

Article

Removal of Ciprofloxacin from Aqueous Solutions by Waste-Pretreated *Ganoderma resinaceum* Biomass: Effect of Process Parameters and Kinetic and Equilibrium Studies

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Abstract

This study explores the use of the waste-pretreated biomass of the macro-fungus *Ganoderma resinaceum* as a biosorbent for removing ciprofloxacin from aqueous solutions. Batch experiments were conducted to evaluate the biosorption performance of *G. resinaceum* under varying conditions. Key operational parameters—including pH, biosorbent dosage, contact time, and initial ciprofloxacin concentration—were systematically assessed. Equilibrium was reached within 120 min. Equilibrium data were fitted to both Freundlich and Langmuir isotherm models, with the Langmuir model providing a better fit. Under optimal conditions (initial pH of 7.0, 120-min contact time, 1 g/L biosorbent dosage, and ciprofloxacin concentrations ranging from 4 to 20 mg/L), the maximum biosorption capacity was determined to be 18.4 mg/g. Kinetic analysis revealed that the biosorption process followed a pseudo-second-order model. Furthermore, the biomass was characterized before and after biosorption using scanning electron microscopy (SEM) and Fourier transform infrared (FT-IR) spectroscopy. The FT-IR analysis of *G. resinaceum* biomass revealed the presence of hydroxyl, amino, and carbonyl functional groups, which play a crucial role in the binding and biosorption of ciprofloxacin molecules.

Keywords: biosorption process; ciprofloxacin; *Ganoderma* biomass; waste biomass



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1. Introduction

Pharmaceuticals, particularly antibiotics, are increasingly recognized as contaminants of emerging concern (CEC) due to their persistence in the environment and their role in promoting antimicrobial resistance (AMR). These compounds enter aquatic environments through human excretion, agricultural runoff, hospital effluents, and pharmaceutical manufacturing discharges [1,2]. Ciprofloxacin (CIP) is a second-generation fluoroquinolone antibiotic. It exhibits broad-spectrum antibacterial activity, particularly against Gram-negative bacteria such as *Escherichia coli*, *Pseudomonas aeruginosa*, *Salmonella*, and *Haemophilus influenzae*, and it is widely used to treat respiratory and urinary tract infections, skin infections,

infectious diarrhea, and typhoid fever [3,4]. However, it is associated with adverse effects on multiple systems in the human body and can accumulate in animal-derived foods, posing potential health risks to consumers [5,6]. The extensive use of CIP in both clinical and veterinary settings has led to its frequent detection in environmental compartments, including effluents from wastewater treatment plants, surface waters, sediments, and hospital wastewater [7,8]. The environmental presence of CIP contributes to selective pressure on microbial communities, promoting the emergence and proliferation of ciprofloxacin-resistant bacterial strains [2,9]. The reason for the increased concentration of CIP and other antibiotics in the environment is mainly due to their incomplete metabolism and improper elimination. In fact, a significant part of CIP is excreted unchanged, with approximately 65% eliminated through urine and around 25% through feces [10–12]. Drugs are introduced into the water and then end up in the wastewater treatment plants (WWTPs). However, conventional WWTPs were not originally designed to specifically remove pharmaceuticals, resulting in incomplete elimination and subsequent release of these contaminants into surface waters [6,13]. CIP is highly persistent and exhibits poor biodegradability, resulting in low removal efficiencies (below 40%) during conventional biological treatment [14].

Several technologies, such as advanced oxidation [15–17], membrane processes [18–21], and electrochemical oxidation [22,23], have been explored for CIP removal. Despite their advantages, these techniques also present several drawbacks, including the formation of toxic by-products, as well as high operational and maintenance costs [24,25]. Adsorption is widely applied for pharmaceutical removal due to its simplicity, safety, and environmental compatibility; however, conventional adsorbents are often expensive [26,27].

Recently, low-cost biosorbents derived from microbial, macrofungal, and algal biomass have gained attention for their availability and functional surface chemistry, enabling effective removal of pharmaceuticals even at low concentrations [28].

The fungal biomass from *Ganoderma* species has shown considerable promise due to its abundance, low cost, and rich surface chemistry. This biomass has been applied in biosorption studies of dyes, metal ions, and herbicides [29–35]. According to our literature review, only one study has been published specifically addressing the removal of pharmaceuticals, particularly progesterone, using *Ganoderma* biomass and its biochar as biosorbents [36].

The aim of the present study is to evaluate for the first time the biosorption potential of waste-pretreated *G. resinaceum* biomass for CIP removal from model aqueous solutions in a batch mode.

The optimum biosorption conditions were determined as a function of pH, biomass dosage, contact time, and initial CIP concentration. Equilibrium and kinetics studies were also carried out.

2. Materials and Methods

2.1. Reagents

All chemicals used in the study (ciprofloxacin hydrochloride monohydrate, $C_{17}H_{18}FN_3O_3 \cdot HCl \cdot H_2O$, NaOH, HCl, KCl, and NaClO) were purchased from Sigma-Aldrich (St. Louis, MO, USA) of analytical reagent grade and were used without further purification.

