A comparative Isothermal and Kinetic Study of the Adsorption of Lead (II) from Solution by Activated Carbon and Bentonite

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ABSTRACT

This work is aiming to study and compare the removal of lead (II) from simulated wastewater by activated carbon and bentonite as adsorbents with particle size of 0.32-0.5 mm. A mathematical model was applied to describe the mass transfer kinetic.

The batch experiments were carried out to determine the adsorption isotherm constants for each adsorbent, and five isotherm models were tested to choose the best fit model for the experimental data. The pore, surface diffusion coefficients and mass transfer coefficient were found by fitting the experimental data to a theoretical model. Partial differential equations were used to describe the adsorption in the bulk and solid phases. These equations were simplified and then solved using a technique with finite elements and orthogonal collection method for the bulk fluid and intraparticle phases, respectively. The results obtained from this work show that the equilibrium adsorption isotherms are favorable, and fitted well using the Freundlich model for activated carbon and bentonite. The activated carbon has a high value of pore diffusion coefficient, D_p, while bentonite has a high value of surface diffusion coefficient, D_s. This indicates that the pore diffusion controls the adsorption process for activated carbon, and the surface diffusion controls the adsorption process for bentonite. The activated carbon was more efficient than bentonite in removing of lead (II) from simulated industrial wastewater.

Key words: adsorption, lead (ll), activated carbon, bentonite, isotherm, kinetic

دراسة مقارنة ازالة الرصاص من محلول بواسطة الكاربون المنشط والبنتونايت

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الخلاصة

يهدف هذا البحث الى دراسة ومقارنة ازالة الرصاص من مياه الصرف بواسطة الكاربون المنشط والبنتونايت كمواد ممتزه بحجم حبيبي 0.32-0.5 ملم . تم تطبيق موديل رياضي لوصف عملية انتقال الماده بالاضافه الى اجراء تجارب الدفعات لتحديد ثوابت موديلات الامتزاز لكل ماده ممتزه وقد استخدم خمسه موديلات وتم فحصها لبيان الافضل تطابقا مع النتائج العمليه .

وتم ايضا ايجاد معامل الانتشار السطحي والمسامي, ومعامل انتقال الماده عن طريق مطابقة النتائج العمليه مع الموديل النظري وتم ايضا استخدام معادلات تفاضلية جزئيه لوصف عمليه الامتزاز وجرى تبسيط هذه المعادلات وحلها باستخدام تقنيه finite element و orthogonal collection عن طريق برنامج MATLAB . النتائج العمليه اثبتت ان معامل الانتشار المسامي D_P كان اعلى قيمة للكاربون المنشط منه للبنتونايت وان معامل الانتشار السطحي D_S كان اعلى قيمة للبنتونايت منه للكاربون المنشط وهذا يؤشر بان الانتشار المسامي هوالمتحكم في عمليه الامتزاز على الكاربون وان الانتشار السطحي هو المتحكم في عملية الامتزاز على البنتونايت وقد اثبتت النتائج ايضا بان الكاربون المنشط اكثر كفاءة من البنتونايت في ازالة الرصاص من مياه الصرف .

الكلمات الرئيسية: امتزاز, رصاص, كاربون منشط, بنتونايت, ايزوثيرم, حركي.

1. INTRODUCTION

Heavy metals as pollutants, are not biodegradable, and tend to accumulate in living organisms, **Kobya, et al., 2005.** Lead is one such heavy metal, and can be introduced into liquid wastes from different industries. In water, lead tends to accumulate in aquatic organisms through the food chain and by direct uptake. Lead can damage practically all tissues, particularly the kidneys and the immune system. Intense exposure to high levels of lead (from 10 to 20 g/day) causes encephalopathy with the following symptoms: vertigo, insomnia, migraine, irritability and even convulsions, seizures and coma. There is a critical need to treat wastewater so as to bring the concentration of toxic elements below the recommended release limit, **Smith, 1992.**

