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Biomass-Derived Biochar for The Removal of Textile Dyes from Aqueous Solutions: A Review

Indurti Srijayathi,[1] *Megana Boddupally,*[1] *Md. Atif Qaiyum* [1*] *Gubbala V. Ramesh*[2†]

[1]Department of Chemical Engineering, Chaitanya Bharathi Institute of Technology, Osman Sagar Rd, Kokapet, Gandipet, Telangana, 500075, India.
[2]Department of Chemistry, Chaitanya Bharathi Institute of Technology, Osman Sagar Rd, Kokapet, Gandipet, Telangana, 500075, India.

Abstract. The textile industry discharges massive volumes of coloured wastewater containing hazardous dyes. Traditional treatment methods such as coagulation, flocculation, membrane filtration and advanced oxidation often face high costs, high sludge production and low efficiency for recalcitrant dyes. Biomass-derived biochar has emerged as a low-cost adsorbent made by pyrolysing agricultural or industrial wastes. This review critically surveys recent literature on biochar for textile dye removal. It analyses sustainable feedstocks, structure–property relationships, comparative adsorption performance, adsorption mechanisms, practical constraints and research gaps. Data show that biochars activated with acids or metals can achieve removal capacities above 200 mg/g for certain cationic dyes, while even pristine biochars can remove anionic dyes at moderate capacities. However, reported adsorption capacities vary widely due to feedstock heterogeneity, pyrolysis conditions and experimental design. Regeneration efficiency often declines sharply after several cycles, and most studies use synthetic dye solutions rather than real textile effluents. The review identifies research gaps such as the lack of standardised testing, limited continuous-flow experiments and insufficient techno-economic analysis. Future work should integrate biochar adsorption with renewable energy and advanced oxidation, explore hybrid composites, and evaluate performance under realistic conditions.

1. Introduction

Dyes are widely used in textiles, paper, cosmetics, plastics and food industries [1]. Global dye production has increased with industrialisation, and large volumes of coloured wastewater are released into water bodies [2]. The kitchen waste article estimated that 1.3 billion tonnes of kitchen waste are generated annually and more than 85 000 tonnes of methylene blue dye are released into aquatic systems each year [2]. Many synthetic dyes are toxic, carcinogenic and resistant to biodegradation. Acid Red 73, for example, is a persistent azo dye that can cause toxicity, mutagenicity and carcinogenicity [1]. Malachite green

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* Corresponding author: qaiyumatif0@gmail.com
† Corresponding author: venkataramesh_chm@cbit.ac.in

residues can persist in fish for 44–342 days [3]. Methyl orange and methylene blue can cause skin, eye and respiratory irritation and have chronic toxicity [2]. The release of dye-laden effluents therefore poses serious ecological and health risks.

Conventional dye removal methods include coagulation–flocculation, membrane filtration, ion exchange, chemical oxidation, biological treatment and adsorption. Each technique has limitations. [4,5] Coagulation–flocculation generates large volumes of sludge. Membrane filtration suffers from fouling, high capital cost and disposal of concentrate streams [2]. Ion exchange and advanced oxidation require expensive resins or reagents. Biological treatment is slow and often ineffective for synthetic dyes because of their complex aromatic structures [6] . Adsorption using activated carbon is effective but expensive and not always sustainable.

Biochar is a carbonaceous material produced by pyrolysis or hydrothermal carbonisation of biomass under limited oxygen. It has been promoted as a low-cost alternative to activated carbon. [7,8] Biochar can be made from agricultural residues, forestry wastes, sewage sludge or industrial by-products. Its porous structure, surface functional groups and surface area can be tuned by feedstock selection, pyrolysis temperature and post-treatment. In recent years, hundreds of studies have investigated biochar for dye removal. However, the literature is fragmented, with inconsistent experimental conditions, varying degrees of characterisation and little attention to real wastewater. A comprehensive, critical synthesis is needed to identify patterns, limitations and future directions. This review follows seven thematic sections that explore dye pollution challenges, sustainability of biomass-derived biochar, structure–property relationships, comparative adsorption performance, adsorption mechanisms, practical constraints and research gaps. An abstract, introduction and conclusion provide context and summarise findings.