2.2. Biosorbent Preparation

G. resinaceum biomass was produced by submerged cultivation as described earlier in a previous work [37]. The fungal biomass investigated in this work is indeed a secondary by-product generated during mushroom cultivation for biotechnological production of polysaccharides and triterpenoids. After the recovery of these bioactive compounds the

remaining solid fraction was separated from the liquid phase by centrifugation at 4000 rpm (MPW-260R) for 10 min, washed with deionized water, and subsequently dried at 80 °C for 24 h in a drying oven. To oxidize the pigments, the biomass was treated with 1% NaClO [38]. After oxidation, the biomass was again washed with deionized water, dried, and stored at 4 °C until further use.

2.3. Characteristics of Pretreated *G. resinaceum* Biomass

2.3.1. Scanning Electron Microscopy

The morphological characteristics of unloaded and CIP-loaded *G. resinaceum* biomass were examined using scanning electron microscopy (SEM) (Prisma™ E SEM, Thermo Fisher Scientific, Waltham, MA, USA) equipped with an Energy Dispersive X-ray (EDX) analyzer (Thermo Fisher Scientific, Waltham, MA, USA) for elemental analysis. The samples were loaded on an aluminum sample holder, and the images were obtained at 10 kV acceleration voltage and magnification 800× using an ETD (Everhart–Thornley) detector (Thermo Fisher Scientific, Waltham, MA, USA).

2.3.2. Fourier Transform Infrared Spectroscopy

Fourier transform infrared (FT-IR) spectra of unloaded and CIP-loaded biomass were recorded at a 4000–400 cm^{−1} wavenumber range at a scan rate of 180 scans and a spectral resolution of 2 cm^{−1} using a Bruker Alpha II FT-IR spectrometer with a Diamond Crystal ATR (Bruker Corporation, Billerica, MA, USA).

2.3.3. pH at the Point of Zero Charge

The pH at the point of zero charge (pH_{pzc}) of the biomass was determined using the solid addition method [39,40]. For this purpose, 0.01 M KCl solutions were prepared and adjusted to an initial pH range of 2.0–10.0 using 0.1 M HCl or 0.1 M NaOH solutions. Subsequently, 0.1 g of biomass was added to Erlenmeyer flasks containing 50.0 mL of the pH-adjusted KCl solutions. The suspensions were continuously stirred at 25 °C for 48 h. The final pH of each solution was measured using a pH meter (Tecnal® TEC-2, Technal Scientific Equipment, Piracicaba, SP, Brazil). The pH_{pzc} of biomass was determined from the plot of ΔpH (pH_{final} − pH_{initial}) vs. pH_{initial}.

2.4. CIP Analysis

CIP solutions of different concentrations were prepared by appropriately diluting a 100 mg/L stock solution with deionized water. All solutions were prepared daily. The CIP concentration in the solutions before and after biosorption was quantified using a UV/VIS spectrophotometer Ultrospec 3300 (Amersham Biosciences, Piscataway, NJ, USA) at 274 nm by the calibration curve method [41].

2.5. Biosorption Studies

The effect of the following process parameters was investigated: the initial pH of aqueous solutions (4.0–9.0), mass of biosorbent (0.5–4 g/L), initial CIP concentration (4–20 mg/L), and contact time (10–120 min). The biosorption experiments were conducted in batch mode, in a series of Erlenmeyer flasks, each covered and sealed with aluminum foil to prevent decomposition of CIP. The pH values of the working CIP solutions were adjusted with 0.1 M HCl or 0.1 M NaOH. A measured quantity of biomass was added to 100 mL of each working CIP solution at the designated pH. The flasks were placed in a shaker at 150 rpm at a constant temperature. At the end of predetermined time intervals, the contents of flasks were centrifuged at 4000 rpm (MPW-260R) for 10 min, and the residual

concentration in the supernatants was determined as listed above. The removal efficiency (%) of CIP on *G. resinaceum* biomass was calculated using the following formula:

$$R = \frac{(C_0 - C_t)}{C_0} \times 100, \quad (1)$$

where C_0 and C_t are, respectively, the initial and at time t concentrations of CIP, mg/L.

CIP biosorption capacity (mg/g) in unit time was calculated from the following equation:

$$q = \frac{(C_0 - C_t)V}{W}, \quad (2)$$

where V is the volume of CIP solution (L); W is the mass of the biosorbent (g).

All measurements were performed in triplicate; average values are presented.

3. Results and Discussion

3.1. Biomass Characterization

3.1.1. SEM/EDX Analysis

The SEM micrographs of the unloaded and CIP-loaded biomass are shown in Figures 1a and 1b, respectively.

