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# Biosorption of Arsenic (III) and Arsenic (V) from Aqueous Solutions: Equilibrium and Kinetic Studies using Mangrove Leaf Biomass (Avicennia marina)

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### ABSTRACT

Due to the gravity of arsenic contamination and its potentially detrimental effects on public health, extensive endeavors have been undertaken to treat water sources harboring these hazardous pollutants. This study focuses on investigating the kinetics and equilibrium of biosorption of arsenic (III) and As(V) from aqueous solutions using mangrove leaf biomass (Avicennia marina). Optimal biosorption conditions were determined by considering the optimum pH, initial concentration of adsorbate, contact time, and adsorbent dose. Adsorption models like Langmuir's and Freundlich's were employed to describe the adsorption and determine the corresponding constants of the isotherm. The isothermal data for biosorption demonstrated acceptable compatibility for the Langmuir and Freundlich isothermal models. The maximum adsorption capacity of As(III) and As(IV) onto mangrove leaf biomass (Avicennia marina) was found to be 90.9 and 76.9 mg/g for As (III) and As (V), respectively. Moreover, the correlation between the kinetic experimental data and the second-order kinetic model was significant, indicating its suitability in describing adsorption.

# **INTRODUCTION**

The rapid progress of technology in modern times has brought about detrimental consequences for the natural environment, notably industrial pollution. An important aspect of this pollution is the transportation of heavy metals between soil and water, leading to metal contamination (Younis, 2020; Younis et al., 2023a; Younis et al., **2023b**). When present at high concentrations, these metals significantly impact various components of natural ecosystems (Soliman et al., 2020; El-Naggar, et al., 2021; Taher et al., 2023; Younis et al., 2023). Industries such as smelters, metal refineries, and mining operations are recognized as major contributors to releasing metals into the environment (Hanafy et al., 2021; Younis et al., 2022).

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Heavy metals are highly toxic, making their presence in the environment a major cause for alarm. Threatening human, animal, and ecological health, bioaccumulation of these metals throughout the food chain, even at relatively low concentrations. Recognizing the gravity of the situation, The USEPA made a list of organic and inorganic pollutants found in wastewater that are extremely dangerous to human health back in 1978 (**Kumar** *et al.*, **2006**).

Arsenic contamination stands out as one of our time's toughest environmental problems to solve. Countless people worldwide drink water contaminated by natural sources, relying on it as their sole supply of potable water. Alarmingly increased arsenic levels have been detected in water across several countries. This increased reliance on water resources for drinking purposes has led to a surge in health problems, encompassing various ailments such as neurological disorders, skin diseases, digestive disorders, and renal and cardiovascular diseases (**Berg** *et al.*, **2001; Nordstrom, 2002**). Arsenic is widely recognized as highly toxic to humans and animals and a potential carcinogen (**Thomas** *et al.*, **2001; Boddu** *et al.*, **2008**).

Moreover, recent studies have revealed that even at very low concentrations, arsenic can act as an endocrine disruptor (**Stoica** *et al.*, **2000**). In response to the situation's urgency, several regulatory bodies have lowered the arsenic MCL for drinking water from 50 to 10  $\mu$ g/L (**Smith** *et al.*, **2002**). Consequently, there is a pressing need to create economically viable and uncomplicated methods for removing arsenic from groundwater to ensure safe drinking water supplies (**Gibb** *et al.*, **2011**, **Sarı** *et al.*, **2011**).

Arsenic exhibits two oxidation states: arsenic (III) and arsenic (V) (**Pokhrel and Viraraghavan, 2006**). It infiltrates the aquatic system through diverse pathways, such as geochemical interactions and industrial waste discharge. Additionally, the application of arsenic-based pesticides in agriculture contributes to its presence. Arsenic contamination of underground water represents a critical public health concern in numerous countries, including Bangladesh, India, China, Iranian Kurdistan, Argentina, Canada, and the United States of America (IARC, 2012). Given that groundwater serves as the primary drinking water source in these regions, the contamination of arsenic poses a grave threat (Pal *et al.*, 2002).

Humans' health can suffer irreparable and devastating harm from chronic exposure to arsenic. Arsenic poisoning manifests in various ways, encompassing skin lesions, mucous membranes and gastrointestinal tract impairment, and respiratory, circulatory, and nervous system disturbances (**Murugesan** *et al.*, **2006**).

Numerous studies have provided evidence of the successful removal of hazardous pollutants from wastewater, particularly arsenic removal, through various techniques.