2. Textile dye pollution: challenges beyond conventional adsorbents

Textile dyes can be classified into several structural classes, including azo dyes, anthraquinone dyes, triphenylmethane dyes, phthalocyanine dyes and sulphur dyes. Azo dyes, characterised by the –N=N– azo linkage, account for more than 60 % of commercial dyes and include reactive, direct and acid dyes. Anthraquinone dyes contain a fused aromatic system with carbonyl groups and are responsible for blue and green colours. Triphenylmethane dyes, such as malachite green and crystal violet, have three benzene rings connected to a central carbon and generate intense colours but are highly toxic. Phthalocyanine dyes have complex macrocyclic structures coordinated to metals and are used for blue and green shades. Sulphur dyes are produced by vulcanising aromatic amines and produce dark colours. The diversity of structures explains why different dyes interact differently with adsorbents and why no single treatment can remove all dyes effectively [1].

The environmental impacts of dye pollution extend beyond colour and toxicity. Dyes reduce photosynthetic activity by blocking light, lower dissolved oxygen through microbial degradation and produce toxic metabolites that accumulate in sediments and biota. Some dyes contain heavy metals such as chromium, copper or cobalt that can leach into water, adding metal toxicity to the dye pollution problem. The persistence of dyes in sediments and groundwater threatens drinking water resources and aquatic food webs. Biotic degradation products of azo dyes can form aromatic amines, some of which are carcinogenic. These impacts highlight the urgency of developing effective and sustainable treatment methods.

Recalcitrance of dyes arises from their aromatic and heterocyclic structures, resonance stabilisation and substituent groups that confer solubility and lightfastness. Reactive dyes form covalent bonds with fibres and resist hydrolysis. Cationic dyes are attracted to negatively charged surfaces while anionic dyes require protonated or positively charged adsorbents. The presence of bulky groups, such as in triphenylmethane dyes, affects diffusion into micropores. Understanding these structural factors helps explain why some adsorbents perform better with specific dye classes and guides the design of tailor-made biochars.

The textile industry generates vast quantities of wastewater containing a mixture of dyes, salts, surfactants and auxiliaries. In China alone, textile and dyeing factories discharge 2.0–2.3 billion tonnes of wastewater each year and dyeing effluent contributes 70–80 % of the total [3]. (Figure 1) Many dyes have aromatic, heterocyclic or azo structures that confer intense colours and chemical stability. Acid dyes carry sulfonic acid groups, while basic dyes such as methylene blue and crystal violet are cationic. Reactive dyes form covalent bonds with fibres and are hard to remove. Their structural diversity leads to different behaviours in water and on adsorbents.

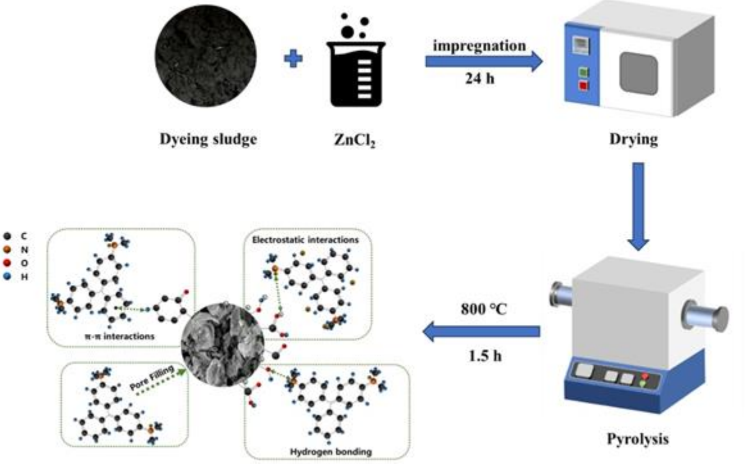

Fig. 1. Schematic illustration of $ZnCl_2$-assisted impregnation and pyrolysis of dyeing sludge to produce porous carbon, highlighting the processing steps (impregnation, drying, and pyrolysis at 800 °C) and the dominant adsorption interactions (π–π interactions, electrostatic attraction, pore filling, and hydrogen bonding). [3]

Uncontrolled discharge of dye-laden wastewater affects aquatic ecosystems by reducing light penetration, inhibiting photosynthesis and causing oxygen depletion. Many dyes or their degradation products are toxic, carcinogenic, mutagenic or endocrine-disrupting. For example, Acid Red 73 has stable diazo and disulfonic structures that resist biodegradation [1]. Malachite green is toxic to fish and mammals and persists for months [3]. The presence of dyes can also interfere with drinking water treatment and reduce aesthetic quality.

