

Adsorptive removal of benzo(ghi)perylene from vehicle-wash wastewater using eucalyptus wood waste-derived biochar

Mohd Abul Hasan 

Received: 18 July 2026 / Accepted: 12 October 2026
© The Author(s) 2026

Abstract

This study reports the preparation of eucalyptus waste-derived biochar (EWB) and its application in the sustainable removal of benzo[ghi]perylene (B[ghi]P) from vehicle wash wastewater (VWW). The EWB was prepared by pyrolysis under a nitrogen atmosphere, followed by activation with 1 M HCl to improve its adsorption performance. The prepared EWB was characterized by surface area analysis (SAA), Fourier transform infrared spectroscopy (FTIR), scanning electron microscopy (SEM), and transmission electron microscopy (TEM). Batch adsorption experiments were conducted to evaluate the effects of initial B[ghi]P concentration, contact time, pH, temperature, and adsorbent dosage on the adsorption process. The optimal adsorption conditions were: initial B[ghi]P concentration of 40 mg L⁻¹, contact time of 120 min, temperature of 60 °C, adsorbent dosage of 0.8 g, and pH of 6. Under these conditions, the maximum removal efficiency of EWB reached 98%. The adsorption process best fits the pseudo-second-order kinetic model, while the Langmuir isotherm model provides the best statistical fit for the equilibrium adsorption data. Furthermore, the EWB maintained an initial adsorption efficiency of over 84% after six consecutive adsorption-desorption cycles, indicating its potential for reusability. The observed adsorption efficiency and reusability suggest that the EWB may have the potential to treat wastewater contaminated with PAHs. However, further research is needed to determine its practical and sustainable applications, addressing issues such as energy requirements, production costs, environmental impact, and scalability.

Keywords Biochar · Eucalyptus wood waste derived biochar · Benzo(ghi)perylene · Vehicle-wash wastewater · Adsorptive removal

Introduction

Due to their widespread presence in flora, soil, sediments, the atmosphere, and aquatic systems, polycyclic aromatic hydrocarbons (PAHs) are among the primary organic contaminants strictly regulated in the environment^{1,2}. These compounds range in molecular weight from 128 to 278 Daltons and consist of at least two fused aromatic rings, encompassing a wide range of chemicals³. Although thousands of PAHs are known, only a few

This Article in Press is shared early to give you faster access to new research. It is citable and carries a permanent DOI. The final edited version will replace it automatically.



are routinely monitored. The U.S. Environmental Protection Agency (USEPA) has listed 16 priority polycyclic aromatic hydrocarbons (PAHs) as potentially hazardous substances, including benzo[ghi]perylene (B[ghi]P), dibenz[a, h]anthracene, indeno[1,2,3-cd]pyrene, benzo[a]pyrene, benzo[k]fluoranthene, benzo[b]fluoranthene, chrysene, benz[a]anthracene, pyrene, fluoranthene, anthracene, phenanthrene, fluorene, acenaphthene, acenaphthylene, and naphthalene⁴. Among them, B[ghi]P is a high molecular weight polycyclic aromatic hydrocarbon that is usually associated with vehicle exhaust emissions and petroleum-derived pollutants, and is therefore a pollutant of concern in VWW.

Sources of PAHs include pyrolysis, petroleum, and biogenic processes, which are associated with both natural phenomena and human activities⁵. Natural sources include synthesis by microorganisms and terrestrial plants, as well as volcanic activity, all of which contribute to the release of PAHs into the environment. However, human activities are the primary source of PAHs emissions, with internal combustion engines, power generation and heating, coal tar pitch and bitumen production, carbon black manufacturing, oil spills, fossil fuel combustion, waste incineration, agricultural activities, vehicle emissions, and military operations being significant sources^{6,7}. Due to their inherent chemical stability, PAHs are persistent environmental pollutants with broad biotoxicity, making their remediation a global concern. Car wash wastewater is considered a significant secondary source of PAHs, as pollutants deposited on vehicle surfaces are flushed into the drainage system during the washing process.

Compared to tricyclic PAHs, PAHs with four or more aromatic rings tend to be less volatile, more lipophilic, and less water-soluble, which significantly affects their environmental fate and bioavailability⁸. B[ghi]P is a hexacyclic high molecular weight PAHs with extremely low water solubility, high chemical stability, and strong hydrophobicity, making it highly persistent in the environment and difficult to remove by conventional treatment methods⁹. These high molecular weight PAHs are also known for their long-distance migration capabilities as well as their mutagenicity, carcinogenicity, genotoxicity, and teratogenicity⁹. In addition, they are associated with endocrine disruption when their concentrations can rapidly exceed regulatory thresholds. The carcinogenicity of PAHs increases with increasing molecular weight^{10,11}. Human exposure to these compounds is associated with an increased risk of cancer in multiple organs, including the skin, stomach, bladder, lungs, scrotum, esophagus, breast, kidneys, prostate, and pancreas⁷. In addition, PAHs are suspected of having endocrine-disrupting and immunosuppressive effects, which further exacerbates their toxicological effects^{12,13}.

PAHs are solid compounds with high melting and boiling points, low vapor pressures, and very low water solubility. Their antioxidant and reducing properties increase with increasing molecular weight, while their solubility decreases with increasing number of aromatic rings. However, their high lipophilicity makes them readily soluble in organic solvents. These compounds also exhibit photosensitivity, electrical conductivity, corrosion resistance, and thermal stability^{14–17}. Despite environmental and health concerns, PAHs have commercial applications in lubricants, pharmaceuticals, agrochemicals, dyes, and photographic materials. Certain specific PAHs are used as intermediates in multiple industries: acenaphthene in dyes and pesticides, pyrene in pigments, anthracene in wood preservatives, fluoranthene in agrochemicals, fluorene in pharmaceuticals, and phenanthrene in resins^{18–20}. PAHs are also found in asphalt, roof tar, electronics, and liquid crystal technology.

Growing concerns about the environmental and health risks of PAHs have spurred research into various soil and water remediation technologies^{21,22}. However, many methods are limited by high costs, operational complexity, and the generation of toxic byproducts such as trihalomethanes and haloacetic acids, which pose additional health risks^{23–25}. Given the toxicity, mutagenicity, and carcinogenicity of PAHs, the development of effective removal strategies remains a top priority²⁶. Among existing treatment technologies, adsorption has attracted considerable attention due to its low cost, ease of operation, high efficiency, and environmental sustainability^{26–28}. Recent research has focused on the development of innovative, sustainable adsorbents for removing various pollutants from wastewater, including dyes, heavy metals, antibiotics, and hydrocarbons^{29–33}. Advanced materials such as modified carbon, metal-organic frameworks, nanocomposites, and bio-based fibers have demonstrated high efficiency in adsorption, catalytic degradation, and electrochemical treatment^{34–39}. Research has also

emphasized the role of physicochemical properties, process optimization, and regeneration potential in improving treatment performance^{40–44}. Environmentally friendly materials such as nanocellulose lithium montmorillonite beads, chitosan-alginate biocomposites, and urea-formaldehyde resin composites have also shown excellent adsorption efficiency and reusability^{45–48}. Recent studies have demonstrated the significant potential of biomass-derived and waste-based adsorbents for removing various pollutants from water bodies. Ghazoui et al.⁴⁹ reported the efficient adsorption of dyes and antibiotics using biomimetic carbon foam, while Ghazoui et al.⁵⁰ highlighted recent advancements in functionalizing waste biomass-derived biochar to improve water treatment applications. Similarly, Raif et al.⁵¹ demonstrated the adsorption potential of agricultural waste, including argan oil, date palm, and olive residue, for removing methylene blue. These studies highlight the growing importance of low-cost, sustainable biomass-derived materials as highly efficient adsorbents in water and wastewater treatment.

Several reviews have highlighted the rapid development of nanocomposite adsorbents and hybrid materials, including the combination of metal or metal oxide nanoparticles with ionic liquids, which can be used for the effective removal of dyes and phenolic pollutants^{52–57}. These studies collectively demonstrate that adsorption and photocatalysis technologies have made significant progress in the removal of pollutants such as methylene blue, allura red, 4-nitrophenol, sunset yellow, and Congo red. However, despite these advances, research on the sustainable removal of high molecular weight PAHs, particularly B[ghi]P, using low-cost EWB derived from lignocellulosic biomass remains limited. Current research focuses on optimizing adsorption media and understanding the environmental factors that can enhance PAHs removal^{17,58–60}. Therefore, this study aims to prepare EWB and evaluate its sustainable removal performance of B[ghi]P from VWV using adsorption. In addition, the effects of adsorption parameters, kinetics, and isotherms behavior were investigated to evaluate the adsorption mechanism of the EWB and its potential applications in wastewater treatment.

Novelty and research gap

B[ghi]P is a persistent and toxic PAHs commonly released from vehicle emissions and found in VWV. Although many studies have reported the removal of PAHs using different adsorbents, limited research has focused on the removal of B[ghi]P using low-cost EWB prepared from eucalyptus wood waste. This study addresses this research gap by preparing EWB and evaluating its performance for the removal of B[ghi]P from VWV. The effects of adsorption conditions, including initial concentration, contact time, pH, temperature, and adsorbent dosage, were investigated. In addition, adsorption kinetics, and isotherms studies were carried out to understand the adsorption process. The results demonstrate that EWB is a sustainable, low-cost, and effective adsorbent for the removal of B[ghi]P from VWV.

Materials and methods

In this study, waste eucalyptus wood was converted into EWB, which was then used as an adsorbent to remove B[ghi]P from VWV. The following methods were employed to achieve this objective.

Sample collection

VWV samples were collected in pre-washed 2.5-liter amber glass bottles and transported to the laboratory for analysis. Eucalyptus wood samples were collected from abandoned eucalyptus trees near the local botanical garden. Since only discarded wood was collected, without disturbing the living trees, no special permission was required. Experienced botanists identified the plant material based on morphological characteristics and with reference to standard taxonomic keys, confirming it belonged to the genus “Eucalyptus”. The identified wood samples were then cleaned and processed for biochar preparation.

Synthesis of EWB

The waste eucalyptus samples were first thoroughly washed with deionized water to remove any attached impurities, then air-dried and cut into small pieces. About 5 g of the prepared eucalyptus waste were placed in a steel reactor and pyrolyzed in a tube furnace (OTF-1200X, MTI Corporation, USA) at 450 °C under a continuous nitrogen atmosphere for 1 h. After cooling to room temperature, the resulting EWB was chemically activated with 1 M HCl at a solid-to-liquid ratio of 1:10 (g: mL). The mixture was kept at room temperature (around 25 °C) for 2 h for acid activation. After activation, the EWB was separated from the acid solution and repeatedly washed with deionized water until the final washing water had a pH of about 6.5–7.0. Then, the activated EWB was dried, ground into a fine powder, sieved through a 100-mesh sieve (approximately 150 µm), and stored in a clean glass bottle for subsequent adsorption experiments.