These include Fe-electrocoagulation/co-precipitation (Hansen *et al.*, 2006), metal oxide adsorption using activated alumina-packed beds (Hlavay and Polyák, 2005), reverse osmosis and nanofiltration (Urase *et al.*, 1999), ion-exchange resin (Vaaramaa and Lehto, 2003), continuous fixed bed adsorption (Maiti *et al.*, 2012) and nanomaterial adsorption (Younis *et al.*, 2019; Saleh *et al.*, 2019; Younis *et al.*, 2020; Younis *et al.*, 2023c). However, the benefits of these methods' superior performance and efficiency may be outweighed by the fact that they frequently require costly maintenance and use mineral adsorbents.

On the other hand, biosorption has emerged as an effective method for removing arsenic ions from aqueous solutions (Gerente *et al.*, 2010; Ayub, and Raza, 2021; Ayub *et al.*, 2022). This technique offers several notable advantages, including the reusability of biomaterial, low operating costs, enhanced selectivity for specific target metals, the ability to remove heavy metals from effluent regardless of their toxicity, short operation times, and the absence of secondary compound production that may pose toxicity concerns (Kamala *et al.*, 2005; Gupta, and Rastogi, 2008).

Mangrove leaf biomass stands out among biological materials as a promising biosorbent due to its economical accessibility, relatively large surface area, and strong binding affinity (**Elnaggar** *et al.*, **2022**). The biosorption capability of mangrove leaf biomass primarily stems from the structural composition of amino, hydroxyl, carboxyl, and sulfate functional groups found in its cell walls. These functional groups possess the ability to serve as metal binding sites through mechanisms such as ion exchange, complexation, and electrostatic attraction.

This research aimed to explore the biosorption potential of mangrove leaf biomass (*Avicennia marina*) for As (III) and As(V) ions using the batch biosorption technique to remove them from aqueous solutions. A comprehensive literature review determined that there is a lack of extensive research on the biomass of mangrove leaves and its ability to biosorb As (III). Consequently, this novel material was selected as the biosorbent in this study due to its natural composition, easy availability, and cost-effectiveness for removing dissolved metal ions. Langmuir and Freundlich isotherm models were used to analyze the obtained equilibrium data. Furthermore, the kinetics of the biosorption mechanisms for both As (III) and As(V) were investigated.

## MATERIALS AND METHODS

### 2.1 Biomass Preparation

In this study, the mangrove leaf biomass (*Avicennia marina*) samples were obtained from the coastal waters of the Egyptian Red Sea. Deionized water was used to cleanse the collected samples of any lingering ions or debris like sand or dirt. Subsequently, the biomass was heated after being sun-dried for two days at 105°C for 60 minutes until a

consistent sample weight was achieved. Afterwards, the mangrove leaf biomass was dried and sieved after being finely chopped for further use in the study.

# **2.2 Batch Biosorption Protocol**

Batch equilibrium adsorption isotherm studies were conducted to investigate As (III) and As(V) adsorption behavior in aqueous solutions. The experimental setup involved maintaining a constant solution volume of 100 mL in flasks containing 0.1 g of mangrove leaf biomass while varying a 10–100 mg/L arsenic (III) and arsenic (V) concentration. Sodium hydroxide or nitric acid of 1.0 M was used to achieve the desired pH levels in the solutions to ensure the appropriate pH conditions. The pH levels (from 3 to 9) were measured precisely using a pH meter. All of the batches were tested at ambient temperatures.

To achieve equilibrium, the system was subjected to continuous agitation at an agitation speed of 150 rpm for a duration of 120 minutes. Following the equilibration period, the biosorbent was separated from the solution by filtration using Whatman 45 mm filter paper. The resulting filtrate was then analyzed to determine the concentration of metals present.

A mass balance approach was employed to quantify the amount of metal adsorbed per unit mass of (qe). The equation used for this calculation is as follows:

$$q_e = \frac{Ci - Ce}{m} V \tag{1}$$

Where Ci and Ce are the initial and equilibrium concentrations of the metal (mg/L), and qe is the amount of metal adsorbed per unit mass of mangrove leaf biomass (g). Biosorbent's dry mass (m) is contrasted with the solution volume (V).

The biosorption percentage of As (III) and (V) was calculated according to the following formula:

Biosorption % of As = 
$$\frac{(Ci - Cf)}{Ci} * 100$$
 (2)

Where Ci (mg/L) represents the initial concentration of As(III) and  $C_f$  (mg/L) represents the final concentration of As.

