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Straw-derived biochar for sustainable chromium remediation: Mechanisms, modifications, and reusability

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ABSTRACT

The contamination due to heavy metals poses a significant threat to both the environment and human health. Among the heavy metal pollutants, hexavalent chromium [Cr(VI)] presents a significant risk to human health due to its high toxicity and carcinogenic nature. Due to its high solubility and carcinogenicity, the regulatory limit for chromium in drinking water has been decreased to ≤25 µg/L. This review primarily focuses on recent developments in chromium removal from aquatic systems using various straw biochar materials. It discusses the modulation of the physicochemical properties of the straw biochar materials under different pyrolysis conditions and surface modification techniques. Consequently, it outlines the various surface modification strategies adopted by various researchers to acquire optimum chromium removal from the water systems. The adsorption capacities of the straw biochar systems range from ~20 to 450 mg/g, with the feedstock pyrolysis range from 400°C to 700°C. The modified biochar outperforms the pristine materials under acidic conditions, with the adsorption following monolayer Langmuir adsorption and pseudo-second-order kinetics. These modified biochar materials also promote the conversion of the Cr(VI) to the less toxic and more stable trivalent chromium at the redox-active sites on the biochar surface. In addition, the straw biochar materials can retain ~70–90% of adsorption capacity even after 3–5 regeneration cycles. Overall, this review discusses straw-derived biochar as an effective and economically viable material for chromium detoxification. Despite this, there is a need for further research on realistic wastewater matrices to translate the laboratory results into real field applications.

1. INTRODUCTION

With the growth of population, urbanization, and global industrialization, heavy metal usage has increased, which poses a serious threat to the environment and human health. Among the heavy metals, chromium or chromates are seen to be present both in water and wastewater and are often cumulative, non-degradable, and latent [1-4]. The various sources of heavy metal production units that lead to the contamination of the aquatic environments are metal plating, painting, printing, petroleum refining, mining, batteries, smelting, pesticides, fertilizers, etc. There is a class of carcinogenic metals, such as arsenic, cadmium, chromium (especially hexavalent), nickel, cobalt, lead, etc., which are non-biodegradable and persist and accumulate in living organisms, thus contributing to the process of bio-magnification [5,6]. Among the carcinogenic metals, chromium has a high teratogenicity because it exists in an array of oxidation states. Within these oxidation states, hexavalent chromium [Cr(VI)] and trivalent chromium [Cr(III)] have been identified as a top-priority contaminant by many countries, including U.S. Environmental Protection Agency and the safety of drinking water standard for chromium is 100 µg/L, while the World Health Organization (WHO) has fixed the permissible Chromium in drinking water to 100 µg/L. In addition, the International Agency for Research on Cancer considers chromium to be one of the carcinogenic elements. It has been reported that laborers with dermal exposure to chromium develop lung cancer [7]. The new limit set by WHO, which has to be complied with by 2036, is 25µg/L [2].

Chromium exists in nature mainly with the valence states of Cr(VI) and Cr(III). Normally, Cr(III) is considered to be less toxic than Cr(VI). The Cr(III) exists in the form of chromium(III) hydroxide, while Cr(VI) mainly exists in the forms of compounds, such as $Cr_2O_7^{2-}$, $HCrO_4^-$, and CrO_4^{2-}. These compounds often exhibit higher solubility and mobility in water and often represent a serious risk of carcinogenicity, persistence, and bioaccumulation [8]. The toxicity of chromium depends majorly on the exposure duration. Due to major

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toxicity, it has been seen that longer exposure can lead to eye, skin, and respiratory diseases [9]. Therefore, removal of Cr(VI) or conversion of hexavalent chromium [Cr(VI)] to trivalent chromium [Cr(III)] in aquatic environments is required to decrease the risk posed by this heavy metal and its compounds. There are several techniques that are adopted for the removal of heavy metals from the environment. Among them, chemical precipitation, membrane separation, adsorption, and like methods are used to reduce chromium contamination from the aquatic environments [4]. Adsorption is often considered an efficient method for the remediation of Cr(VI) from water. This method mainly relies on transferring the contaminant from the liquid phase to the surface of the solid either by physio or chemisorption.

The removal of contaminants from the water systems has been achieved with the aid of many low-cost materials, such as industrial by-products, agricultural waste, inactive or dead microorganisms, biomass, etc. [1,10,11]. These materials are environmentally friendly and renewable because of its high abundance, economical nature, ease of availability, recyclability, high loading capacity and less sensitivity to toxic materials, and effective performance [1,12,13]. Several researchers have found eco-friendly, cost-effective, and highly efficient adsorbents, such as nanoparticles, activated carbon, macromolecules, biochar, etc. Among them, waste-based biochar is a preferred adsorbent because of its environmental benefits and cost-effective nature. Biochar can be generated from several sources, such as industrial and agricultural residues, through the pyrolysis process without or in limited supply of oxygen [3,14]. Biochar-based technologies are efficient and promising in the removal and remediation of heavy metals from the aquatic environment. The physicochemical properties of biochar, such as surface area, porosity, and oxygen-rich functional organic groups, contribute toward the efficiency in the removal of contaminants from the aquatic environments [2,15,16]. There are several types of agricultural straw biomass, which are often regarded as an affordable and sustainable feedstock. These straw biomasses are rich in cellulose, hemicellulose, lignin, and silicon dioxide, which helps in developing highly efficient biochar materials for environmental remediation.

Many reviews have addressed the issue of chromium remediation by biochar from a variety of sources of biomass, and many primarily focused on the adsorption capacities or modified biochar. In contrast, this review concentrates on biochar materials from straw, a unique class of lignocellulosics that are high in cellulose and silica but low in lignin, and that can be effective, sustainable, and inexpensive adsorbents for the detoxification of chromium. This review presents a unique synthesis of the two mechanisms underpinning the efficiency of straw biochar: Adsorption and reduction, demonstrating the unique role that straw biochar can play in converting the toxic Cr(VI) species into a less toxic, Cr(III) form. This review synthesizes the feedstock composition, pyrolysis conditions, surface-modification method, and mechanistic pathways into a broader framework for adsorption-reduction synergies that moves beyond simply presenting a summary of facts from other papers. Moreover, it discusses perspectives surrounding sustainability, the scalability of deployment in wastewater systems, and the zero-waste circular bioeconomy potential of straw-derived biochar by discussing its regeneration potential.

2. METHODS

The method adopted for generating the literature for this review is outlined in the following subsections.

2.1. Search Strategy

A web-based study detailing the remediation of chromium with the aid of straw biochar from aquatic environments was employed, and the time span was from 2010 to 2025 [Figure 1]. Google Scholar, PubMed, Scopus, and ScienceDirect databases were included for carrying out the literature search.

The following key terms with the Boolean operators were used to perform the literature search: "Straw AND Biochar AND Chromium adsorption," "Rice AND Straw AND Biochar AND Chromium adsorption," "Wheat AND Straw AND Biochar AND Chromium adsorption" and "Corn AND Straw AND Biochar AND Chromium adsorption."

2.2. Screening Criteria

There were two phases in screening the literature. Initially, the title and the abstract of the publications were screened and, thereafter, suitable articles were downloaded for critical study. The following inclusion and exclusion criteria were taken.

2.2.1. Inclusion criteria

The articles and/or book chapters containing adsorption of chromium and/or reduction of Cr(VI) using straw-derived biochar were included in the study.

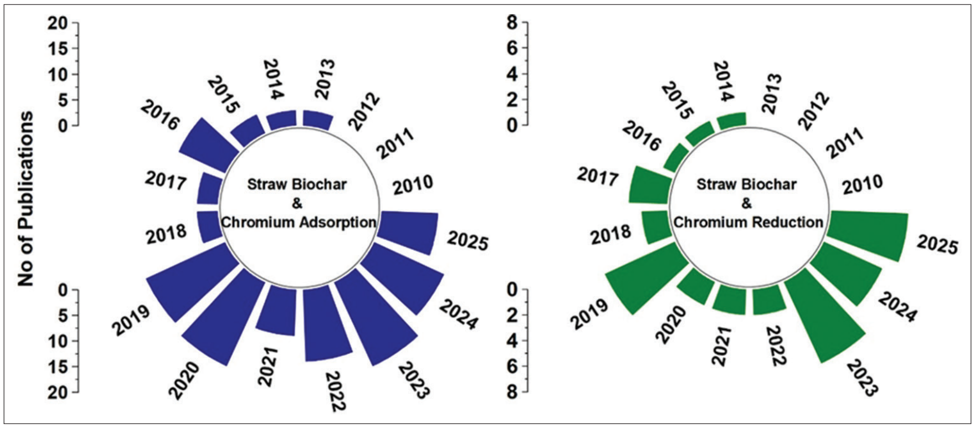


Figure 1: Trends in publications related to straw-derived biochar for chromium remediation and reduction. The left panel (blue) represents the number of publications per year (2010–2025) focusing on straw biochar and chromium adsorption, while the right panel (green) shows publications addressing straw biochar and chromium reduction.