Characterization of EWB

Laboratory prepared EWB were characterized using SEM, TEM, FTIR, and specific surface area analysis. The surface morphology of the EWB was observed using a scanning electron microscope manufactured by JEOL Ltd. (Tokyo). The internal microstructure of EWB was characterized using TEM. A small amount of powdered EWB was dispersed in anhydrous ethanol and sonicated for 10–15 min to obtain a homogeneous suspension. A drop of the suspension was placed on a carbon-coated copper grid and air-dried at room temperature. TEM images were obtained using a JEOL JEM-2100 transmission electron microscope at an accelerating voltage of 200 kV. FTIR spectra were measured using a Nicolet iS10 Fourier transform infrared spectrometer (Cary 630, Agilent Technologies, USA) with a scanning range of 4000–400 cm⁻¹. Average pore size, total pore volume, and specific surface area were determined using a Quantachrome NOVA2200e specific surface area and porosity analyzer. The zeta potential of EWB was measured at different pH values, and the pH value at which the zeta potential crossed zero was determined as the zero charge point.

Adsorption experiments

This study employed batch adsorption experiments to investigate the adsorption performance of EWB on B[ghi]P in VWW. The effects of EWB dosage (0.2, 0.3, 0.4, 0.5, 0.6, 0.7 and 0.8 g), contact time (15, 30, 45, 60, 90, and 120 min), pH (1–11), and B[ghi]P concentration (10, 20, 30, and 40 mg L⁻¹) were examined. Due to the extremely low aqueous solubility of B[ghi]P, a concentrated acetonitrile stock solution was prepared and then artificially added to the collected wastewater to obtain the initial concentration of 10–40 mg L⁻¹ required for batch adsorption experiments. To minimize solvent effects, the volume of acetonitrile added to each treatment group was kept as small and constant as possible, and a corresponding solvent blank containing the same amount of acetonitrile but without B[ghi]P was used as a control. The total volume of each experimental sample was 100 mL. All B[ghi]P adsorption and kinetic experiments were conducted in 500 mL Erlenmeyer flasks, with samples stirred at 150 rpm using a magnetic stirrer at 30 °C, 40 °C, 50 °C, and 60 °C until adsorption equilibrium was reached. All experiments were repeated three times, and only the average value is reported.

Analyses and calculations

The adsorption performance of EWB for the removal of B[ghi]P from VWW was evaluated by calculating the equilibrium adsorption capacity (q_e , mg g⁻¹), percentage removal (%), and root mean square error (RMSE). The corresponding equations used for these calculations are presented in Table 1. The initial (C_0) and equilibrium (C_e) concentrations of B[ghi]P were used to determine the equilibrium adsorption capacity and percentage removal, while RMSE was employed to evaluate the goodness-of-fit between the experimental adsorption data and the values predicted by the adsorption models.

Table 1 Equations used for calculating the equilibrium adsorption capacity (q_e), percentage removal (%), and root mean square error (RMSE) for the adsorption of B[ghi]P onto EWB.

Parameter	Equation
Equilibrium adsorption capacity	$q_e = \frac{(C_0 - C_e)}{M} \times v \quad (1)$
Percentage removal	$\text{Removal (\%)} = \frac{(C_0 - C_e)}{C_0} \times 100 \quad (2)$
Root mean square error (RMSE)	$\text{RMSE} = \sqrt{\frac{1}{N-2} \sum_{i=1}^N (q_{e,\text{exp}} - q_{e,\text{calc}})^2} \quad (3)$

Note: q_e = equilibrium adsorption capacity (mg g^{-1}); C_0 = initial B[ghi]P concentration (mg L^{-1}); C_e = equilibrium B[ghi]P concentration (mg L^{-1}); V = solution volume (L); M = mass of EWB (g); N = number of experimental data points; $q_{e,\text{exp}}$ = experimental equilibrium adsorption capacity (mg g^{-1}); and $q_{e,\text{calc}}$ = model-calculated equilibrium adsorption capacity (mg g^{-1}).

Table 2 Kinetic equations of the pseudo-first-order, pseudo-second-order, and Elovich models for the adsorption of B[ghi]P onto EWB.

Model	Equation
Pseudo-first order ⁶¹	$\ln(q_e - q_t) = \ln q_e - k_1 t \quad (4)$
Pseudo-second order ⁶²	$\frac{t}{q_t} = \frac{t}{q_e} + \frac{1}{k_2 q_e^2} \quad (5)$
Initial adsorption rate ⁶³	$h = k_2 q_e^2 \quad (6)$
Elovich ²¹	$q_t = 1/b \ln(ab) + 1/b \ln(t) \quad (7)$

Note: q_e = adsorption capacity at equilibrium (mg g^{-1}); q_t = adsorption capacity at time (t) (mg g^{-1}); t = adsorption time (min); k_1 = pseudo-first-order rate constant (min^{-1}); k_2 = pseudo-second-order rate constant ($\text{g mg}^{-1} \text{min}^{-1}$); h = initial adsorption rate for the pseudo-second-order model ($\text{mg g}^{-1} \text{min}^{-1}$); α = initial adsorption rate in the Elovich model ($\text{mg g}^{-1} \text{min}^{-1}$); and β = Elovich constant (g mg^{-1}).

First, the UV-Vis absorption spectrum of B[ghi]P in the wavelength range of 300–420 nm was scanned using a Shimadzu UV-1700 UV-Vis spectrophotometer. The maximum absorption wavelength (λ_{max}) was observed at 375 nm, which was subsequently used for quantitative analysis. Calibration standard solutions of B[ghi]P with concentrations ranging from 10 to 40 mg L^{-1} were prepared using stock solutions, corresponding to the concentration range used in the adsorption experiments. A calibration curve was constructed by plotting absorbance (A) against B[ghi]P concentration (C), described by a linear regression equation $A = 0.0215 C + 0.006$, with a coefficient of determination $R^2 = 0.998$. After adsorption, the adsorbent was separated from the aqueous phase, and the absorbance of the resulting solution was measured at 375 nm using an appropriate blank solution. The residual concentration of B[ghi]P was determined using the calibration equation. Since no comprehensive matrix-specific recovery/interference study and method-specific limits of detection (LOD) and quantification (LOQ) were carried out, these are considered the limitations of the existing analytical method. The adsorption kinetics of B[ghi]P on EWB were analyzed using pseudo-first-order, pseudo-second-order, and Elovich kinetic models (Table 2). The corresponding kinetic parameters, including rate constants (k_1 and k_2), initial adsorption rate (h), Elovich constants (α and β), and equilibrium adsorption capacity (q_e), were determined. The initial adsorption rate (h) was calculated using the pseudo-second-order kinetic model ($h = k_2 q_e^2$). These kinetic parameters were used to compare the adsorption rates of different models, and the kinetic model that best described the experimental data was determined.

To investigate the adsorption equilibrium, we employed four isothermal models: the Langmuir model, the Freundlich model, the Temkin model, and the Harkins-Jura model (Table 3). These models were used to evaluate the equilibrium adsorption behavior of B[ghi]P on EWB. To avoid dimensional inconsistencies in the parameters of the Temkin model, we used its empirically linearized form.

Table 3 Adsorption isotherm models and equations used to describe the adsorption of B[ghi]P onto EWB.

Model	Equation
Langmuir ⁶⁴	$\frac{C_e}{q_e} = \frac{1}{KQ_{\max}} + \frac{C_e}{Q_{\max}}$ (8)
Freundlich ⁶⁵	$\frac{C_e}{q_e} = \frac{1}{KQ_{\max}} + \frac{C_e}{Q_{\max}}$ (9)
Temkin ⁶⁶	$q_e = B_T \ln K_T + B_T \ln C_e$ (10)
Harkins-Jura ⁴	$\frac{1}{q_e^2} = \frac{B}{A} - \frac{1}{A} + \log C_e$ (11)

Note: C_e = equilibrium concentration of B[ghi]P (mg L^{-1}); q_e = equilibrium adsorption capacity (mg g^{-1}); q_m = maximum adsorption capacity (mg g^{-1}); K_L = Langmuir constant (L mg^{-1}); K_F = Freundlich constant; n = Freundlich adsorption intensity parameter; K_T = empirical Temkin equilibrium binding constant (L mg^{-1}); B_T = empirical Temkin fitting coefficient; and A and B = Harkins–Jura constants.

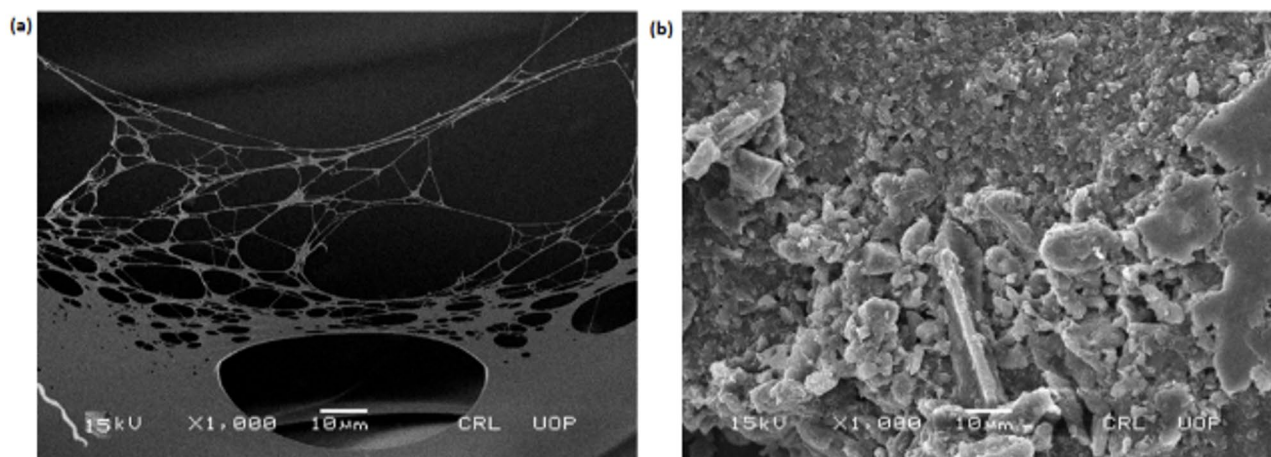


Fig. 1 SEM images of EWB before and after adsorption of B[ghi]P from VWV: **(a)** fresh EWB before adsorption and **(b)** EWB after adsorption, showing surface deposition and partial pore blockage due to the adsorption of B[ghi]P.

Results and discussion

Characterization of EWB

In the laboratory, the properties of EWB made from eucalyptus wood waste were examined using SEM, TEM, FTIR, and specific surface area.