The quantification of metal content in the supernatant was performed using ICP-MS (XSeriesII Thermo Scientific Inc., USA). The adsorption kinetics were investigated by sampling the system at regular intervals of 5, 10, 20, 40, 60, 80, and 100 minutes.

All of the batch experiments were done in triplicate, and the recorded data represent the average values. Results were presented with error bars to highlight the inherent uncertainty in the data.

The maximum arsenic sorption capacity of the biosorbent was estimated using the Langmuir sorption model. The equation for the Langmuir isotherm can be expressed as:

$$\frac{C_e}{Q_e} = \frac{1}{Q_{max}K_L} + \frac{C_e}{Q_{max}}$$
(3)

The adsorption energy is denoted by the Langmuir constant K (L/mg), and the maximum monolayer adsorption capacity of the adsorbent is denoted by Qmax. Following is a linearization of the Langmuir model that was used to fit the experimental data:

The Freundlich model, on the other hand, is described by the equation:

$$Log Q_e = log K_f + \frac{1}{n} Log C_e$$
(4)

The adsorption capacity of the adsorbent is represented by the Freundlich constant, Kf (mg/g), and the adsorption intensity is represented by the Freundlich exponent, 1/n (dimensionless).

## **RESULTS AND DISCUSSION**

### 3.1. pH Influence

Solution pH value holds significant importance as a governing parameter in sorption of metals. The impact of pH on the intake of metals is attributed to the biosorbent's surface functional groups and the metal's solution chemistry (**Rahman, and Islam, 2009**). In this study, As(III) biosorption onto mangrove leaf biomass was studied, and its relationship to pH was over a pH range of 3 to 9. Figure 1 displays the outcomes. The graph shows that the maximum biosorption efficiency of 80% was achieved at pH 6 for As(III). Consequently, all subsequent As(III) biosorption experiments were conducted at pH 6.

In addition, the graphs show that the sorption capacity of mangrove leaf biomass (*Avicennia marina*) for As(V) is pH-dependent, with an increase in sorption capacity observed as the pH decreases.

Several studies investigating the influence of pH on the sorption of As(III) using different sorbents have yielded consistent results: at this value, biosorption is at its peak (**Pokhrel and Viraraghavan, 2008**). Adsorption onto mangrove leaf biomass (*Avicennia marina*) as a mechanism for removing As(V) from an aqueous medium may be attributed to interactions with unprotonated amino groups. The biosorbent may contain hydroxyl groups, some of which may also participate in coordination with the adsorbate.



Figure 1. pH Influence on As(III) and As(V) removal during batch experiments with reaction conditions of 10 mg/L As, 0.1 g/50 mL adsorbent dose, and 40 minutes.

The dominant species responsible for the sorption of As(III) are considered to be the monoanionic form  $H_2AsO_3$  and the neutral form ( $H_3AsO_3$ ), which replace hydroxyl ions or water molecules (**Biswas** *et al.*, **2008**). Arsenite's speciation in an aqueous medium between pH 3 and 9 is controlled by the following reaction (**Reed** *et al.*, **2000**):

## $H_3AsO_3 \rightarrow H_2AsO_3^- + H$

It was observed that these arsenite species do not interact with the adsorbent via electrostatic forces. However, they were found to interact with unprotonated amino

groups present on the adsorbent. Additionally, certain hydroxyl groups within the sorbent were observed to coordinate with the sorbate, as reported by **Muñoz** *et al.* (2002).

The lower biosorption yield observed above pH 7 can be explained by the fact that OH ions predominate at alkaline pH, which is antagonistic to the  $H_2AsO_3$  species of arsenic (**Pokhrel and Viraraghavan, 2008**). Additionally, the biomass's carboxyl, hydroxyl, and amide groups become negatively charged under alkaline conditions. As a result, the sorption efficiency decreases because of the repulsion between the negatively charged biomass and the anions  $H_2AsO_3$  and  $HAsO_3$  (**Rahaman** *et al.*, **2008**).

Generally, at pH 6, the surface charge of dry mangrove biomass can be influenced by the presence of functional groups on its surface. These functional groups, such as hydroxyl (-OH), carboxyl (-COOH), and amino (-NH2) groups (**Yang** *et al.*, **2013**), can have different pKa values, which determine their ionization behaviour at a given pH. For example, the pKa of amino groups ranges from 8 to 11.

If the pH is below the pKa of a functional group, it will be protonated and carry a positive charge. Conversely, if the pH is above the pKa, the functional group will be deprotonated and carry a negative charge. The overall surface charge of the biomass will depend on the balance between the different protonated and deprotonated functional groups.