2.2.2. Exclusion criteria

Studies in which biochar materials are used for chromium remediation from soil.

2.3. Data Retrieval

The information pertaining to the inclusion criteria was retrieved from the databases and categorized in a structural manner. This was done by directly citing the reported values across each study.

3. CHROMIUM SOURCES AND CHEMISTRY

In the environment, chromium contamination may occur due to man-made activities as well as natural causes. The distribution of chromium in land and aquatic environments is heavily dependent on phenomena, such as adsorption-desorption, redox reactions, and dissolution-precipitation [17]. The main source of chromium is chromite ores, which are formed by the segregation of magma. Chromium is also found in the atmosphere due to volcanic eruptions, fly ash distribution due to combustion of coal, etc. The Sukinda region in Odisha, India, has one of the highest chromite ore deposits, and the water released from these mining activities heavily pollutes the nearby water sources [Figure 2].

Cr(III) and Cr(VI) primarily exist in aquatic systems. Cr(VI) is known to be dangerous because of its carcinogenic, mutagenic, and oxidizing properties relative to the Cr(III) [19-21]. The existence of trivalent and Cr(VI) ions in water is majorly dependent on its redox potential and the pH of the solution. Under acidic conditions, the Cr(III) ion is more stable due to its high redox potential, while in basic conditions the decrease in the redox potential makes the Cr(VI) ion more stable. Hence, from a thermodynamic perspective, chromium is highly stable in alkaline conditions. At various pH values, Cr(VI) exists in different forms, such as H_2CrO_4, $HCrO_4^-$, CrO_4^{2-}, HCr_2O^{7-} and $Cr_2O_7^{2-}$ [22]. When the pH is a range between 0 and 2, it produces $Cr(III)^+$ species (Cr(III) ion), which is very stable. At pH higher than 6, the Cr(III) precipitates as chromium hydroxides [Figure 3] [23,24] and as the pH rises to 6–8, it gives chromium dihydroxide, while above pH 9, the transition starts and the precipitate solubilizes to form chromium tetrahydroxide complex [25]. The conversion of trivalent to Cr(VI)is not usual by natural oxidants and requires strong oxidants, ultraviolet (UV) light and high temperature at specific pH conditions [26]. Although, Cr(III) ion is required in trace amounts for human health, continuous exposure and inhalation of Cr(VI) ion can lead to several diseases. Hence, as discussed previously, biochar is an environmentally friendly and economically viable adsorbent that can be used for the removal of chromium from contaminated water environments.

4. BIOCHAR PRODUCTION AND PHYSICOCHEMICAL TUNING

4.1. Biochar as a Green, Tunable Adsorbent

Biochar is a porous, carbon-rich solid material with a high degree of aromatization that is produced by the thermal decomposition of biomass from plant or animal waste under oxygen-free or limited oxygen conditions. These biochar materials have varied physical and chemical properties, which depend on the feedstock biomass, pre-activation conditions, pyrolysis parameters, etc. [Figure 4] [27-29]. Among all the physicochemical parameters, the surface area and porosity of biochar are directly proportional to the available active sites on the surface of the biochar [29,30]. This enhances the biochar properties, such as cation exchange capacity (CEC), water holding capacity, and adsorption capacity [29-32]. Tomczyk *et al.* have outlined that the physicochemical properties of biochar depend on the pyrolysis temperature and the feedstock [33].

4.2. Pyrolysis Temperature/Time

Pyrolysis temperature is considered the predominant parameter influencing biochar physicochemical characteristics [29]. Slow pyrolysis is a reliable, energy-efficient process that heats biomass slowly (5–20°C/min) between 300°C and 700°C for several hours, producing the highest biochar yield (35–50%) along with bio-oil and syngas. Its advantages include low equipment needs and simple handling, but it has uneven heat distribution and longer processing times [34]. Fast pyrolysis rapidly heats the biomass at higher heating rates and temperatures, producing mostly bio-oil (50–60%) with lower biochar yield (15–25%) and short vapor residence time [35]. Flash pyrolysis uses ultrafast heating (>1000°C/s) and high temperatures (900–1300°C) to maximize bio-oil yield (up to 75%), with very short residence times (<10 s) [36]. Microwave-assisted pyrolysis efficiently converts biomass using electromagnetic heating, which results in a rapid reaction and improved product quality, such as cleaner bio-oil and greater energy efficiency. Hydrothermal carbonization treats wet biomass under pressure at 170–260°C, producing hydrochar

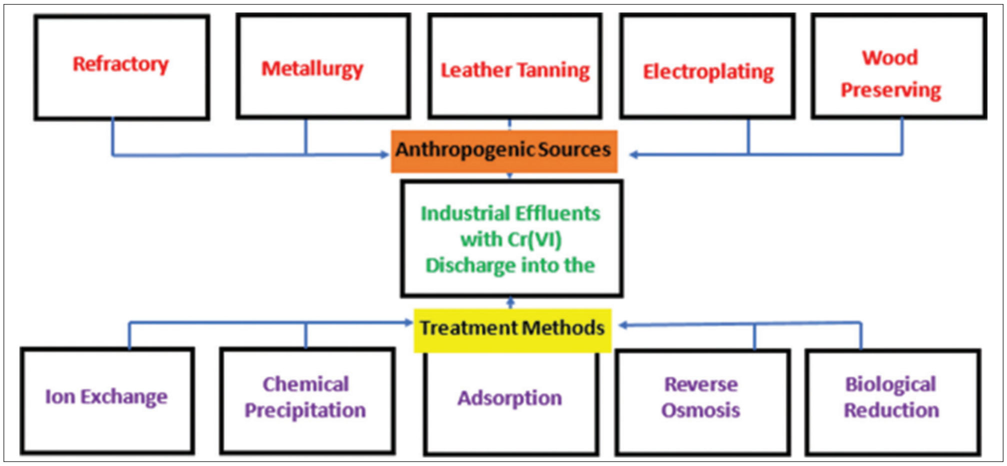


Figure 2: Hexavalent chromium effluence and current management. The diagram illustrates major sources of chromium pollution in the environment and also highlights common treatment methods [18].

with improved fuel quality without drying feedstock. Torrefaction is mild pyrolysis at 200–300°C for 15–120 min, yielding carbon-rich solids with improved bio-oil quality using less energy [37-39]. Each method's product yield and quality depend on feedstock, temperature, heating rate, and residence time, allowing flexibility for targeted biochar or bio-oil production [Figure 5]. In summary, the pyrolysis parameters suitable for producing high porosity biochar are moderate temperature (400–700°C), low heating rate (5–30°C/min), long residence time (30–120 min), atmospheric pressure, moderate gas flow rate (50–150 mL/min) [29]. Hence, from the above discussion, it is clear that the biochar yield and its physicochemical properties are highly dependent on the pyrolysis parameters.

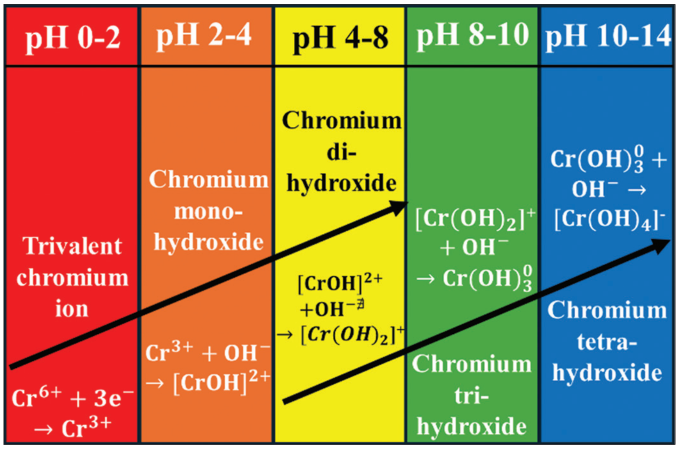


Figure 3: Speciation reactions of chromium at different pH levels. The diagram illustrates the transformation of chromium species from trivalent chromium ions (Cr^{3+}) at acidic pH (0–2) to various hydroxide forms as the pH increases from 2 to 14 [24].

4.3. Biochar Physicochemical Characteristics

The physicochemical properties of biochar significantly influence its adsorption behavior and the removal of pollutants from aquatic environments [33,40]. The most critical parameters that influence the adsorption behavior of the biochar materials are surface area, pore size distribution and its volume, functional groups and CEC, volatile matter, carbon content, and ash content [Table 1] [29,40,41].