Conventional treatment methods are often inadequate. Coagulation–flocculation can remove suspended solids and some dyes but is less effective for soluble dyes and generates sludge requiring disposal. Advanced oxidation processes break down dyes through reactive radicals, but the required chemicals (e.g., ozone, Fenton reagents) are expensive and energy-intensive. Membrane filtration can physically separate dyes but suffers from membrane fouling and high operating pressure [2]. Biological treatments using bacteria, fungi or algae are promising but are limited by the toxicity and recalcitrance of dyes [9]. Adsorption onto activated carbon is effective but commercial activated carbon is costly and requires regeneration. There is a need for low-cost, efficient and sustainable adsorbents.

Biochar derived from wastes has drawn attention because it utilises abundant biomass and can be produced locally. However, dye adsorption onto biochar depends on many parameters including biochar surface chemistry, solution pH, ionic strength, dye concentration and contact time. Data from the literature show significant variation in adsorption capacities and removal efficiencies. To critically assess biochar as an alternative, we must analyse why some biochars perform well and others do not, and identify the factors that limit practical application.

3. Why biomass-derived biochar? A sustainability–performance perspective

Pyrolysis of biomass produces three products: biochar, bio-oil and syngas. The bio-oil and syngas can be used as fuels to offset the energy required for pyrolysis. In pyrolysis of sewage sludge, the process also immobilises heavy metals and organic pollutants, reducing environmental risks while generating adsorbent. The OAE study argued that converting dyeing sludge into biochar recovers resources and reduces the volume of hazardous waste disposed of [3]. Using syngas to heat the reactor can close the energy loop and improve the overall sustainability of biochar production.

The availability of feedstocks is vast. Agricultural residues such as rice straw, wheat straw, corn stover, soybean stalks, coconut shells, and fruit peels are produced in millions of tonnes annually. Forestry wastes include sawdust, bark and pruning residues. Industrial wastes include paper mill sludge, distiller's grains and textile sludge. Kitchen waste, as highlighted in the dynamic removal study, represents a large but underutilized resource [2]. Collecting and processing these wastes require logistics and storage infrastructure, and seasonal availability may affect supply. Decentralised pyrolysis units near waste sources could reduce transportation costs and provide local employment.

After using biochar as an adsorbent, it can often be returned to soil as a conditioner. Adsorbed dyes may degrade slowly in soil or be mineralised by soil microbes. Biochar improves soil structure, water retention, nutrient retention and microbial activity. Applying biochar to saline soils improved wheat growth and chlorophyll content [10]. (Figure 2) These agronomic benefits close the loop between waste valorisation, water treatment and agriculture. However, the stability of adsorbed dyes in soil and potential leaching need further investigation.

Using biochar for dye removal offers sustainability benefits. Biochar production converts agricultural waste, forestry residues and sewage sludge into useful sorbents. It reduces waste disposal problems, recovers energy in the form of syngas or bio-oil and sequesters carbon. For instance, the kitchen waste article emphasised that pyrolysis of organic waste can mitigate odours and greenhouse-gas emissions while producing adsorbents [2]. Biochar can also be applied to soils to improve fertility, water retention and carbon content after its adsorptive service. Sulfuric-acid-treated kitchen-waste biochar not only removed crystal violet dye but also improved wheat growth under saline conditions [10]. Biochar addition increased chlorophyll content and mitigated salt stress, demonstrating a circular benefit of using waste-derived adsorbents.

