

Sulphate removal from synthetic acid mine drainage by iron-impregnated coconut shell biochar: role of iron retention and surface accessibility

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Abstract

Acid mine drainage (AMD) carries high sulphate loads that lime neutralisation removes poorly. Iron-modified biochars are promising anion adsorbents, but the roles of iron loading and iron accessibility are rarely separated. In this study, coconut shell biochar produced at 650°C was impregnated with $\text{FeCl}_3 \cdot 6\text{H}_2\text{O}$ at precursor-to-biochar mass ratios of 0.5, 0.7 and 1.0 g g^{-1} (FeBC-0.5, FeBC-0.7 and FeBC-1.0). The materials were characterised by AAS, XRD and FTIR and tested for sulphate removal from synthetic AMD (initial sulphate 1958.6 mg L^{-1} ; $\text{pH} \approx 2.5$). The best material was then examined in a randomised factorial design with dose (2–6 g L^{-1}) and contact time (30–120 min) as factors. Iron retention levelled off above a ratio of 0.7, so FeBC-0.7 and FeBC-1.0 held the same total iron (13.90 and 13.88 mg g^{-1}). However, FeBC-0.7 showed the strongest Fe–O/Fe–OH absorption and removed more sulphate (90.2% versus 83.7% at 2 g L^{-1}). No crystalline iron oxide was detected, which indicates finely dispersed, X-ray-amorphous iron (oxyhydr)oxide. At 4 g L^{-1} , FeBC-0.7 removed $90.93 \pm 1.75\%$ of sulphate, compared with $79.54 \pm 1.08\%$ for the pristine biochar, and reached $92.55 \pm 0.32\%$ at 60 min. Uptake was fast and followed pseudo-second-order kinetics. The Freundlich model described the equilibrium data better than the Langmuir model. Iron accessibility, and not iron loading alone, therefore controls the performance of Fe-biochar, and precursor dosed beyond the retention limit is wasted. FeBC-0.7 is a promising low-cost first stage for sulphate attenuation in AMD treatment trains.

1. Introduction

Acid mine drainage (AMD) is one of the most serious and persistent environmental problems associated with coal and metal mining. It forms when sulphide minerals, principally pyrite (FeS_2), are exposed to air and water. The resulting effluent has a very low pH, a high sulphate (SO_4^{2-}) concentration and elevated levels of dissolved metals such as Fe, Mn, Zn and Cu (Cheng et al. 2026; Gao et al. 2026; Makwinja et al. 2026). Lime neutralisation effectively raises the pH and precipitates metals. However, it removes sulphate poorly, generates large volumes of sludge and incurs high operating costs (Skousen et al. 2019). Sulphate, therefore, remains a critical residual contaminant after neutralisation. This has prompted recent interest in technologies that move beyond simple neutralisation towards sulphate removal and resource recovery (Jegatheesan et al. 2026; Wang et al. 2026). Biological routes based on sulphate-reducing bacteria have been widely studied. Their efficiency, however, depends strongly on operating conditions and is difficult to sustain at the low pH of AMD (Srinivasan et al. 2025). Low-cost, robust materials that can reduce the sulphate load in AMD are therefore needed.

Biomass-derived biochar has emerged as a promising adsorbent owing to its surface area, porosity and oxygen-rich surface functional groups (Cao et al. 2026; Setyawan et al. 2024). Biochar produced from agricultural residues is particularly attractive in tropical countries, where feedstocks such as coconut shells are abundant, inexpensive and renewable (Salazar et al. 2026; Sam and Ragavendran 2026; Sarkar et al. 2026). Pristine coconut shell biochar has already been used to lower sulphate and cadmium in AMD (Fahrudin et al. 2024), and mineral slags have been screened as sulphate adsorbents for the same

purpose (Zheng et al. 2020). Nevertheless, pristine biochar generally shows a limited affinity for anions such as sulphate. Its carbon surface carries few positively charged sites, so the electrostatic attraction of anions is weak (Hinsene et al. 2026; Kapoor and Zdart 2024; Zhao et al. 2026). Overcoming this limitation requires chemical modification of the biochar surface, for example, by surface functionalisation or by the deposition of metal and metal oxide phases. Both routes have enhanced the uptake of charged pollutants on waste-derived biochar (Nessiani et al. 2025; Sam and Ragavendran 2026).

Iron impregnation is an effective route to enhance the remediation performance of biochar (Ahmad et al. 2022; de Souza Borges et al. 2025; Mishra and Kumar Tiwari 2025; Yan et al. 2025; Yang et al. 2025; Zhou et al. 2026). Deposited iron oxides and hydroxides introduce positively charged sites and Fe–OH groups. These sites markedly increase the uptake of both heavy-metal cations and oxyanions such as sulphate, phosphate and arsenate (Bagherifam and Kazemian 2026; Manawi et al. 2024; Wang et al. 2021). Importantly, the reactivity of such composites often depends on how iron is dispersed and speciated on the carbon surface rather than on the absolute iron loading. Amorphous iron (oxyhydr)oxides, fine crystalline phases and framework-derived iron clusters expose very different densities of accessible Fe sites (Bauer et al. 2026; Hu et al. 2026; Mo et al. 2025). Our previous work on anaerobic digestion showed the same principle, in that the Fe-bearing minerals in biochar, rather than the carbon matrix alone, controlled microbial activity and conversion performance (Sugiarto et al. 2021; Sugiarto et al. 2025). Despite this, most Fe-biochar adsorption studies still optimise the total iron loading and report a single pollutant. The link between iron retention, surface Fe–OH availability and sulphate removal under AMD conditions has rarely been examined. In particular, it remains unclear whether adding more iron precursors beyond the biochar's retention capacity creates more reactive sites or simply wastes reagents.

In the present work, coconut shell biochar produced by slow pyrolysis was impregnated with $\text{FeCl}_3 \cdot 6\text{H}_2\text{O}$ at three precursor-to-biochar mass ratios and evaluated for sulphate removal from synthetic AMD. The iron content, crystalline phases, and surface functional groups of the materials were determined by AAS, XRD, and FTIR, respectively, and were related to their sulphate removal. The best material was then examined in batch experiments arranged in a randomised factorial design of adsorbent dose and contact time, and the data were analysed with isotherm and kinetic models. The specific objectives were (i) to determine how iron retention responds to the precursor ratio, (ii) to test whether sulphate removal follows total iron loading or the accessibility of surface Fe–OH sites, and (iii) to propose a removal mechanism consistent with the characterisation, performance and model data. Understanding how iron accessibility, rather than iron abundance, controls sulphate removal will enable researchers and end-users alike to tailor low-cost biochar adsorbents for the sustainable remediation of AMD.

2. Experimental

2.1 Materials

Coconut shell, used as the biomass precursor, was obtained locally in Malang, East Java, Indonesia. Ferric chloride hexahydrate ($\text{FeCl}_3 \cdot 6\text{H}_2\text{O}$, analytical grade) was used as the iron precursor, and barium chloride (BaCl_2) was used for the turbidimetric determination of sulphate. Synthetic AMD was prepared by dissolving predetermined amounts of FeSO_4 , ZnSO_4 , and Na_2SO_4 in distilled water to reproduce the elevated metal and sulphate concentrations characteristic of acid mine drainage. The solution was acidified with HCl to an initial pH of 2.5 ± 0.5 , within the range commonly reported for acidic mine waters and synthetic AMD. The resulting synthetic AMD contained 142.38 mg L^{-1} Fe and 48.80 mg L^{-1} Zn, while the measured initial sulphate concentration was 1958.6 mg L^{-1} .

2.2 Biochar production and iron impregnation

The cleaned coconut shells were size-reduced and converted to biochar by slow pyrolysis in a sealed reactor under oxygen-limited conditions. The reactor was heated at approximately $10^\circ\text{C min}^{-1}$ to 650°C and maintained at this temperature for 20 min. After pyrolysis, the resulting char was rapidly cooled to minimise further oxidation. The biochar was subsequently ground and sieved to obtain particles smaller than 1.0 mm before further use. This unmodified material was designated as pristine biochar and used as the control.

Iron loading was carried out by wet impregnation. The pristine biochar was ground using a mortar and dispersed in distilled water at a biochar-to-water ratio of 1:3 (w/v), followed by magnetic stirring at 400 rpm for 15 min. Ferric chloride hexahydrate ($\text{FeCl}_3 \cdot 6\text{H}_2\text{O}$) was subsequently added at precursor-to-biochar mass ratios of 0.5, 0.7, and 1.0 g g^{-1} . Based on the Fe mass fraction in $\text{FeCl}_3 \cdot 6\text{H}_2\text{O}$, these ratios corresponded to nominal Fe additions of 103.3, 144.6, and $206.6 \text{ mg Fe g}^{-1}$ biochar, respectively. The suspensions were further stirred at 600 rpm for 15 min, then aged at room temperature for 24 h to promote interaction between Fe species and the biochar surface and pore structure. After ageing, the solids were separated from the impregnation solution by filtration and washed repeatedly with distilled water until the washings approached neutral pH, thereby removing loosely bound Fe species and residual chloride. The washed solids were subsequently oven-dried at 105°C for 24 h. The resulting materials were designated FeBC-x, where x represents the $\text{FeCl}_3 \cdot 6\text{H}_2\text{O}$ -to-biochar mass ratio, namely FeBC-0.5, FeBC-0.7, and FeBC-1.0. Figure 1 summarises the preparation, characterisation, and adsorption-testing procedure.

2.3 Characterisation

The total iron content of the pristine and Fe-modified biochars was determined by atomic absorption spectroscopy (AAS; Shimadzu AA-7800) following acid digestion of the solid samples. Approximately 0.5 g of dried biochar was transferred into a digestion vessel and treated with 9 mL concentrated HNO_3 and 3 mL concentrated HCl. The samples were digested in a closed-vessel microwave system by heating to approximately 175°C , followed by a short holding period at this temperature, in accordance with the general procedure of USEPA Method 3051A. After cooling, the digests were filtered, diluted to a known volume with deionised water, and analysed for Fe by AAS. USEPA Method 3051A permits digestion with

HNO₃ alone or an HNO₃-HCl mixture and is widely used for metal extraction from solid environmental matrices prior to atomic spectrometric analysis.