The surface area and porosity of biochar materials vary with the pyrolysis temperature as well as the type of feedstock. Biochar develops a higher specific surface area and well-defined pores at higher pyrolysis temperatures. However, excessively high temperatures may lead to the collapse of pores and reduce the surface area of the biochar due to the fusion of carbon structures and the breakdown of organic matter. This breakdown and fusion may lead to the creation of micropores [42-45]. The increase in the surface area of biochar materials at higher pyrolysis conditions is due to the disintegration of aliphatic alkyls, ester groups, and the vulnerability of the aromatic lignin core [46,47]. Sun *et al.* reported from BET analysis that the surface area of straw biochar varies between 50 and 300 m^2/g, depending on pyrolysis temperature. At 500–600°C, well-developed micropores and mesopores form due to volatile release, enhancing adsorption potential [48]. Similarly, Li *et al.* with the aid of BET analysis of corn straw-derived biochar produced at 550°C, found a surface

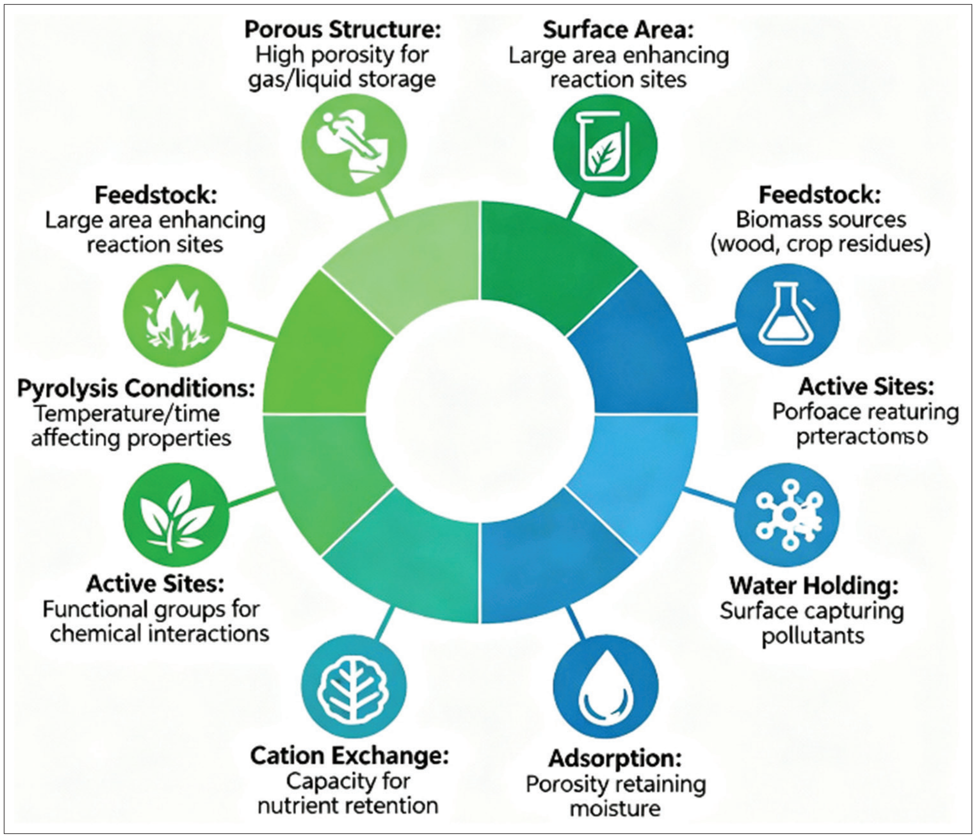
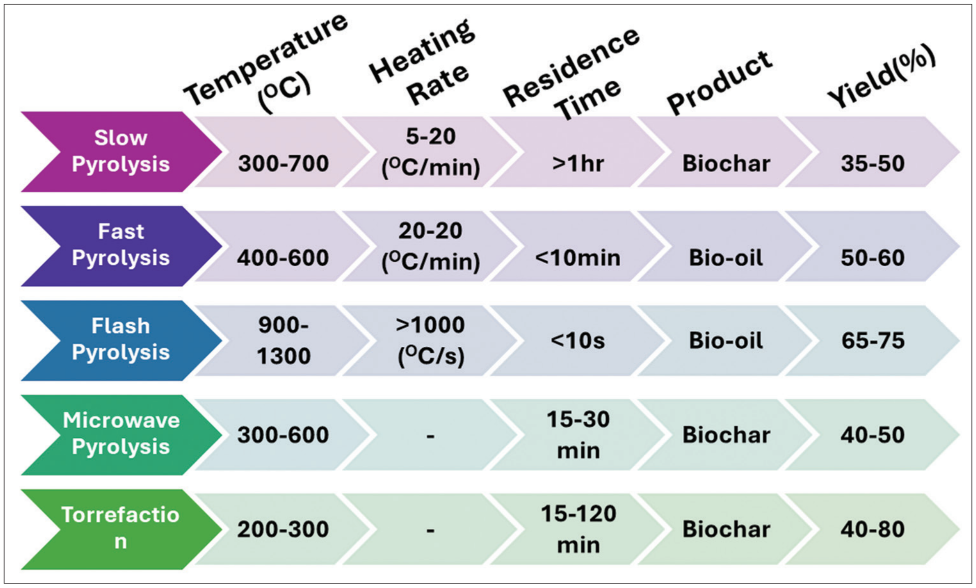


Figure 4: Physicochemical properties of biochar and the factors affecting these properties. The diagram illustrates how feedstock type, pyrolysis conditions, and physicochemical features determine biochar's reactivity and efficiency.



	Temperature (°C)	Heating Rate	Residence Time	Product	Yield(%)
Slow Pyrolysis	300-700	5-20 (°C/min)	>1hr	Biochar	35-50
Fast Pyrolysis	400-600	20-20 (°C/min)	<10min	Bio-oil	50-60
Flash Pyrolysis	900-1300	>1000 (°C/s)	<10s	Bio-oil	65-75
Microwave Pyrolysis	300-600	-	15-30 min	Biochar	40-50
Torrefaction	200-300	-	15-120 min	Biochar	40-80

Figure 5: Comparison of different pyrolysis processes used for biochar and bio-oil production. This figure summarizes the operational parameters and product yields of major pyrolysis techniques.

Table 1: The variation of physicochemical properties of biochar under different pyrolytic conditions.

Physio-chemical properties	Low pyrolysis temperature (<400°C)	Moderate pyrolysis temperature (400–700°C)	High pyrolysis temperature (>700°C)
Surface area and porosity	Low, poorly developed	High, well-developed micropores	Reduced due to pore collapse
Functional groups	High acidic groups	Balanced acidic/basic groups	Dominant basic groups
Elemental composition	High H/C and O/C ratios	Moderate H/C and O/C ratios	Low H/C and O/C ratios
pH and ash	Neutral-to-slightly acidic, low ash	Alkaline, moderate ash	Strongly alkaline, high ash

H/C: Hydrogen/Carbon, O/C: Oxygen/Carbon

area of ~110 m^2/g and micropore volume of 0.006 cm^3/g [49]. However, during pyrolysis, the degradation of cellulose causes the formation of amorphous carbon structures. It has been reported that an amorphous carbon structure leads to the formation of micropores [33,50,51]. At higher temperatures, the volatile matter within the biochar materials is released, resulting in an increase in pore volume. Somboon *et al*. found that the total pore volume of rice straw biochar can be significantly higher than that of pristine biochar, which was mainly due to the development of mesopores and ultra-micropores [52].

Heating the biomass between 350°C and 650°C leads to the disintegration and rearrangement of the chemical bonds, which may lead to the introduction of new functional groups like carboxyl, lactone, lactol, quinine, anhydride, phenol, ether, pyrone, pyridine, and pyrrole [53] as shown in Figure 6.

The analysis of the biochar surface with the aid of Fourier Transform Infrared (FTIR) spectroscopy divulges the different organic functional groups present on the biochar. Most of these functional groups are dominated by different oxygen-containing hydrocarbons, which reflect the carbohydrate structure of cellulose and hemicelluloses [55-57]. The disintegration of cellulose, hemi-cellulose, and lignin present in the biomass leads to the formation of new organic bonds [33]. The biochar surface exhibits high hydrophobicity with structured carbon layers at temperatures between 600°C and 700°C. However, biomass having lower hydrogen and oxygen contents may lead to dehydration and deoxygenation [58,59].

The type of feedstock determines the CEC of biochar. At lower pyrolysis temperatures, CEC is higher, while at higher temperatures, CEC is lower. The oxygen-containing functional groups and mineral components on the biochar surface influences biochar surface charge [60,61]. Wang *et al*. found that biochar derived from rice straw produced at moderate pyrolysis temperatures (400–500°C) exhibits CEC values of 18–45 $cmol^{(+)}$/ kg, which is mainly attributed to the presence of oxygen rich functional groups [62]. In another work they also found that biochar produced from multi-feedstock, such as rice, wheat, and maize possess 14–35 $cmol^{(+)}$/ kg CEC, which exceeds wood-based biochar due to more mineral-rich and functionalized structure [62].

The pH of biochar is a key physicochemical property that affects its role in the adsorption of pollutants. Biochar pH depends primarily on feedstock composition, pyrolysis temperature, and post-treatment conditions [33,44,63]. Most biochars are alkaline, with pH values typically ranging between 7 and 10, acting as natural liming agents that can raise the pH of acidic soils. However, depending on the feedstock and production conditions, biochar pH can vary widely from 3 to 12.0, as found across different biomass origins and pyrolysis regimes [33,64]. Sakhiya *et al*. reported that the pH of the rice straw increases with the increase in the pyrolysis temperature from 8.2 to 11.4, which may be due to decomposition of organic acids and the accumulation of basic metal oxides [65].