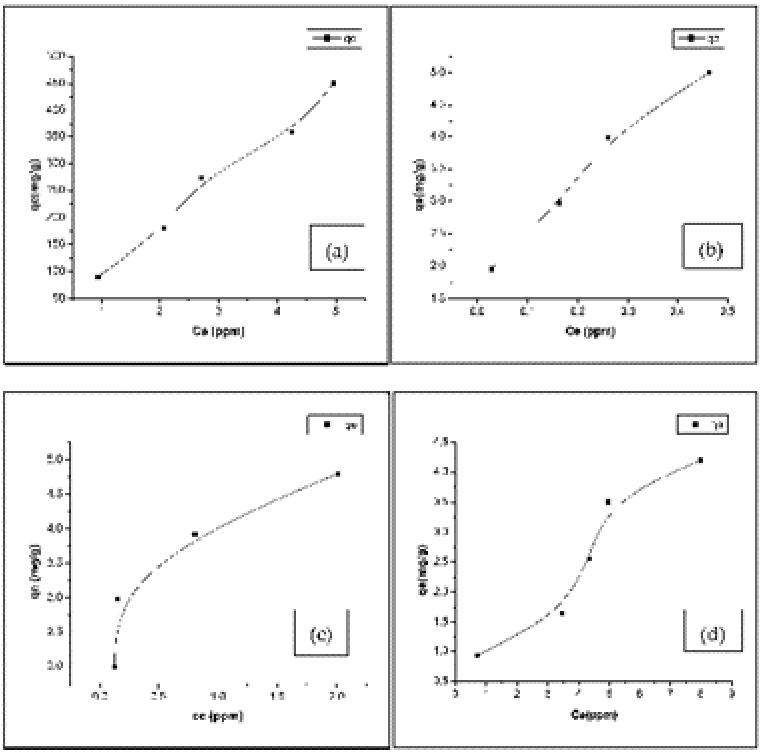

Fig. 2. Adsorption isotherms of CV dye onto biochars at room temperature: (a) H2SO4-biochar, (b) Ag-biochar, (c) Fe-biochar, and (d) non-activated biochar. [10]

Activated carbon typically costs USD 1–3 per kilogram. Biochar made from agricultural wastes can cost as little as USD 0.1–0.5 per kilogram depending on feedstock and production method. The price of biochar is significantly lower than that of activated carbon and that pyrolysis at about 700 °C gives biochar with good sorption properties [11]. Hydrothermal carbonisation at 180 °C followed by activation can also produce efficient sorbents with lower energy consumption. However, pyrolysis is energy-intensive, and the economic viability depends on scale, feedstock logistics, energy recovery and co-product utilisation. A life-cycle assessment is needed to quantify environmental and economic benefits.

Sustainability also relates to performance and stability. Biochar's performance can be enhanced by chemical activation, heteroatom doping and composite formation. For example, $ZnCl_2$-activated dyeing sludge biochar showed high surface area (495 m^2/g) and a maximum malachite green adsorption capacity of 224 mg/g [3]. Sulfuric-acid-treated kitchen-waste biochar achieved 100 % removal of crystal violet with a capacity of 450 mg/g [11]. Metal impregnation or nanoparticles can introduce redox activity that aids adsorption and catalytic degradation. However, modifications may increase cost and complicate regeneration. A sustainability perspective requires balancing performance gains against additional chemical inputs and energy use.

Finally, biochar use aligns with circular economy principles. Many developing countries produce large volumes of agricultural waste such as rice husk, peanut shells, sunflower husks, sugarcane bagasse and kitchen waste. Converting these wastes into adsorbents provides local solutions for wastewater treatment and reduces reliance on imported materials. However, feedstock composition varies widely, and there is a need for local characterisation and optimisation. The success of biomass-derived biochar in dye removal therefore depends on both sustainability and performance considerations.

4. Structure–property relationships governing dye adsorption

Pyrolysis parameters beyond temperature also influence biochar properties. Slow pyrolysis, with low heating rates (5–10 °C min^{-1}) and long residence times (30–60 minutes), produces more biochar with higher fixed carbon and surface functional groups. Fast pyrolysis, with high heating rates (>100 °C s^{-1}) and short residence times (<2 seconds), yields more bio-oil and less biochar. Hydrothermal carbonisation operates in subcritical water at 150–250 °C and produces hydrochar with different properties. Heating rate affects pore development: slow heating allows for gradual release of volatiles and formation of micropores, while fast heating favours mesopores. Residence time influences the degree of carbonisation and aromaticity. Researchers must tailor pyrolysis conditions to balance yield and performance.

Heteroatom doping beyond nitrogen is an emerging strategy. Phosphorus doping introduces acidic sites that can attract basic dyes and increases hydrophilicity. Sulphur doping can create electron-rich thiophene-like structures that participate in π–π interactions and redox reactions. Boron doping can alter electronic distribution and improve adsorption of certain anionic dyes. Co-doping with multiple heteroatoms may create synergistic effects. However, such modifications often require additional chemicals and may not be sustainable. [12, 13] (Figure 3) Physical modifications such as ball milling, ultrasound and steam explosion can increase surface area, expose internal pores and introduce defects that enhance adsorption. Ball milling reduces particle size, increases external surface area and can induce mechanochemical reactions with additives. Ultrasonic treatment can clean and open pores. Steam explosion disrupts the lignocellulosic matrix and increases porosity. Combining physical modifications with chemical activation may further improve performance but raises questions about cost and scalability.