Crystalline phases were analysed by X-ray diffraction (XRD; PANalytical X'Pert PRO) using Cu K α radiation ($\lambda = 1.5406 \text{ \AA}$) over a 2θ range of $10\text{--}80^\circ$, with a step size of 0.02° at 25°C . Surface functional groups were characterised by Fourier-transform infrared spectroscopy (FTIR; Shimadzu IRSpirit) over $400\text{--}4000 \text{ cm}^{-1}$, using Happ-Genzel apodisation, 20 scans, and a spectral resolution of 4 cm^{-1} . The pristine biochar and the selected FeBC-0.7 sample were additionally re-analysed by XRD and FTIR under the same instrumental conditions to directly assess structural and surface-chemical changes associated with iron impregnation.

2.4 Batch sulphate removal experiments

Sulphate removal was evaluated in batch mode (Zheng et al. 2020). A preliminary screening experiment was first conducted to identify the most suitable Fe-modified biochar for subsequent testing. Pristine biochar and the three FeBC materials (FeBC-0.5, FeBC-0.7 and FeBC-1.0) were separately contacted with synthetic AMD at an adsorbent dose of 2 g L^{-1} for 60 min at room temperature. Each treatment was performed in triplicate. After contact, the suspensions were separated, and the residual sulphate concentration was determined. The sulphate removal efficiency and apparent uptake capacity of each material were then compared with its total iron content and with its surface Fe-O/Fe-OH response in the FTIR spectra to select the most effective precursor-to-biochar ratio (Table 3).

The selected adsorbent (FeBC-0.7) was subsequently evaluated using a randomised two-factor factorial design, with adsorbent dose ($2, 4$ and 6 g L^{-1}) and contact time ($30, 60$ and 120 min) as the experimental factors. Each treatment combination was performed in triplicate, resulting in 27 experimental runs. Pristine biochar was additionally evaluated as the control at a fixed dose of 4 g L^{-1} at each contact time. A separate kinetic run was performed with FeBC-0.7 at 4 g L^{-1} under the same conditions, with samples withdrawn at $2, 5, 10, 15, 30, 60$ and 120 min . For each run, the required mass of adsorbent was added to 250 mL of synthetic AMD and agitated continuously at approximately 200 rpm and $25 \pm 2^\circ\text{C}$. The synthetic AMD was strongly acidic, with an initial pH of approximately 2.5 . After treatment, the final pH was recorded; values ranged from 2.01 to 2.80 for FeBC-0.7 treatments and from 2.34 to 3.15 for the pristine-biochar controls. The suspensions were subsequently filtered through a $0.45 \text{ }\mu\text{m}$ membrane filter to separate the adsorbent from the aqueous phase. Residual sulphate was quantified by the BaCl₂ turbidimetric method, following the principle of Standard Method 4500-SO₄²⁻ E and EPA Method 375.4. Briefly, BaCl₂ was added to the appropriately diluted filtrate to form a BaSO₄ suspension, and the resulting turbidity was measured spectrophotometrically at 420 nm against the corresponding sample blank. Reagent blanks and an adsorbent-free synthetic-AMD blank were processed in parallel to account for background absorbance and any non-adsorptive sulphate loss. Sulphate concentrations were determined from an external calibration curve. The removal efficiency (R , %) and apparent uptake capacity (q_e , mg g^{-1}) were calculated from the initial (C_0 , mg L^{-1}) and residual

(C_e , mg L^{-1}) sulphate concentrations, solution volume (V , L) and adsorbent mass (m , g) using Eqs. (1) and (2):

$$R = (C_0 - C_e)/C_0 \times 100 \quad (1)$$

$$q_e = (C_0 - C_e)V/m \quad (2)$$

2.5 Isotherm and kinetic modelling

The equilibrium data were analysed with the linearised Langmuir (Eq. 3) and Freundlich (Eq. 4) models. The equilibrium uptake at each dose was taken as the mean uptake over the three contact times, because the kinetic run showed that more than 95% of the plateau uptake was reached within 30 min (Section 3.8). The Langmuir model assumes monolayer adsorption on a homogeneous surface with a finite number of identical sites. The Freundlich model describes adsorption on a heterogeneous surface with a distribution of site energies (Awad and Ibrahim 2024; Usman et al. 2025). The kinetics at 4 g L^{-1} were evaluated with the pseudo-first-order (PFO, Eq. 5) and pseudo-second-order (PSO, Eq. 6) models. Both models were fitted to the q_t - t data by nonlinear least-squares regression. The PSO model was also evaluated in its linear form (Eq. 6), because linearisation can bias the fitted parameters and inflate R^2 (Tran et al. 2017). The PFO model assumes that the uptake rate is proportional to the difference between the equilibrium and instantaneous uptakes, whereas the PSO model assumes that the rate is proportional to the square of that difference (Han et al. 2026; Ma et al. 2026).

$$C_e/q_e = 1/(K_L q_m) + C_e/q_m \quad (3)$$

$$\log q_e = \log K_F + (1/n) \log C_e \quad (4)$$

$$q_t = q_e[1 - \exp(-k_1 t)] \quad (5)$$

$$q_t = k_2 q_e^2 t / (1 + k_2 q_e t); t/q_t = 1/(k_2 q_e^2) + t/q_e \quad (6)$$

Here, q_m (mg g^{-1}) is the maximum monolayer capacity, K_L (L mg^{-1}) is the Langmuir constant, K_F ($(\text{mg g}^{-1})(\text{L mg}^{-1})^{1/n}$) and $1/n$ are the Freundlich constants, q_t (mg g^{-1}) is the uptake at time t (min), and k_1 (min^{-1}) and k_2 ($\text{g mg}^{-1} \text{ min}^{-1}$) are the PFO and PSO rate constants. The goodness of fit was judged from the coefficient of determination (R^2), the root-mean-square error (RMSE) between measured and predicted q_t , and the agreement between the calculated and experimental q_e values (Simonin 2016; Tran et al. 2017). Results are reported as mean $\pm$ standard deviation ($n = 3$). The effects of adsorbent dose, contact time and their interaction on sulphate removal efficiency were evaluated by two-way analysis of variance (ANOVA) at a significance level of $\alpha = 0.05$. The screening results of FeBC-0.7 and FeBC-1.0 were compared with Welch's t-test. ChatGPT (OpenAI) and Claude (Anthropic) were used for language editing and for checking the internal consistency of values reported in the text, tables and figures. The authors verified all outputs and take full responsibility for the content.

3. Results and discussion

3.1 Iron content and retention

The total iron contents determined by AAS are listed in Table 1. The control contained only $1.25 \pm 0.08 \text{ mg g}^{-1}$ (0.125 wt%) of intrinsic iron, whereas impregnation raised the iron content by about an order of magnitude. The iron content increased from $11.90 \pm 0.20 \text{ mg g}^{-1}$ in FeBC-0.5 to $13.90 \pm 0.23 \text{ mg g}^{-1}$ in FeBC-0.7, but did not increase further in FeBC-1.0 ($13.88 \pm 0.18 \text{ mg g}^{-1}$). Doubling the precursor ratio from 0.5 to 1.0 therefore raised the iron content by only about 17%. The incremental gain fell from 2.00 mg g^{-1} between FeBC-0.5 and FeBC-0.7 to a negligible -0.02 mg g^{-1} between FeBC-0.7 and FeBC-1.0. The net retention efficiency, defined as the iron added by impregnation divided by the nominal iron supplied, decreased from 10.3% at a ratio of 0.5 to 8.7% at 0.7 and 6.1% at 1.0 (Table 3). Iron uptake had thus reached a plateau at a ratio of about 0.7. Once the anchoring sites on the biochar surface approach saturation, additional Fe^{3+} is only weakly held and is largely removed during washing rather than being stably incorporated. Similar behaviour has been reported for other iron-impregnated carbons in which the precursor supply exceeds the available anchoring sites (Capobianco et al. 2020; Torboli et al. 2024).

The total iron content, however, reveals nothing about the form in which iron is deposited. FeBC-0.7 and FeBC-1.0 retained the same amount of iron, yet a higher bulk iron content does not necessarily yield more reactive sites. As the XRD and FTIR results below show, the distribution and speciation of iron on the biochar surface, rather than its bulk quantity, governed the subsequent sulphate removal. A similar dependence on iron distribution rather than bulk loading has been reported for magnetic and metal–organic framework-modified biochars, in which iron (oxyhydr)oxide precipitation reorganises the pore network and changes the density of reactive sites (Bagherifam and Kazemian 2026; Bauer et al. 2026).

Table 1
Total iron content of the control and iron-impregnated biochars determined by AAS (mean $\pm$ SD, n = 3). Precursor ratio = $\text{FeCl}_3 \cdot 6\text{H}_2\text{O}$:biochar mass ratio

Sample	Precursor ratio (g g^{-1})	Total Fe content (mg g^{-1})	Total Fe content (wt%)
Control	0	1.25 ± 0.08	0.125 ± 0.008
FeBC-0.5	0.5	11.90 ± 0.20	1.190 ± 0.020
FeBC-0.7	0.7	13.90 ± 0.23	1.390 ± 0.023
FeBC-1.0	1.0	13.88 ± 0.18	1.388 ± 0.018

3.2 Crystalline phases

The XRD patterns of the control and the three FeBC samples are shown in Fig. 2, and the principal reflections are listed in Table 2. All four samples showed essentially the same set of reflections. A sharp,

intense peak at $2\theta \approx 26.6^\circ$ ($d \approx 3.35 \text{ \AA}$) is assigned to the (101) reflection of quartz ($\alpha\text{-SiO}_2$), which originates from the mineral fraction of the coconut shell. It is superimposed on a broad hump centred at about $20\text{--}26^\circ$, attributed to the (002) reflection of turbostratic, partially amorphous carbon. The persistence of this hump confirms that the disordered carbon framework was preserved after impregnation (Zhou et al. 2026). Weaker reflections at $2\theta \approx 20.8^\circ$, 36.5° , 50.1° , 60.0° and 68.1° were also indexed to quartz. The enlarged views in Fig. 2b and c show that the quartz (100) and (101) reflections did not shift with iron loading ($20.82\text{--}20.86^\circ$ and $26.53\text{--}26.60^\circ$, respectively). Iron was therefore not incorporated into the quartz lattice.