The adsorption of As(III) and As(IV) onto the surface of the biomass can potentially alter the pKa values of the functional groups due to changes in the local chemical environment. The interaction between the arsenic species and the surface functional groups can result in the transfer of electrons and modification of the ionization behavior of the functional groups (**Ranjan** *et al.*, 2022).

# **3.2. Impact of Contact Time**

The rate of biosorption plays a crucial role in the design of batch sorption experiments (**Anayurt** *et al.*, **2009**). To assess contact time's impact on biosorption, the experiment was conducted under the following conditions: 10 mg/L of metals, 0.1 g/50 mL of biomass, and pH 6. Arsenic ions were absorbed onto mangrove leaf biomass (*Avicennia marina*) for 40 minutes, and this was sufficient to reach equilibrium. Subsequently, further increases in contact time did not result in significant changes in adsorption. Hence, the equilibrium values for arsenic were determined at 40 minutes, as depicted in Figure 2.



Figure 2. Contact time influence on As(III) and As(V) removal during batch experiments with reaction conditions of 10 mg/L As, 0.1 g/50 mL adsorbent dose at pH 6.

### 3.3. Dosage Effects of Adsorbents

Figure 3 illustrates the influence of adsorbent dosage on the percentage of removal at equilibrium conditions. The results demonstrate that As(III) and As(V) adsorption depends on the dosage of the adsorbent. As the amount of adsorbent used keeps increasing from 0.01 to 0.2 g, the amount of As(III) and As(V) absorbed also increases. The results showed that the percentage of As(III) and As(V) removal increased from 48% to 82% and from 45% to 77%, respectively, with increased biomass dosage.

The enhanced biosorption achieved by increasing the biomass concentration is attributable to various factors, including the greater surface area of the biomass, improved solute availability, electrostatic interactions, and possible interference between binding sites (**Kumar** *et al.*, **2010**). The elimination of metals followed a similar pattern using *Azadirachta indica* as an adsorbent (**Pokhrel and Viraraghavan, 2008**).



Figure 3. Adsorbent dose influence on the removal of As (III) and As(V) during batch experiments with reaction conditions of 10 mg/L As at pH 6 for 40 minutes.

### **3.4. Initiation Concentration Impact**

The impact of the initial concentration of As(III) on removal efficiency is depicted in Figure 4. The batch trials were carried out with a fixed biosorbent dosage of 0.1 g/50 mL and a pH value of 6. As anticipated, a rise in the initial concentration of As(III) is observed, while maintaining all other parameters at a constant level, decreasing the removal percentage. The results revealed a rapid decline in removal efficiency from approximately 80 to 31% and 75 to 28% when As(III) and As(V) concentrations were elevated from 10 to 200 mg/L. The decline in adsorption % can be explained by the limited surface area available to accommodate a larger quantity of metal present in the solution. While arsenic equilibrium absorption tends to increase with increasing concentration, the adsorption percentage tends to decrease. At lower concentrations, ions of arsenic in the solution can effectively incite binding site interaction, resulting in a higher percentage of adsorption compared to higher arsenic ion concentrations. Higher concentrations, however, lead to saturation of the adsorption sites and decreased adsorption yields (**Amin et al., 2006; Mamisahebei et al., 2007**).



Figure 4. Initial adsorbate concentration influence on the removal of As(III) and As(V) during batch experiments with reaction conditions of 0.1 g/50 mL adsorbent dose, and 40 minutes at pH 6.

Therefore, to increase the purification yield, it is advisable to dilute wastewater that contains high concentrations of metal ions. By doing so, the available adsorption sites can accommodate a greater proportion of the target metal, improving the overall effectiveness of the adsorption.

## 3.5. The Isotherms of Adsorption

Using the Langmuir and Freundlich adsorption isotherms, we correlated the amount of metal adsorbed per unit mass of mangrove leaf biomass, qe (mg/g), with the equilibrium liquid-phase concentration, Ce (mg/L). The Langmuir equation is given by

$$qe = \frac{Q1*Ce}{1+b*Ce} \tag{5}$$

The Langmuir constants are denoted by Q1 (mg/g) and b (L/mg). Q1 is the monolayer coverage equivalent of metal ion adsorption. Figure 5 depicts the linear plot of 1/qe vs

1/Ce, which was used to calculate the parameters. For the time being, we can write the Freundlich adsorption isotherm as:

$$q_e = k * C_e^{(1/n)}$$
 (6)

In which k and n are constant values specific to the absorbent-adsorbate system. The model constants and Table 1 provide values for the regression coefficient ( $R_2$ ).