Conventional techniques for removing dissolved heavy metals from wastewater includes chemical precipitation, adsorption, electrolytic recovery, ion-exchange, chelation and solvent extraction or liquid membrane separation **,Gõksungur, et al., 2005.** Adsorption is one of the methods commonly used to remove heavy metal ions from various aqueous solutions. The efficiency of adsorption relies on the capability of the adsorbent to adsorb metal ions from solutions onto its surfaces. Different naturally occurring adsorbents, like activated carbon, tea waste, egg shells, mineral mixtures, are used as adsorbents for industrial waste management. Furthermore, bentonite has been used as an adsorbent for the removal of Pb (II), Fe (II), Zn (II) and Cr (III), **Tahir, and Rauf, 2004.**

2. ADSORPTION ISOTHERM MODELS

The study of adsorption isotherms is fundamental for supplying essential information required for designing the sorption process. The adsorption of a substance from one phase to another leads to a thermodynamically defined distribution of that substance between the phases as the system reaches a state of equilibrium, **Naiya, et al., 2009.** This distribution can be expressed in terms of adsorption isotherms. In this investigation, Langmuir, Freundlich, Radk-Prausnitz, Reddlich-Peterson, and a Combination of Langmuir-Freundlich isotherm models were used to analyze the equilibrium data.

2.1 Langmuir Model

In 1916, Irving Langmuir published a new model isotherm for gas or liquid adsorbed onto solid, which resulted in the model being named after him. The Langmuir adsorption model is valid for single-layer adsorption. It is based on the assumption that maximum adsorption corresponds to a saturated monolayer of solute molecules on the adsorbent surface. The energy of adsorption is constant, and there is no transmigration of adsorbate in the plane of the surface. The Langmuir isotherm equation is

$$q_e = \frac{q_m b C_e}{1 + b C_e} \tag{1}$$

where q_e (mg/g) is the amount of solute adsorbed per unit weight of adsorbent at equilibrium, C_e (mg/l) is the equilibrium concentration of the solute in the bulk solution, q_m is the maximum adsorption capacity, and b is the constant related to the free energy of adsorption. The Langmuir isotherm is used most frequently to describe the adsorption isotherm which is limited by the assumptions of uniform energies of adsorption on the surface of the adsorbent. It is based on four assumptions **,Lucas and Cocero, 2004,** namely:

- The surface of the adsorbents is uniform, that is, all the adsorption sites are equivalent.
- There is no interaction between molecules adsorbed on neighboring sites.
- All adsorption occurs through the same mechanism.

- Molecules are adsorbed at the defined sites on the adsorbent's surface.

2.2 Freundlich Model

The first mathematical fit to an isotherm was published by Freundlich and Kuster in 1894. Freundlich showed that adsorption from a solution could be expressed by the empirical formula, **Lucas and Cocero, 2004.**

$$q_e = kC_e^{1/n} \tag{2}$$

Where k (mg/g)(l/mg)$^{1/n}$ is the Freundlich sorption coefficient and n is an empirical coefficient indicative of the intensity of the adsorption.

2.3 Radk-Prausnitz Isotherm Model

The Radk-Prausnitz isotherm model **, Radke and prausnitz, 1972** was expressed empirically by the following equation:

$$q_e = \frac{K_{RP} C_e}{1 + \left(\frac{K_{RP}}{F_{RP}}\right) C_e^{1-N_{RP}}} \tag{3}$$

Where K_{RP}, F_{RP} and N_{RP} are the model parameters which are obtained by a nonlinear statistical fit of the equation to the experimental data. The Radk-Prausnitz equation has several important properties making it suitable for use in many adsorption systems.

2.4 Reddlich-Peterson Model

The Reddlich-Peterson isotherm model, **Lucas and Cocero, 2004** was expressed by the following equation:

$$q = \frac{A_R C_e}{1 + B_R C_e^{m_R}} \tag{4}$$

where; A_R, B_R and m_R are the model parameters.