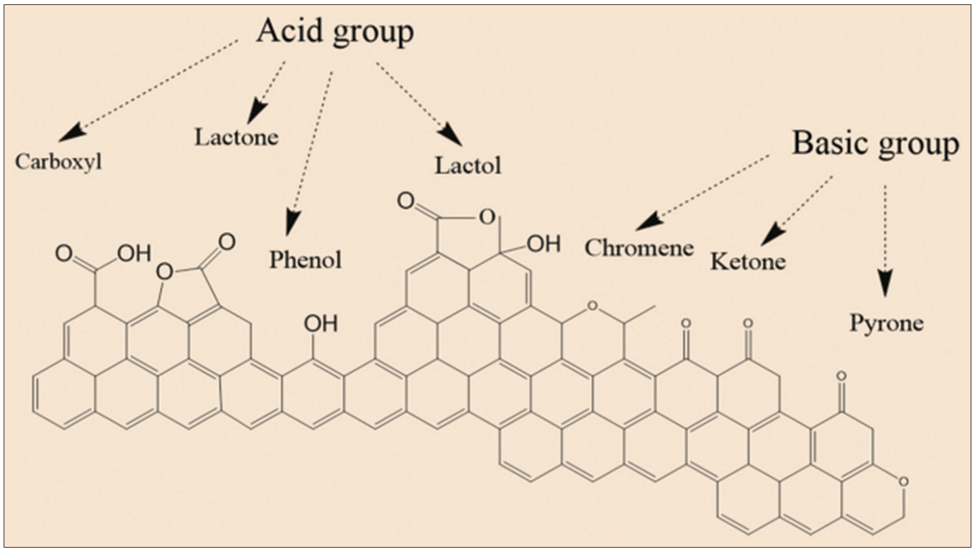


Figure 6: Functional groups present on the surface of the biochar. The figure depicts the major acidic and basic surface functional groups responsible for biochar's chemical reactivity and adsorption capacity [54].

5. MECHANISTIC PATHWAYS OF CHROMIUM ADSORPTION BY STRAW BIOCHAR

In recent years, adsorption has become the main method for the removal of Cr(VI) from the aquatic environments [4,66,67]. Chromium is absorbed onto the surface of the biochar via physisorption and chemisorption. The major mechanism operating for the adsorption of the chromium ion by both physiosorption and chemisorption depends on the surface area, porous structure, presence of functional groups, pore volume of biochar, and the chromium interaction with the biochar surface through H-bonding, surface complexation, cation exchange, ion exchange, and electrostatic interaction [68-70]. The adsorption of chromium is significantly influenced by adsorption parameters, biochar preparation temperature, feedstock, and solution pH [68,71,72]. When the biochar's surface is positively charged, it attracts anionic species, such as Cr(VI) through electrostatic attraction [3]. On the biochar surface, alkali or alkaline-earth cations (e.g., calcium ion, potassium ion [K^+], magnesium ion) can be replaced by chromium cations, endowing the biochar with higher CECs. The CECs of the biochar materials decreases as pyrolysis temperature rises above 350°C [73,74]. Cation exchange is limited in the acidic conditions, although the exchange with cations, such as calcium or potassium is a major uptake mechanism, even in acidic conditions [3]. The chromium species present above pH 8 is mainly CrO_4^{2-}.The dominant chemical reactions in the solution are as follows [75]

$$Cr_2O_7^{2-} + H_2O \rightarrow 2HCrO_4^-$$

$$HCrO_4^- \rightarrow CrO_4^{2-} + H^+$$

$$2CrO_4^{2-} + 2H^+ \rightarrow Cr_2O_7^{2-} + H_2O$$

$$Cr_2O_7^{2-} + 14H^+ + 6e^- \rightarrow 2Cr^{3+} + 2H_2O$$

$$CrO_4^{2-} + 4H_2O + 3e^- \rightarrow Cr(OH)_3 + 5OH^-$$

A key pathway is based on oxygen-containing functional groups, such as –OH, –COOH, and phenolic moieties, which act as electron donors and play a crucial role in contributing to Cr(VI) removal by adsorption/reduction. Besides the mechanisms of electrostatic attraction, precipitation, complexation and ion exchange, optimization of initial adsorption parameters also plays a vital role in the adsorption of chromium ions [4,76]. FTIR and X-ray Photoelectron Spectroscopy analysis show the existence of functional groups, such as C-H and –OH groups, and most of the Cr(VI) is reduced to Cr(III) ions, which remain static on the surface of the biochar [3,77,78]. The electrostatic force, ion exchange, and surface complexation processes may contribute toward the reduction of hexavalent to Cr(III) [3,79]. A possible mechanism of chromium adsorption is outlined in Figure 7.

6. MODIFICATION STRATEGIES AND PERFORMANCE GAINS

The adsorption capacity of biochar materials can be enhanced by modifying them through various physical and chemical processes. A list of methods used to modify the straw biochar material to improve its chromium adsorption capacity has been outlined in Table 2.

The physical modification of biochar can be achieved through ball milling, sonication, steam activation, and UV radiation, among other methods. Among the physical modification methods, ball milling is an eco-friendly and low-cost technology that can induce both structural and chemical modifications in biochar. This process modifies the particle size and enhances the surface area of the biochar. The reduction in particle size and increase in surface area lead to an increase in chromium adsorption [83]. The UV irradiation also enhances the pore size distribution of the biochar materials by inducing the formation of micropores, which has also been shown to enhance the chromium removal ability of biochar materials [84]. Steam activation is another way to modify the biochar. Super-heated steam leads to the development of crystalline structures and surface oxides. In addition, the steam also facilitates the formation of meso- and micropores, which may aid in the adsorption process [85]. Although no specific studies have been carried out for the removal of chromium with the aid of the steam-activated biochar materials.

In addition to physical activation, chemical activation is also widely used to modify the biochar materials to enhance their chromium adsorption behavior. Chen *et al*. synthesized two types of biochar

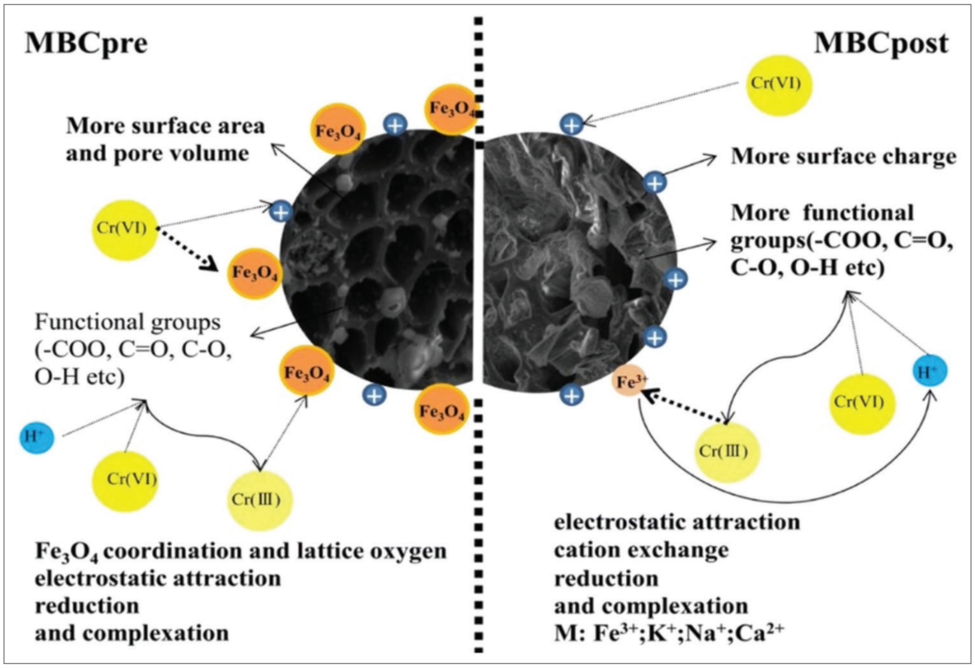


Figure 7: Possible mechanism of chromium adsorption on biochar. The illustrations depict multiple interaction mechanisms, including electrostatic attraction, ion exchange, surface complexation, pore filling, and redox reactions between hexavalent chromium and trivalent chromium [80-82].

Table 2: Feedstock type, modification methods, and adsorption performance for chromium removal.