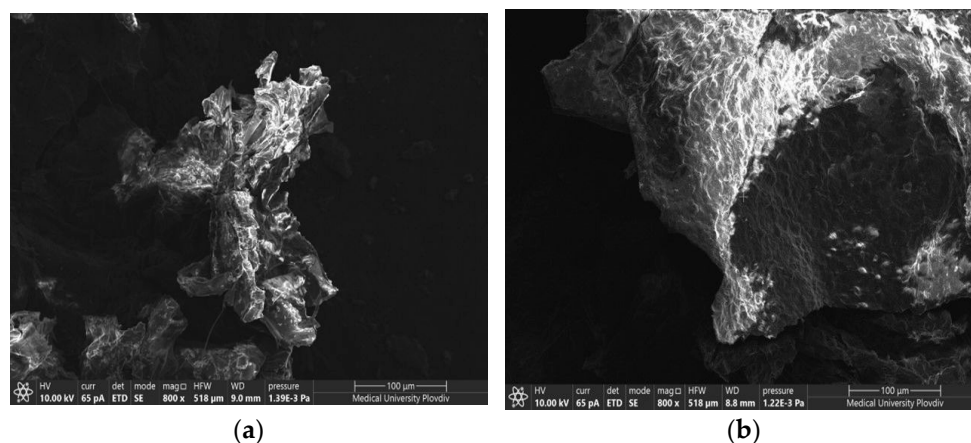


Figure 1. SEM micrographs of *G. resinaceum* biomass (magnification $\times 800$): (a) Unloaded biomass; (b) CIP-loaded biomass.

The unloaded biomass shows an irregular, porous, and rough surface structure with folded and wrinkled areas. These features indicate a heterogeneous surface morphology that could provide a large surface area and multiple sites favorable for biosorption. The presence of cavities and fibrous structures suggests the complex composition of the fungal cell wall. In contrast, the CIP-loaded biomass showed a smoother and denser surface morphology with a noticeable reduction in surface porosity. The partially filled voids and compact texture suggest biosorption/deposition of CIP molecules on the biomass surface.

Figure 2 shows the EDX spectra for *G. resinaceum* biomass before and after biosorption.

The EDX spectrum of the unloaded fungal biomass (Figure 2a) displayed dominant signals corresponding to carbon (~ 0.28 keV) and oxygen (~ 0.52 keV), consistent with the organic composition of the biomass. A minor signal attributable to nitrogen (~ 0.39 keV) was also detected but not labeled due to its low intensity. After CIP biosorption (Figure 2b), the intensities of oxygen and nitrogen peaks slightly increased. A new signal at approximately 0.68 keV was assigned to fluorine, indicating the biosorption of CIP on the biomass.

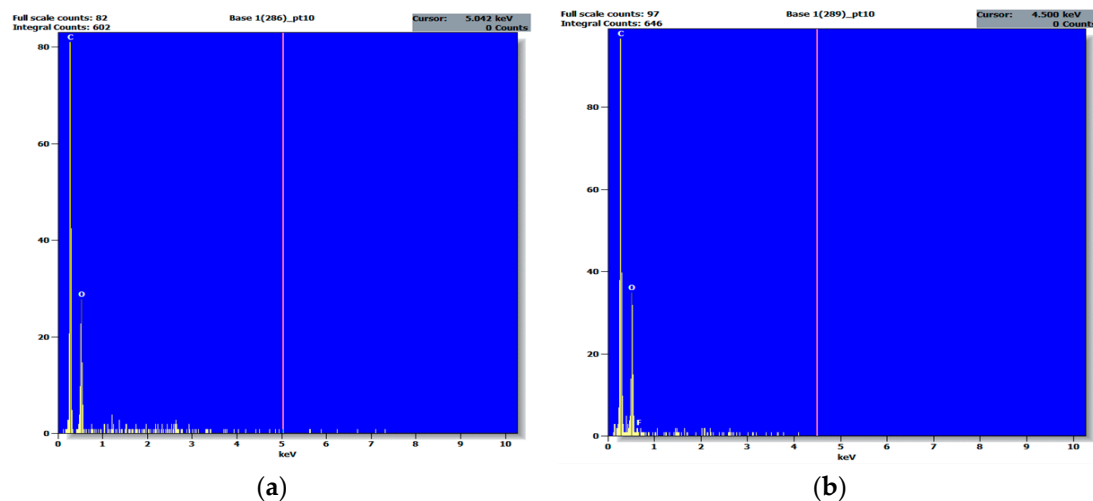


Figure 2. EDX spectra of *G. resinaceum* biomass: (a) Unloaded biomass; (b) CIP-loaded biomass.

3.1.2. FT-IR Analysis

FT-IR spectroscopy was used to elucidate the functional groups present in the *G. resinaceum* biomass before and after CIP biosorption (Figure 3).