No reflection attributable to a crystalline iron oxide was resolved in any FeBC sample. The weak peak at $2\theta \approx 36.5^\circ$ ($d \approx 2.46 \text{ \AA}$) coincides with the quartz (110) reflection. It appeared with comparable intensity in the control and in all FeBC samples and therefore cannot be assigned to an iron oxide. The diagnostic reflections of magnetite ($2\theta \approx 35.4^\circ$ and 62.6°), hematite ($2\theta \approx 33.2^\circ$ and 35.6°) and goethite ($2\theta \approx 21.2^\circ$ and 33.2°) were absent from every pattern. A very weak feature near $2\theta \approx 39.4^\circ$ ($d \approx 2.29 \text{ \AA}$) appeared in the three FeBC samples but not in the control (Fig. 2a; Table 2). It lies close to the quartz (102) reflection (39.46°). However, its absence from the control suggests a minor contribution from a poorly ordered iron-bearing phase. Because no second diagnostic reflection of such a phase was resolved, the feature was not assigned. The impregnated iron therefore remained predominantly X-ray-amorphous or highly dispersed at all precursor ratios. It should be noted that the iron contents ($1.19\text{--}1.39 \text{ wt\% Fe}$, or about $1.7\text{--}2.0 \text{ wt\%}$ expressed as Fe_2O_3) are close to the usual detection limit of laboratory XRD for minor crystalline phases. The absence of iron oxide reflections thus indicates that any crystalline iron domains were too small or too scarce to be detected, rather than proving that crystalline phases were entirely absent.

The absence of long-range crystalline order does not preclude adsorption activity. Amorphous and poorly crystalline iron (oxyhydr)oxides expose many under-coordinated surface Fe sites and usually present a larger reactive surface area than well-crystallised phases. They can therefore be more, rather than less, reactive towards oxyanions such as sulphate (Bagherifam and Kazemian 2026; Wang et al. 2021). A similar interplay between poorly ordered and crystalline iron phases has been reported for ferrihydrite-bearing biomass composites used to treat AMD (Hu et al. 2026). The chemical state of the dispersed iron was therefore examined further by FTIR (Section 3.3).

Table 2
Principal XRD reflections of the control and FeBC samples

Observed 2θ (°)	d (Å)	Assignment (phase, hkl)	Quartz reference 2θ (°)	Relative intensity, I/I_{101} (%)	Detected in samples
20.82–20.86	4.255–4.263	Quartz, SiO ₂ (100)	20.86	19	All
26.53–26.60	3.348–3.357	Quartz, SiO ₂ (101)	26.64	100	All
36.48–36.52	2.458–2.461	Quartz, SiO ₂ (110)	36.54	10	All
50.08–50.12	1.819–1.820	Quartz, SiO ₂ (112)	50.14	7	All
59.98–60.02	1.540–1.541	Quartz, SiO ₂ (211)	59.96	5	All
68.08–68.12	1.375–1.376	Quartz, SiO ₂ (203)	68.14	6	All
≈ 20–26 (broad)	≈ 3.4–4.4	Turbostratic carbon, C (002)	–	Broad	All
≈ 39.4 (weak)	≈ 2.29	Very weak feature close to quartz (102); absent from the control; not assigned	≈ 39.46	Very weak	FeBC-0.5, FeBC-0.7 and FeBC-1.0 (not in control)

3.3 Surface functional groups

The FTIR spectra of the control and the three FeBC samples are presented in Fig. 3. All samples showed the skeletal absorptions typical of carbonaceous biochar. These include a strong C–O/Si–O–Si stretching band near 1044 cm^{-1} , a band near 1700 cm^{-1} (C = O) with a shoulder at about $1580\text{--}1620\text{ cm}^{-1}$ (aromatic C = C and adsorbed H₂O), weak aliphatic C–H stretching near 2920 cm^{-1} and a broad O–H stretching band at $3200\text{--}3600\text{ cm}^{-1}$. The oxygen-containing surface chemistry of the biochar was therefore retained after impregnation (Fahrudin et al. 2024; Kerr et al. 2025). In addition, the FeBC samples showed stronger low-wavenumber absorptions at about $416\text{--}497\text{ cm}^{-1}$ than the control, in which only weak Si–O bending and metal–O bands appeared in this region. These bands are assigned to Fe–O vibrations and show that iron was introduced onto the biochar surface in all impregnated samples (Capobianco et al. 2020; Manawi et al. 2024; Torboli et al. 2024).

The spectra nevertheless differed with the precursor ratio. FeBC-0.7 showed the most intense Fe-related absorptions, both near 465 cm^{-1} and in a distinct band near 681 cm^{-1} that may arise from surface Fe–O/Fe–OH vibrations of iron oxyhydroxide-like species. This band is not taken as evidence of crystalline

goethite or magnetite, because XRD showed no diagnostic reflections of these phases. Instead, it indicates that FeBC-0.7 carried the largest population of surface-accessible Fe–O and Fe–OH sites.

FeBC-1.0 was prepared with the highest precursor ratio, but its Fe–O bands were no stronger than those of FeBC-0.5 and clearly weaker than those of FeBC-0.7. Excess precursor may therefore have promoted iron aggregation or deposition in less accessible positions rather than creating more reactive surface sites. The FTIR data thus indicate that sulphate removal was not governed by total iron loading alone. It also depended on the distribution and accessibility of Fe–O/Fe–OH sites, which bind sulphate through ligand exchange and surface complexation on iron (oxyhydr)oxides (Manawi et al. 2024; Peak et al. 1999). Because the spectra are offset for clarity, band intensities are compared here qualitatively, relative to the C–O/Si–O–Si band of each spectrum.

3.4 Selection of the optimal FeBC sample

The characterisation and screening results are compared in Table 3. FeBC-0.7 and FeBC-1.0 contained virtually identical total iron (13.90 and 13.88 mg g^{-1}), both more than FeBC-0.5 (11.90 mg g^{-1}). Their sulphate removal, however, differed clearly. At 2 g L^{-1} , FeBC-0.7 removed $90.2 \pm 1.6\%$ of sulphate, compared with $83.7 \pm 1.9\%$ for FeBC-1.0 (Welch's t-test, $p \approx 0.01$), $73.6 \pm 2.1\%$ for FeBC-0.5 and $46.8 \pm 2.4\%$ for the control. Two comparisons explain this ranking. Between FeBC-0.5 and FeBC-1.0, which showed Fe–O bands of similar intensity, removal increased with total iron. Between FeBC-0.7 and FeBC-1.0, which held the same total iron, removal followed the intensity of the surface Fe–O/Fe–OH bands. Total iron therefore sets the upper limit of reactive sites, whereas surface accessibility determines how many of them take part in sulphate removal. FeBC-0.7 combined the highest retained iron with the strongest surface Fe–O/Fe–OH response and was selected for all subsequent experiments. This selection reflects the central finding of this work: at a given iron content, the accessibility of surface iron determines the sulphate removal potential of the material.

Table 3

Preliminary screening of the control and FeBC samples at an adsorbent dose of 2 g L^{-1} (mean $\pm$ SD, $n = 3$)

Criterion	Control	FeBC-0.5	FeBC-0.7	FeBC-1.0
Total Fe content (mg g^{-1})	1.25 ± 0.08	11.90 ± 0.20	13.90 ± 0.23	13.88 ± 0.18
Nominal Fe supplied (mg Fe g^{-1} biochar)	–	103.3	144.6	206.6
Net Fe retained (mg Fe g^{-1} biochar)	–	10.65	12.65	12.63
Fe retention efficiency (%)	–	10.3	8.7	6.1
Relative Fe–O/Fe–OH spectral response (FTIR)	Weak	Moderate	Strongest	Moderate
Sulphate removal efficiency (%)	46.8 ± 2.4	73.6 ± 2.1	90.2 ± 1.6	83.7 ± 1.9
Apparent sulphate uptake, q (mg g^{-1})	458.3 ± 23.5	720.8 ± 20.6	883.3 ± 15.7	819.7 ± 18.6
Screening outcome	Not selected	Not selected	Selected	Not selected

The screening values in Table 3 were used only to select the most suitable precursor ratio. They were obtained in a separate run from the factorial experiment and are therefore not directly comparable with the factorial control at 4 g L^{-1} (Section 3.5). The net Fe retained is the total Fe content minus the intrinsic Fe of the control (1.25 mg g^{-1}). The retention efficiency is the net Fe retained divided by the nominal Fe supplied.

3.5 Sulphate removal by FeBC-0.7

The sulphate removal by FeBC-0.7 is presented in Fig. 4. Across all treatments, FeBC-0.7 removed 84.81–92.55% of the sulphate initially present in the synthetic AMD ($C_0 = 1958.6 \text{ mg L}^{-1}$). Two-way ANOVA showed that adsorbent dose ($F(2,18) = 152.35, p < 0.0001$) and contact time ($F(2,18) = 77.46, p < 0.0001$) both had significant effects on removal. Their interaction was also significant ($F(4,18) = 24.22, p < 0.0001$), so the effect of contact time depended on the dose. At 4 g L^{-1} , the control removed $79.54 \pm 1.08\%$ of sulphate, whereas FeBC-0.7 removed $90.93 \pm 1.75\%$ on average and reached a maximum of $92.55 \pm 0.32\%$ at 60 min. The residual sulphate at this optimum was about 146 mg L^{-1} . This is below the sulphate limit of 300 mg L^{-1} for Class II water set by Indonesian Government Regulation No. 22/2021. These values compare favourably with those reported by Fahrudin et al. (2024) for pristine coconut shell biochar applied to AMD. Iron impregnation thus yields a competitive adsorbent for sulphate-laden mine water.

The effect of adsorbent dose is shown in Fig. 4a. Averaged over contact time, removal increased from $87.51 \pm 2.17\%$ at 2 g L^{-1} to $90.93 \pm 1.75\%$ at 4 g L^{-1} and $91.16 \pm 0.72\%$ at 6 g L^{-1} . The gain was therefore

marked between 2 and 4 g L⁻¹ but marginal between 4 and 6 g L⁻¹. Once enough active sites have been supplied relative to the fixed amount of sulphate, further adsorbent adds little uptake. A similar non-linear dependence of removal on dose has been attributed to the supply of active sites exceeding the demand set by the adsorbate concentration (Kerr et al. 2025). In practice, a dose of 4 g L⁻¹ is the most efficient choice. It captures almost all of the removable sulphate without the reagent penalty of the higher dose.