Feed-stock	Modifications	Pyrolysis temperature	Adsorption capacity	pH	Dominant removal mechanism	Isotherm models	Adsorption kinetics	References
Peanut straw	Na_2SO_3/$FeSO_4$	500°C	99.7%	2.0–2.4	Precipitation, surface complexes			[92]
Wheat straw	H_3PO_4	250°C	188.1 mg/g	1.8~3.1	Complexation and electrostatic interaction	Langmuir model	Pseudo-second order	[93]
Wheat straw	-	650°C	24.6 mg/g	2	Anionic and cationic adsorption combined with Cr(VI) species reduction	Langmuir isotherm model	Pseudo-second order	[94]
Rice straw	H_3PO_4	700°C	96.91 mg/g	2	Electrostatic attraction	Langmuir isotherm	Pseudo-second order	[95]
Rice straw	KOH	105±5°C	40.32 mg/g	2	Diffusion	Langmuir and the Freundlich isotherms	-	[96]
Wheat straw	-	400°, 450°, 500° and 550°C	89.22 mg/g	1	Binding, reduction, electrostatic attraction	Langmuir isotherm	Pseudo-second order	[97]
Rice straw		400°C	26.31 mg/g	5–7	Electrostatic attraction or complexation	Langmuir and Freundlich	Pseudo-second order	[98]
Brown teff straw	NaOH, H_3PO_4 and $ZnCl_2$	-	73.8, 89.3, 88.9 mg/g	2.0	Reduction, complexation	Langmuir isotherm model	Pseudo-second order	[99]
Rice straw	$FeSO_4$.$7H_2O$	100°, 300°, 400°, 500° and 700°C	2.2, 3.8, 6.4, 6.2, and 5.8 mg/g	4	Monodentate interaction			[100]
Rice straw	Epichlorohydrin and diethylenetriamine	-	140.39 mg/g	2	Adsorption-reduction and electrostatic attraction	Freundlich isotherm	Elovich model	[101]
Corn straw	HCl and HF	300° , 500° and 700°C	33.3,22.03, 26.18 mg/g	7	Electrostatic attraction	Langmuir isotherm model	-	[102]

(*Contd*...)

Table 2: (*Continued*).

Feed-stock	Modifications	Pyrolysis temperature	Adsorption capacity	pH	Dominant removal mechanism	Isotherm models	Adsorption kinetics	References
Rice straw	HCl, NaOH	300°, 500° and 700°C	89.90 mg/g	2	Electrostatic attraction	Freundlich isotherm		[103]
Corn straw	KOH	450°, 650° and 800°C	175.44 mg/g	2	On exchange, the reduction of Cr(VI) and the complexation reaction	Langmuir adsorption isotherm model	Pseudo-second order model	[104]
Maize straw	Washed with deionized water	300°C	>99.9%	2	Electrostatic attraction of Cr(VI), Cr(VI) reduction, and subsequent Cr(III) complexation	Langmuir adsorption isotherm model	Pseudo-second order model	[105]
Rice straw	$FeCl_3$	400°C	94.95 mg/g	2	Reduction	Langmuir-Freundlich isotherm model	Pseudo-second order model	[106]
Wheat straw and corn straws	washed by distilled water	Ball milling and sieved size 100 mesh	WS_ Cr(VI)=125.6 Cr(III)=68.9, CS_ Cr(VI)=87.4 Cr(III)=62.3 mg/g	3	Electrostatic attraction, chemical adsorption	Langmuir isotherm	Pseudo-second order	[86]
Corn straw	HNO_3, H_2SO_4 and H_3PO_4	500°C	BCN, BCS, and BCP were 11.24, 8.05, and 9.58 mg/g	2	Electrostatic attraction, electrons (e^-) and hydrogen ions (H^+)	Freundlich isotherm	Pseudo-second order	[88]
Wheat straw	Washed by distilled water	550°C	194, 183 mg/g	5.1	Electrostatic attraction, Chemisorption, Surface complexion,	Langmuir and Freundlich isotherm	Pseudo-second order	[87]
Flax straw	$FeSO_4{\cdot}7H_2O$	300°C	186.99 mg/g	3	Precipitation, reduction and complexation	Langmuir, Freundlich, and Sips models	Pseudo-second order	[89]
Banana leaf, cassava straw, Chinese fir straw, corn straw, taro straw	$MgCl_2$	430°C	BSB, CSB, TSB, FSB, CFSB 114.64, 75.56, 50.21, 17.21, 20.21, and 40.12 mg/g, respectively	2	Electrostatic attraction, complexation, and functional group bonding	Langmuir model	Pseudo-second order	[90]
Corn straw	$FeCl_3$ and $KMnO_4$	400°C	118.03 mg/g	2	Electrostatic attraction, reduction, complexation	Langmuir model	Pseudo-second order	[91]
Corn straw	$FeCl_3{\cdot}6H_2O$	350°, 500°, and 650°C	361.011 mg/g	4.5	Electrostatic attraction, physical adsorption, ion exchange, or co-complexation	Langmuir model	Pseudo-second order	[82]
Corn straw	KOH	800°C	450.43 mg/g	2	Electrostatic attraction, complexation, ion exchange, and reduction action	Langmuir, Freundlich, and Sips models	Avrami fractional-order model	[107]
Brassica straw	$FeCl_3{\cdot}6H_2O$ and $FeCl_2{\cdot}4H_2O$	600°C	434.85 mg/g	1	Physisorption, electrostatic interaction, chelation of functional groups and reduction-coupled surface precipitation	Freundlich model	Pseudo-second order	[108]
Water hyacinth waste straw	$FeCl_3$	60°C	8.78 mg/g	5.52	Surface precipitation, complexation, and reduction processes	Langmuir model	Pseudo-second order	[109]

(*Contd...*)

Table 2: (*Continued*).

Feed-stock	Modifications	Pyrolysis temperature	Adsorption capacity	pH	Dominant removal mechanism	Isotherm models	Adsorption kinetics	References
Flax straw and corn waste	Washed by distilled water	500°C	97% and 32%	4.0.	Chelate (complexes)	-	-	[110]
Rice straw	HCl and NaOH	550°C	59 mg/g	3	Electrostatic attraction	Langmuir, Freundlich, and Sips models	Pseudo-second order	[111]
Corn straw	$FeSO_4 \cdot 7H_2O$		33.09 mg/g	2	Adsorption, chemical reduction–oxidation reaction, and co-precipitation	Freundlich model	Pseudo-second order	[112]
Rice straw	HNO_3		60.204 mg/g	3	Electrostatic interaction, reduction, complexion exchange, and pore adsorption	Freundlich model	Pseudo-second order	[113]
Rice straw	$Fe(NO_3)_3$	450°C	99.5%	2	Electrostatic adsorption and the redox reaction, reduction process	Langmuir model	pseudo-second order	[114]
Corn straw	Fe_3O_4	500°C	138.8 mg/g	6	Ion exchange, complexation, and reduction reaction	Langmuir model	Second order	[115]
Rice straw	H_2SO_4		95.57%	2	Surface complexation, Hydrogen bonding, electrostatic attraction, and intraparticle diffusion	Freundlich, Sips, and Toth model	Pseudo-second order and Elovich Kinetic study	[116]
Wheat straw	KOH		96.08 mg/g	2	Electrostatic attraction and complexation	Langmuir model	Pseudo-second order	[117]
Corn straw	HCl, NaOH, and $FeCl_3$		45%	3	Adsorption, reduction, and co-precipitation	Langmuir model	Pseudo-second order	[118]
Rice straw	$FeCl_3 \cdot 6H_2O$ and $FeSO_4 \cdot 7H_2O$	600°C	66.10 mg/g	2	Electrostatic attraction, reduction, and complexation	Langmuir model	Pseudo-second order	[119]
Wheat straw	Agate balls	300°C	52.21 mg/g	2	Electrostatic attraction, redox, and complexation	Langmuir model	Pseudo-second order	[120]
Wheat straw	$FeSO_4 \cdot 7H_2O$	500°C	147.93 mg/g	7	Pore adsorption, chemical reduction, chelation reactions, and electrostatic interactions	Langmuir model	pseudo-second order	[121]
Rice straw	$FeCl_3$, $FeSO_4$ and $(NH_4)_2Fe(SO_4)_2$	500°C	21.98 mg/g	4	Electrostatic attraction, reduction, and Complexation	Freundlich model	pseudo-second order	[122]

Cr(VI): Hexavalent chromium, Cr(III): Trivalent chromium, BCN: Biochar treated with HNO3, BCS: Biochar treated with H2SO4, BCP: Biochar treated with H3PO4, BSB: Banana straw, CSB: Cassava straw, MSB: Corn straw, TSB: Taro straw, CFSB: Camellia oleifera shell biochar

materials from wheat and corn straw and washed them with distilled water to remove the ash debris from the biochar surface [86]. These biochar materials exhibited significant removal capacity of chromium from water and followed the Langmuir monolayer adsorption isotherm. The adsorption capacity for wheat and corn straw biochar was found to be 125.6 mg/g [Cr(VI)], 68.9 mg/g [Cr(III)], 87.4 mg/g [Cr(VI)], and 62.3 mg/g [Cr(III)], respectively. The reaction kinetics of chromium removal followed pseudo-second order, and the maximum removal was obtained at an acidic pH 3 [86]. Similarly, another study carried out with wheat straw biochar and washed with distilled water exhibited adsorption capacities of 194 and 183 mg/g at ~pH 5 but followed both Langmuir and Freundlich isotherms [87]. Xu *et al*. modified corn straw biochar with nitric acid, sulphuric acid, and phosphoric acid and found that the adsorption capacities enhanced upon modification by 11.24, 8.05, and 9.58 mg/g, respectively [88]. The adsorption process followed the multilayer Freundlich adsorption isotherm at pH 2. In another study carried out by Fengfeng *et al*., iron treated flax straw biochar showed a loading capacity of 186.99 mg/g at pH 3 and followed pseudo-second-order kinetics [89]. The adsorption process follows the Langmuir, the Freundlich, and the Sips isotherm models. Banana, cassava, Chinese fir, corn, and taro straw derived biochar materials modified with magnesium chloride exhibited adsorption capacities between 17.21 and 114.64 mg/g and followed a monolayer adsorption process under acidic conditions [90]. Zhu *et al*. activated corn straw biochar with iron and oxidising agent potassium permanganate, which enhanced the adsorption capacity to 118.03 mg/g and followed monolayer adsorption with pseudo-second-order kinetics [91].