2.5 Combination of Langmuir-Freundlich Model

The Sips model for single component adsorption, **Sips, 1984,** is

$$q_e = \frac{b q_m C_e^{1/n}}{1 + b C_e^{1/n}} \tag{5}$$

3. KINETIC STUDY

3.1 Intraparticle Transport

Intraparticle transport is also known as internal mass transfer. After the adsorbate passes through the hydrodynamic boundary layer, they are transported through the adsorbent to the available adsorption sites. Intraparticle transport may occur by molecular diffusion through the solution in the pores (pore diffusion), or by the diffusion along the adsorbent surface (surface diffusion) or both combined, pore and surface diffusion.

3.1.1 Pore diffusion

This is the diffusion of molecules in the liquid that fills the pores. The adsorbate first diffuses through the liquid filling the pores and is then adsorbed. The driving force for pore diffusion is the concentration gradient of the liquid in the pore, **Ivars, 1976.**

If the adsorbate is desorbed and repeatedly dissolved in the liquid, pore diffusion is prevalent. The diffusion of adsorbate within the pores is described by the pore diffusion coefficient. The unsteady state equation of the diffusion of species through a spherical particle may be written as

$$\rho_p \frac{\partial q_i}{\partial t} + \varepsilon_p \frac{\partial C_{pi}}{\partial t} = D_{pi}\varepsilon_p \left(\frac{\partial^2 C_{pi}}{\partial r^2} + \frac{2}{r}\frac{\partial C_{pi}}{\partial r} \right) \tag{6}$$

where the pore diffusion coefficient, D_{pi}, is assumed to be independent of concentration in this case, **Quek and Al-Duri, 2007.**

3.1.2 Surface diffusion

This is the diffusion of molecules in the adsorbed state. Molecules' migration takes place along the surface when an adjacent adsorption site is available and the molecules have enough energy to leave the site being presently occupied by them. This is described by the surface diffusion coefficient D_s. The driving force is the local adsorbent-phase gradient **,Hand, et al., 1983.**

The mass transport is described by the following unsteady state homogenous diffusion equation **,Fritz, et al., 1981.**

$$\frac{dq_i}{dt} = D_{si}\left(\frac{\partial^2 q_i}{\partial r^2} + \frac{2}{r}\frac{\partial q_i}{\partial r} \right) \tag{7}$$

3.1.3 Pore and surface diffusion

Pore and surface diffusion occur in parallel within the adsorbent particle. The solute diffusion inside the particle is described as

$$\rho_p \frac{\partial q_i}{\partial t} + \varepsilon_p \frac{\partial C_{pi}}{\partial t} = D_{pi}\varepsilon_p \left(\frac{\partial^2 C_{pi}}{\partial r^2} + \frac{2}{r}\frac{\partial C_{pi}}{\partial r} \right) + \rho_p D_{si}\left(\frac{\partial^2 q_i}{\partial r^2} + \frac{2}{r}\frac{\partial q_i}{\partial r} \right) \tag{8}$$

The adsorbent kinetics is governed by external mass transfer and internal diffusion, that is, by the coefficients K_f, D_p and D_s. In order to simplify the simulation, models that take into account only two coefficients were developed.

The effective intraparticle diffusion coefficients D_p and D_s are derived from a typical concentration decay curve for each solute in a certain adsorbent, with the help of an iterative search technique to minimize the sum of the square of residuals. Each iteration involves the numeric solution of a coupled set of partial differential equations with appropriate initial and boundary conditions that describe the film and intraparticle transport mechanisms in batch-wise experiments, **Liu, et al., 2006.**

4. MATHEMATICAL MODEL FOR ADSORPTION

In the present study, a mathematical model for batch adsorption will be employed following the assumption that homogeneity exists throughout the batch system. Accordingly, the model will be referred to as a pore and surface diffusion model. The model equations can be written as follows:

4.1 Mass Balance inside Particle Phase

This equation can be used for intra-particle mass transfer, **Hand, et al., 1983.**

$$\frac{1}{r^2}\frac{\partial}{\partial r}\left[r^2 D_s\rho_p \frac{\partial q}{\partial r}(r,t)+r^2 D_p\varepsilon_p\frac{\partial C_p}{\partial r}(r,t)\right]=\frac{\partial}{\partial t}\left[\rho_p q(r,t)+\varepsilon_p\, C_p(r,t)\right] \tag{9}$$