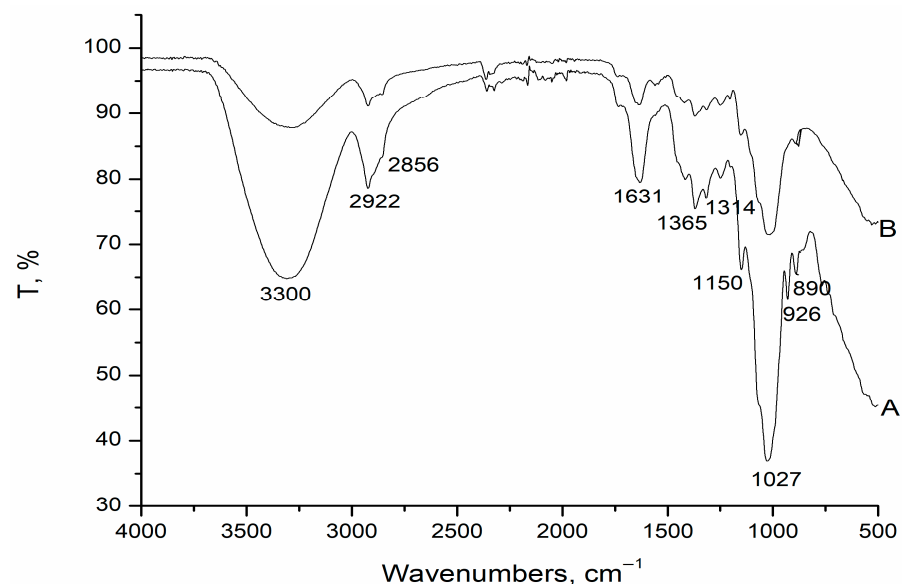


Figure 3. FT-IR spectra of unloaded (A) and CIP-loaded (B) *Ganoderma* biomass.

The FT-IR of the unloaded biomass (A) shows distinct peaks, confirming the complex structure of the fungal cell wall. The broad absorption band in the range of 3500–3100 cm^{-1} likely corresponds to overlapping —OH stretching vibrations from polysaccharides and —NH stretching vibrations from amide groups in proteins [42], making precise identification of the functional groups difficult. Weak bands observed at 2922 and 2856 cm^{-1} can be attributed to asymmetric and symmetric deformation of —CH_2 groups, respectively. The amide I band, associated with C=O stretching in carboxyl and amide groups, appears at 1631 cm^{-1} , while the less pronounced band at 1524 cm^{-1} corresponds to the amide II band [31,42]. A weak amide III band, representing C—N bond vibrations, is observed at 1314 cm^{-1} [43]. Additional bands are seen at 1150 and 1027 cm^{-1} (C—O stretching of carboxyl groups and bending vibrations of —OH groups) [33,36]. Bands between 950 and 750 cm^{-1} are attributed to the anomeric configurations of polysaccharides, consisting of glucans with various glycosidic linkages, for example, $(1 \rightarrow 3)\text{—}\alpha\text{—glucans}$, $(1 \rightarrow 6)\text{—}\beta\text{—}$

glucans, and hetero-glucans [43,44]. Therefore, the FT-IR analysis of biomass revealed the presence of hydroxyl, carboxyl, and amino groups on the biomass surface, which are potential sites for CIP binding. After CIP biosorption, significant alterations were observed in the FT-IR spectrum (B). The broad O–H/N–H stretching band around 3300 cm^{-1} decreased in intensity and shifted slightly to 3281 cm^{-1} . The peaks at 2922 and 2856 cm^{-1} exhibited minor shifts and a reduction in intensity. The bands at 1631 , 1365 , 1314 , 1150 , and 890 cm^{-1} also show significant reductions in intensity. Notably, a marked decrease was observed at 1150 and 890 cm^{-1} , while the peak at 926 cm^{-1} disappeared entirely. These spectral changes indicate that hydroxyl, carbonyl, and amine groups on the *G. resinaceum* biomass surface play a crucial role in CIP biosorption.

3.1.3. Point of Zero Charge

The point of zero charge (pH_{pzc}) refers to the pH at which the net surface charge of a biomass is zero [29]. When the pH of a solution is below the pH_{pzc} , the biomass surface tends to carry a net positive charge. Conversely, at pH values above the pH_{pzc} , the surface charge becomes predominantly negative. As shown in Figure 4, the *G. resinaceum* biomass exhibited a pH_{pzc} of 6.1.

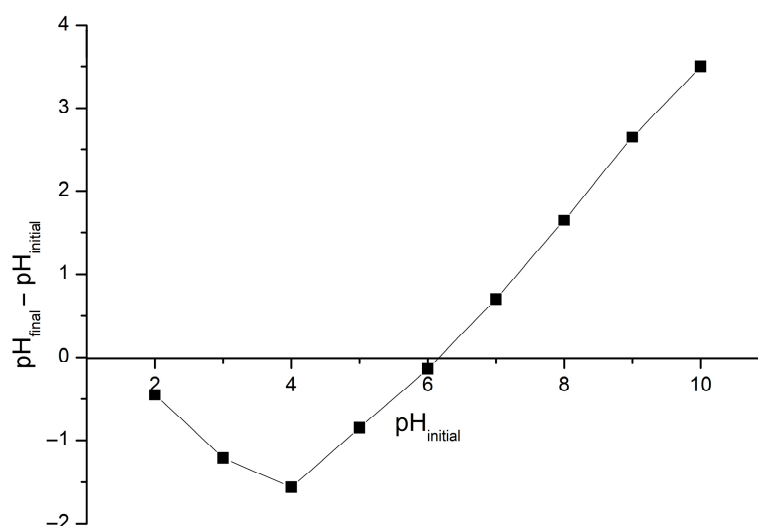


Figure 4. Point of zero charge of the pretreated *Ganoderma* biomass.