Wheat straw biochar modified with lithium bromide showed a loading capacity of 56.34 mg/g, while that treated with iron(III) chloride and hydrochloric acid exhibited a removal efficiency of 91.2% under acidic conditions. Qu *et al*. prepared corn straw biochar treated with potassium hydroxide, which showed a notably high adsorption capacity of 450.43 mg/g modelled by the Avrami fractional-order kinetic model [107], while Jian *et al*. modified with the same with ferric chloride.$\cdot 6H_2O$ and achieved 361.01 mg/g at pH 4.5. Both of the adsorption processes were described by Langmuir and pseudo-second-order kinetics [82]. Similarly, Chu *et al*. Fe_3O_4-modified corn straw showed 138.8 mg/g adsorption capacity and followed the Langmuir adsorption isotherm model and pseudo-second order kinetics [115]. In another group of studies, researchers modified rice straw with various iron salts and found that the adsorption capacity of the iron-modified biochar materials ranged from 66.10 to 104.53 mg/g under acidic conditions. The adsorption process for the iron-modified biochar materials followed the monolayer Langmuir adsorption isotherm and pseudo-second order kinetics [114,119,123]. Similarly, Luo *et al*. modified Brassica straw with ferrous and ferric salts and found that the adsorption capacity under pH conditions enhanced drastically to 434.85 mg/g under acidic pH [108]. Although the adsorption kinetics follow pseudo-second order, the adsorption process follows a multilayer Freundlich adsorption isotherm. In another study, Ayele *et al*. prepared acid and base-modified biochar with Teff straw and found that the adsorption efficiency reached ~95% at pH 2 and follows a monolayer adsorption process [124]. Similarly, Venkatraman *et al*. modified wheat straw biochar with iron and found the maximum adsorption capacity to be 147.93 mg/g at pH 7. The adsorption isotherm process followed Freundlich, Sips, and Toth isotherm models and pseudo-second order-Elovich kinetics [116]. Similarly, in another study, wheat straw treated with base showed a loading capacity of 96.08 mg/g at pH 2 [117].

All the studies discussed above are enlisted in Table 2. All the above studies clearly revealed that the adsorption capacity of the biochar materials depends on the type of straw biomass, the modifications executed on this biomass, and the conditions of the adsorption process. In addition, these studies collectively highlight that straw-derived biochar, especially that modified with Fe-based and alkali-acidic compounds, exhibits high Cr(VI) adsorption efficiencies, predominantly governed by the Langmuir isotherm and pseudo-second-order kinetics. The table shows straw biochars contain more silica and oxygenated family groups, primarily facilitating Cr(VI) removal through surface complexation and electrostatic adsorption. Multiple studies confirm that functional groups on rice and wheat straw biochars (hydroxyl [–OH], carboxylic acid [–COOH], Si–O–H) facilitate the electrostatic attraction or complexation of chromium species, thereby aiding in total Cr(VI) removal mechanisms that rely on adsorption and surface complexation. When straw biochars are activated in an acid solution, the surface area increases significantly and is enriched with oxygenated functional groups, such as –COOH and –OH groups, etc. This increased oxygenated surface area leads to a greater adsorption potential for Cr(VI), thereby resulting in a positively charged biochar that promotes electrostatic attraction of negatively charged Cr(VI) species ($HCrO_4^-$, CrO_4^{2-}). Therefore, this acid treatment process results in a strong adsorption capacity by increasing physical adsorption sites and oxygenated chemical binding groups [3,116].

7. ROLE OF OPERATING PARAMETERS ON THE ADSORPTION OF CHROMIUM

The operation parameters during the process of adsorption, that is, pH conditions, initial chromium concentrations, biochar dosage, contact time, temperature, ionic strength of the solution, and interfering ions, play a vital role in the adsorption process.

Generally, biochar materials used for the remediation of chromium have a zero point charge pH_{PZC} values between 4 and 12 and can be classified as basic or acidic [125]. Both the ionization of chromium species and the surface charge of biochar at different pH levels play a vital role in the adsorption process of chromium onto the biochar surface [8]. The biochar surface is protonated at pH 2–4, promoting electrostatic interaction with negatively charged species of chromium ($HcrO_4^-$ and $Cr_2O_7^{2-}$) [126,127]. As the pH increases, biochar becomes more negatively charged, which favors adsorption of cationic Cr (III) species but inhibits Cr(VI) uptake. The redox conversion of Cr(VI) to Cr(III) typically occurs under moderately low pH conditions, which enhances both electron transfer and binding strength [4]. Therefore, from the above discussion, it is quite clear that the surface charge on the biochar surface, the pH conditions and the type of chromium species present in the aquatic systems play a vital role in the removal of chromium from the wastewater. In addition to the pH of the solution, the initial chromium concentration also plays an important role in the process of adsorption.

At low initial Cr(VI) concentrations (<60 mg/L), removal efficiency increases with concentration due to abundant free adsorption sites. However, at higher concentrations (>60–100 mg/L), competition for active sites reduces efficiency, even though total adsorption per gram increases. Yang *et al*. found that for the optimal adsorption of Cr(VI) onto corn straw biochar, it occurred between 40 and 80 mg/L of Cr(VI), which balances the site availability and ionic competition [128,129]. Apart from the initial concentration, the dosage of biochar material should also be considered. When the biochar dose is increased, it initially enhances Cr(VI) removal efficiency due to the increased availability of active sites. However, beyond an optimal value, removal performance plateaus or even decreases per unit mass because of unsaturated adsorption sites, particle aggregation, and decreased surface accessibility [128,129]. Kokab *et al*. suggested that the decrease in adsorption with increasing biochar dosage may be due to the saturation of the active adsorption sites on the biochar surface. In addition, this decline could result from a longer diffusion path caused by the aggregation of biochar particles, which limits effective contact between the adsorbate and adsorbent [129].

The adsorption efficiency is typically determined by the contact time between the adsorbate and the adsorbent. Conversely, equilibrium achievement and the adsorption kinetics are affected by the contact time. Most of the chromium adsorption kinetics onto the biochar surface follow pseudo-second-order models. The collision between the adsorbate and the adsorbent increases with the temperature, which also enhances the diffusion of the chromium ions into the active site of the biochar materials [130-132]. In general terms, the chromium adsorption onto the biochar surface increases with an increase in temperature, but at excessively high temperatures, desorption and thermal transformation of the active groups take place, which decreases the adsorption efficiency [130-132]. The wastewater containing chromium ions from the industries is often laden with salts, other cations, anions, and organic compounds. These inorganic ions and organic compounds often compete with the chromium ions for the active sites on the biochar surface. The presence of salts can often compress the electrical double layer on charged biochar surfaces, which can hamper the chromium adsorption onto the biochar surface [130-132]. The presence of anions, such as sulfate, nitrate, or phosphate can lead to competitive adsorption with chromium ions, thereby decreasing chromium adsorption efficiency.

The sensitivity of the process is elevated toward variations in pH, followed by initial concentration, temperature, and adsorbent dose. In addition, iron impregnation or acid activation increases the robustness of the process by providing redox-active and oxygenated functional sites that alleviate pH dependency and broaden the operating range [109,133,134]. These interlinked parameters define the optimal chromium removal conditions, typically at acidic pH, moderate initial concentration, controlled biochar dosage, and equilibrium contact, ensuring maximum adsorption and redox conversion efficiency.

