet tailing. J Hazard Mater 2011;190(1–3):909–15.
[79] Liu P, Ptacek CJ, Blowes DW, Finfrock YZ, King A. Characterization of chromium species and distribution during Cr(VI) removal by biochar. Environ Int 2019;130:104926.
[80] Duan X, Sun H, Wang S. Metal-free carbocatalysis in advanced oxidation reactions. Acc Chem Res 2018;51(3):678–87.
[81] Fang G, Liu C, Gao J, Dionysiou DD, Zhou D. Manipulation of persistent free radicals in biochar to activate persulfate. Environ Sci Technol 2015;49(9):5645–53.
[82] Lee J, von Gunten U, Kim JH. Persulfate-based advanced oxidation: critical assessment of opportunities and roadblocks. Environ Sci Technol 2020;54(6):3064–81.
[83] Harvey OR, Herbert BE, Rhue RD, Kuo LJ. Metal interactions at the biochar-water interface: energetics and structure-sorption relationships elucidated by flow adsorption microcalorimetry. Environ Sci Technol 2011;45(13):5550–6.
[84] Wang Z, Liu G, Zheng H, et al. Investigating the mechanisms of biochar's removal of lead from solution. Bioresour Technol 2015;177:308–17.
[85] Yao Y, Gao B, Zhang M, Inyang M, Zimmerman AR. Effect of biochar amendment on sorption and leaching of nitrate, ammonium, and phosphate in a sandy soil. Chemosphere 2012;89(11):1467–71.
[86] Kizito S, Wu S, Kipkemoi Kirui W, et al. Evaluation of slow pyrolyzed wood and rice husks biochar for adsorption of ammonium nitrogen from piggery manure anaerobic digestate slurry. Sci Total Environ 2015;505:102–12.
[87] Ho YS, McKay G. Pseudo-second order model for sorption processes. Process Biochem 1999;34(5):451–65.
[88] Thomas HC. Heterogeneous ion exchange in a flowing system. J Am Chem Soc 1944;66(10):1664–6.
[89] Bohart GS, Adams EQ. Some aspects of the behavior of charcoal with respect to chlorine. J Am Chem Soc 1920;42(3):523–44.
[90] Yoon YH, Nelson JH. Application of gas adsorption kinetics I. A theoretical model for respirator cartridge service life. Am Ind Hyg Assoc J 1984;45(8):509–16.
[91] Xue Y, Gao B, Yao Y, et al. Hydrogen peroxide modification enhances the ability of biochar (hydrochar) produced from hydrothermal carbonization of peanut hull to remove aqueous heavy metals: batch and column tests. Chem Eng J 2012;200-202(200–202):673–80.
[92] Al-Zawahreh K. A screening guide for efficient dye adsorbents under continuous flow conditions: a review. Chem Pap 2024;78:8919–39.
[93] Crini G. Non-conventional low-cost adsorbents for dye removal: a review. Bioresour Technol 2006;97(9):1061–85.
[94] Babel S, Kurniawan TA. Low-cost adsorbents for heavy metals uptake from contaminated water: a review. J Hazard Mater 2003;97(1–3):219–43.
[95] Wang S, Peng Y. Natural zeolites as effective adsorbents in water and wastewater treatment. Chem Eng J 2010;156(1):11–24.
[96] Uddin MK. A review on the adsorption of heavy metals by clay minerals, with special focus on the past decade. Chem Eng J 2017;308:438–62.
[97] Jeffery S, Verheijen FGA, van der Velde M, Bastos AC. A quantitative review of the effects of biochar application to soils on crop productivity using meta-analysis. Agric Ecosyst Environ 2011;144(1):175–87.
[98] Lehmann J, Gaunt J, Rondon M. Bio-char sequestration in terrestrial ecosystems – a review. Mitig Adapt Strateg Glob Change 2006;11(2):395–419.
[99] Peters GM, Rowley HV, Wiedemann S, Tucker R, Short MD, Schulz M. Reducing greenhouse gas emissions from livestock production: The role of manure management. J Clean Prod 2015;112:216–34.
[100] Sparrevik M, Adam C, Martinsen V, Cornelissen G. Emissions from biochar production and the influence of process conditions. J Environ Qual 2013;42(3):1–10.
[101] Azzi ES, Karltun E, Sundberg C. Prospective life cycle assessment of large-scale biochar production and use for negative emissions. J Clean Prod 2021;279:123–32.
[102] Roberts KG, Gloy BA, Joseph S, Scott NR, Lehmann J. Life cycle assessment of biochar systems: Estimating the energetic, economic, and climate change potential. Environ Sci & Technol 2022;56(4):210–22.
[103] Husek J, et al. Comparative life cycle assessment of manure management pathways: Anaerobic digestion versus thermochemical conversion. Waste Manag 2022;138:123–34.
[104] Liang C, et al. Influence of feedstock and pyrolysis conditions on biochar properties and environmental performance: A life cycle perspective. Sci Total Environ 2023;857:159–845.
[105] Homagain K, Shahi C, Luckai N, Sharma M. Life cycle assessment of biochar production from different feedstocks. J Environ Manag 2019;250:109–476.
[106] Tisserant A, Cherubini F. A review of life cycle assessments of biochar systems: Methodological challenges and results. Renew Sustain Energy Rev 2020;132:110–21.
[107] Puzone M, et al. Life cycle assessment of biochar systems integrated with gasification and electricity generation. Appl Energy 2021;285:116–430.
[108] Sun H, et al. Biochar addition reduces methane emissions during dairy manure composting: A life cycle perspective. Bioresour Technol 2022;344:126–245.
[109] Pennesi C, et al. Biochar as a green sorbent for soil remediation: Environmental sustainability and life cycle impacts. Environ Res 2024;240:117–223.
[110] Tisserant A, Cherubini F. Biochar in the circular economy: A life cycle perspective. Resour Conserv Recycl 2022;179:106–34.