The effect of contact time is shown in Fig. 4c. Averaged over dose, removal was already 90.58% at 30 min, reached 90.83% at 60 min and decreased slightly to 88.19% at 120 min. The control followed the same pattern at a lower level. Near-maximum removal within 30–60 min indicates fast initial uptake driven by abundant vacant sites, followed by a rapid approach to equilibrium (Nguyen et al. 2026; Xiang et al. 2026). The small decrease at 120 min may reflect partial desorption of weakly bound sulphate. However, it was not observed in the separate kinetic run at 4 g L⁻¹ (Section 3.8), in which uptake levelled off between 60 and 120 min. The decrease is therefore regarded mainly as run-to-run variation rather than systematic desorption. Because of the significant interaction, the best single condition in the factorial experiment was identified from the individual treatments: 4 g L⁻¹ and 60 min gave the highest removal (92.55 ± 0.32%).

The contribution of iron impregnation is most evident in the direct comparison between the control and FeBC-0.7 (Fig. 4a). At the same dose of 4 g L⁻¹, FeBC-0.7 removed 90.93 ± 1.75% of sulphate, compared with 79.54 ± 1.08% for the control. Even at 2 g L⁻¹, FeBC-0.7 (87.51 ± 2.17%) outperformed the control dosed at twice that amount. Impregnation, therefore, created additional sites that were scarce in the unmodified biochar. These sites are most likely associated with dispersed iron species and surface Fe–O/Fe–OH groups, which interact more strongly with sulphate than the carbonaceous and mineral components of the control. This enhancement is consistent with reports that FeCl₃ impregnation markedly improves the performance of biochar and with the broader literature on iron-modified and magnetic biochars for removing anions and metals from mine water (Bagherifam and Kazemian 2026; Mo et al. 2025).

The corresponding uptake capacities are shown in Fig. 4b. The control gave an average q_e of 389.44 ± 5.31 mg g⁻¹. FeBC-0.7 gave q_e values of 857.00 ± 24.40, 445.25 ± 9.33 and 297.57 ± 2.27 mg g⁻¹ at 2, 4 and 6 g L⁻¹, respectively. The highest capacity at the lowest dose is expected, because q_e is normalised by the adsorbent mass. A lower dose therefore gives a higher calculated capacity even when the overall removal is not at its maximum (Qiao et al. 2026; Rai and Thengane 2025; Su et al. 2025). The dose of 4 g L⁻¹ offered the best balance between high removal and efficient use of the adsorbent and was therefore used for the kinetic analysis. These capacities should be read as apparent uptake capacities, because they may include removal pathways other than surface adsorption (Section 3.9).

3.6 Structural and functional comparison of the control and FeBC-0.7

To confirm that the improved performance of FeBC-0.7 originates from the deposited iron, the control and FeBC-0.7 were compared directly by XRD and FTIR under identical conditions. The XRD patterns are shown in Fig. 5. Both materials retained the dominant reflection at $2\theta \approx 26.5^\circ$ (26.548° for the control and 26.526° for FeBC-0.7), assigned to quartz (101) overlapping the (002) band of turbostratic carbon (Section 3.2). The other quartz reflections inherited from the mineral matter of the coconut shell were also retained. FeBC-0.7 again showed the very weak feature near $2\theta \approx 39.4^\circ$ ($d \approx 2.29 \text{ \AA}$), which was absent from the control. As discussed in Section 3.2, this feature lies close to quartz (102) ($\approx 39.46^\circ$) and could not be assigned to hematite, maghemite or any other iron phase. No diagnostic reflection of magnetite, maghemite, hematite or goethite was resolved in either material. Consistent with Section 3.2, the deposited iron is therefore interpreted as a dispersed, predominantly X-ray-amorphous iron (oxyhydr)oxide.

The FTIR comparison in Fig. 6 supports this interpretation at the level of surface chemistry. The control showed only weak metal–O/Si–O absorptions near 476 and 503 cm^{-1} , consistent with an essentially unmodified carbon matrix. After impregnation, FeBC-0.7 developed a clear Fe–O band at 465 cm^{-1} and a distinct Fe–O/Fe–OH band near 681 cm^{-1} , both much weaker in the control. It also showed a relatively stronger C = O band at 1713 cm^{-1} , which suggests mild surface oxidation during impregnation in the acidic FeCl_3 solution. The broad O–H stretching envelope at $3200\text{--}3600 \text{ cm}^{-1}$ was retained but did not intensify relative to the C–O/Si–O–Si band. This region is dominated by adsorbed water and carbon-bound hydroxyls, so it cannot resolve Fe–OH groups on its own. The appearance of the Fe–O and Fe–O/Fe–OH bands after impregnation, and their weakness in the control, links the structural modification to the gain in sulphate removal (Manawi et al. 2024; Wang et al. 2021).

3.7 Adsorption isotherms

The equilibrium data were fitted with the linearised Langmuir and Freundlich models (Fig. 7). The Freundlich model gave a better fit ($R^2 = 0.901$) than the Langmuir model ($R^2 = 0.680$). The Langmuir regression also returned a negative slope, which yields a negative q_m and is therefore physically meaningless. The better fit of the Freundlich model is consistent with sulphate uptake on an energetically heterogeneous surface bearing sites of different affinity. Such heterogeneity is expected for FeBC-0.7, which combines carbonaceous porosity, quartz-bearing mineral matter, dispersed amorphous iron and surface Fe–OH groups.

Two limitations must be noted. First, the equilibrium points were generated by varying the adsorbent dose at a single initial concentration, yielding only three points. Second, the Freundlich exponent ($1/n = 2.63$; $K_F = 4.5 \times 10^{-4} (\text{mg g}^{-1})(\text{L mg}^{-1})^{1/n}$) exceeds unity, which describes an apparently unfavourable, S-shaped isotherm over this narrow C_e range. This is atypical for surface adsorption. It suggests that the data may include non-equilibrium effects or removal pathways other than adsorption, such as surface precipitation (Section 3.9). The Freundlich parameters are therefore best regarded as indicative of surface heterogeneity, not as precise equilibrium constants (Tran et al. 2017). A preference for the

Freundlich model has also been reported for hop-derived and other engineered biochars in which precipitation and exchange processes coexist (Salazar et al. 2026).

3.8 Adsorption kinetics

The kinetics of sulphate uptake by FeBC-0.7 at 4 g L^{-1} are shown in Fig. 8, and the fitted parameters are listed in Table 4. Uptake was very fast. About 77% of the plateau uptake was reached within 2 min, almost 90% within 5 min and more than 95% within 15 min. Uptake then rose slowly from 439.2 mg g^{-1} at 15 min to 455.3 mg g^{-1} at 120 min, which corresponds to 93.0% removal. The uptake at 60 min (453.2 mg g^{-1} , 92.55%) matches the factorial maximum at the same dose and contact time. The nonlinear PFO model described the data poorly ($R^2 = 0.864$; $\text{RMSE} = 12.9 \text{ mg g}^{-1}$). It overestimated the uptake at 5–15 min, levelled off at 442.05 mg g^{-1} and therefore underestimated the plateau by about 13 mg g^{-1} (Fig. 8a). A single first-order step thus cannot describe both the rapid initial uptake and the slow approach to equilibrium.

The PSO model described the data much better. The nonlinear fit gave $R^2 = 0.9995$ and $\text{RMSE} = 0.76 \text{ mg g}^{-1}$, with residuals below 1.5 mg g^{-1} at every sampling time. The linearised PSO plot (Fig. 8b) gave a calculated q_e of 457.69 mg g^{-1} and $k_2 = 3.63 \times 10^{-3} \text{ g mg}^{-1} \text{ min}^{-1}$ ($R^2 > 0.9999$). Both PSO fits returned a calculated q_e within 0.6% of the experimental plateau (455.33 mg g^{-1}). From the linear fit, the initial uptake rate, $h = k_2 q_e^2$, was about $760 \text{ mg g}^{-1} \text{ min}^{-1}$, and the half-life, $1/(k_2 q_e)$, was only about 0.6 min. The very high linear R^2 should be read with caution. When uptake is close to equilibrium at most sampling times, the t/q_t versus t plot is dominated by the t term and returns a high R^2 almost by construction (Simonin 2016; Tran et al. 2017). The nonlinear fit and the RMSE are therefore the more reliable criteria, and both favour the PSO model. A good PSO fit is often taken to indicate chemisorption, which is consistent with sulphate binding to Fe sites through ligand exchange (Ai et al. 2023; Hnamte and Pulikkal 2022; Imreová et al. 2024). However, the PSO fit alone does not prove the mechanism. The same shape also arises when a fast surface reaction is followed by slower pore diffusion or surface precipitation, which matches the two-stage uptake observed here (Section 3.9). The PSO result is therefore treated as supporting, rather than decisive, evidence of chemisorption.

Table 4

Kinetic parameters for sulphate uptake by FeBC-0.7 at 4 g L⁻¹ ($C_0 = 1958.6$ mg L⁻¹; experimental plateau $q_{e,exp} = 455.33$ mg g⁻¹ at 120 min)

Model	Fitting method	$q_{e,cal}$ (mg g ⁻¹)	Rate constant	R^2	RMSE (mg g ⁻¹)
PFO	Nonlinear (Eq. 5)	442.05	$k_1 = 0.749$ min ⁻¹	0.864	12.9
PSO	Nonlinear (Eq. 6)	457.99	$k_2 = 3.52 \times 10^{-3}$ g mg ⁻¹ min ⁻¹	0.9995	0.76
PSO	Linear, t/q_t vs t (Eq. 6)	457.69	$k_2 = 3.63 \times 10^{-3}$ g mg ⁻¹ min ⁻¹	> 0.9999 ^a	1.22

^a R^2 of the linear t/q_t - t regression. RMSE was calculated from q_t for all fits.

3.9 Proposed mechanism of sulphate removal

The results above show that iron impregnation markedly enhanced sulphate removal and that, at a given iron content, the enhancement tracked the accessibility of surface iron. Based on the characterisation, performance and model data, a removal mechanism is proposed in Fig. 9 and discussed below.

The mechanism begins with the creation of iron-bearing sites during impregnation. When the biochar contacts the FeCl₃ solution, Fe³⁺ hydrolyses and condenses on the carbon surface. This forms iron (oxyhydr)oxide domains that terminate in surface hydroxyl (Fe–OH) groups, as represented by Eq. (7):



FTIR showed new Fe–O and Fe–O/Fe–OH bands after impregnation (Section 3.6), consistent with the formation of surface Fe–OH sites. XRD indicated that the iron was deposited as a finely dispersed, predominantly X-ray-amorphous phase (Sections 3.2 and 3.6). Three complementary pathways then transfer sulphate from the AMD to the FeBC-0.7 surface (Fig. 9).