8. ADSORPTION–REDUCTION SYNERGY IN CR(VI) DETOXIFICATION

The alteration of straw-based biochars substantially changes their surface chemistry, electronic structure, and reactivity to chromium species in aqueous systems. Among the modification strategies, metal impregnation and alkali activation are the most commonly applied to fine-tune the adsorption capacity and redox behavior of the biochar. The introduction of metal atoms – such as iron, can lead to the formation of catalytic redox centers that can better facilitate electron transfer to oxidize or reduce Chromium species. In contrast, alkali treatment enhances surface area, porosity, and cation-exchange capacity through the deprotonation of acidic surface groups. The modification of biochar with Fe produces redox-active metal sites (ferrous ion [Fe^{2+}]/ferric ion [Fe^{3+}]) that promote catalysis of electron transfer reactions that enhance the reduction of toxic Cr(VI) to less toxic Cr(III) [3]. Notably, Fe modification represents a significant contribution to the synergistic removal mechanisms, as it couples adsorption with redox transformation. This ultimately enhances the effectiveness of detoxification beyond the capabilities of adsorption alone [3]. Conversely, the influence of alkali-treated straw biochar shows a more negative surface charge due to deprotonation of acidic functional groups and surface alkalinity, which increases their cation-exchange capacity [4]. For this reason, alkali treatment has been shown to enhance the binding of positively charged species but shows a weaker redox potential in comparison to Fe modification. Alkali treatment does not result in the introduction of new or enhancement of existing redox-active centers [4]. As a result, alkali activation mainly enhances physical adsorption and ion-exchange removal mechanisms, but not the reductive transformation of Cr(VI). The combination of physical adsorption (i.e., electrostatic attraction, pore trapping) and chemical reduction (i.e., surface-mediated electron transfer) enhances the efficient detoxification of Cr(VI) [4].

The adsorption-reduction of Cr(VI) takes place in four steps [Figure 8].

Step 1: Initial adsorption – Cr(VI) species ($HcrO_4^-$, CrO_4^{2-}) are captured on the biochar surface via functional groups such as –OH, –COOH, and Fe–O

Step 2: Surface-mediated reduction – adsorbed Cr(VI) is reduced to Cr(III) by redox-active groups and nano-metal oxides, which remain active even at neutral pH

Step 3: Stabilization and ion exchange – Cr(III) forms stable $Cr(OH)_3$ precipitates or binds to biochar surfaces; counterions (e.g., K^+) are released to maintain charge balance

Step 4: Synergy – adsorption enriches Cr(VI) at reactive sites, enhancing local concentration and optimizing the reduction process for effective detoxification.

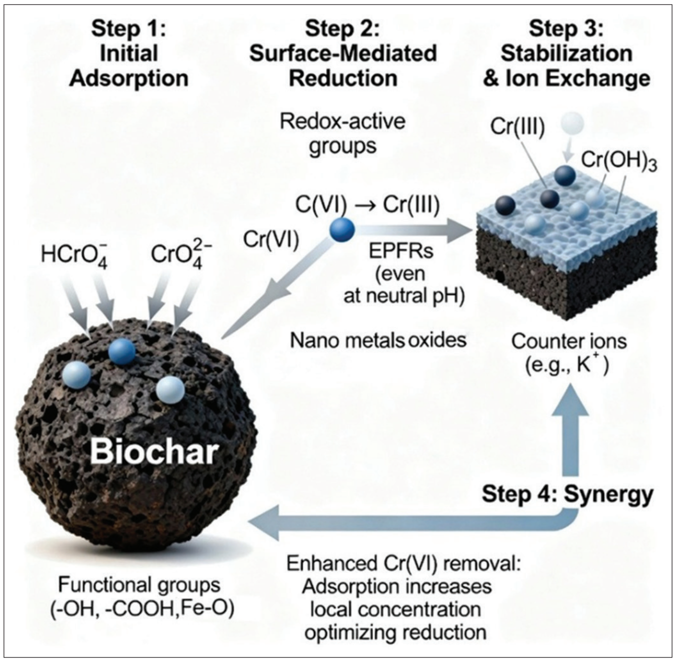


Figure 8: Schematic illustration of the adsorption–reduction synergy mechanism for hexavalent chromium removal by biochar.

9. REGENERATION METHODS

The sustainable use of biochar through the process of reactivation or regeneration significantly decreases the cost of the wastewater treatment process. The greater recyclability and reusability of biochar endow it with more economic efficiency in the field of wastewater treatment [135,136]. Several regeneration techniques have been developed for the regeneration of biochar materials, which invariably depend on the type of biochar, its physicochemical characteristics, and the type of pollutant.

Regeneration of chromium-loaded biochar is essential for sustainable reuse in wastewater treatment. Multiple studies have investigated different types of regeneration techniques [Figure 9] – chemical, thermal, and hybrid – targeting both desorption efficiency and structural preservation as outlined in Figure 9. Table 3 outlines some of the regeneration methods adopted for chromium adsorbed straw biochar materials.

Chemical regeneration is primarily used, and the regeneration process depends on the interaction forces between the biochar surface and the pollutant molecule [138]. In the process of regeneration, various solvents and chemical reagents are used to desorb the adsorbate from the biochar surface [138]. In the chemical regeneration technique, various acids such as hydrochloric acid, nitric acid, sulfuric acid, and organic acids are used for the regeneration of the adsorbent and the recovery of pollutants [138]. In this technique, the regeneration efficiency depends on the solubility of adsorbates in the solvents, and care is also taken to ensure that the removal reagents do not alter the chemical structure of the adsorbent. There are several solvents (inorganic/organic) that are used to desorb both inorganic/organic adsorbates in the chemical regeneration technique [138]. Ma *et al*. used sodium hydroxide as an eluent for the regeneration of chromium, which showed good initial regeneration efficiency at 91.57% in the first cycle, but efficiency declined significantly to 52.39% by the fourth cycle, indicating progressive loss of active sites or fouling during regeneration [104]. In

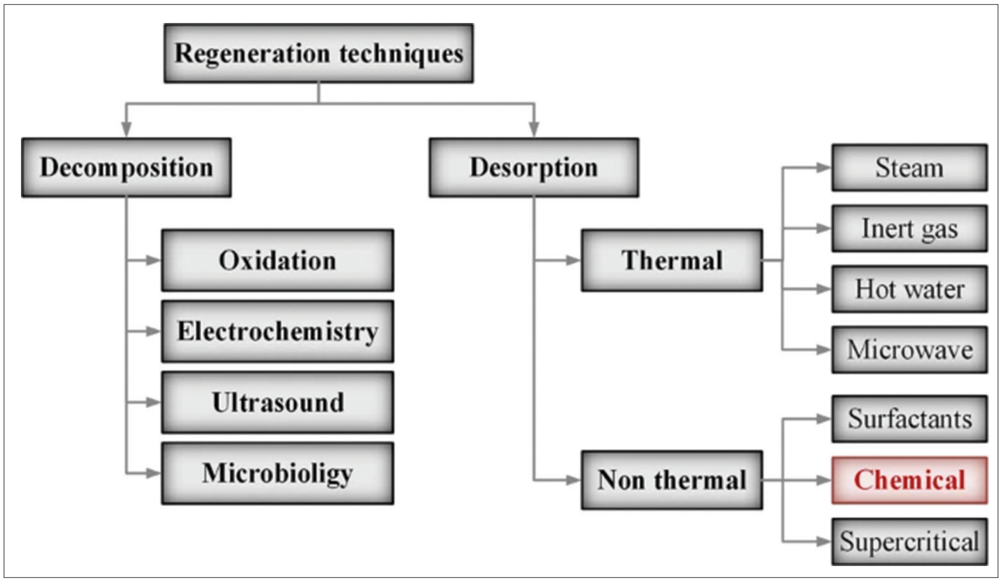


Figure 9: Different regeneration techniques proposed for adsorbents' reactivation. The diagram illustrates two main regeneration pathways – decomposition and desorption used to restore the adsorption capacity of spent biochar [137].

Table 3: Different eluents and efficiency for adsorbent regeneration using the chemical regeneration technique.

Adsorbent material	Eluents used	No. of regeneration cycle	Regeneration efficiency	References
Corn straw	1M NaOH	4	1st cycle=91.57% 2ndcycle and 3rd cycle=70.65% 4thcycle=52.39%	[104]
Wheat straw and Corn straw	1M KOH and 0.1M HNO$_3$	10	WS for Cr (VI) and Cr (III) reach 125.6 and 68.9 mg/g, and that of CS for Cr (VI) and Cr (III) can reach 87.4 and 62.3 mg/g in every cycle	[86]
Rice straw	1 M NaOH	2	1st cycle=5.70, 2nd cycle=6.31 and 3rd cycle=5.22 mg/g	[81]
Rice straw	0.3 N HCl	3	1st cycle=95.57%, 2nd cycle=92.17% and 3rd cycle=74.17%	[116]
Rice straw	0.5 mol/L NaOH	3	With three cycles, the efficiency decreased	[103]
Rice straw	0.1 M NaOH	4	With the 4th cycle, the removal rate was 75.12%	[106]
Corn straw	0.05 M EDTA	8	97.6, 96.8, 95.7, 84.5, 78.6, 70.1, 58.3, and 42.3% with first, second, third, fourth, fifth, sixth, seventh and eighth cycle, respectively	[115]

Cr (VI): Hexavalent chromium, Cr (III): Trivalent chromium

addition to acids and alkalis, reducing and complexing agents are also used in the regeneration process.