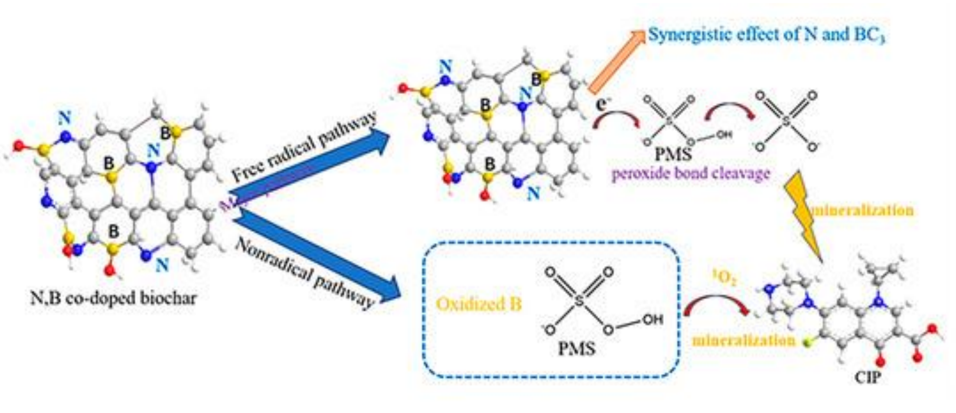

Fig. 3. Illustration of the degradation mechanism for Ciprofloxacin (CIP) by N, B co-doped biochar. [13]

Biochar properties are primarily controlled by feedstock composition, pyrolysis temperature, heating rate, residence time and activation method. Lignocellulosic biomass consists of cellulose, hemicellulose and lignin. Lignin-rich feedstocks often yield biochar with higher aromaticity, ash content and stability, whereas cellulose-rich feedstocks may produce biochar with more oxygenated functional groups. The carbon, hydrogen and nitrogen content of the feedstock influence the development of surface functional groups that interact with dyes.

Pyrolysis temperature strongly affects surface area, porosity and surface chemistry. Tang et al. reported that increasing pyrolysis temperature from 200 °C to 600 °C increases specific surface area but decreases oxygen-containing functional groups; at temperatures above 700 °C surface area may continue to increase but many functional groups are lost, reducing adsorption of crystal violet [14]. Biochars produced at moderate temperatures (400–600 °C) therefore balance surface area and functional groups. Extreme temperatures may produce graphite-like structures with high stability but fewer active sites. The same study showed that pH influences adsorption: crystal violet removal increased up to pH 6–7 but decreased at highly alkaline or acidic conditions [14].

Feedstock composition and nitrogen content can also enhance adsorption. The oil-seed shell study found that pyrolysis of peanut-shell feedstock at 900 °C (PNS900) produced biochar with higher nitrogen content and graphitic structures compared with sunflower shell biochar (SFS900); PNS900 contained 11.4 % more carbon and exhibited enhanced stability [15]. Despite its very low BET surface area (2.1 m^2/g), PNS900 removed 100 % of the anthraquinone dye Remazol Brilliant Blue at pH 6.5, whereas SFS900 with 85.7 m^2/g required a higher dosage to achieve the same removal [15]. These results highlight that functional groups and electronic properties, rather than surface area alone, control adsorption. The same article reported that Congo Red adsorption peaked at around 89 % and decreased above pH 5.5 because negatively charged biochar surfaces repel anionic dye molecules [8-2]. Equilibrium was reached within 5 min for PNS900 but took 30 min for SFS900, indicating faster diffusion and adsorption in the nitrogen-rich biochar [15]. Mechanistic analysis suggested that π–π stacking, hydrogen bonding and anion–cation interactions contributed to dye binding [15].