4.2 Mass Balance in Fluid-Bulk Phase

The solute concentration $C(t)$ in the vessel is given by following equation **,Ping and Guohua, 2001.**

$$\frac{dC(t)}{dt}=-\frac{3k_f W_o}{RV_L\rho_p}\left[C(t)-C_p\left(r=R_p,t\right)\right] \tag{10}$$

4.3 Initial and Boundary Conditions

The initial condition for Eq. (9) is:

$$C_p(r,0)=0 \tag{11}$$

The initial condition for Eq. (10) is:

$$C(t)=C_o \tag{12}$$

The boundary conditions for Eq. (9) are:

$$\frac{\partial}{\partial r}\left[q(r,t)+\frac{\varepsilon_p}{\rho_p}C_p(r,t)\right]=0 \qquad \text{at } r=0,\ t>0$$

$$D_s\rho_p\frac{\partial q(r,t)}{\partial r}+D_p\varepsilon_p\frac{\partial C_p(r,t)}{\partial r}=k_f\left[C(t)-C_p(r,t)\right] \qquad \text{at } r=R_p,\ t>0 \tag{13}$$

Coupling equation between Eqs. (9) and (10) and solving the system of partial differential equations numerically will result in the unsteady state in concentration of the solute in the vessel.

The bulk-fluid phase and particle equations for the batch adsorber model are first discretized using finite element (FE) and orthogonal collocation (OC) methods, respectively, **Fig.1**.The resulting ordinary differential equations (ODE) system is solved using an existing ODE solver provided by MATLAB (V-7.9), **Eggers, 2000.**

5. EXPERIMENTAL MODEL AND PROCEDURES

5.1 Materials and Methods

Granular activated carbon (GAC) was used as the first adsorbent in the present work. It was supplied by Unicarbo (Italy), to the Iraqi local markets. The granulated activated carbon was crushed, sieved into 0.32–0.5 mm, (De = 0.0004 m). The required sieve fraction was removed and washed with distilled water to remove fines. The GAC was washed, then the wet GAC was dried in an oven that was maintained at 100^{o}C for 24 hours, after which the GAC was kept in a desiccator for experimental use.

Local bentonite (calcium base) was used as the second adsorbent; it was supplied by the State Company of Geological Survey and Mining as pieces of rocks. These rocks were crushed to granules of different sizes, and then sieved using sieves to produce granules of sizes 0.32–0.5 mm, (D_e = 0.0004 m). The granules of bentonite were dried at 100^{o}C for 1 hour before being used.

The salt of $Pb(NO_3)_2$ (BDH, England) was used in the preparation of the stock solutions by dissolving it at a known concentration in distilled water. The solutions used for the study were obtained by dilution of the stock solution to the required concentration. The initial pH of each of the solutions was adjusted to a pH of 4 by the addition of HNO_3 or NaOH solution. The initial concentration of the metal ion, and the corresponding concentrations after fixed the time periods, were measured by an Atomic Absorption Spectrophotometer (type: ACCUSYS 211).

5.2 Equilibrium Isotherm Experiments

Two sets of adsorption experiments were carried out, activated carbon and bentonite, each one separately. An aqueous solution of 800 mg/l Pb (II) was prepared with a pH of 4. The Pb (II) adsorption isotherm equilibrium data were obtained using the batch process technique in several glass bottles of 250 ml volume at room temperature. Each flask was filled with 100 ml of prepared solutions, initially containing 2 g/100 ml of an adsorbent, activated carbon or bentonite.

The glass bottles were then placed on a shaker and agitated continuously at 300 rpm for 3 hours at room temperature (20^{o}C) to achieve equilibrium concentrations. The adsorption equilibrium concentration could be established and then the product solution was filtered with filter paper (type: Whatman 542, England) to remove any remaining adsorbent particles in the solution. The adsorbate concentrations of the liquid samples were then analyzed by means of an Atomic Absorption Spectrophotometer (type: ACCUSYS 211). The amount of adsorbate adsorbed was calculated according to the following equation:

$$q_e = \frac{V_L\left(C_o - C_e\right)}{W} \tag{14}$$

Where C_o and C_e are the initial and equilibrium concentrations of Pb (II) in the solution (mg/l), V_L is the volume of solution in the flask (L) and W is the mass of the adsorbent in the flask (g). The adsorption isotherms were obtained by plotting the weight of solute adsorbed per unit weight of adsorbent q_e against the equilibrium concentration of Pb (II) in the solution C_e.