3.2. Biosorption Studies

3.2.1. Effect of Process Parameters on the Effectiveness of Ciprofloxacin Removal from Aqueous Solutions

The pH of the solution plays a crucial role in the biosorption process, as it influences both the ionization state of the biosorbate and the surface charge of the biosorbent. Many pharmaceuticals contain acidic and/or basic functional groups, allowing them to exist in anionic, cationic, neutral, or zwitterionic forms depending on the solution's pH. Therefore, the pH effect should be examined for each specific biosorbent–biosorbate system [28,40]. In this study, the pH effect on CIP removal was investigated over a pH range of 4.0 to 9.0.

CIP is an amphoteric compound with two main pK_a values: 5.90 ± 0.15 (for the carboxylic acid group) and 8.89 ± 0.11 (for the basic N-moiety) [45]. Consequently, it can exist as a cation—at $\text{pH} < 5.90$, a zwitterion—at pH between 5.90 and 8.89, and an anion—at $\text{pH} > 8.89$ (Figure 5).

The removal efficiency of CIP on *Ganoderma* biomass was found to be pH dependent (Figure 6).

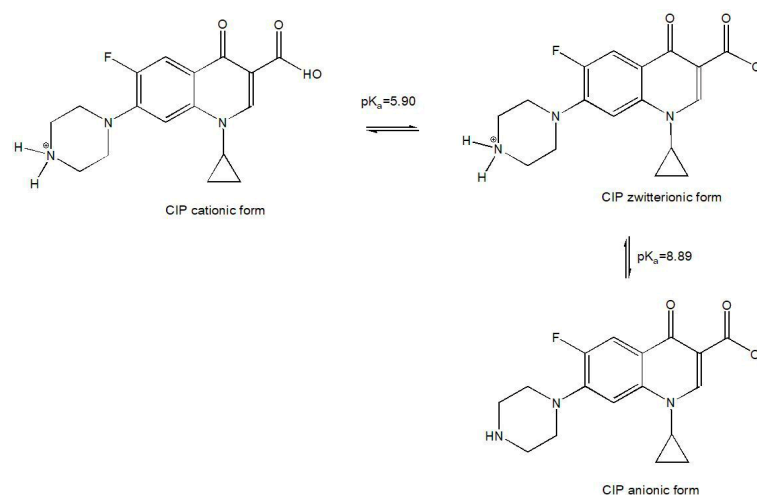


Figure 5. Ionic forms of CIP.

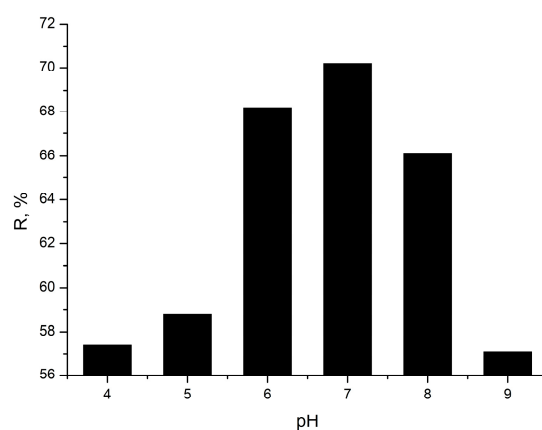


Figure 6. Effect of pH on the removal of CIP on *G. resinaceum* biomass (CIP concentration: 12 mg/L; biomass: 1 g/L; contact time: 120 min; temperature: 25 ± 0.2 °C).

The removal efficiency increased from 57.4% at pH 4.0 to a maximum of 70.2% at pH 7.0, then declined to 57.1% at pH 9.0. At pH values 4.0 and 5.0, electrostatic repulsion between the positively charged biosorbent surface and cationic CIP species led to reduced biosorption. As the pH increased beyond the pH_{pzc} , the surface became negatively charged, promoting interactions with zwitterionic ciprofloxacin through electrostatic interactions. At pH values greater than 8.0, CIP becomes negatively charged, leading to increased electrostatic repulsion with the biosorbent surface and thus decreased removal efficiency. It should be noted that non-electrostatic interactions are also possible, such as the formation of hydrogen bonds between electronegative atoms (e.g., N, F and O) in the CIP molecule and hydroxyl groups (-OH) on the biomass surface; a similar conclusion was made by other researchers [46,47]. Mirizadeh et al. [48] observed the highest CIP removal by magnetic chitosan/microalgae biocomposites at pH values around 6. At pH values greater than 6.0, the biosorption remained significant due to hydrogen bonding between CIP and biosorbent oxygen atoms. Recently, Salah et al. [49] reported a decrease in CIP removal efficiency by *Chlorella vulgaris* and *Synechocystis* sp. biomasses at pH values above 5.9, which was attributed to prevailing electrostatic repulsion between the CIP molecules and the algal biomass surfaces. In their study, maximum removal was observed at around pH 3.0, where electrostatic attraction dominated the biosorption process.