Figure 5. Langmuir Adsorption Isotherms for Arsenic (III) and Arsenic (V) on Mangrove Leaf Biomass

The dimensionless parameter RL is determined by the Langmuir constant, b, which is related to the energy of adsorption, using the equation:

$$\mathbf{R}_{\mathrm{L}} = \frac{1}{(1 + \mathbf{b} * \mathbf{Co})} \tag{7}$$

Where Co represents the initial concentration of the metal ion (mg/L), and b is the Langmuir constant (L/mg). The  $R_L$  values indicate whether the isotherm is irreversible

 $(R_L = 0)$ , favorable  $(0 < R_L < 1)$ , linear  $(R_L = 1)$ , or unfavorable  $(R_L > 1)$ . The calculated  $R_L$  values at different initial concentrations of As(III) and As(V) (Table 3) indicate that the adsorption is favorable, as the RL values for both As(III) and As(V) lie between 0 and 1.

Adsorption model	Parameters	As (III)	As(V)
Langmuir	Q <sub>max</sub>	90.9	76.9
	K <sub>L</sub>	0.0055	0.001
	R <sub>L</sub>	0.94	0.98
	R <sup>2</sup>	0.99	0.99
Freundlich	1/n	1	0.95
	K <sub>F</sub>	0.5	1.30
	$R^2$	0.99	0.99

Table 1. Modelling Adsorption Isotherm Parameters for Arsenic (III) and Arsenic (V)

Results from this study's comparison to earlier work show that mangrove leaf biomass has a significantly better adsorption capacity than many other adsorbents when it comes to removing As(III) and As(V). In comparison to activated alumina, which has an adsorption capacity of 3.5 and 15.9 mg/g for As(III) and As(V), respectively (**Lin and Wu, 2001**), mangrove leaf biomass shows an adsorption capacity of 90.9 and 76.9 mg/g. Both the Langmuir and the Freundlich adsorption models provide a good fit to the experimental data (Figure 6).



Figure 6. Langmuir Adsorption Isotherms for Arsenic (III) and Arsenic (V) on Mangrove Leaf Biomass

### **3.6.** Kinetics of Adsorption

Understanding the adsorption rate is of utmost significance when designing batch adsorption systems and determining the optimal operating conditions for large-scale processes. Two kinetic models are employed to gain a deeper understanding of the kinetics involved in adsorption data: the pseudo-first-order and the pseudo-second-order models. These models provide valuable insights into the uptake of solutes, thereby facilitating the precise estimation of reactor volume. Now, let us delve into the intricacies of these models to comprehend their functionality and significance in adsorption kinetics.

### **3.6.1.** The Pseudo-First-Order Equation

Typically, the pseudo-first-order equation, originally proposed by Lagergren and Kungliga, is conventionally expressed in the following manner:

$$\frac{\mathrm{d}qt}{\mathrm{d}t} = K_1 \left( q - q_t \right) \tag{8}$$

In this context, the variables q and qt denote the sorption capacities at equilibrium and time t, respectively, measured in milligrams per gram (mg/g). Additionally, k1 represents the rate constant associated with pseudo-first-order sorption, expressed in units of inverse

minutes (1/min). Upon performing the integration of the equation and imposing the relevant boundary conditions (namely, the requirement that the quantity q is equal to zero at time t equals zero), the resultant form of the equation after integration can be expressed as follows:

$$Log (q - qt) = log (q) - (k_1/2.303) * t$$
 (9)

However, Equation (9) deviates from a genuine first-order equation in two distinct manners:

- 1. The parameter k1(q qt) does not accurately reflect the quantity of available sites.
- 2. In contrast to a genuine first-order sorption reaction, it is common for the parameter log(q) to not align with the intercept of a graph illustrating log(q qt) against t.

To utilize Equation (9) for the analysis of experimental data, it is essential to have knowledge of the equilibrium sorption capacity, denoted as q. Nevertheless, in several instances, the exact value of q is unknown, and due to the relatively slower nature of chemisorption, the amount adsorbed usually remains significantly lower than the equilibrium amount. Despite the fact that the pseudo-first-order equation proposed by Lagergren typically provides a good fit for the initial 20-30 minutes of the process of adsorption, it may not be suitable for the entire contact time range. Furthermore, it becomes necessary to extrapolate the experimental data to  $t = \infty$  and consider q as a variable obtained through trial and error. Consequently, it is worth noting that in over 50% of the literature references, the authors did not directly measure an isotherm of equilibrium when analyzing sorption kinetics (**King et al., 2007**).