5.3 Kinetic Experiments

The aim of the kinetic adsorption experiments of Pb (II) onto activated carbon and bentonite was to find the external mass transfer coefficient, k_f, the pore and the surface diffusion coefficients. A solution was prepared containing the Pb (II) concentration of 800 mg/l; 100 ml of solution was placed in several glass bottles each of 250 ml volume. The adsorbents, each one separately, were accurately weighed for the desired amount of 0.2, 0.4, 0.6, 1, 1.5, 2, 3, 4 g, and placed in several individual bottles. The bottles were then placed on a shaker for agitating, and then analyzed at various time intervals. The above experiment was repeated for each adsorbent at variable speeds of 100, 150, 200, 250 and 300 rpm, and then a sample of the solution was filtered with filter paper (type: Whatman 542, England), the residual concentrations of Pb (II) in solution was determined using atomic absorption spectrophotometer (type: ACCUSYS 211). During these experiments, the solute concentration decreased with time, the rate of adsorption of solute was measured by monitoring the solute concentration as a function of time. The desired weight of the adsorbent to reach an equilibrium concentration of C/C_o = 0.05 was calculated from the isotherm equation and mass balance equation for solute. The desired speed was obtained by repeating the experiments for each component with variable speeds (100, 150, 200, 250 and 300 rpm). The maximum speed at which the concentration ratio C/C_o remains constant at 0.05 was taken as the desired speed. The approach of $C/C_o = 0.05$ (equivalent to 95% removal efficiency) refers to the optimum operational durability and sustainability of the adsorption process.

6. RESULTS AND DISCUSSION

6.1 Isotherms Study

6.1.1 Determination of contact time

The plots of Pb (II) concentrations versus time for activated carbon and bentonite are presented in **Fig. 2**. On the basis of this plot, an equilibrium time of 40 min was chosen for the two adsorbents.

6.1.2 Determination of adsorbent dosage

The plots of Pb (II) concentrations versus various adsorbent quantities for activated carbon and bentonite are presented in **Fig. 3**. On the basis of this plot, the best weight that gave the higher removal was 2 g and 4 g for activated carbon and bentonite, respectively.

6.1.3 Estimation of adsorption isotherms constants

The constants of five isotherm models for activated carbon and bentonite were determined. The parameters for each model were obtained using a nonlinear statistical fit of the equations to the experimental data. All parameters with correlation coefficients are summarized in **Table. 1**, it is clear that the Freundlich isotherm model provided the best correlation compared with other isotherm models for Pb (II) sorption on the activated carbon and bentonite **Fig.4**.

From **Table 1** and **Fig. 4**, it can be concluded, that

- The equilibrium isotherm for activated carbon and bentonite are of a favorable type (i.e., somewhat convex upward).
- The sorption capacity varied from 80 mg/g to 20 mg/g, and from 52 mg/g to 17 mg/g for activated carbon and bentonite, respectively, **Table 2**. The average lead sorption capacity was the highest for activated carbon, and the lowest for bentonite.

6.2 Kinetic Study

6.2.1 Determination of diffusivity coefficients

The D_s and D_p for Pb (II) can be determined by matching experimental concentration curves with predicted concentrations at the optimum agitation speed. This can be generated by a numerical solution of the batch adsorber model. The matching requires values of the following parameters:

6.2.2 The optimum weight of adsorbent

The amount of adsorbent used for adsorption of Pb (II) was determined from the equilibrium-related concentration of $C_e/C_o = 0.05$ using the Freundlich equation as follows:

$$q_e = \frac{V_L(C_o - C_e)}{W_o} = KCe^{1/n} \qquad \text{where} \quad W_o = \frac{V_L(C_o - C_e)}{KCe^{1/n}} \tag{15}$$

The optimum weights W_o of activated carbon and bentonite were found to be 2 g and 4 g, respectively, under experimental conditions.