The first pathway is pore filling and physical adsorption within the carbonaceous matrix. The porosity of the coconut shell biochar allows the sulphate-bearing solution to penetrate the pore network, where weak interactions retain part of the anions. Together with the native surface groups, this pathway accounts for much of the baseline removal by the control. Pore filling has likewise been recognised as a weak-interaction pathway operating alongside stronger complexation in multifunctional magnetic biochars (Mo et al. 2025).

The second pathway is inner-sphere surface complexation through ligand exchange at Fe–OH sites. This is the pathway that iron impregnation adds. Under the acidic conditions of AMD, the surface hydroxyls

are protonated to Fe-OH_2^+ . The aquo ligand is then exchanged for sulphate to form an inner-sphere Fe-O-SO_3^- complex, as shown in Eq. (8):



Ligand exchange forms a chemical bond between a sulphate oxygen and a surface iron atom and is therefore a chemisorption process. It is consistent with the good PSO fit and with the surface heterogeneity captured by the Freundlich model. Inner-sphere sulphate complexes on iron (oxyhydr)oxides are well documented for goethite, particularly at pH values below the point of zero charge, as in AMD (Peak et al. 1999). Analogous inner-sphere complexation of oxyanions on iron-modified biochar has been demonstrated for arsenic removal from AMD (Wang et al. 2021).

The third pathway is electrostatic attraction. Protonation of Fe-OH and other oxygenated groups gives the FeBC-0.7 surface a net positive charge under acidic conditions. The resulting Coulombic attraction draws sulphate towards the surface and concentrates it near the active sites, which facilitates ligand exchange. This contribution explains why the control, whose surface carries fewer positive sites, removed less sulphate than FeBC-0.7 (Yu et al. 2025).

However, a site balance shows that these three pathways cannot fully account for the measured uptake. The iron added by impregnation in FeBC-0.7 was 12.65 mg g^{-1} ($0.227 \text{ mmol g}^{-1}$). The iron-attributed increase in uptake at 4 g L^{-1} , defined as the difference between FeBC-0.7 and the control, was approximately 55.8 mg g^{-1} (0.58 mmol g^{-1}). This corresponds to about 2.6 mol of sulphate per mole of added iron. The ratio exceeds the 1:1 stoichiometry of monodentate ligand exchange, even if every iron atom were exposed at the surface. The apparent uptake of the control (389.44 mg g^{-1} , about 4.1 mmol g^{-1}) is also high for an unmodified carbon. Additional pathways, therefore, probably contribute. These may include surface precipitation or co-precipitation of sulphate-bearing phases, such as schwertmannite-like basic ferric sulphates or gypsum, depending on the AMD composition, and retention of sulphate in the pore water (Bigham et al. 1996). The final pH supports this view. Ligand exchange releases water or hydroxyl and would raise the pH, yet the final pH with FeBC-0.7 (2.01–2.80) was generally lower than with the control (2.34–3.15). This acidification is consistent with proton-releasing hydrolysis and precipitation of Fe(III)-sulphate phases at the biochar surface. The slow uptake stage after 15 min (Section 3.8) is also consistent with such a precipitation or pore-diffusion contribution. These pathways do not alter the main conclusion, which rests on the comparison between FeBC samples of equal iron content. They do mean that the q_e values should be interpreted as apparent uptake capacities.

Taken together, these pathways explain the improvement observed in the factorial experiment. The control removed $79.54 \pm 1.08\%$ of sulphate at 4 g L^{-1} through pore filling, native surface interactions and possibly precipitation. FeBC-0.7 added ligand exchange and electrostatic attraction at dispersed Fe-OH sites and removed $90.93 \pm 1.75\%$ on average at the same dose, with a maximum of $92.55 \pm 0.32\%$ at 60

min. Figure 9 therefore summarises the central conclusion of this work. The dispersion and surface speciation of iron, and the Fe–OH chemistry that accompanies it, rather than the absolute iron content alone, determine the sulphate removal enhancement of Fe-biochar.

3.10 Implications for AMD remediation

The results carry two broader implications. First, the characterisation, screening, isotherm, kinetic and mechanistic evidence consistently shows that the functional quality of an iron-modified biochar depends on how iron is distributed and speciated at the surface, and not on iron loading alone. FeBC-0.7 carried the same total iron as FeBC-1.0 but outperformed it, because more of its iron was exposed as accessible surface Fe–O/Fe–OH sites. Because iron retention plateaued at a ratio of 0.7, raising the precursor ratio to 1.0 consumed about 43% more $\text{FeCl}_3 \cdot 6\text{H}_2\text{O}$ without adding iron or reactive sites. For scale-up, the precursor ratio should therefore be optimised against retention and surface accessibility rather than against nominal loading.

Second, the performance achieved here is relevant to the sustainable management of mine water. Using coconut shell waste and an inexpensive iron salt, FeBC-0.7 removed more than 92% of sulphate from synthetic AMD with an initial concentration close to 2000 mg L^{-1} . It thereby addresses the main weakness of conventional AMD treatment, namely, poor sulphate removal. Established approaches such as chemical neutralisation and precipitation, membrane filtration, ion exchange, and reverse osmosis each face constraints with sulphate-rich mine water. These include large sludge volumes, high energy and reagent demands, membrane fouling and high operating costs (Jegatheesan et al. 2026; Roa et al. 2025; Skousen et al. 2019). A low-cost adsorbent derived from waste biomass therefore offers a complementary route to sulphate attenuation. Real AMD, however, contains competing anions and a wider range of dissolved metals than the synthetic feed used here. FeBC-0.7 is therefore best regarded as a low-cost first stage within an integrated AMD treatment train, placed upstream of resource-recovery or biological polishing units (Feng et al. 2026; Jegatheesan et al. 2026; Vega Garzon et al. 2026). Its regeneration, iron and chloride leaching and performance under continuous flow must be established before such use.

4. Conclusions

Coconut shell biochar was impregnated with $\text{FeCl}_3 \cdot 6\text{H}_2\text{O}$ at three precursor ratios and evaluated for sulphate removal from synthetic AMD. Iron retention plateaued above a precursor-to-biochar ratio of 0.7, so FeBC-0.7 and FeBC-1.0 held the same total iron (13.90 and 13.88 mg g^{-1}). Nevertheless, FeBC-0.7 removed more sulphate in the screening test (90.2% versus 83.7% at 2 g L^{-1}), and this advantage was consistent with its stronger surface Fe–O/Fe–OH absorption. XRD showed no crystalline iron oxide, which indicates that the active iron was finely dispersed and predominantly X-ray amorphous. At 4 g L^{-1} , FeBC-0.7 removed $90.93 \pm 1.75\%$ of sulphate on average, compared with $79.54 \pm 1.08\%$ for the pristine biochar, and reached $92.55 \pm 0.32\%$ at 60 min. This lowered the residual sulphate to about 146 mg L^{-1} .

Uptake was fast and followed PSO kinetics, and the Freundlich model described the equilibrium data better than the Langmuir model. These results show that iron accessibility, and not iron loading alone, governs the enhancement of sulphate removal by Fe-biochar. Future work should validate FeBC-0.7 with real AMD, quantify iron and chloride leaching, test regeneration over several cycles and evaluate its performance in continuous-flow columns.

Declarations

Competing Interests

The authors have no relevant financial or non-financial interests to disclose.

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Author Contributions

Yusron Sugiarto: Conceptualisation, Methodology, Supervision, Writing – original draft, Writing – review & editing, Funding acquisition. Tine Aprianti: Investigation, Formal analysis. Sri Rizki Putri Primandari: Investigation, Formal analysis. Akhmad Adi Sulianto: Methodology, Validation. Ahmad Zikrillah: Investigation, Data curation, Writing – original draft. Mingming Zhu: Supervision, Writing – review & editing.

Data Availability

The data supporting the findings of this study are available from the corresponding author upon reasonable request.