Figure 7: Pseudo-Second-Order Kinetic Plots at As (III) and As (V) Concentration: 10 mg/L; pH: 6; Biomass Concentration: 0.1 g/50 mL)

The slope of the plot between log (q - qt) and time, t, can be used to calculate the pseudofirst-order rate constant,  $k_1$ . Table 2 displays the estimated values of  $k_1$  as well as their related linear regression correlation coefficient values. In particular, the linear regression correlation coefficients, or  $R_2$ , for As (III) and As (V) are discovered to be 0.74 and 0.69, respectively. These results show that the pseudo-first-order model accurately predicts the adsorption kinetics of both arsenic species.

## 3.6.2. The Equation of Pseudo-Second Order

The pseudo-second-order chemisorption kinetic rate equation is as follows when the sorption rate adheres to a second-order mechanism:

$$\frac{dqt}{dt} = k(q - qt)^2$$
(10)

In this context, the symbols q and qt stand for the sorption capacity at equilibrium and time t, respectively (both expressed in mg/g), while k stands for the pseudo-second-order sorption rate constant (expressed in g/mg min). The integral form of Equation (10), taking into account the boundary conditions (qt=0 at t=0), is as follows:

$$\frac{1}{(q-qt)} = \frac{1}{q} + \mathbf{K}_{t} \tag{11}$$

This equation can be rearranged to:

$$q_{t} = \frac{t}{\frac{1}{Kq^2} + (\frac{t}{q})}$$
(12)

which can be expressed in linear form as:

$$\frac{t}{qt} = \frac{1}{k q^2} + \frac{1}{q} q$$
 (13)

The variables q and qt reflect the solute amounts adsorbed at equilibrium and at any time t, respectively (in mg/g), while the variable t denotes the contact time (in minutes) in the particular setting. Estimating an effective q is avoided in equation (13). Plotting t/qt against t according to Equation (13) should produce a linear relationship if pseudo-second-order kinetics are applicable, allowing for the derivation of q and k from the slope and intercept of the plot (Figure 7) without the need for prior parameter knowledge.

Table 2: Kinetic Parameters Derived from Pseudo-First Order and Pseudo-Second-Order Models

Caulerpa	First- order			Second-order		
prolifera	<b>K</b> <sub>1</sub>	q <sub>e</sub>	$\mathbf{R}^2$	K <sub>2</sub>	<b>q</b> <sub>e</sub>	$\mathbf{R}^2$
As(III)	0.0054	1.728	0.74	12.21937	4.3478	0.996
As(V)	0.0056	1.7629	0.69	9.384084	4.1666	0.996

Table 2 presents the values of the pseudo-second-order rate constant, denoted as  $k_2$ , as well as the estimated q value and the accompanying linear regression correlation coefficient values, denoted as  $R^2$ . It is worth noting that the  $R^2$  consistently demonstrates higher values (0.996 for both As (III) and As (V)) across all baseline arsenic concentrations. The high  $R^2$  values observed in this study provide strong evidence that the adsorption data is in close agreement with the pseudo-second-order kinetics model. This finding further supports the hypothesis that chemisorption is the dominant mechanism driving the adsorption process.

### 4. Conclusion:

The current study aimed to examine the adsorption properties of both As(III) and As(V)on the biomass of mangrove leaves (Avicennia marina) under conditions of both equilibrium and dynamic adsorption. The impact of pH on the degree of adsorption was investigated. The equilibrium adsorption data were subjected to fitting using the Langmuir and Freundlich isotherm models, and afterwards, the isotherm parameters were assessed. All of the models effectively depicted the experimental data. The sorbent's monolayer adsorption capacity for both As(III) and As(V) was found to be around 90.9 mg/g for each species. The adsorption capacity of As(III) was found to be larger than that of As(V), which can be attributed to the speciation of arsenic at a pH of 6.0. The concentration of arsenic employed in this investigation to evaluate the efficacy of the biosorbent was approximately 10 mg/L. Additional adsorption tests should be undertaken in a cyclic manner, employing lower concentrations, in order to effectively utilize this adsorbent for the purpose of arsenic remediation in wastewater treatment. The findings of the current study suggest that the biosorbent obtained from the biomass of Avicennia marina, a type of mangrove leaf, shows promise in its ability to effectively remove arsenic from wastewater.

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