6.2.3 The best agitation speed

The efficiency of removal of Pb (II) is shown in **Figs.5** and **6** for activated carbon and bentonite, respectively, at different agitation speeds of 100, 150, 200, 250 and 300 rpm. The best agitation speed that achieved $C_e/C_o = 0.05$ is 300 rpm; at speed above 300 rpm, the removal efficiency is more than 95%, with possible pulverization of activated carbon and bentonite, and in this case, both adsorbents were in powder form rather than granular. On the basis of the above information, the maximum agitation speed for activated carbon and bentonite was regarded as 300 rpm.

6.2.4 External mass transfer coefficient

The external mass transfer coefficient (k_f) in the batch adsorber was determined from the concentration decay curves at optimum speed at the initial rate data, using the following equation [31]:

$$k_f = -\frac{R_p \rho_p V_L}{3 W_o t} \ln\left(\frac{C(t)}{C_o}\right) \qquad 0 < C(t)/C_o < 1.0 \qquad (16)$$

Where R_p is the radius of particles (m), ρ_P is the particle density (Kg/m^3), V_L is the volume of solution (L), t is the time (min), C_o and C(t) are the solute concentration at time zero and time (t), respectively. For accurate estimation of k_f, samples were taken at 2, 4, 5 and 7 minutes and analyzed immediately, **Table 3**.

The average calculated values of k_f for adsorbents were found to be 6.05×10^{-5} and 2.65×10^{-5} m/s for activated carbon and bentonite, respectively. This indicates that the rate of mass transfer of activated carbon is higher than bentonite. In other words, lead is adsorbed by activated carbon at a higher rate than bentonite.

6.2.5 Determination of pore and surface diffusion coefficients

The pore and surface diffusion coefficients for the two adsorbents were obtained from the numerical solution of Eqs. (9) and (10), with the initial and boundary conditions that describe the film and intraparticle transport mechanism in batch-wise experiments, using the external mass transfer coefficient described in the previous section.

D_s and D_p were obtained from the typical concentration decay for each adsorbent by an iterative search technique for the minimization of the difference between experimental and predicted data from a batch model. It is clear from **Figs.7** and **8**, that for activated carbon and bentonite, respectively, there is good matching between experimental and predicted data, as can be seen from the correlation coefficients presented in **Table. 4**.

Examining **Table 4**, it can be seen that activated carbon has the largest value of D_p compared with bentonite, this indicates that the pore diffusion controls the adsorption process for activated carbon. While bentonite has the largest value of D_s, this indicates that the surface diffusion controls the adsorption process for bentonite.

7. CONCLUSION

The activated carbon was more efficient than natural bentonite for removing lead ions from industrial wastewater. The experimental data fitted well to the Freundlich equation, with good correlation coefficients 0.9850 and 0.9598 for activated carbon and bentonite, respectively. There is good matching between the experimental and predicted data in batch experiments using a pore-surface diffusion model for the two adsorbents. Hence, the transfer of solute within the adsorbents is controlled by pore diffusion for activated carbon and by surface diffusion for bentonite. The average calculated value of k_f for activated carbon is higher than bentonite which indicates that the rate of mass transfer with activated carbon is higher than bentonite. In other words, lead is adsorbed by activated carbon at a higher rate than bentonite. The mathematical model which includes mass transfer coefficient, isotherm model constants, pore and surface diffusion coefficients provides a good description of the adsorption process for this study.

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9.NOMENCLATURE

FE	Finite elements
OC	Orthogonal collection
D_s	Surface diffusion
D_p	Pore diffusion
C_e	Concentration of Pb(II) remaining in solution at equilibrium (mg/l)
K_f	External mass transfer coefficient
q_e	Pb(II) concentration sorbed on the adsorbent (mg/g)

Table 1. Parameters of isotherm models for adsorption of Pb(II) using activated carbon and bentonite.