Activation and modification further tune biochar properties. Acid treatments such as sulfuric acid, nitric acid or phosphoric acid increase surface area and introduce sulfonic or carboxylic groups that enhance cationic dye adsorption. Alkali activators such as KOH and NaOH create micro–mesoporous structures and oxygenated sites beneficial for anionic dyes. Metal impregnation with Fe, Zn, Cu or Mn introduces catalytic sites that can adsorb and oxidise dyes. $ZnCl_2$ activation of dyeing sludge biochar produced a surface area of 495 m^2/g and improved pore development [3]. Sulfuric-acid-treated kitchen-waste biochar achieved a capacity of 450 mg/g for crystal violet and 100 % removal efficiency [10]. Metal-doped biochars such as magnetite composites have shown 99.99 % removal of methylene blue and good reusability [16].

Post-functionalisation with polymers or biopolymers can also enhance adsorption. The chitosan-impregnated sugarcane bagasse biochar had a maximum adsorption capacity of 170 mg/g for Congo Red and removed more than 95 % of dye at pH 3[6]. Chitosan introduces amine groups that interact with anionic dyes and overcome the negative surface charge of pristine bagasse biochar. However, polymer impregnation may block pores and reduce surface area, so a balance is needed.

The structure–property relationship is thus complex. High surface area alone does not guarantee high adsorption capacity; the type and density of functional groups, aromaticity and electronic properties are equally important. Moderate pyrolysis temperatures often yield biochar with a balanced combination of surface area and functional groups. Nitrogen-doped and graphitic structures can enhance electron donor–acceptor interactions with dyes. Chemical activation and modification significantly improve performance but increase cost and potential environmental impact. Understanding these relationships helps design biochars tailored for specific dyes and wastewater matrices.

5. Comparative adsorption performance across biomass feedstocks

Comparative study of rice-residue biochars developed from rice varieties (Vietnamese IR50404 and Japanese Koshihikari) demonstrated that adsorption properties differ based on feedstock type. Biochars made using the IR50404 rice variety displayed 1.5 and 1.7 times greater adsorption of a cationic dye and anionic dye respectively compared with the Koshihikari biochar. The differences in adsorption were due to differences in ash content and alkalinity of the Vietnamese rice straw used as feedstock; the ash contributed to negatively charged sites to attract cationic dye molecules and increased pore size to promote adsorption of anionic dye molecules. Adsorption of both types of dyes was affected by pH: alkaline environments promoted adsorption of cationic dye but inhibited adsorption of anionic dye. This research demonstrates that all biomass is not the same; rice variety, production environment, and ash composition influence adsorption performance. [17] (Figure 4)
Comparing acid-treated and metal-modified biochars reveals trade-offs. In the Materials study, sulfuric-acid-treated biochar achieved 450 mg/g capacity for crystal violet with 100 % removal, whereas silver-modified biochar achieved similar removal but only 5 mg/g capacity [10]. The acid treatment increased surface area and introduced sulfonic groups that provide many active sites, while silver nanoparticles occupied surface area without contributing significantly to adsorption. This shows that modifications must be chosen carefully, and metal impregnation may not always be advantageous. Differences between dynamic and static experiments also matter. In batch systems, equilibrium can be reached relatively quickly because mass transfer is not limited by flow. In dynamic packed-bed systems, breakthrough curves depend on flow rate, bed height and mass transfer coefficients. The dynamic study of MB and MO showed a large difference in removal efficiency because of kinetic and diffusion limitations [2]. Therefore, translating batch results to continuous systems requires caution.

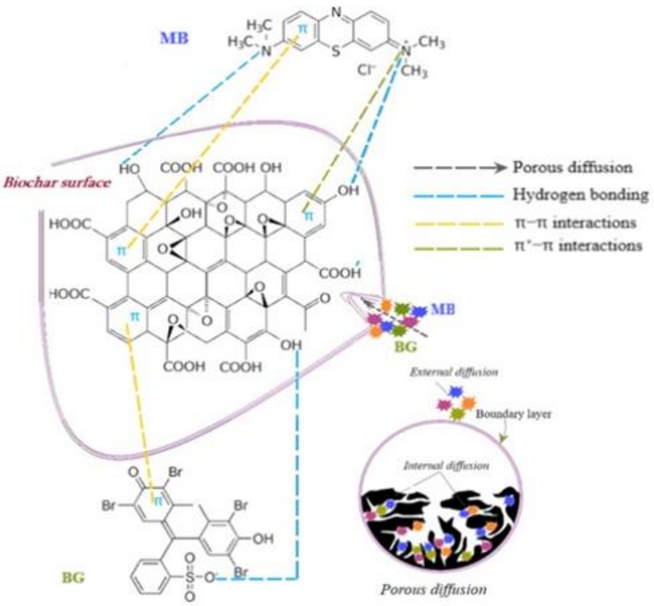

Fig. 4. Possible physical adsorption mechanism of MB and BG onto biochar surface. [17]