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Figures

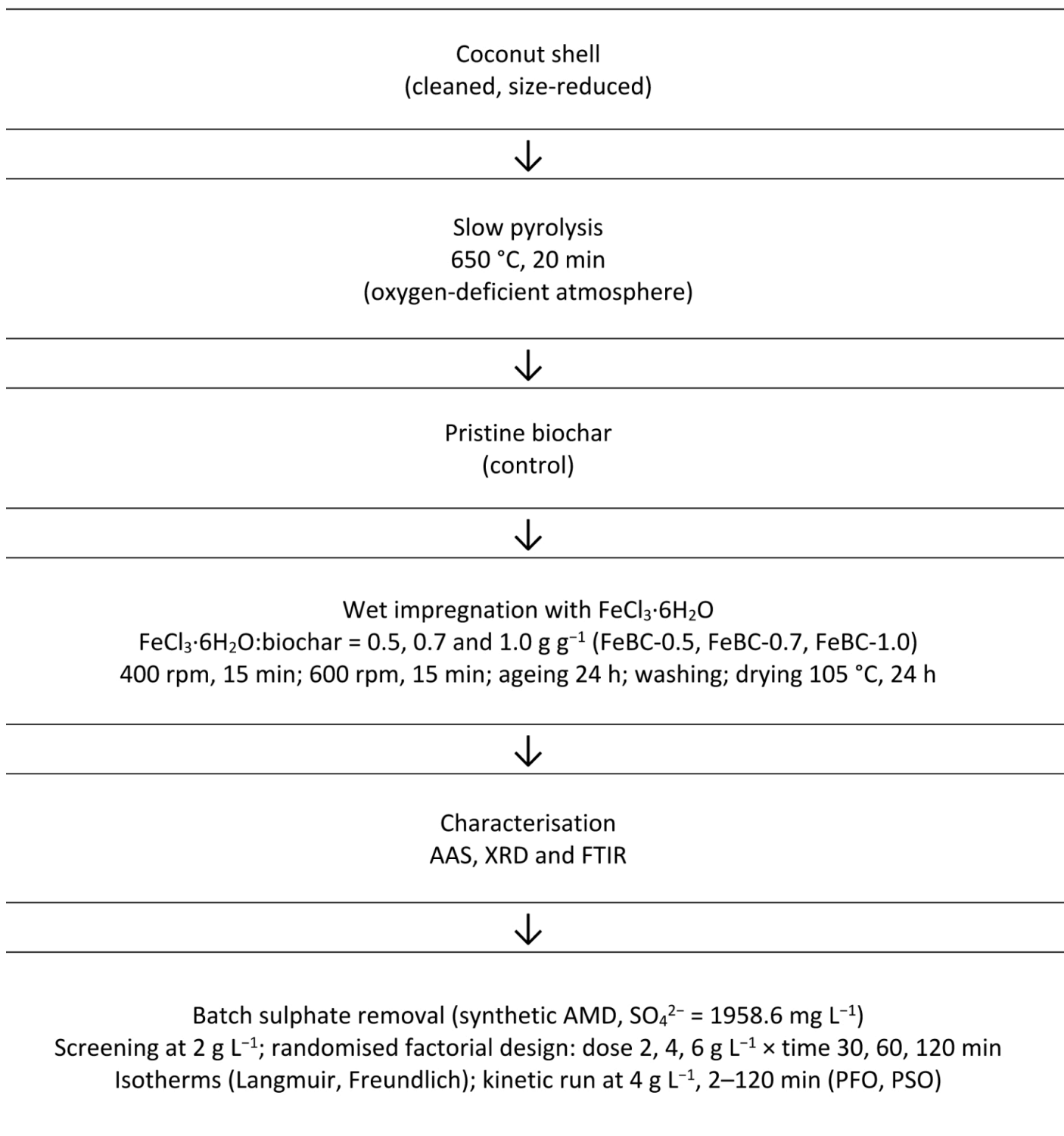


Figure 1

Preparation of the iron-impregnated coconut shell biochars (FeBC-x) and the batch sulphate removal tests

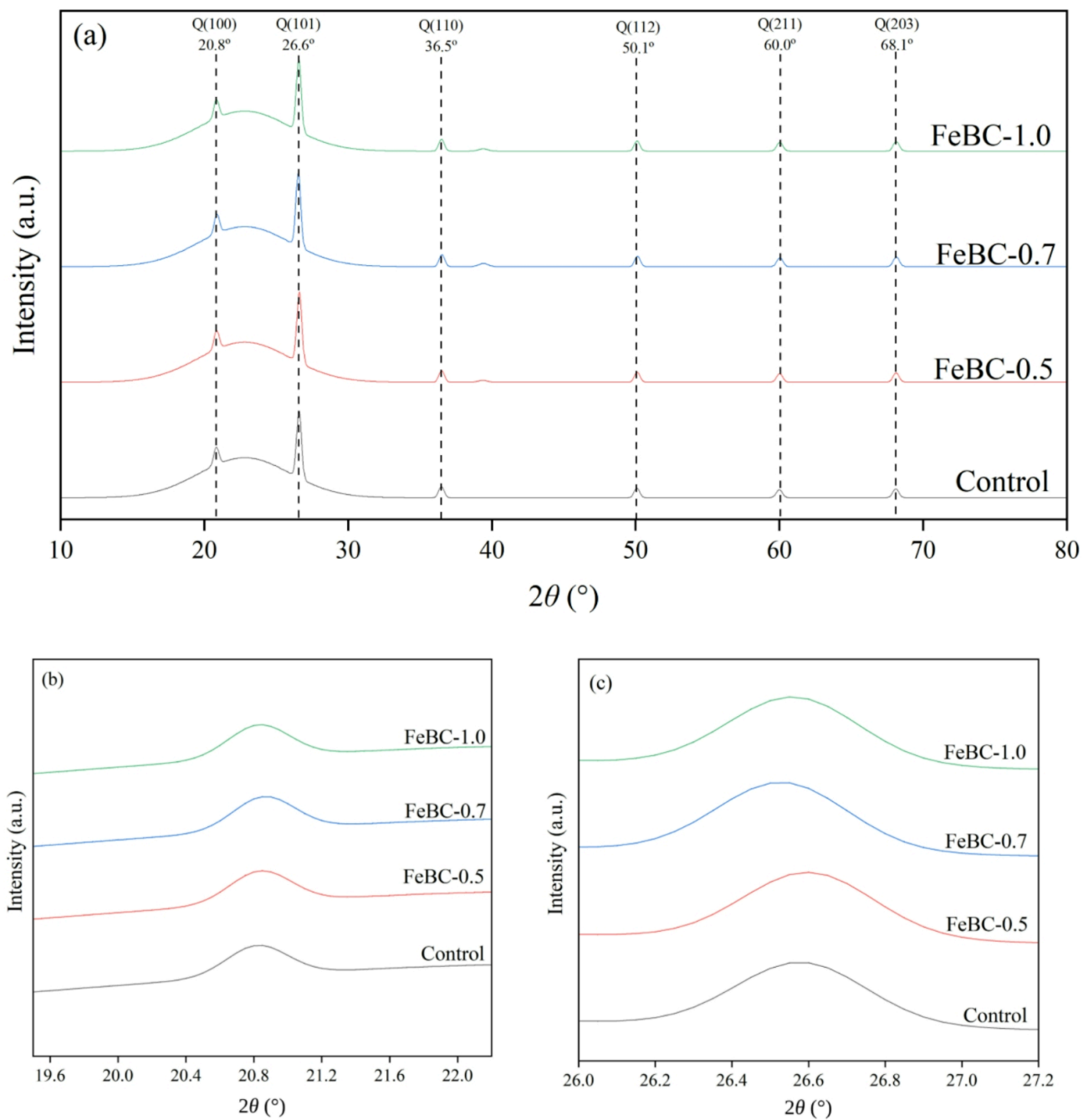


Figure 2

XRD patterns of the control and FeBC samples prepared at $\text{FeCl}_3 \cdot 6\text{H}_2\text{O}$:biochar mass ratios of 0.5, 0.7 and 1.0: (a) full patterns, and enlarged views of the quartz reflections at (b) 20.8° and (c) 26.6° .

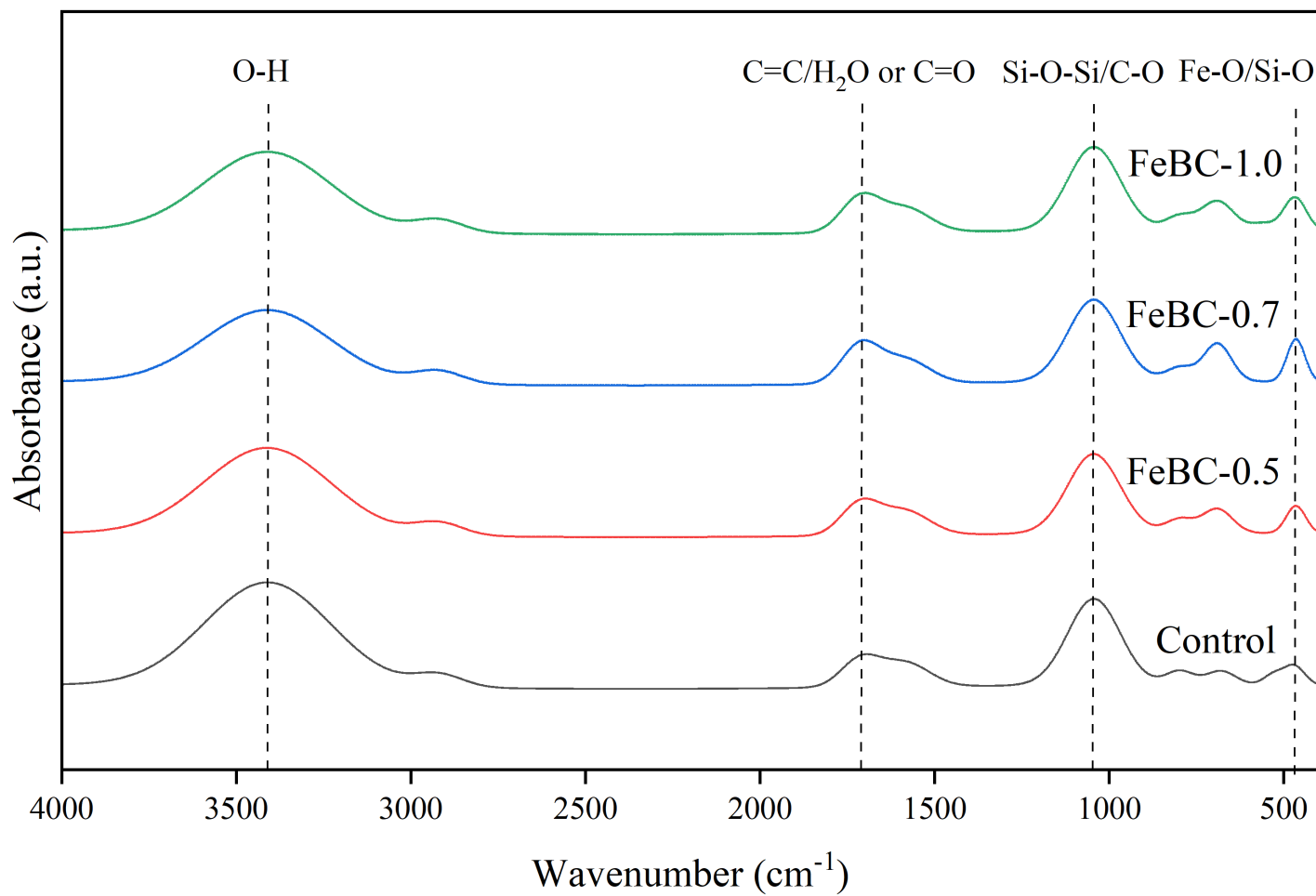


Figure 3

FTIR spectra of the control and FeBC samples prepared at $\text{FeCl}_3 \cdot 6\text{H}_2\text{O}$:biochar mass ratios of 0.5, 0.7 and 1.0