Model	Parameters	Activated Carbon	Bentonite
Langmuir Eq. (1)	q_m,(mg/g)	81.7	60.8
	b, L/mg	0.0214	0.0046
	Correlation coefficient	0.9689	0.9089
Freundlich Eq. (2)	K,(mg/mg)(L/mg)$^{1/n}$	0.0147	0.0053
	n,	3.8714	3.0450
	Correlation coefficient	0.9850	0.9598
Radk-Prausnitz Eq. (3)	K_{RP}, L/mg	0.9765	0.5772
	F_{RP},	0.0006	39.0473
	N_{RP},	1.6369	0.1388
	Correlation coefficient	0.9547	0.9001
Reddlich-Peterson Eq. (4)	A_R, L/mg	0.5823	0.5739
	B_R,	12405.15	0.0033
	m_R,	-1.0419	1.0811
	Correlation coefficient	0.9460	0.9140
Combination of Langmuir-Freundlich Eq. (5)	q_m, (mg/g)	108.0789	110.4549
	b, $(m^3/kg)^{1/n}$	0.0016	0.0107
	n,	1.2719	1.1736
	Correlation coefficient	0.9476	0.9200

Table 2. Experimental lead sorption capacity values (x/m) for activated carbon and bentonite.

Adsorbent	**mg lead sorbed per g adsorbent (X/M)**	
	Rang	**Mean**
Activated carbon	80 – 20	50
Bentonite	52 – 17	34.5

Table 3. Values of K_f at different times.

Adsorbent	**t, min**	**C/Co**	**K_f, m/s**
Activated carbon	2	0.73	6.73×10^{-5}
	4	0.55	6.39×10^{-5}
	5	0.5	5.93×10^{-5}
	7	0.43	5.15×10^{-5}
Bentonite	2	0.8	2.34×10^{-5}
	4	0.59	2.77×10^{-5}
	5	0.52	2.74×10^{-5}
	7	0.4	2.75×10^{-5}

Table 4. Values of (D_p) and (D_s) and their correlation coefficients.

Adsorbent	**D_p,m^2/s**	**D_s, m^2/s**	**Correlation Coeff.,R^2**
Activated carbon	$4.45x10^{-8}$	$1.95x10^{-11}$	0.9793
Bentonite	$2.21x10^{-13}$	$3.33x10^{-9}$	0.9896

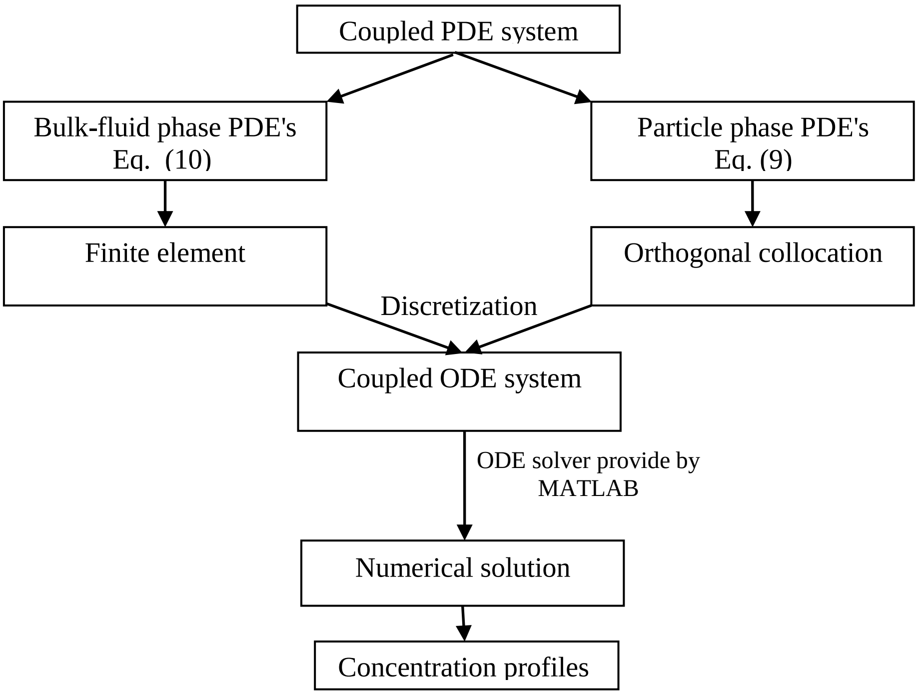

Figure 1. Flow diagram of the numerical method used to solve the PDE system in the model.