6. Adsorption mechanisms: universal trends vs dye-specific behaviour

Structural differences among dye classes explain their adsorption behaviour. Anthraquinone dyes have planar aromatic structures with carbonyl groups that can form hydrogen bonds and π–π interactions with biochar. Azo dyes contain azo linkages and often have sulfonic acid groups that increase solubility but reduce aromaticity. Triphenylmethane dyes have a bulky, non-planar structure that may prevent them from entering micropores, making mesoporous biochar more suitable for their adsorption. Cationic triphenylmethane dyes like malachite green and crystal violet can interact with negatively charged biochars via electrostatic attraction and cation–π interactions. Knowing how chemical structures on a biochar surface, specifically electron withdrawing groups and their effect on electron density and π-π interactions, will help in designing biochars that have adequate pore size and functional group characteristics. [18]

Chemical composition of solution can affect the mechanism of adsorption. Ionic strength can change the thickness of the electrical double layer surrounding biochar particles which can alter the degree of electrostatic attraction/repulsion between dye molecules and biochar particles. For example, high salt concentration can either interfere with dye molecules by competing for adsorption sites or can shield the charge on the surface of the biochar particles resulting in less adsorption. Multivalent cationic species (i.e. Ca2+, Mg2+) can bind with dye molecules and thus affect the adsorption of dyes. Also, natural organic matter can occlude pores and/or compete for adsorption sites. The effect of temperature on adsorption is seen as it affects both the enthalpy and kinetics of adsorption; i.e. endothermic adsorption is increased at higher temperatures whereas exothermic adsorption can be reduced. Thus, to provide an accurate assessment of biochar performance, the evaluation of performance should be conducted using representative water chemistries rather than laboratory-controlled conditions.

Biochar adsorption of dyes involves multiple physical and chemical interactions. The dominant mechanisms depend on dye type, biochar surface chemistry, solution pH and ionic strength. Universal mechanisms include electrostatic attraction, π–π electron donor–acceptor interactions, hydrogen bonding and pore filling. For cationic dyes such as methylene blue, crystal violet, basic blue and malachite green, electrostatic attraction occurs between the positively charged dye molecules and negatively charged oxygen-containing groups (e.g., carboxyl, phenolic, carbonyl) on the biochar surface. π–π stacking occurs between the aromatic rings of the dye and the aromatic structures of biochar. Pore filling allows dye molecules to diffuse into mesopores and micropores.

For anionic dyes such as Congo Red, Acid Red 73 and Eriochrome Black T, the interactions are different. Negatively charged dyes are repelled by negatively charged biochar surfaces at neutral or alkaline pH. Acidic conditions protonate biochar functional groups and reduce electrostatic repulsion. Hydrogen bonding between the sulfonic acid groups of dyes and hydroxyl or amine groups on biochar becomes important. Functionalisation with amine-containing polymers such as chitosan can create positive sites that attract anionic dyes [6]. The mandarin-peel MBCOT biochar used triethylenetetramine to introduce amine groups and achieved high adsorption of Acid Red 73[1].

Isotherm models provide insight into adsorption behavior. The Langmuir model assumes monolayer adsorption on homogeneous sites and finite adsorption capacity. Many studies report Langmuir fitting for cationic dyes. In the dynamic removal study, methylene blue followed a Langmuir isotherm, indicating monolayer coverage [1]. The same study reported that methyl orange followed a Freundlich isotherm, suggesting multilayer adsorption on heterogeneous sites. Freundlich constants also provide information on surface heterogeneity and adsorption intensity.