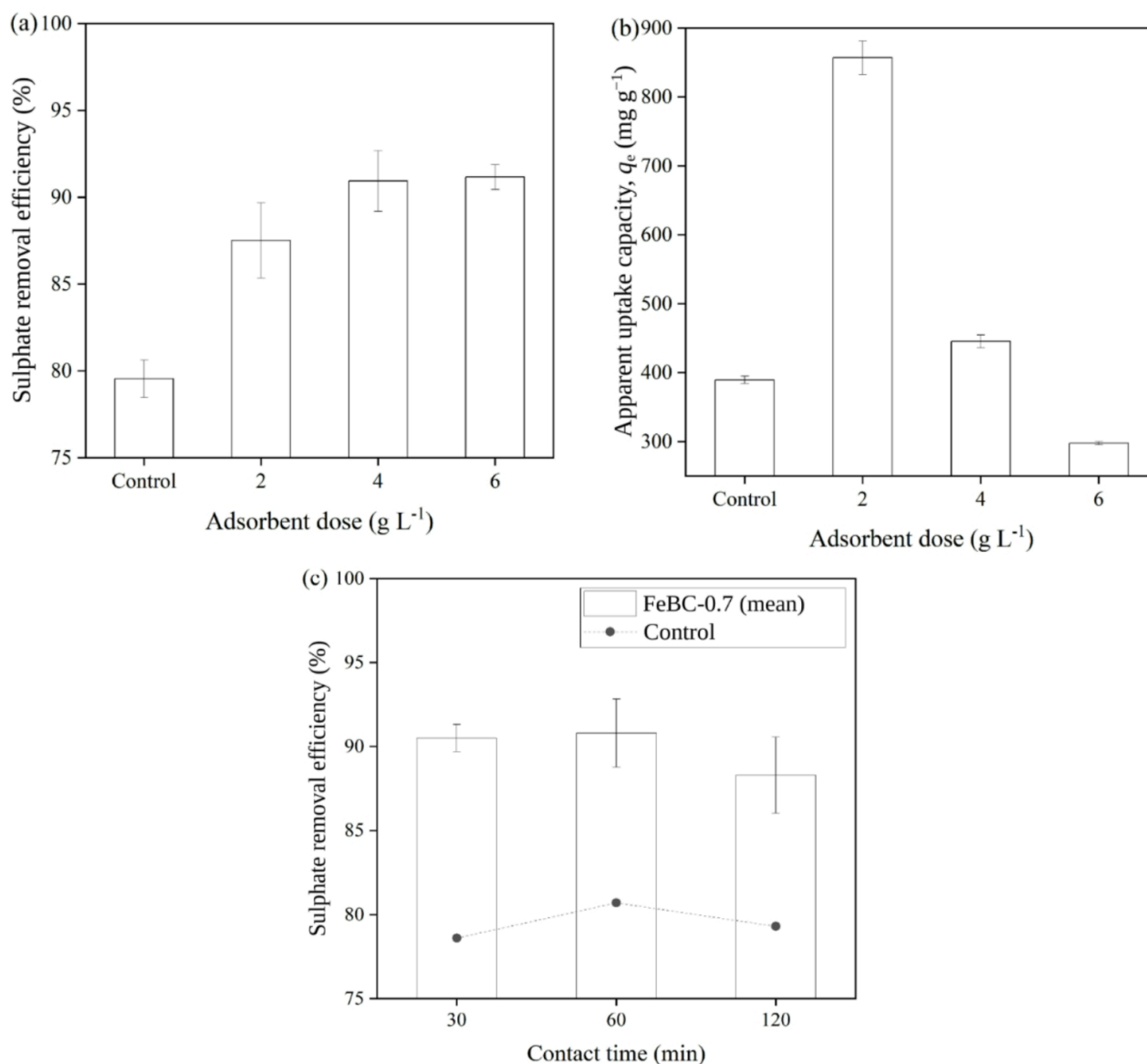


Figure 4

Sulphate removal by the control and FeBC-0.7: (a) removal efficiency of the control (4 g L⁻¹) and of FeBC-0.7 at 2, 4 and 6 g L⁻¹, averaged over contact time; (b) the corresponding apparent uptake capacity (q_e); and (c) effect of contact time on removal by FeBC-0.7 (averaged over dose) and by the control. Error bars show standard deviations.

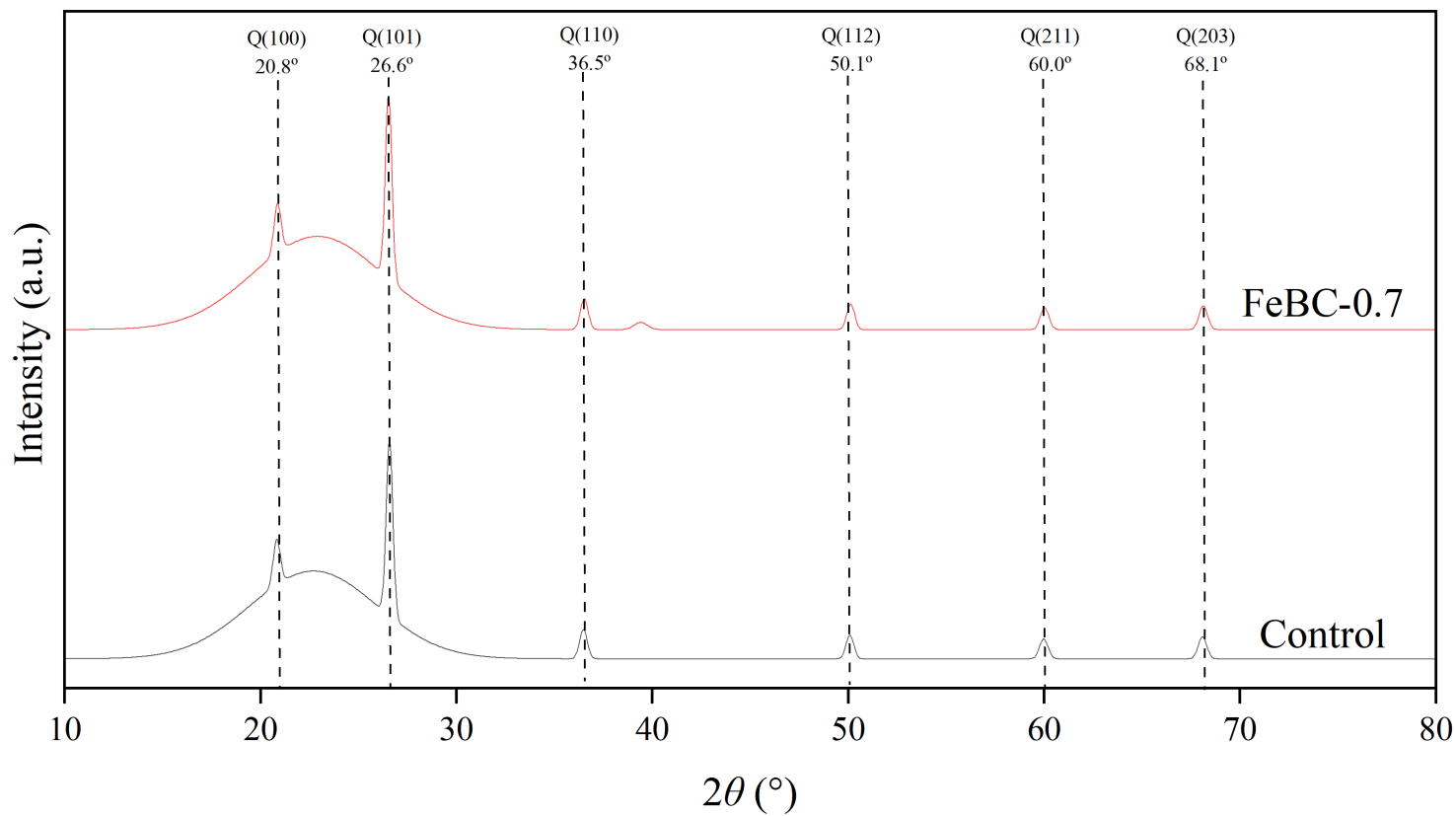


Figure 5

XRD patterns of the control and FeBC-0.7.

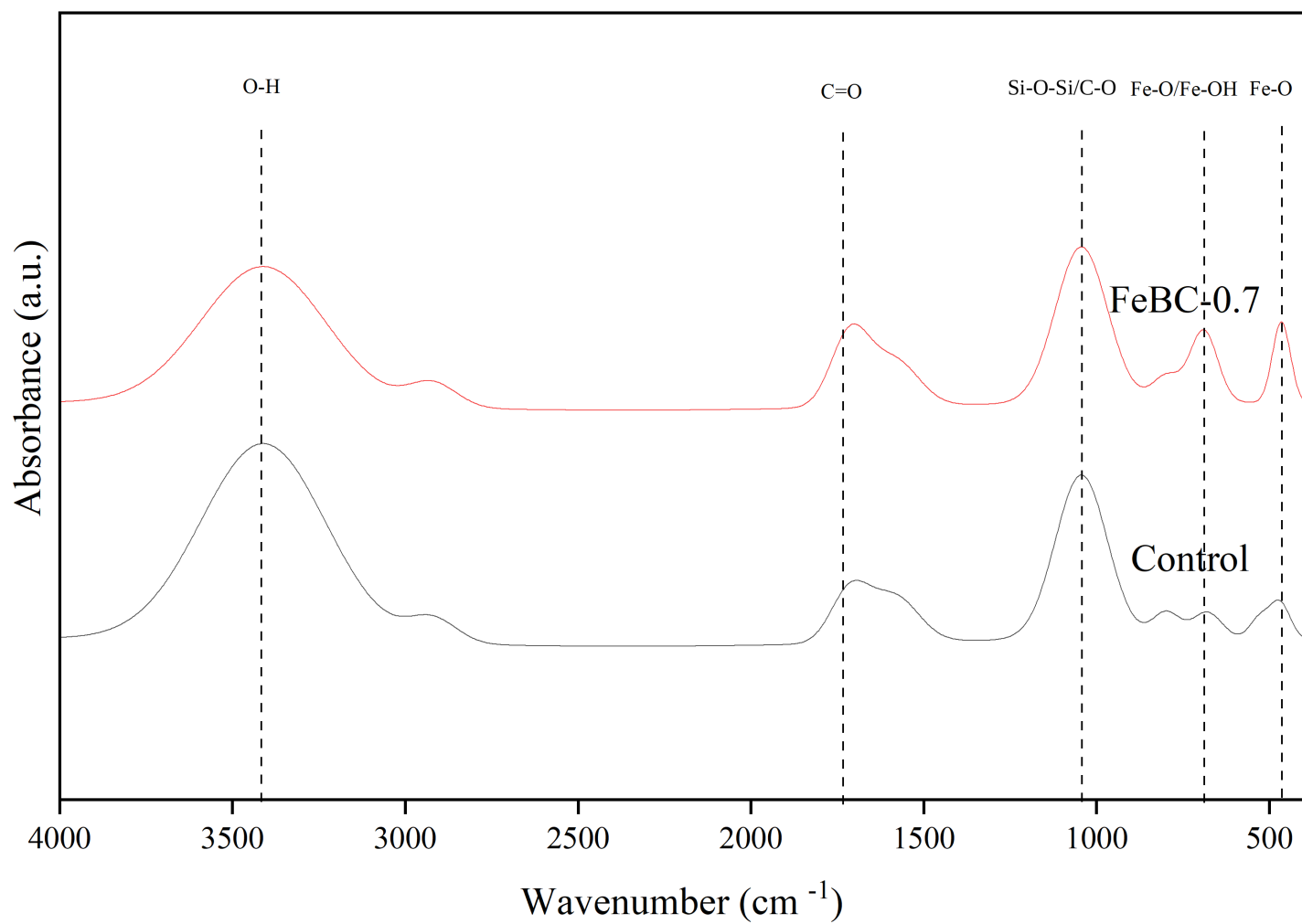


Figure 6

FTIR spectra of the control and FeBC-0.7.