SEM analysis

Figure 1 shows SEM images of EWB before and after adsorption of B[ghi]P in VWV. Before adsorption (Fig. 1a), the EWB exhibits a highly porous and heterogeneous surface morphology, containing numerous interconnected pores, cavities, and irregular channels. The rough surface texture and well-developed pore network are typical characteristics of EWB prepared by pyrolysis, endowing it with a large specific surface area and abundant adsorption sites for pollutants. These structural features promote the diffusion and retention of B[ghi]P molecules through pore filling, surface adsorption, and hydrophobic interactions. Following the adsorption process (Fig. 1b), the surface morphology of the EWB underwent significant changes. Its porous structure became less defined, with some pores partially or completely blocked by deposits. The surface became denser due to the accumulation of adsorbed B[ghi]P and other organic components from the VWV. Furthermore, aggregated particles and irregular deposits appeared on the surface, indicating that adsorption occurred not only on the external surface of the EWB but also within its pore structure. The reduced pore visibility and the appearance of the surface coating suggest that available adsorption sites were effectively occupied during the adsorption process. The

observed changes in surface morphology provide strong visual evidence for the successful adsorption of B[ghi]P onto EWB. The high porosity of the fresh EWB, along with the significant pore filling and surface deposition after adsorption, indicates that the physical properties of the EWB play a crucial role in enhancing adsorption performance. These findings are consistent with the proposed adsorption mechanism and confirm that EWB is a highly efficient and sustainable adsorbent for the removal of B[ghi]P from VWW.

TEM analysis

Figure 2 presents the TEM images of EWB before and after the adsorption of B[ghi]P from VWW. The fresh EWB (Fig. 2a) exhibits a heterogeneous carbonaceous structure with a rough surface and visible porous regions. The porous structure and irregular morphology provide abundant active sites for the adsorption of B[ghi]P molecules. These structural characteristics are favorable for the diffusion of contaminants into the EWB matrix, thereby enhancing its adsorption capacity. After the adsorption process (Fig. 2b), noticeable changes in the EWB microstructure were observed. The surface became comparatively denser, and several porous regions appeared partially filled or covered by adsorbed materials. The increased contrast and irregular deposits indicate the successful attachment of B[ghi]P molecules onto the EWB surface and within its pore structure. These morphological changes provide clear evidence that the adsorption process occurred effectively through surface adsorption and pore-filling mechanisms, confirming the potential of EWB as a sustainable adsorbent for the removal of B[ghi]P from VWW.

FTIR analysis

Figure 3 shows the FTIR spectra of EWB before and after B[ghi]P adsorption. Fresh EWB exhibits characteristic absorption bands at approximately 3420 cm^{-1} (O–H stretching vibration), 2920 cm^{-1} (C–H stretching vibration), 1710 cm^{-1} (C=O stretching vibration), 1600 cm^{-1} (aromatic C=C stretching vibration), and 1100 cm^{-1} (C–O stretching vibration), indicating the presence of various surface functional groups. After B[ghi]P adsorption, the positions and intensities of these absorption bands changed slightly, indicating that they interacted with B[ghi]P. Only repeatable FTIR peak shifts are considered, while small shifts of $2\text{--}5\text{ cm}^{-1}$ are cautiously interpreted and not considered as direct evidence of chemical interactions. The changes in the O–H, C=O, and C–O absorption bands suggest that oxygen-containing surface groups may play a role through weak intermolecular interactions. The changes in the aromatic C=C bonds may be related to the $\pi\text{--}\pi$ interactions between the aromatic domains of EWB

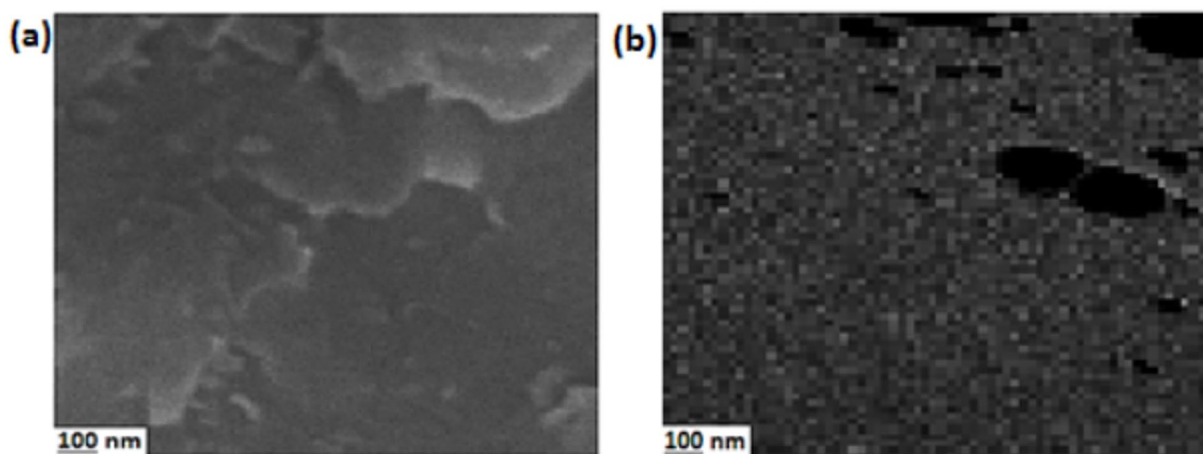


Fig. 2 TEM images of EWB before and after adsorption of B[ghi]P from VWW: (a) fresh EWB before adsorption and (b) EWB after adsorption, showing the deposition of B[ghi]P on the EWB surface and changes in the pore structure.

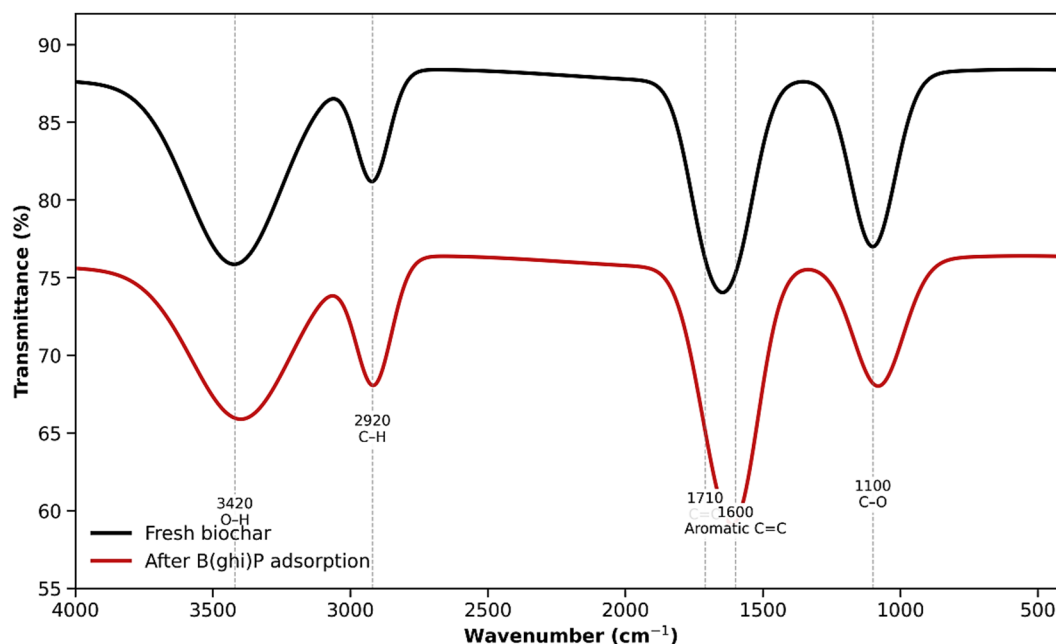


Fig. 3 FTIR spectra of EWB before and after the adsorption of B[ghi]P from VWW, illustrating the changes in surface functional groups associated with the adsorption process.

and the fused aromatic rings of B[ghi]P. The changes in the C-H region may also reflect the interaction between B[ghi]P and the hydrophobic carbonaceous domains of EWB. Given the highly hydrophobic and non-ionizing nature of B[ghi]P, hydrophobic interactions and van der Waals forces may further promote its adsorption. Overall, the FTIR observations support the combined promotion of B[ghi]P adsorption by surface functional groups, π - π interactions, hydrophobic interactions, and van der Waals forces; however, these spectral changes should be considered as supplementary evidence rather than definitive evidence of a specific adsorption mechanism.

Surface area analysis

The surface area characteristics of EWB were determined using N_2 adsorption-desorption analysis at 77.35 K (Fig. 4). The BET specific surface area of EWB was $42.68 \text{ m}^2 \text{ g}^{-1}$, the BJH specific surface area was $35.24 \text{ m}^2 \text{ g}^{-1}$, the total pore volume was $0.038 \text{ cm}^3 \text{ g}^{-1}$, and the average pore radius was $17.85 \text{ \AA}$. Compared to many high-porosity or activated biochar adsorbents, the relatively small BET specific surface area of EWB indicates that its adsorption performance cannot be attributed solely to its specific surface area. The relatively low specific surface area and pore volume may limit the number of accessible adsorption sites, which could be partly responsible for the lower adsorption capacity of EWB. However, the adsorption of B[ghi]P is also influenced by pore accessibility and the surface chemistry of the adsorbent. Therefore, the adsorption behavior of EWB may be influenced by both its porous structure and surface interactions, including π - π interactions, hydrophobic interactions, van der Waals forces, and interactions with surface functional groups. After B[ghi]P adsorption, the measured BET and BJH specific surface areas, pore volumes, and average pore radii all decreased. These changes are consistent with the fact that adsorbed B[ghi]P molecules occupy or partially block accessible pores, providing evidence that porous structures participate in the adsorption process.

Fig. 4 N₂ adsorption isotherm of EWB measured at 77.35 K for the determination of BET surface area and pore structure characteristics.