Kinetic models distinguish between rate-limiting steps. The pseudo-first-order model implies physisorption with diffusion control, while the pseudo-second-order model implies chemisorption where chemical bonding limits the rate. Many studies report pseudo-second-order kinetics for dye adsorption on biochar. For example, the magnetite biochar composite and giant reed biochar both followed pseudo-second-order kinetics [16], suggesting chemisorption. The chitosan-bagasse composite also followed pseudo-second-order kinetics and an endothermic process [6]. In contrast, some low-capacity adsorption processes may follow pseudo-first-order kinetics.

pH significantly influences adsorption mechanisms. For cationic dyes, removal efficiency often increases with pH because deprotonated biochar surfaces carry negative charges that attract positively charged dye molecules. For anionic dyes, removal is higher at acidic pH because protonation reduces repulsion. The oil-seed shell study showed that Congo Red adsorption decreased above pH 5.5[15]. The mandarin-peel biochar achieved optimal Acid Red 73 removal at pH 1.5[23]. The dynamic removal study observed that MO adsorption was lower than MB partly due to pH and electrostatic interactions [2]. These findings highlight the need to consider pH adjustment in practical applications.

Functional groups introduced through activation or modification create specific adsorption sites. Acid treatments produce –COOH, $–SO_3H$ and –OH groups that donate electrons or form hydrogen bonds. Alkaline treatments create hydroxyl groups and increase negative surface charge. Nitrogen doping introduces pyridinic and graphitic nitrogen that enhance π–π interactions and electron donor–acceptor complexes [15]. Metal doping introduces redox-active sites that may catalyse dye degradation in addition to adsorption. Composite formation with polymers such as chitosan provides amine groups that interact with anionic dyes. These interactions can be summarised as universal trends: cationic dyes favour negative and aromatic surfaces, anionic dyes favour protonated or amine-functionalised surfaces, and large aromatic dyes benefit from π–π interactions and pore filling.

7. Critical research gaps and future directions

Machine-learning and data-driven approaches are emerging as powerful tools for rational biochar design. By integrating datasets on biochar physicochemical properties, dye molecular features and adsorption performance, models such as artificial neural networks, random forests and support-vector machines can predict adsorption capacity and optimise preparation conditions. These approaches can reveal non-intuitive relationships and reduce experimental trial-and-error. However, their reliability depends strongly on data quality, diversity and transparency. The lack of standardised datasets and the underreporting of negative results increase the risk of overfitting and limit model transferability.

Future research must also address multi-pollutant removal. Real textile wastewater contains metals, pharmaceuticals, nutrients and microplastics in addition to dyes. Designing multifunctional biochars requires understanding competitive and cooperative adsorption. For instance; iron oxide modified biochars that have been demonstrated to both remove dye pollutants and phosphate from water as well as biochar-based magnetic composites that will allow for the physical separation of the adsorbed materials. Treatment methods combining adsorption (which includes biochar) with biodegradation or advanced oxidation technologies may potentially provide a wider range of treatment options.

While an abundance of literature exists in this area, there is still much work to be done. The variability of the feedstocks used to produce biochars along with variable experimental conditions (i.e. different pH's, temperatures etc.) hinders comparisons between studies. As such, standard procedures for producing biochar (characterizing), and for conducting adsorption experiments need to be developed. Currently, there is very little mechanistic understanding of the process at the molecular level and needs to be studied using various forms of advanced spectroscopy and modeling. Finally, regeneration costs, techno economic feasibility, lifecycle analysis and regulatory frameworks all need to be assessed before large scale deployments are possible.

8. Conclusion

Biochar made from biomass is one potential, though still in its infancy, solution for removing textile dyes. A key review of this subject emphasizes that biochar's adsorption performance is influenced by much more than just surface area. The many variables include; feedstock composition, temperature during pyrolysis, functional groups found on the surface, and electronic properties. These variables lead to a great deal of variability among results reported within literature. For example, when produced under moderate to high temperatures, activated or chemically modified biochars have shown good adsorption capacity for cationic dyes; however, anionic dyes are much harder to remove and require the biochar to be specifically designed to interact with them. Successful lab-based trials rarely translate into effective field-based applications due to low regeneration stability, usage of artificial dye solutions, lack of studies performed using continuous flow, and real wastewater streams. Cost-effectiveness is directly related to the cost of the raw materials, lifespan of the biochar adsorbent, and how efficiently it may be regenerated. The sustainability of biochar as a treatment method will depend on the amount of energy required to produce it, the chemicals used to modify it, and the environmental impacts associated with its production and disposal. In order to progress biochar from an exciting adsorbent to a viable treatment option, there needs to be standardized testing methods developed, additional research conducted to improve the mechanistic understanding of how biochar functions, performance evaluations that accurately reflect real-world applications, economic and life cycle assessments, and development of hybrid approaches utilizing multiple treatment technologies.

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