10. REAL FIELD CHALLENGES AND COST-BENEFIT ANALYSIS

Chromium rarely appears on its own in actual effluents; it is typically accompanied by additional cationic and anionic species that impact Cr uptake and reduction on biochar surfaces. Studies on competitive adsorption have found that coexisting metal ions, such as Lead(II) ion (Pb^{2+}), Copper(II) ion, Nickel(II) ion, Zinc(II) ion, and Cadmium(II) ion (Cd^{2+}) reduce Cr(VI) adsorption capacity significantly, primarily due to accommodation of active oxygen- and amino-functional groups and distortion of the electrostatic balance (toxicity) of the adsorbent surface. Dong *et al*. (2018) found that bi-functional ions Pb^{2+} and PO_4^{3-} decreased Cr(VI) removal by 21.8% and 12.4%-, respectively, through the formation of competing Pb–O and P–NH$_2$ complexes for active sites, while multivalent ions also precipitated a compression of the electrical double layer on the adsorbent, restricting the diffusion of anions [139]. In addition, other reports show that cationic competition responds to ionic radius and hydration energy – larger ions such as Pb^{2+} and Cd_2^+ display stronger binding to negatively charged functional groups and therefore reduce the availability of redox-active Fe^{2+}/Fe^{3+} sites to facilitate Cr(VI) reduction. Coexisting anions impact exclusion regarding adsorption equilibria and are in direct competition with CrO_4^{2-} or $HcrO_4^-$ species. Phosphate and sulfate ions exhibit similar effects on Cr(VI) equilibrium as observed for cations.

Ageing of biochar changes the accessibility of pores and redox chemistry because of surface oxidation, mineral fouling, and dissolution/re-precipitation of Fe phases, which usually means a shift in removal changes from reductive pathways to physio-sorption. The aging effect on Fe-modified biochars may lead to the loss of the electron-donor species and the Fe oxides. This leads to a decrease in the reduction of Cr(VI) and decreased redox activity. In real effluents, performance is normally decreased compared to clean lab (upstream) conditions because of competition from anions (SO_4^{2-}, Cl^-), organic matter coatings, and pH buffers. In the case of Fe-modified rice-straw biochar, very good near-quantitative removal of Cr(VI) was judged through actual industrial wastewater. However, a much higher dose/contact time is required than seen in research using synthetic waters, which is mainly due to the matrix effects. In mechanistic type work, where electron-donating moieties in the biochars (and Fe(II) on the surface) converted the Cr(VI) to Cr(III). In addition, kinetics also slow down as co-ions start competing for sites and due to fouling of biochar.

Agricultural residues such as rice or wheat straw, which are locally available at around USD 20 t^{-1}, serve as low-cost feedstock for biochar production. In a small-scale rotary kiln processing 2 t of dry straw/day, the yield is approximately 40% biochar by mass, with a production cost (including transport, drying, pyrolysis, crushing, and activation) of about USD 1.10 kg^{-1}. With an adsorption capacity of 100 mg Cr(VI) g^{-1}, a treatment plant processing 10 m^3/day of wastewater containing 100 mg/L of Cr(VI) would remove roughly ~0.5 kg of chromium daily. This value may vary depending on the adsorption and the loading capacity of the biochar. At a bed loading of 10 g biochar L^{-1} (100 kg/day), the annual media requirement is approximately 7.3 t, considering five regeneration cycles with 90% capacity retention, resulting in a total biochar cost of around USD 8,030/year. This translates to a media cost of USD 2.2 m^{-3} of treated water or roughly USD 22 kg^{-1} of Cr removed. Accounting for a 10% performance loss per regeneration cycle, the adjusted cost rises marginally to USD 2.7 m^{-3} (USD 26.8 kg^{-1} Cr). Compared with activated carbon, which typically offers half the adsorption capacity at a similar or higher unit price, straw-biochar reduces media cost by nearly 50%. In addition, feedstock availability, ease of regeneration, and negligible disposal cost – since Cr(VI) is reduced to stable Cr(III) – further enhance its cost-effectiveness. Even under conservative assumptions (higher feedstock transport or reduced capacity), treatment costs remain below USD 3.5 m^{-3}, confirming that straw-derived biochar is a sustainable and economically attractive solution for chromium-contaminated water remediation.

11. CRITICAL ANALYSIS AND FUTURE RESEARCH DIRECTIONS

A successful implementation of straw-derived biochar for the remediation of chromium from aqueous environments needs an understanding of the key factors influencing its performance. The three most widely used straws to generate biochar for the remediation and reduction of chromium VI from aqueous solutions are wheat, rice, and corn straw. Figure 10 shows a comparison between the adsorption capacities of biochar materials generated from the three straws. Corn straw biochar exhibits the highest maximum adsorption capacity with a wide distribution range. The outliers in the corn straw biochar show that tuning the physicochemical and experimental conditions can help in achieving higher adsorption efficiencies. Biochar from wheat straw has a relatively higher median adsorption capacity than biochar from rice straw. The moderate interquartile range indicates that wheat straw biochar has consistent performance. The rice straw biochar shows a low median adsorption capacity with a narrow spread [Figure 10]. This reflects uniform adsorption performance for rice straw biochar materials. Hence, of all three straw biochar materials used for the adsorption and reduction of Cr(VI), corn biochar needs further investigation due to its large spread. The main limitations identified across the studies include issues such as low inherent efficiency of pristine biochar, variability in performance, scale-up feasibility, long-term stability and regeneration of straw-derived biochar, interference from co-existing ions/organics in real wastewaters, and environmental risk of spent adsorbents.

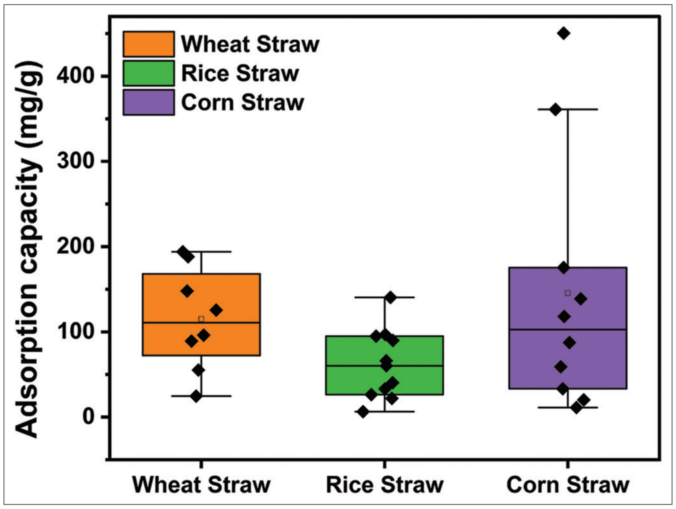


Figure 10: Distribution of adsorption capacity (mg/g) for wheat, rice and corn straw biochar materials. The boxes represent the interquartile range, and the central line indicates the median values. Individual data points show the adsorption capacities from different studies.

It is necessary to focus some specific future research such as enhancing modification techniques, improving performance under environmentally relevant conditions (e.g., neutral pH), mechanistic investigations under realistic matrices, scaling up applications, systematic pilot-scale and field studies, development of multifunctional/bi-functional straw biochar composites, standardized regeneration protocols, and life cycle/cost–benefit analyses to bridge the gap between laboratory performance and practical implementation.

12. CONCLUSION

In this review, straw-derived biochar has emerged as a sustainable, low-cost, and tunable adsorbent with promising potential for chromium remediation from contaminated water. Its performance arises from a combination of physicochemical processes, including adsorption, surface reduction, ion exchange, electrostatic attraction, and complexation, all influenced by feedstock composition, pyrolysis parameters, and activation/modification techniques. Iron-, magnesium-, and acid-modified straw biochars demonstrated substantial synergistic effects due to enhanced surface charge, functional group density, and redox-active sites, enabling efficient Cr(VI) reduction and Cr(III) immobilization even under variable pH conditions. The adsorption followed predominantly Langmuir isotherm and pseudo-second-order kinetics. Regeneration via acid or alkali eluents, complemented with reducing agents, such as oxalic acid, can effectively restore surface functionality for multiple reuse cycles with good retention of adsorption capacity.

13. ACKNOWLEDGMENTS

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14. AUTHORS' CONTRIBUTIONS

All authors made substantial contributions to conception and design, acquisition of data, or analysis and interpretation of data; took part in drafting the article or revising it critically for important intellectual content; agreed to submit to the current journal; gave final approval of the version to be published; and agreed to be accountable for all aspects of the work. All the authors are eligible to be an author as per the International Committee of Medical Journal Editors (ICMJE) requirements/guidelines.

15. CONFLICTS OF INTEREST

The authors report no financial or any other conflicts of interest in this work.

16. ETHICAL APPROVALS

This study did not involve human participants, animals, or any sensitive personal data; therefore, ethical approval and informed consent were not required.

17. DATA AVAILABILITY

Data will be made available on request.

18. PUBLISHER'S NOTE

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