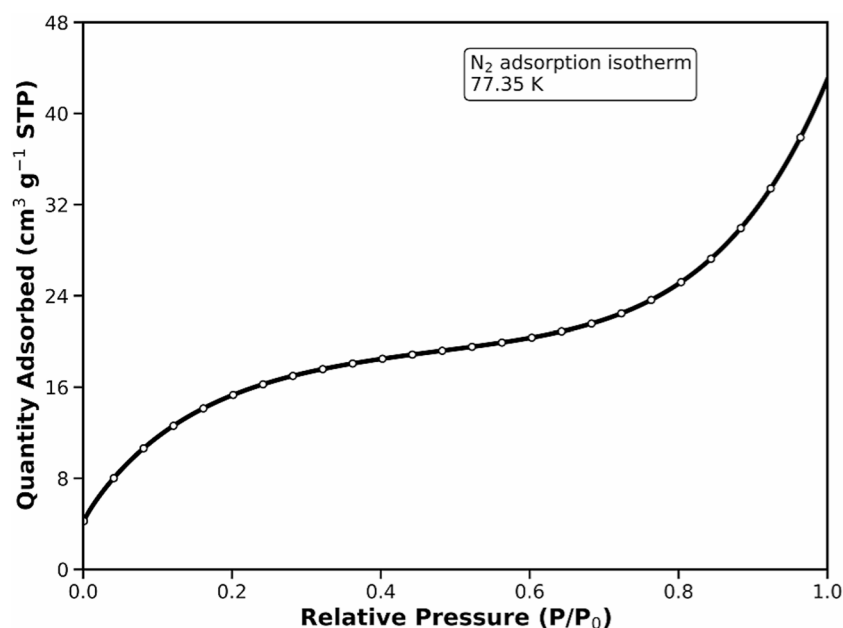
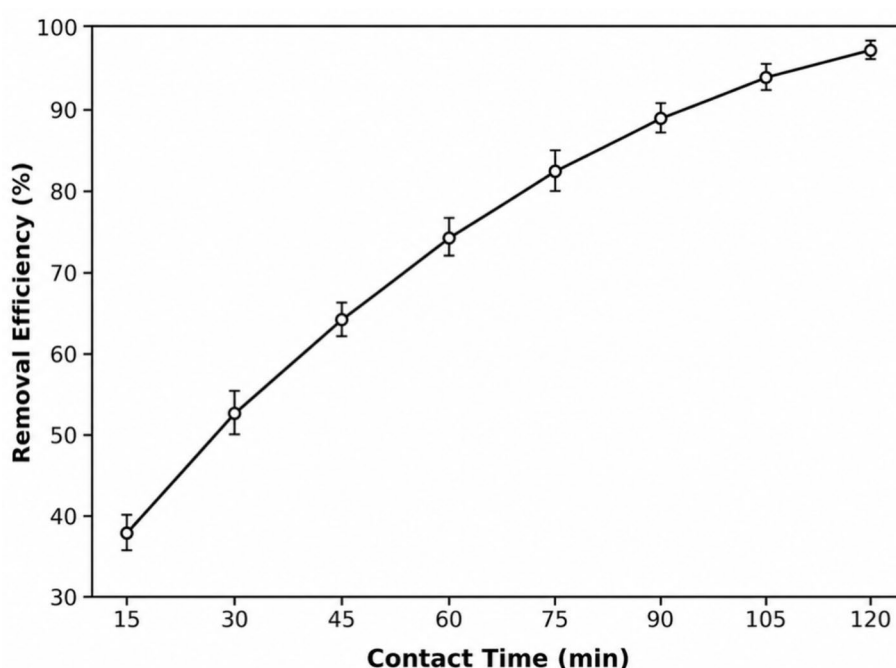


Fig. 5 Effect of contact time on the adsorption of B[ghi]P from VWW using EWB. Experimental conditions: initial B[ghi]P concentration = 40 mg L⁻¹, temperature = 60 °C, pH = 6, and adsorbent dose = 0.8 g.



Investigation of the adsorption parameters

The effects of the initial B[ghi]P concentration, adsorbent dosage, temperature, pH, and contact time were systematically investigated to determine the optimum conditions for the maximum adsorption of B[ghi]P onto EWB.

Effect of contact time

To evaluate the effect of contact time on the adsorption of B[ghi]P in VWW, we conducted a series of batch adsorption experiments with contact times ranging from 15 to 120 min. As shown in Fig. 5, the adsorption efficiency

increased with increasing contact time, reaching a maximum at 120 min. The rapid adsorption observed in the initial stage can be attributed to the presence of numerous active sites on the surface of EWB. As the adsorption process proceeds, these active sites are gradually occupied by B[ghi]P molecules, leading to a slower adsorption rate. At 120 min, the adsorption process reached equilibrium, indicating that most adsorption sites had been occupied, and further extending the contact time did not significantly increase the adsorption capacity.

Effect of temperature

As shown in Fig. 6, the adsorption of B[ghi]P from the VWW was evaluated in the temperature range of 30–60 °C. The adsorption efficiency increased with increasing temperature and reached its maximum at 60 °C. The increase in adsorption efficiency at higher temperatures may be attributed to the enhanced mobility of B[ghi]P molecules, which facilitates their migration from the solution to the adsorption sites accessible on the EWB. Higher temperatures may also enhance mass transfer between the adsorbate and the adsorbent. Additional experiments conducted at temperatures above 60 °C showed no further improvement in adsorption efficiency; therefore, 60 °C was chosen as the optimal temperature under the conditions studied. The absolute standard thermodynamic parameters (ΔG° , ΔH° , and ΔS°) were not calculated because a suitable dimensionless equilibrium constant was not established. Similar temperature-dependent adsorption behavior has been reported in previous studies⁶¹.

Effect of Adsorbent dose

Figure 7 illustrates the effect of adsorbent dosage on the removal efficiency of B[ghi]P from VWW using EWB. The removal efficiency increased with increasing EWB dosage from 0.2 g to 0.8 g, reaching a maximum at 0.8 g. This improvement can be attributed to the increased number of active adsorption sites and surface area resulting from higher EWB dosage, thus enhancing the interaction between B[ghi]P molecules and the EWB surface. At lower dosages, the limited number of active sites restricted the adsorption process, leading to lower removal efficiency. With increasing adsorbent dosage, more binding sites became available, promoting B[ghi]P adsorption. However, beyond the optimal dosage of 0.8 g, further improvements in removal efficiency were generally very limited, as most available adsorption sites were already occupied, leading to surface saturation.

Fig. 6 Effect of temperature on the removal efficiency of B[ghi]P from VWW using EWB. Experimental conditions: initial B[ghi]P concentration = 40 mg L⁻¹, contact time = 120 min, pH = 6, and adsorbent dose = 0.8 g.

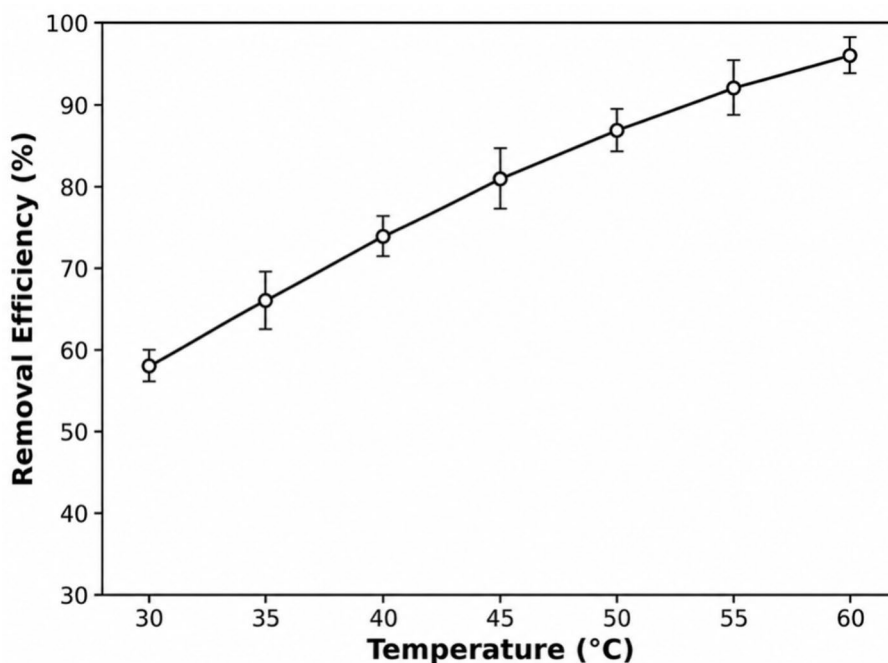


Fig. 7 Effect of adsorbent dosage on the removal efficiency of B[ghi]P from VWW using EWB. Experimental conditions: initial B[ghi]P concentration = 40 mg L⁻¹, contact time = 120 min, temperature = 60 °C, and pH = 6.

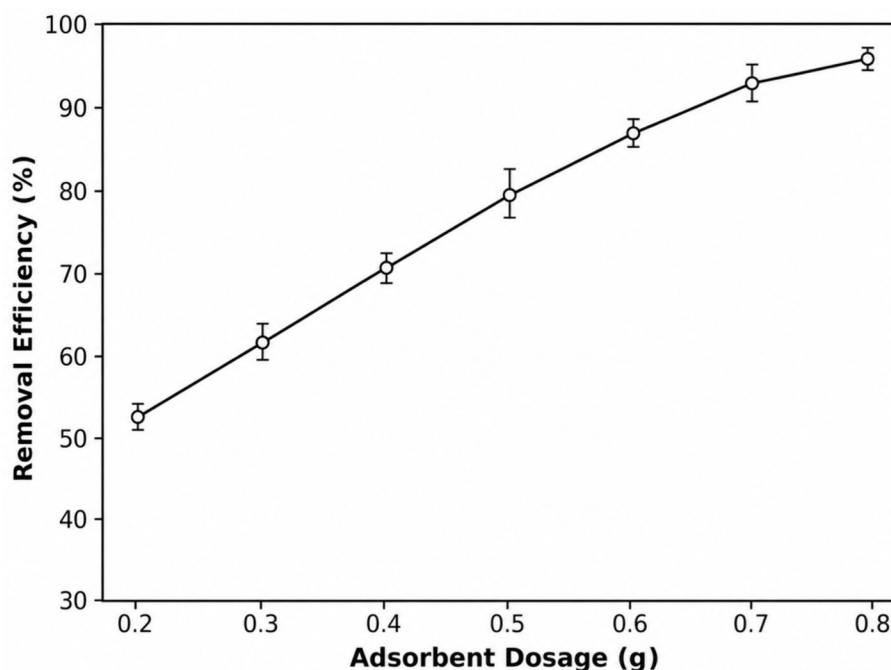
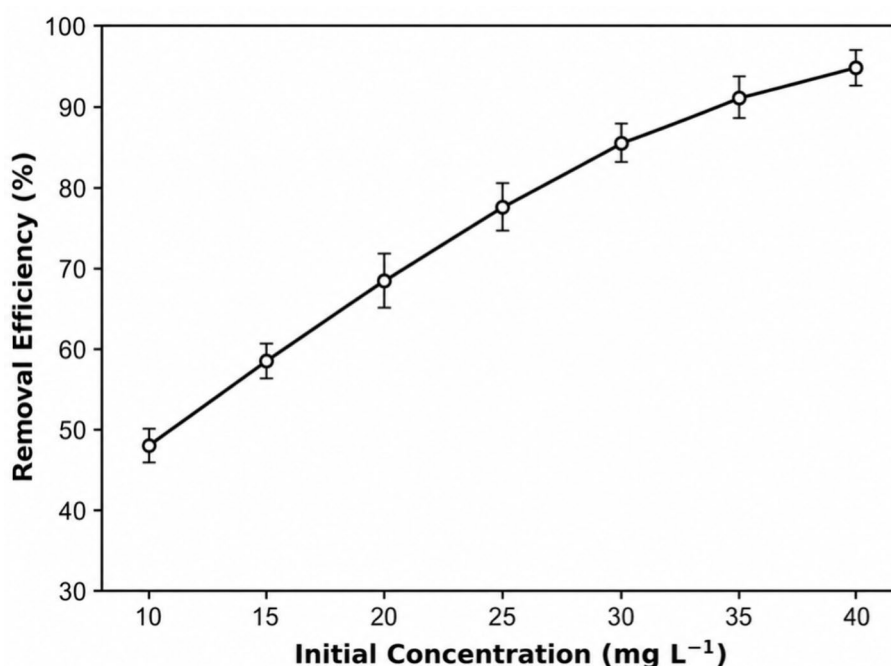


Fig. 8 Effect of the initial B[ghi]P concentration on the removal efficiency from VWW using EWB. Experimental conditions: contact time = 120 min, temperature = 60 °C, pH = 6, and adsorbent dose = 0.8 g.