The next studies in this research were conducted at pH 7.0.

The effect of biomass dosage on CIP biosorption was studied over a range of 0.5 to 4 g/L, as shown in Figure 7.

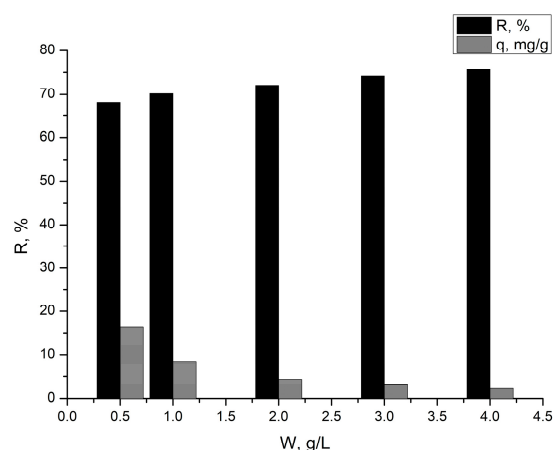


Figure 7. Effect of biomass concentration on CIP removal efficiency (CIP concentration: 12 mg/L; pH: 7.0; contact time: 120 min; temperature: 25 ± 0.2 °C).

An increase in biomass dosage from 0.5 to 4 g/L resulted in a rise in removal efficiency, from 68.1% to 75.8%. This improvement can be attributed to the increased surface area and the greater availability of binding sites for CIP molecules at higher biomass concentrations. However, the biosorption capacity (uptake) decreased significantly, from 16.34 to 2.27 mg/g. This inverse relationship is likely due to the aggregation of biomass particles at higher concentrations, which can reduce the effective surface area and cause site overlapping or saturation [31,50]. From an economic point of view, all subsequent experiments were conducted using a biomass concentration of 1 g/L to minimize material costs while maintaining effective performance.

The effect of contact time on the removal efficiency of CIP by *Ganoderma* biomass was examined at three initial concentrations over time intervals up to 120 min. As illustrated in Figure 8, the biosorption rate increased rapidly during the initial phase, followed by a slower uptake as contact time progressed, eventually reaching equilibrium.

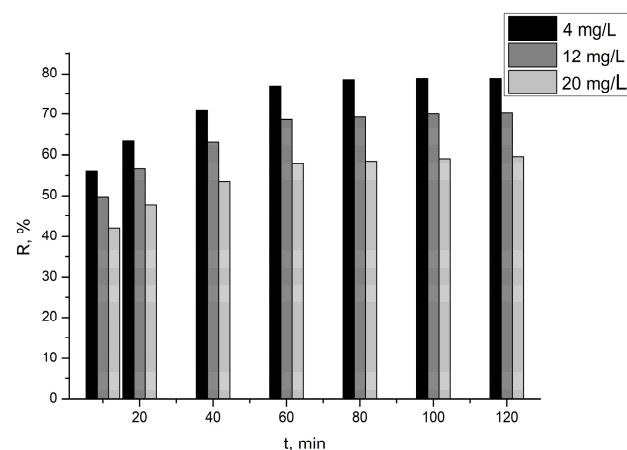


Figure 8. Effect of contact time on the removal of CIP on *Ganoderma* biomass (initial concentration: 4, 12, and 20 mg/L; pH: 7.0; biomass dosage: 1 g/L; temperature: 25.0 ± 0.2 °C).

An increase in the initial concentration of CIP generally resulted in decreased removal efficiency. This trend is likely due to the progressive saturation and blockage of available active sites on the biomass surface. At a contact time of 60 min, removal efficiencies of 77.0%, 68.7%, and 57.9% were recorded for the 4, 12, and 20 mg/L concentrations, respectively. Equilibrium was achieved at 120 min, with maximum removal efficiencies of 78.8%, 70.2%, and 59.6% for the respective concentrations. It was also observed that the initial CIP concentration had no significant effect on the time required to reach equilibrium, except at

the lowest concentration (4 mg/L), where equilibrium was attained more quickly, within 60–70 min. The present findings were in good agreement with earlier reports for various alternative biomasses [48,49]. Results of the present study indicated the efficiency of *G. resinaceum* biomass for rapid CIP uptake in short periods of contact time.