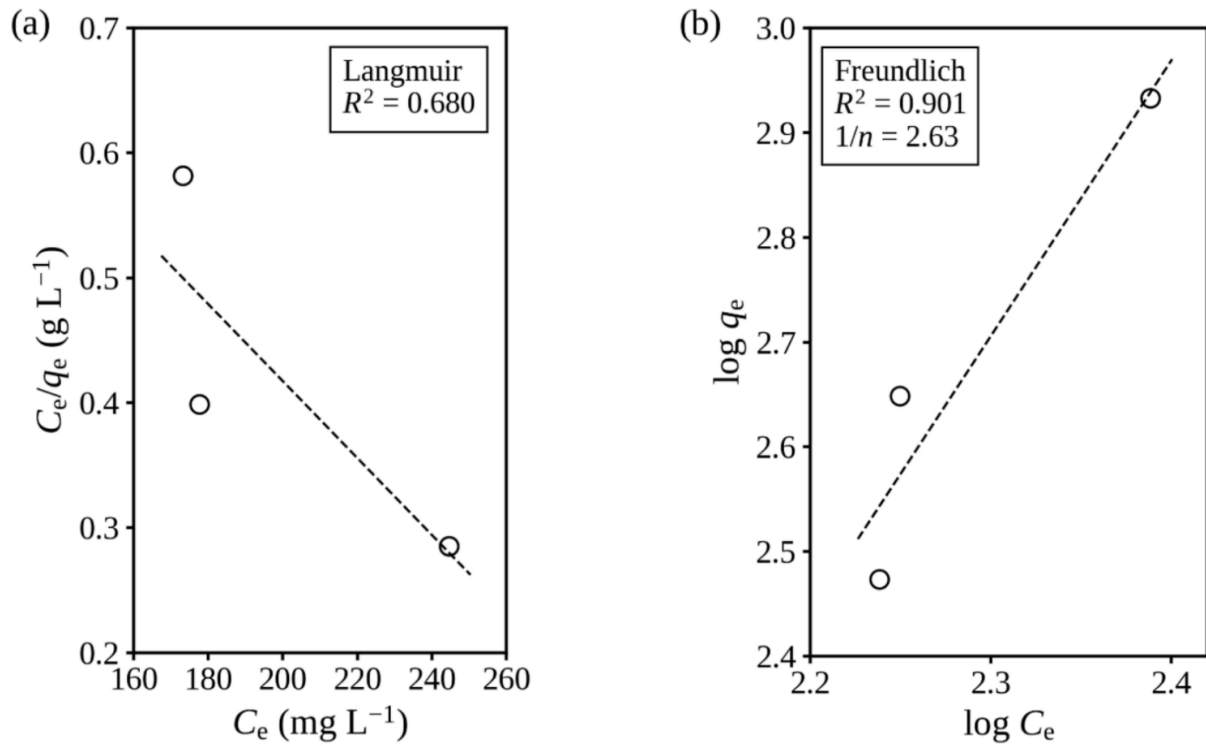


Figure 7

Linearised isotherm plots for sulphate uptake by FeBC-0.7: (a) Langmuir and (b) Freundlich. The equilibrium uptake at each dose (2, 4 and 6 g L^{-1}) is the mean over the three contact times

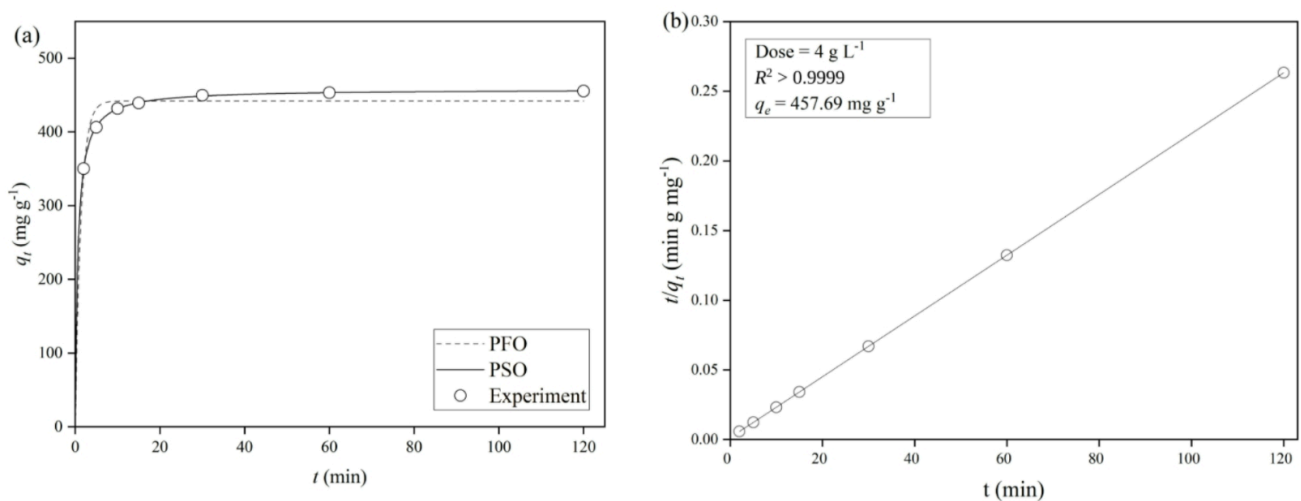


Figure 8

Kinetics of sulphate uptake by FeBC-0.7 at 4 g L⁻¹ (C₀ = 1958.6 mg L⁻¹): (a) measured q_t with the nonlinear PFO and PSO fits; and (b) linearised PSO plot, t/q_t versus t .

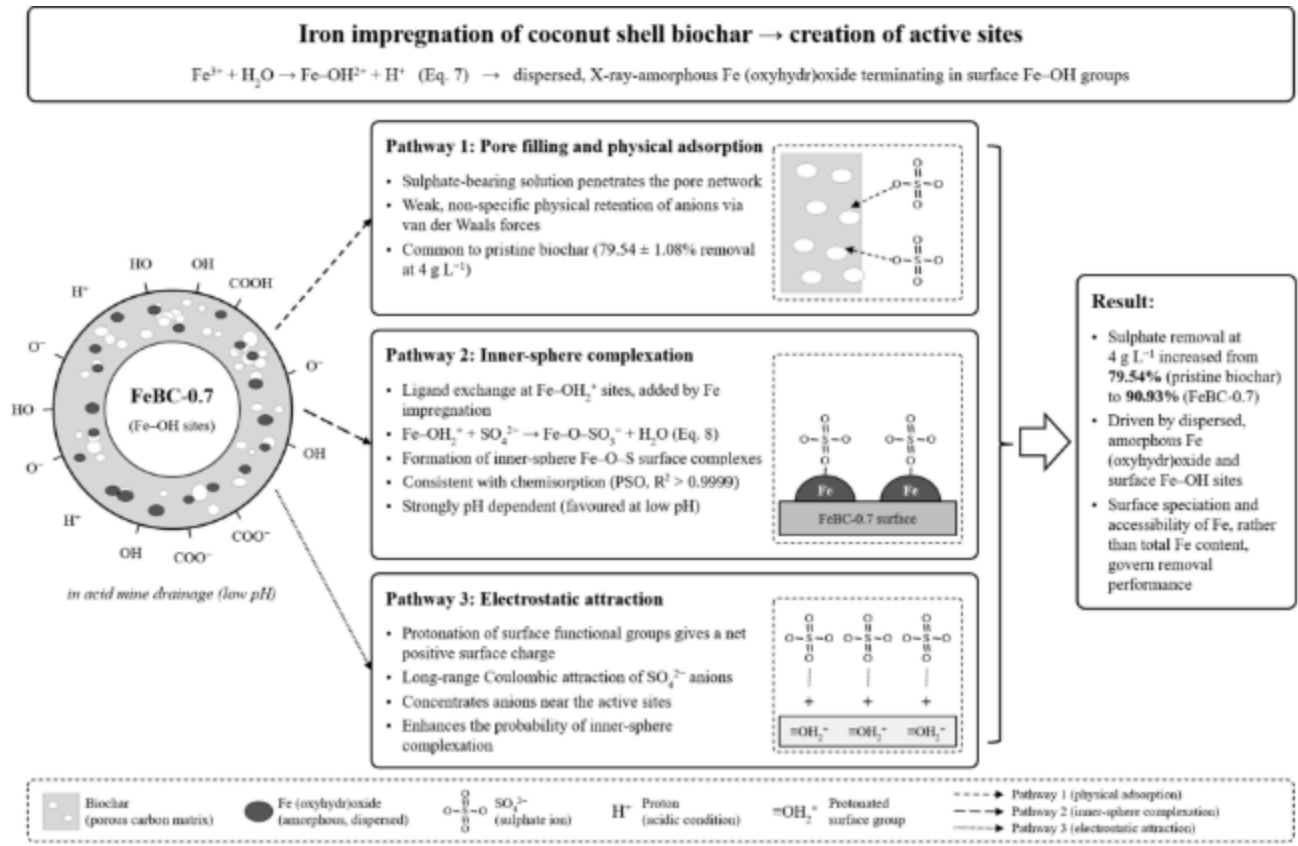


Figure 9

Proposed mechanism of sulphate removal from synthetic AMD by FeBC-0.7.