Effect of initial B[ghi]P concentrations

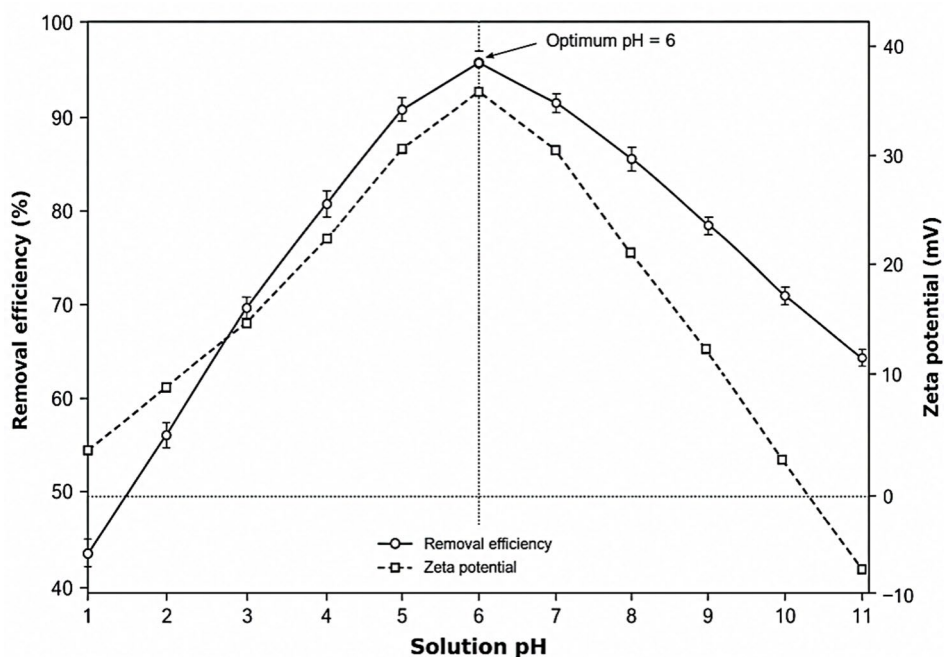
Figure 8 illustrates the effect of initial B[ghi]P concentration on the removal of B[ghi]P from VWW by EWB. The initial concentration varied from 10 mg L⁻¹ to 40 mg L⁻¹, and the results showed that the adsorption efficiency increased with increasing B[ghi]P concentration, reaching a maximum at 40 mg L⁻¹. At lower concentrations, the number of available B[ghi]P molecules was relatively small, resulting in low removal efficiency. With increasing initial concentration, the probability of B[ghi]P molecules interacting with the active adsorption sites on the EWB

surface also increased, thereby enhancing adsorption. However, at higher concentrations, the adsorption process gradually approached equilibrium as the available active sites on the EWB was gradually occupied, limiting further adsorption. These results indicate that the adsorption capacity of EWB is strongly influenced by the initial B[ghi]P concentration, consistent with previous findings reported in the literature⁶¹.

Effect of pH

The pH of the solution is one of the most important factors affecting the adsorption of B[ghi]P on EWB, as it influences the surface properties of the adsorbent and the interaction between the adsorbent and adsorbate⁶¹. Figure 9 shows the effect of solution pH (1–11) on the B[ghi]P removal rate and the corresponding EWB zeta potential. The initial pH of the solution was adjusted using 0.1 M HCl or 0.1 M NaOH. The removal rate increased with increasing pH, reaching a maximum at pH 6, and then gradually decreased with further increases in pH. At low pH values, protonation of oxygen-containing functional groups may alter the surface properties of EWB, thereby reducing adsorption efficiency. Since B[ghi]P is neutral and non-ionizable, direct competition between H^+ ions and B[ghi]P molecules is unlikely. When the pH increases to 6, the interaction between the EWB surface and B[ghi]P molecules becomes more favorable, resulting in the highest removal efficiency. The zeta potential of EWB exhibits a significant pH dependence, remaining positive over most of the studied pH range and reaching a maximum at approximately pH 6. The zeta potential gradually decreases with increasing pH, crossing zero at approximately pH 10.3. This zero point represents the isoelectric point and can approximately indicate the surface charge transition of the EWB. Maximum removal of B[ghi]P was observed at pH 6. Since B[ghi]P is a neutral and non-ionizable polycyclic aromatic hydrocarbon, its maximum adsorption at pH 6 cannot be directly attributed to electrostatic attraction; similarly, the decrease in adsorption at higher pH values cannot be entirely attributed to electrostatic repulsion. Instead, pH-dependent variations in the EWB surface properties may influence adsorption. Hydrophobic interactions, π - π interactions, pore filling, van der Waals forces, and interactions with surface functional groups may all contribute to the adsorption of B[ghi]P.

Fig. 9 Effect of solution pH on the removal efficiency of B[ghi]P from VWV and the zeta potential of EWB. Experimental conditions: initial B[ghi]P concentration = 40 mg L^{-1} , contact time = 120 min, temperature = 60°C , and adsorbent dose = 0.8 g.



Adsorption Kinetics

To better evaluate the adsorption kinetics of B[ghi]P on EWB, we fitted the experimental data using a pseudo-first-order kinetic model, a pseudo-second-order kinetic model, and an Elovich kinetic model. The corresponding linear regression plots are shown in Figures S1-S3, and the calculated kinetic parameters are summarized in Table 4. As shown in Figure S1, the pseudo-first-order model fits the experimental data well, with an R^2 of 0.967 and an RMSE of 0.120. The calculated equilibrium adsorption capacity ($q_{e,cal}$) is 2.316 mg g^{-1} , while the experimental value ($q_{e,exp}$) is 1.96 mg g^{-1} , with a relative error of approximately 18.2%. The calculated pseudo-first-order rate constant (k_1) is 0.028 min^{-1} . Among the three kinetic models evaluated, the pseudo-second-order model showed the best fit to the experimental adsorption data (Figure S2), the highest correlation coefficient ($R^2 = 0.9992$), and the lowest RMSE (0.038). The calculated equilibrium adsorption capacity ($q_{e,cal}$) was 2.230 mg g^{-1} , while the experimental value ($q_{e,exp}$) was 1.96 mg g^{-1} . The relative difference between the two was approximately 13.8%, indicating that the model-predicted adsorption capacity was in good agreement with the experimental value. The calculated pseudo-second-order rate constant (k_2) was 0.015 $\text{g mg}^{-1} \text{min}^{-1}$, and the initial adsorption rate (h) was 0.0737 $\text{mg g}^{-1} \text{min}^{-1}$. The h value represents the initial mass-normalized adsorption rate of B[ghi]P on EWB and provides an additional quantitative parameter for describing adsorption kinetics. Overall, the high R^2 value, low RMSE value, and good agreement between calculated and experimental values indicate that the pseudo-second-order model best describes adsorption kinetics among the models examined. However, the excellent statistical fit of the pseudo-second-order model should not be interpreted as conclusive evidence of chemisorption or other specific adsorption mechanisms, as kinetic model fit alone cannot reliably distinguish between physisorption and chemisorption processes.

Among the three models, the Elovich model (Figure S3) showed the worst statistical fit, with an R^2 of 0.776 and an RMSE of 0.294. The calculated Elovich parameters were $\alpha = 6.26 \text{ mg g}^{-1} \text{min}^{-1}$ and $\beta = 0.620 \text{ g mg}^{-1}$. Although the Elovich equation is often used as an empirical model for adsorption on heterogeneous surfaces, its relatively low R^2 and high RMSE indicate that, under the studied experimental conditions, its mathematical description of B[ghi]P adsorption on EWB is not satisfactory. Overall, the comparison of the three kinetic models shows that the pseudo-second-order model provides the best statistical description of B[ghi]P adsorption on EWB, with an R^2 of 0.9992 and an RMSE of 0.038. Furthermore, the equilibrium adsorption capacity calculated by the pseudo-second-order model was 2.230 mg g^{-1} , which is close to the experimental value of 1.96 mg g^{-1} , with a relative error of approximately 13.8%. In contrast, the pseudo-first-order model had a lower R^2 value (0.967), a higher RMSE value (0.120), and a larger deviation between the calculated and experimental values; while the Elovich model showed the worst statistical fit. However, the superior performance of the pseudo-second-order model mainly reflects its ability to effectively fit the observed kinetic data mathematically, rather than proving a specific adsorption mechanism. Further physicochemical and surface characterization evidence is needed to determine the main adsorption mechanism of B[ghi]P on EWB.

Table 4 Kinetic parameters obtained from the pseudo-first-order, pseudo-second-order, and Elovich models for the adsorption of B[ghi]P from VWV onto EWB.

Kinetic model	Regression equation	Slope	Intercept	k_1 (min^{-1})	k_2 (g mg^{-1} min^{-1})	α (mg g^{-1} min^{-1})	β (g mg^{-1})	$q_{e,cal}$ (mg g^{-1})	h (mg g^{-1} min^{-1})	R^2	RMSE
Pseudo-first-order	$y = -0.0122x + 0.3647$	-0.0122	0.3647	0.028	—	—	—	2.316	—	0.967	0.120
Pseudo-second-order	$y = 0.4484x + 13.5692$	0.4484	13.5692	—	0.015	—	—	2.230	0.0737	0.9992	0.038
Elovich	$y = 1.613x + 2.187$	1.613	2.187	—	—	6.26	0.620	—	—	0.776	0.294

Adsorption isotherms

This study evaluated the equilibrium adsorption behavior of B[ghi]P on an EWB using Langmuir, Freundlich, Temkin, and Harkins-Jura isothermal adsorption models. The corresponding linearization plots are shown in Figures S4-S7, and the calculated model parameters, correlation coefficients, and RMSE values are summarized in Table 5. Among the evaluated isothermal adsorption models, the Langmuir model showed the best fit to the equilibrium adsorption data, as evidenced by its correlation coefficient ($R^2 = 0.9992$) and root mean square error (RMSE=0.034). The calculated maximum adsorption capacity (q_m) and Langmuir constant (K_L) were 1.96 mg g^{-1} and 0.500 L mg^{-1} , respectively. These statistical parameters indicate that, among the evaluated models, the Langmuir equation provides the best mathematical description of the equilibrium adsorption data of B[ghi]P on an EWB under the studied experimental conditions. However, good statistical agreement with the Langmuir model should not be interpreted as definitive evidence of monolayer adsorption, energy-equivalent adsorption sites, or homogeneous adsorbent surfaces, as these mechanisms and surface features cannot be determined solely by isotherm fitting. Although the estimated maximum adsorption capacity of EWB for B[ghi]P (1.96 mg g^{-1}) is relatively limited compared to some previously reported PAHs adsorbents, its removal rate can still reach approximately 98% under optimized experimental conditions. Utilizing eucalyptus waste as a biochar precursor also provides an opportunity for the resource utilization of biomass waste. However, the adsorption capacity should be considered in conjunction with the energy required for pyrolysis and the chemical input required for HCl activation. Therefore, further optimization, cost assessment, nonlinear isotherm analysis, and scale-up studies using actual wastewater are needed to more comprehensively evaluate the practical application value of EWB in treating PAHs-contaminated wastewater.