3.2.2. Kinetic Modeling

The biosorption kinetics data corresponding to CIP biosorption on *Ganoderma* biomass were fitted to Lagergren's pseudo-first order and Ho's pseudo-second-order models [50–52]. The linear form of the pseudo-first-order rate equation by Lagergren is given as:

$$\log(q_e - q_t) = \log q_e - \frac{2.303}{k_1} t, \quad (3)$$

where q_t (mg/g) and q_e (mg/g) are the biosorption uptakes at time t and at equilibrium time; k_1 is the rate constant of pseudo-first-order kinetics (min^{-1})

The Ho pseudo-second-order equation can be expressed in its linearized form by Equation (4):

$$\frac{t}{q_t} = \frac{1}{k_2 q_e^2} + \frac{1}{q_e} t, \quad (4)$$

where k_2 is the rate constant of pseudo-second-order kinetics (g/mg g)

By plotting $\log(q_e - q_t)$ versus t and t/q_t versus t for different initial CIP concentrations, straight lines were obtained as shown in Figures 9 and 10.

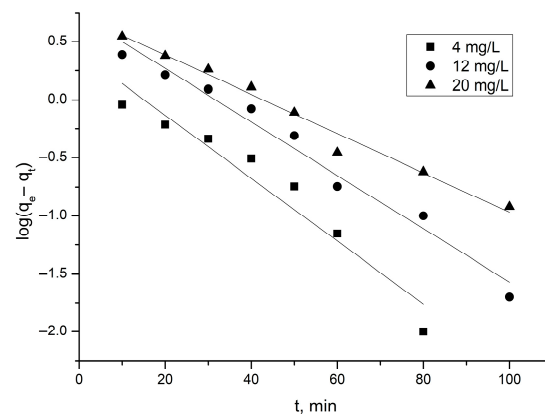


Figure 9. Pseudo-first-order model (pH: 7.0; biomass dosage: 1 g/L; temperature: 25.0 ± 0.2 °C).

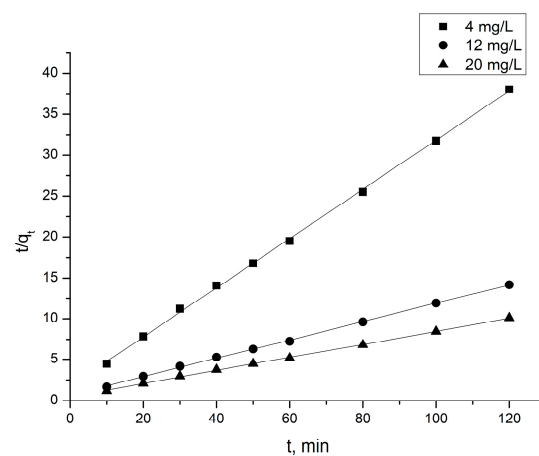


Figure 10. Pseudo-second-order model (pH: 7; biomass dosage: 1 g/L; temperature: 25.0 ± 0.2 °C).

The kinetic constants, calculated from the slopes and intercepts of the plots for both models, along with their corresponding correlation coefficients (R^2 values), are presented in Table 1.

Table 1. Kinetic parameters for the biosorption of CIP on *Ganoderma* biomass.

C_0 , mg/g	q_e^{exp} mg/g	Pseudo-First-Order Kinetic Model			Pseudo-Second-Order Kinetic Model		
		q_e^m , mg/g	k_1 , min^{-1}	R^2	q_e^m , mg/g	k_2 , g/mg min	R^2
4	3.15	2.56	0.0624	0.924	3.33	0.0509	0.999
12	8.42	5.36	0.0529	0.977	8.93	0.0187	0.999
20	11.93	5.28	0.0389	0.981	12.65	0.0130	0.999

The results indicate that the pseudo-second-order kinetic model provides a better fit for the kinetics of CIP biosorption, as evidenced by higher correlation coefficient (R^2) values. Moreover, the equilibrium uptake values (q_e^m) calculated from this model were much closer to the experimental values compared to those obtained from the pseudo-first-order model. These findings suggest that the biosorption of CIP onto *Ganoderma* biomass follows pseudo-second-order kinetics. The pseudo-second-order kinetic model is one of the most widely used models due to its high accuracy in describing the entire range of experimental kinetic data [53,54].

3.2.3. Equilibrium Modeling

Analyzing equilibrium data is essential for the practical design and optimization of biosorption systems. In this study, Freundlich and Langmuir adsorption models were used for the mathematical description of the biosorption equilibrium [55,56]. The Langmuir isotherm model assumes monolayer biosorption onto a surface containing a finite number of identical and energetically equivalent sites. In contrast, the Freundlich model is an empirical equation used to describe biosorption on heterogeneous surfaces, where it does not assume a maximum adsorption capacity. This model is more suitable for systems in which biosorption occurs on sites with varying affinities. The linearized forms of the Langmuir and Freundlich equations are typically expressed as follows:

$$\frac{C_e}{q_e} = \frac{1}{q_m k_L} + \frac{1}{q_m} C_e, \quad (5)$$

$$\log q_e = \log k_F + \frac{1}{n} \log C_e, \quad (6)$$

where q_e —ciprofloxacin uptake at equilibrium (mg/g); q_m —maximum biosorption capacity (mg/g); C_e —equilibrium concentration (mg/L); k_L —Langmuir constant related to the affinity of binding sites (L/mg); k_F —Freundlich constant ((mg/g)(L/mg)ⁿ); n —biosorption intensity (dimensionless).