The Freundlich model also showed a good fit, with $R^2 = 0.9539$ and RMSE=0.069. The calculated Freundlich adsorption strength (n) was 1.76, and the Freundlich constant (K_F) was 0.32. A value greater than 1 for n indicates favorable adsorption of B[ghi]P on EWB. However, compared to the Langmuir model, the Freundlich model had a lower R^2 and a higher RMSE, indicating a relatively weaker statistical descriptive ability for the experimental equilibrium data. These differences in goodness of fit were not used to infer the relative contributions of heterogeneous, multilayer, or monolayer adsorption mechanisms. The Temkin model showed a moderate fit to the experimental data, with $R^2 = 0.8935$ and RMSE=0.132. The empirical Temkin parameters B_T and K_T were 0.27 and 5.43 L mg^{-1} , respectively. These parameters were considered empirical fit parameters and did not assign them any interpretative significance to the heat of adsorption. The relatively low R^2 and high RMSE indicate that the Temkin model has a weaker statistical fit to the experimental data compared to the Langmuir and Freundlich models. Similarly, the Harkins-Jura model showed the worst fit to the adsorption data, with $R^2 = 0.5668$ and RMSE=0.286. The calculated A_H and B_H values were 0.024 and 0.70, respectively. The relatively low R^2 and high RMSE only indicate that the Harkins-Jura model provides a poor statistical representation of the equilibrium data under the studied conditions. These results alone are insufficient to confirm or rule out any specific adsorption mechanism. Overall, the statistical fit order was: Langmuir>Freundlich>Temkin>Harkins-Jura. Therefore, among the isothermal models evaluated in this study, the Langmuir model was determined to be the best fit. This finding can only be explained from the perspective of statistical goodness of fit and does not constitute evidence

Table 5 B[ghi]P adsorption isotherm parameters with their corresponding correlation coefficients.

Isotherm Model	Parameter	R^2 (RMSE)
Langmuir	$q_m = 1.96 \text{ mg g}^{-1}$	0.9992 (0.034)
	$K_L = 0.500 \text{ L mg}^{-1}$	
Freundlich	$N = 1.76$	0.9539 (0.069)
	$K_F = 0.32$	
Temkin	$K_T = 5.43 \text{ L mg}^{-1}$	0.8935 (0.132)
	$B = 0.27$	
Harkins Jura	$A_H = 0.024$	0.5668 (0.286)
	$B_H = 0.70$	

of monolayer adsorption, homogeneous surfaces, or any specific adsorption mechanism. Nevertheless, the high removal efficiency observed under optimized experimental conditions still suggests the potential of EWB in removing B[ghi]P from VWW.

Adsorption mechanism of B[ghi]P

The adsorption of B[ghi]P on EWB is influenced by the adsorbent surface properties, the physicochemical properties of B[ghi]P, and solution conditions. The Langmuir model provides the best statistical description of the equilibrium data; however, a good fit alone cannot definitively prove monolayer adsorption or the homogeneity of the adsorbent surface. The observed changes in FTIR spectra after adsorption indicate that the interaction between EWB and B[ghi]P involves surface functional groups. Oxygen-containing functional groups on the EWB surface may contribute to the formation of weak intermolecular interactions and provide accessible adsorption sites. Furthermore, the aromatic structure of B[ghi]P can interact with the carbonaceous and aromatic domains of EWB through π - π interactions. Because B[ghi]P is highly hydrophobic, neutral, and non-ionizable, its hydrophobic interactions with carbonaceous surfaces are expected to significantly contribute to adsorption, while electrostatic interactions may have a relatively smaller effect. Pore filling and van der Waals dispersion forces may further promote the retention of B[ghi]P in the porous structure of EWB. HCl activation may alter the EWB surface by removing mineral impurities, exposing additional adsorption sites, and changing the accessibility of oxygen-containing surface functional groups, thereby affecting the interaction between B[ghi]P and the adsorbent. In summary, the adsorption of B[ghi]P on EWB is likely influenced by a combination of π - π interactions, hydrophobic interactions, pore filling, van der Waals forces, and interactions with surface functional groups, rather than a single mechanism. Therefore, FTIR observations are considered corroborating evidence of these interactions, rather than definitive evidence of a specific adsorption mechanism.

Comparison of adsorption potential of different adsorbents

Table 6 compares the adsorption performance of EWB with other reported adsorbents for the removal of B[ghi]P. In this study, EWB achieved a removal rate of approximately 98% under optimized experimental conditions. However, caution is needed when directly comparing different adsorbents because of the varying experimental conditions used in reported studies. The use of eucalyptus waste as a raw material and the simple preparation process are potential advantages of EWB. However, this study did not include a detailed economic or environmental assessment. Furthermore, the optimal temperature of 60 °C may increase energy requirements in practical applications. Therefore, further research is required to evaluate the economic and environmental feasibility of EWB for large-scale wastewater treatment.

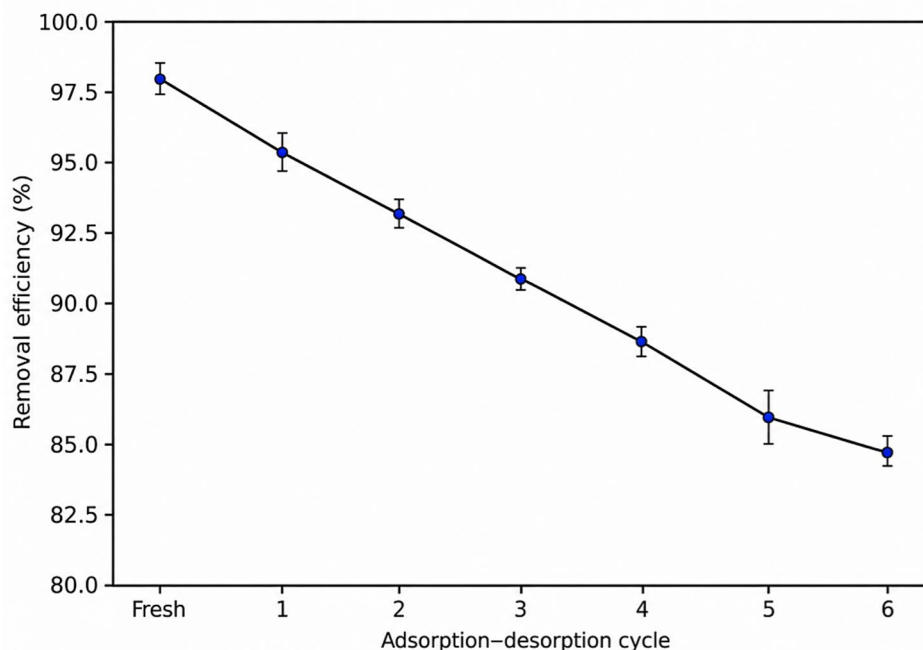
Reusability or regeneration

To assess the reusability of the EWB, six consecutive adsorption–desorption cycles were conducted (Fig. 10). After each adsorption step, the used EWB was separated and treated with 0.1 M HCl under stirring for 30 min to desorb B[ghi]P. The regenerated EWB was then rinsed with deionized water, dried at 60 °C, and reused under the optimized adsorption conditions. Desorption efficiency was evaluated based on the amount of B[ghi]P released

Table 6 Comparison of adsorption performance of various adsorbents for the removal of B[ghi]P.

Materials	B[ghi]P removal efficiency (%)	Isotherm model	References
Composite adsorbent (wPS/C/Bt)	80.36	Langmuir	⁴
Composite adsorbent (wPET/C/Bt)	81.34	Langmuir	⁴
Waste newspaper-derived activated carbon	~91%	Langmuir	⁶⁷
Wood ashes	>99	Freundlich	⁶⁸
EWB	98	Langmuir	This study

Fig. 10 Regeneration and reusability of EWB for the adsorption of B[ghi]P over six consecutive adsorption-desorption cycles. Experimental conditions: initial B[ghi]P concentration = 40 mg L^{-1} , contact time = 120 min, temperature = 60°C , pH = 6, and adsorbent dose = 0.8 g.



into the desorption solution compared with the amount originally adsorbed on the EWB. The removal efficiency gradually decreased from about 98% with fresh EWB to 95.3%, 93.1%, 90.9%, 88.6%, 86.0%, and 84.9% after the first to sixth cycles, respectively. This gradual decline could be due to incomplete desorption of B[ghi]P and partial blocking or loss of available adsorption sites during repeated regeneration. Nevertheless, even after six cycles, the EWB still retained about 84.9% of its removal efficiency, indicating good reusability under the studied conditions. These results suggest that regenerated EWB has potential for repeated removal of B[ghi]P from VWV; however, more long-term regeneration and real-scale studies are needed to evaluate its practical feasibility.

Limitations and future work

This study has several limitations. Adsorption experiments were limited to the adsorption of B[ghi]P under controlled conditions, and the adsorption capacity of EWB was relatively low (1.96 mg g^{-1}) compared to some reported adsorbents. Furthermore, detailed physicochemical characterization of the wastewater was not performed. For B[ghi]P in the VWV matrix, matrix-specific recoveries and interferences, as well as method-specific limits of detection (LOD) and quantitation (LOQ), were not investigated. Therefore, potential matrix effects and related analytical uncertainties cannot be completely ruled out, and the reported removal efficiencies (including approximately 98% removal achieved under optimized conditions) should be interpreted with caution. Kinetic and equilibrium analyses were primarily based on a linearized model, and the optimal temperature of 60°C may affect actual energy consumption. In addition, a detailed economic and environmental assessment is beyond the scope of this study. Future research should focus on the simultaneous removal of multiple preferentially controlled PAHs from well-defined real-world wastewater, improving the adsorption capacity of EWB, applying nonlinear kinetics and isothermal models, and conducting comprehensive analytical validation, including assessments of recovery, interference, LOD, and LOQ, to confirm the quantitative accuracy of observed removal performance. Further research should also evaluate regeneration, cost, energy consumption, and environmental impact. Pilot-scale and large-scale studies under actual operating conditions are needed to assess the practical feasibility of EWB in wastewater treatment.

Conclusion

This study demonstrates the potential of EWB in the effective removal of B[ghi]P from VWW. Under optimized experimental conditions (initial concentration = 40 mg L⁻¹, contact time = 120 min, temperature = 60 °C, pH = 6, and adsorbent dosage = 0.8 g), a removal rate of approximately 98% for B[ghi]P was achieved. Adsorption results indicate that EWB can effectively interact with B[ghi]P, and the observed adsorption behavior may be related to π - π interactions, hydrophobic interactions, pore filling, van der Waals forces, and interactions with surface functional groups. Furthermore, EWB exhibits good regeneration performance, maintaining a removal rate of over 84% after six adsorption-desorption cycles, indicating its potential for reusability. In conclusion, these results demonstrate that eucalyptus waste can be transformed into an effective adsorbent for B[ghi]P removal, providing a valuable basis for further evaluation and development of EWB applications in wastewater treatment.

Supplementary Information The online version contains supplementary material available at <https://doi.org/10.1038/s41598-026-76208-2>.

Author contributions The author conceived and designed the study, collected and analyzed the data, wrote the manuscript, and approved the final version.

Funding The authors extend their appreciation to the Deanship of Research and Graduate Studies at King Khalid University for funding this work through Large Research Project under grant number RGP.2 /44 /47.

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

Declarations

Competing Interest The authors declare no competing interests.

Open Access This article is licensed under a Creative Commons Attribution-NonCommercial-NoDerivatives 4.0 International License, which permits any non-commercial use, sharing, distribution and reproduction in any medium or format, as long as you give appropriate credit to the original author(s) and the source, provide a link to the Creative Commons licence, and indicate if you modified the licensed material. You do not have permission under this licence to share adapted material derived from this article or parts of it. The images or other third party material in this article are included in the article's Creative Commons licence, unless indicated otherwise in a credit line to the material. If material is not included in the article's Creative Commons licence and your intended use is not permitted by statutory regulation or exceeds the permitted use, you will need to obtain permission directly from the copyright holder. To view a copy of this licence, visit <http://creativecommons.org/licenses/by-nc-nd/4.0/>.

References

1. Adeniji, A. O., Okoh, O. O. & Okoh, A. I. Analytical methods for polycyclic aromatic hydrocarbons and their global trend of distribution. In: Recent insights in petroleum science and engineering. p. 393. (2018). <https://doi.org/10.5772/intechopen.71163>
2. El-Bouhy, Z., Reda, R. M., Mohamed, F. A., Elashhab, M. W. & Abdel Rahman, A. N. Polycyclic aromatic hydrocarbons (PAHs) pollution approaches in aquatic ecosystems: perils and remedies using green technologies. *Zagazig Vet. J.* **52** (1), 49–87. <https://doi.org/10.21608/zvjz.2024.239217.1221> (2024).
3. Kumar, B., Verma, V. K., Sharma, C. S. & Akolkar, A. B. Estimation of toxicity equivalency and probabilistic health risk on lifetime daily intake of polycyclic aromatic hydrocarbons from urban residential soils. *Hum. Ecol. Risk Assess.* **21** (2), 434–444. <https://doi.org/10.1080/10807039.2014.921530> (2015).
4. Ilyas, M., Ahmad, W. & Khan, H. Utilization of composite adsorbents for polycyclic aromatic hydrocarbons adsorption from vehicle wash wastewater and aqueous solutions. *Desalin. Water Treat.* **256**, 194–216. <https://doi.org/10.5004/dwt.2022.28413> (2022).

5. Berrios-Rolón, P. J., Cotto, M. C. & Márquez, F. Polycyclic aromatic hydrocarbons (PAHs) in freshwater systems: a comprehensive review of sources, distribution, and ecotoxicological impacts. *Toxics* **13** (4), 321. <https://doi.org/10.20944/preprints202404.1983.v1> (2025).
6. Patel, A. B., Shaikh, S., Jain, K. R., Desai, C. & Madamwar, D. Polycyclic aromatic hydrocarbons: sources, toxicity, and remediation approaches. *Front. Microbiol.* **11**, 562813. <https://doi.org/10.3389/fmicb.2020.562813> (2020).
7. World Health Organization. Human health effects of polycyclic aromatic hydrocarbons as ambient air pollutants: report of the Working Group on Polycyclic Aromatic Hydrocarbons of the Joint Task Force on the Health Aspects of Air Pollution. Copenhagen: WHO Regional Office for Europe; (2021). Available from: <https://iris.who.int/handle/10665/350636> (accessed 29 January 2025).
8. Wild, S. R. & Jones, K. C. Polynuclear aromatic hydrocarbons in the United Kingdom environment: a preliminary source inventory and budget. *Environ. Pollut.* **88** (1), 91–108. [https://doi.org/10.1016/0269-7491\(95\)91052-M](https://doi.org/10.1016/0269-7491(95)91052-M) (1995).
9. Mirzaee, E. & Sartaj, M. Activated carbon-based magnetic composite as an adsorbent for removal of polycyclic aromatic hydrocarbons from aqueous phase: characterization, adsorption kinetics and isotherm studies. *J. Hazard. Mater. Adv.* **6**, 100083. <https://doi.org/10.1016/j.hazadv.2022.100083> (2022).
10. Eisler, R. *Polycyclic aromatic hydrocarbon hazards to fish, wildlife, and invertebrates: a synoptic review* (Fish and Wildlife Service, US Department of the Interior, 1987).
11. Ferreira Azevedo, L. et al. High molecular weight polycyclic aromatic hydrocarbon (HMW-PAH) isomers: unveiling distinct toxic effects from cytotoxicity to oxidative stress-induced DNA damage. *Arch. Toxicol.* **99** (2), 679–687. <https://doi.org/10.1007/s00204-024-03917-w> (2025).
12. US Environmental Protection Agency. *Polycyclic aromatic hydrocarbons (PAHs)-EPA fact sheet* (National Center for Environmental Assessment, Office of Research and Development, 2008).
13. Sun, K. et al. A review of human and animals exposure to polycyclic aromatic hydrocarbons: health risk and adverse effects, photo-induced toxicity and regulating effect of microplastics. *Sci. Total Environ.* **773**, 145403. <https://doi.org/10.1016/j.scitotenv.2021.145403> (2021).
14. Masih, J., Singhvi, R., Kumar, K., Jain, V. K. & Taneja, A. Seasonal variation and sources of polycyclic aromatic hydrocarbons (PAHs) in indoor and outdoor air in a semi-arid tract of northern India. *Aerosol Air Qual. Res.* **12** (4), 515–525. <https://doi.org/10.4209/aaqr.2011.11.0192> (2012).
15. Nzila, A. Biodegradation of high-molecular-weight polycyclic aromatic hydrocarbons under anaerobic conditions: overview of studies, proposed pathways and future perspectives. *Environ. Pollut.* **239**, 788–802. <https://doi.org/10.1016/j.envpol.2018.04.074> (2018).
16. Anyahara, J. N. Effects of polycyclic aromatic hydrocarbons (PAHs) on the environment: a systematic review. *Int. J. Adv. Acad. Res.* **7** (3), 12. <https://doi.org/10.46654/ij.24889849.e7303> (2021).
17. Feng, Y., Li, Z. & Li, W. Polycyclic aromatic hydrocarbons (PAHs): environmental persistence and human health risks. *Nat. Prod. Commun.* **20** (1), 1934578X241311451. <https://doi.org/10.1177/1934578X241311451> (2025).
18. Abdel-Shafy, H. I. & Mansour, M. S. A review on polycyclic aromatic hydrocarbons: source, environmental impact, effect on human health and remediation. *Egypt. J. Pet.* **25** (1), 107–123. <https://doi.org/10.1016/j.ejpe.2015.03.011> (2016).
19. Klaassen, C. D. & Amdur, M. O. (eds) *Casarett and Doull's toxicology: the basic science of poisons* (McGraw-Hill, 2013).
20. Hashemi, N., Pourbabae, A. A., Shariati, S. & Yadzanfar, N. Rapid phenanthrene biodegradation in highly calcareous saline sodic soil using an artificial halophile bacterial consortium. *Int. J. Environ. Sci. Technol.* **22** (3), 1817–1828. <https://doi.org/10.1007/s13762-024-05913-0> (2025).
21. Lamichhane, S., Krishna, K. B. & Sarukkalige, R. Polycyclic aromatic hydrocarbons (PAHs) removal by sorption: a review. *Chemosphere* **148**, 336–353. <https://doi.org/10.1016/j.chemosphere.2016.01.036> (2016).
22. Devendrapandi, G. et al. Innovative remediation strategies for persistent organic pollutants in soil and water: a comprehensive review. *Environ. Res.* **249**, 118404. <https://doi.org/10.1016/j.envres.2024.118404> (2024).
23. Shih, K. L. & Lederberg, J. Chloramine mutagenesis in *Bacillus subtilis*. *Science* **192** (4244), 1141–1143. <https://doi.org/10.1126/science.818709> (1976).
24. Sharma, A., Ahmad, J. & Flora, S. J. Application of advanced oxidation processes and toxicity assessment of transformation products. *Environ. Res.* **167**, 223–233. <https://doi.org/10.1016/j.envres.2018.07.010> (2018).
25. Yang, Y. et al. Ecological risk assessment methods for oxidative by-products in the oxidation degradation process of emerging pollutants: a review. *Sci. Total Environ.* **950**, 175401. <https://doi.org/10.1016/j.scitotenv.2024.175401> (2024).
26. Ilyas, M., Ahmad, W., Khan, H. & Ahmad, I. Application of composite adsorbents prepared from waste PS and PET for removal of Cr and Cu ions from wastewater. *Desalin. Water Treat.* **171**, 144–157. <https://doi.org/10.5004/dwt.2019.24764> (2019).
27. Ren, Y. S. et al. Treatment of dyes contaminated water using biochar derived from eucalyptus wood waste. *ES Mater. Manuf.* **25**, 1236. <https://doi.org/10.30919/esmm1236> (2024).
28. Ren, Y. S. et al. Sustainable adsorbent: activated carbon from waste styrofoam for efficient aluminum removal. *Microsc. Res. Tech.* **88** (4), 1044–1054. <https://doi.org/10.1002/jemt.24770> (2025).