By plotting C_e/q_e versus C_e and $\log q_e$ versus $\log C_e$, straight lines were obtained as shown in Figures 11 and 12.

The corresponding model parameters were calculated and summarized in Table 2.

A brief comparison showed that the Langmuir isotherm model provided a slightly better fit to the equilibrium data than the Freundlich model, as indicated by their respective correlation coefficients. This may suggest that the biosorption of CIP occurs predominantly as a monolayer process, and the surface of *G. resinaceum* comprises both homogeneous biosorption sites and heterogeneous domains with varying affinities [31,53].

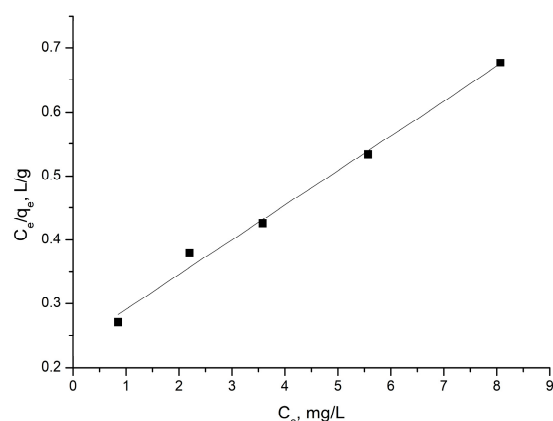


Figure 11. Langmuir isotherm model (initial CIP concentration: from 4 to 20 mg/L; pH: 7.0; biomass dosage: 1 g/L; contact time: 120 min; temperature: 25.0 ± 0.2 °C).

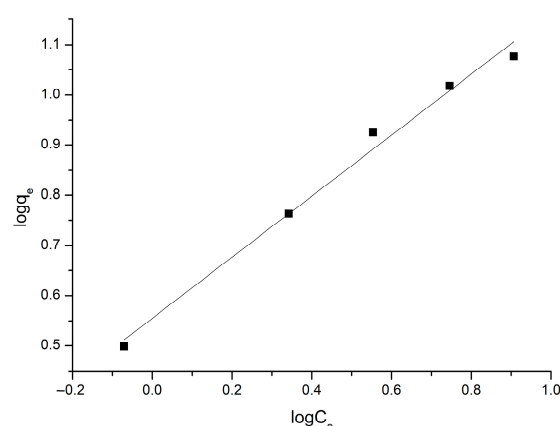


Figure 12. Freundlich isotherm model (initial CIP concentration: from 4 to 20 mg/L; pH: 7.0; biomass dosage: 1 g/L; contact time: 120 min; temperature: 25.0 ± 0.2 °C).

Table 2. Langmuir and Freundlich isotherm parameters for biosorption of CIP on *Ganoderma* biomass.

Langmuir Isotherm Model	Freundlich Isotherm Model
$q_m = 18.4 \text{ mg/g}$ $k_L = 0.229 \text{ L/g}$ $R^2 = 0.992$	$k_F = 3.59 \text{ (mg/g)(L/mg)}^{1.65}$ $n = 1.65$ $R^2 = 0.989$

In comparison to other microbial and algal biomaterials reported in the literature, the maximum biosorption capacity of *G. resinaceum* (18.4 mg/g) for CIP removal demonstrates moderate performance. Raw microbial and algal biomasses, such as activated sludge and waste *Chlorella vulgaris*, have shown considerably lower adsorption capacities, typically ranging from 2.9 to 7.6 mg/g [57,58]. While some chemically modified materials, such as algal-bacterial sludge-derived magnetic composites [59] and algal biochar [60] have exhibited higher capacities (e.g., 81.6–300 mg/g range). The preparation of these biosorbents often involve additional processing steps that increase cost and complexity.

4. Conclusions

An eco-friendly and cost-effective biosorbent was developed through chemical pre-treatment of waste *G. resinaceum* biomass. Its ability to remove CIP from aqueous solutions was tested under various experimental conditions. According to the Langmuir model, the maximum biosorption capacity was determined to be 18.4 mg/g. The biosorption process was relatively rapid, reaching equilibrium within 120 min. Kinetic analysis showed

that CIP biosorption followed a pseudo-second-order model. FT-IR analysis identified –OH, –NH, and C=O groups on the surface of the pretreated biomass, suggesting possible interactions between these functional groups and the CIP molecules. Future studies will focus on immobilizing the biomass on different carriers and exploring its regeneration.

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