29. Din, S. U. et al. Heavy metal contamination in agricultural soils: an environmental and food safety perspective. *Environ. Monit. Assess.* **198** (5), 419. <https://doi.org/10.1007/s10661-026-15311-3> (2026).
30. Ren, Y. S. et al. Eucalyptus wood waste biochar as an eco-friendly adsorbent for arsenic detoxification in wastewater. *Nat. Prod. Res.* 1–6. <https://doi.org/10.1080/14786419.2026.2649923> (2026).
31. Sajjad, M. et al. Synergistic effects of thermal stress and microplastic pollution on seed germination under agricultural soil conditions. *CLEAN. Soil. Air Water.* **54** (3), e70156. <https://doi.org/10.1002/clen.70156> (2026).
32. Malik, S. et al. Fabrication of an electrochemical sensor based on magnetic molecularly imprinted polymer for detection of sunset yellow dye. *Sci. Rep.* **16**, 6369. <https://doi.org/10.1038/s41598-026-38556-x> (2026).
33. dos Santos, L. G. et al. Glycerol-based modified carbons as adsorbents for efficient and sustainable nitrate removal from wastewater. *Energy Environ. Sustain.* **1** (3), 100031. <https://doi.org/10.1016/j.eesus.2025.100031> (2025).
34. Li, Y. et al. Efficient degradation of norfloxacin by synergistic activation of PMS with a three-dimensional electrocatalytic system based on Cu-MOF. *Sep. Purif. Technol.* **356**, 129945. <https://doi.org/10.1016/j.seppur.2024.129945> (2025).
35. Lu, J. Y. et al. Facile microwave-assisted synthesis of Sb₂O₃-CuO nanocomposites for catalytic degradation of p-nitrophenol. *J. Mol. Liq.* **409**, 125503. <https://doi.org/10.1016/j.molliq.2024.125503> (2024).
36. Qin, X., Cui, H. & Zhou, Q. Physisorption behaviors of organochlorine pesticides on the InP₃ monolayer from theoretical insight. *ACS Omega.* **8** (35), 32168–32175. <https://doi.org/10.1021/acsomega.3c04665> (2023).
37. Tong, X. et al. Rapid determination of 14 odorous compounds in drinking water using gas chromatography-mass spectrometry coupled with headspace solid-phase microextraction pretreatment. *Anal. Methods.* **14** (48), 5076–5080. <https://doi.org/10.1039/D2AY01784A> (2022).
38. Zhang, Y. et al. An ionic liquid-modified PVC nanofiber facilitates gold recovery from wastewater by a light-enhancing effect. *Chem. Eng. J.* **505**, 159419. <https://doi.org/10.1016/j.cej.2025.159419> (2025).
39. Shi, L., Li, A., Xu, Y., Yang, H. & Yang, G. A novel ionic-liquid-supported polythieno[3,2-b]thiophene coating for headspace solid-phase microextraction of phenolic compounds from environmental water via gas chromatography-mass spectrometry. *Microchim. Acta.* **192** (5). <https://doi.org/10.1007/s00604-025-07157-2> (2025).
40. Liu, Y., Gao, C., Liu, L., Li, Y. & Guo, X. A photovoltaic powered ocean-based electrochemical system produces highly oxidizing active substances for simultaneous removal of antibiotics and heavy metals from mariculture wastewater. *Water Res.* **286**, 124177. <https://doi.org/10.1016/j.watres.2025.124177> (2025).
41. Jiang, F. et al. Ceramsite catalyst derived from printed circuit board sludge for catalytic ozonation treatment of coking wastewater: performance and mechanism. *J. Hazard. Mater.* 139613. <https://doi.org/10.1016/j.jhazmat.2025.139613> (2025).
42. Ganesan, R., Latha, A., Kavitha, P. & Venkatesan, G. Experimental investigation of wastewater by using novel Borassus flabellifer fiber and Cocos nucifera fiber. *Asian J. Water Environ. Pollut.* **21** (6), 39–47. <https://doi.org/10.3233/AJW240071> (2024).
43. Alshididi, S. S., Al-Mashhadani, M. K. & Abed, I. J. Wastewater treatment by the cyanobacterium species: Synechococcus elongatus as biosorption material for Pb, Cr and Ni heavy metals. *Asian J. Water Environ. Pollut.* **21** (6), 111–118. <https://doi.org/10.3233/AJW240078> (2024).
44. Liu, R. et al. Innovative amorphous calcium carbonate for superior anionic dye adsorption towards near-zero discharge. *Sep. Purif. Technol.* **361**, 131349. <https://doi.org/10.1016/j.seppur.2024.131349> (2025).
45. El Messaoudi, N. et al. Fabrication a sustainable adsorbent nanocellulose-mesoporous hectorite bead for methylene blue adsorption. *Case Stud. Chem. Environ. Eng.* **10**, 100850. <https://doi.org/10.1016/j.csee.2024.100850> (2024).
46. Şenol, Z. M., Arslanoğlu, H., Keskin, Z. S., Mehmeti, V. & El Messaoudi, N. Biosorption of rhodamine B and sunset yellow dyes on cross-linked chitosan-alginate biocomposite beads: experimental and theoretical studies. *Int. J. Biol. Macromol.* **298**, 139264. <https://doi.org/10.1016/j.ijbiomac.2024.139264> (2025).
47. Aasli, B. et al. Synthesis of urea-formaldehyde resin@chitosan composite for the removal of Congo red from an aqueous solution via adsorption: Box-Behnken design optimization. *Int. J. Biol. Macromol.* **315**, 144648. <https://doi.org/10.1016/j.ijbiomac.2025.144648> (2025).
48. Aasli, B. et al. A comprehensive review of the use of urea-formaldehyde resin composites for the adsorption of organic and inorganic pollutants from wastewater. *Nano-Struct Nano-Objects.* **43**, 101495. <https://doi.org/10.1016/j.nanoso.2025.101495> (2025).
49. Ghazoui, M. et al. Multi-target adsorption of dyes and antibiotics on biomimetic carbon foams: mechanisms, column design, and integrated treatment of mixed industrial effluents. *Sep. Purif. Technol.* :136424. (2025).
50. Ghazoui, M. et al. Next-generation biochar adsorbents from waste biomass: functionalization strategies and real-world water treatment applications. *Eng. Environ.* **20** (1), 12 (2026).
51. Raif, F. E. et al. Adsorption of methylene blue onto raw wastes of argan, date, and olive: study of kinetics and isotherms. *Environ. Eng. Sci.* **42** (4), 166–176 (2025).
52. El Messaoudi, N. et al. A comprehensive review on designing nanocomposite adsorbents for efficient removal of 4-nitrophenol from water. *Nano-Struct Nano-Objects.* **40**, 101326. <https://doi.org/10.1016/j.nanoso.2024.101326> (2024).

53. El Messaoudi, N. et al. A critical review of Allura Red removal from water: advancements in adsorption and photocatalytic degradation technologies, and future perspectives. *J. Environ. Chem. Eng.* **12** (6), 114843. <https://doi.org/10.1016/j.jece.2024.114843> (2024).
54. El Messaoudi, N. et al. Advancements in adsorption and photocatalytic degradation technologies of brilliant green from water: current status, challenges, and future prospects. *Mater. Today Chem.* **42**, 102399. <https://doi.org/10.1016/j.mtchem.2024.102399> (2024).
55. El Messaoudi, N. et al. Metal and metal oxide nanoparticles combined with ionic liquids and deep eutectic solvents and their practices in drug extraction and environmental processes: a review. *J. Mol. Liq.* **422**, 126955. <https://doi.org/10.1016/j.molliq.2025.126955> (2025).
56. Miyah, Y. et al. CWPO mechanism for toxic dye degradation in the presence of Cu@FbHAp catalyst: DFT study, performance analysis, response surface methodology, regeneration, and cost estimation. *Results Chem.* **13**, 102038. <https://doi.org/10.1016/j.rechem.2025.102038> (2025).
57. Cigeroğlu, Z. et al. Recent advances in the removal of Sunset Yellow dye from wastewater: a review. *Sustain. Mater. Technol.* **42**, e01187. <https://doi.org/10.1016/j.susmat.2024.e01187> (2024).
58. Kobayashi, T. & Sumida, H. Effects of humic acids on the sorption and bioavailability of pyrene and 1,2-dihydroxynaphthalene. *Soil. Sci. Plant. Nutr.* **61** (Suppl 1), 113–122. <https://doi.org/10.1080/00380768.2015.1024581> (2015).
59. Wang, J., Wang, C., Huang, Q., Ding, F. & He, X. Adsorption of PAHs on the sediments from the Yellow River Delta as a function of particle size and salinity. *Soil. Sediment. Contam.* **24** (2), 103–115. <https://doi.org/10.1080/15320383.2014.920292> (2015).
60. Pathak, S., Sakhiya, A. K., Anand, A., Pant, K. K. & Kaushal, P. A state-of-the-art review of various adsorption media employed for the removal of toxic polycyclic aromatic hydrocarbons (PAHs): an approach towards a cleaner environment. *J. Water Process. Eng.* **47**, 102674. <https://doi.org/10.1016/j.jwpe.2022.102674> (2022).
61. Ilyas, M. et al. Removal of anthracene from vehicle-wash wastewater through adsorption using eucalyptus wood waste-derived biochar. *Desalin. Water Treat.* **317**, 100115. <https://doi.org/10.1016/j.dwt.2024.100115> (2024).
62. Chen, S. et al. Equilibrium and kinetic studies of methyl orange and methyl violet adsorption on activated carbon derived from *Phragmites australis*. *Desalination* **252** (1–3), 149–156. <https://doi.org/10.1016/j.desal.2009.10.010> (2010).
63. Ho, Y. S. & McKay, G. Pseudo-second order model for sorption processes. *Process. Biochem.* **34** (5), 451–465 (1999).
64. Yao, Y. et al. Characterization and environmental applications of clay-biochar composites. *Chem. Eng. J.* **242**, 136–143. <https://doi.org/10.1016/j.cej.2013.12.062> (2014).
65. Lawrinenko, M. & Laird, D. A. Anion exchange capacity of biochar. *Green. Chem.* **17** (9), 4628–4636. <https://doi.org/10.1039/C5GC00828J> (2015).
66. Chaudhary, S. et al. Assessment of hydrochar and porous carbon from *Tectona grandis* seeds for removal of acridine dyes. *Molecules* **30** (19), 3989 (2025).
67. Nahar, A. et al. Waste newspaper driven activated carbon to remove polycyclic aromatic hydrocarbon from wastewater. *Heliyon* **9**, (7). (2023).
68. Pérez-Gregorio, M. R., García-Falcón, M. S., Martínez-Carballo, E. & Simal-Gándara, J. Removal of polycyclic aromatic hydrocarbons from organic solvents by ashes wastes. *J. Hazard. Mater.* **178** (1–3), 273–281. <https://doi.org/10.1016/j.jhazmat.2010.01.073> (2010).

Publisher's note Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.

Authors and Affiliations

Mohd Abul Hasan¹ 

✉ Mohd Abul Hasan
mohad@kku.edu.sa

¹ Department of Civil Engineering, College of Engineering, King Khalid University, Abha, Kingdom of Saudi Arabia