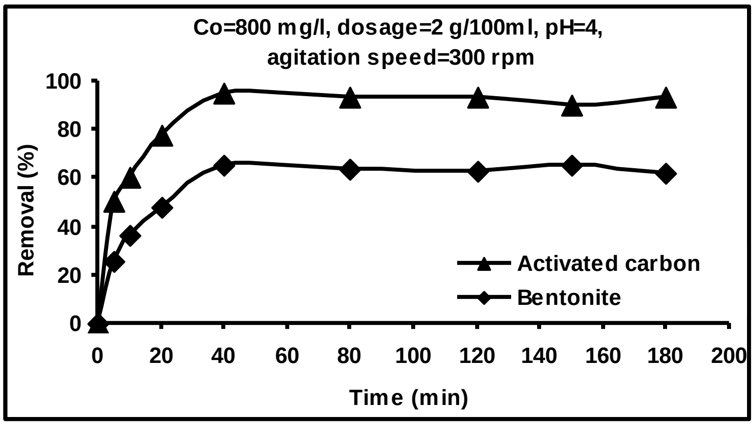

Figure 2. Equilibrium times for the sorption of Pb(II) by activated carbon and bentonite.

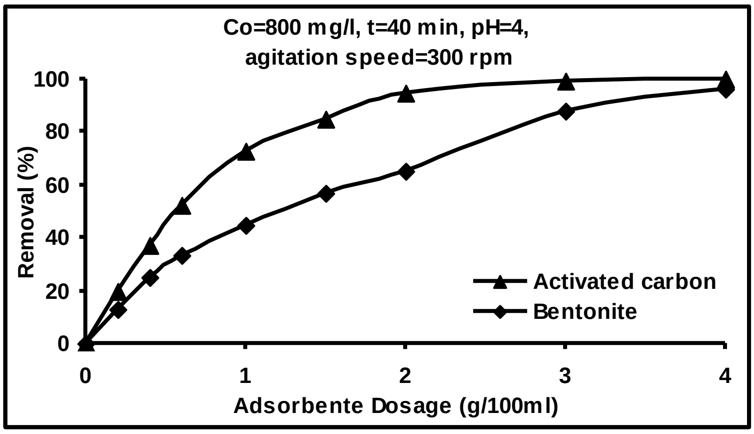

Figure 3. Adsorption of Pb(II) onto activated carbon and bentonite at different dosage.

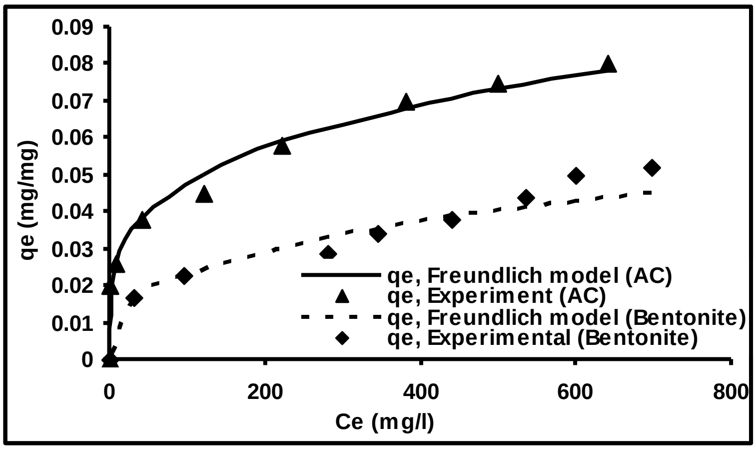

Figure 4. Adsorption isotherms of Pb(II) onto activated carbon and bentonite.

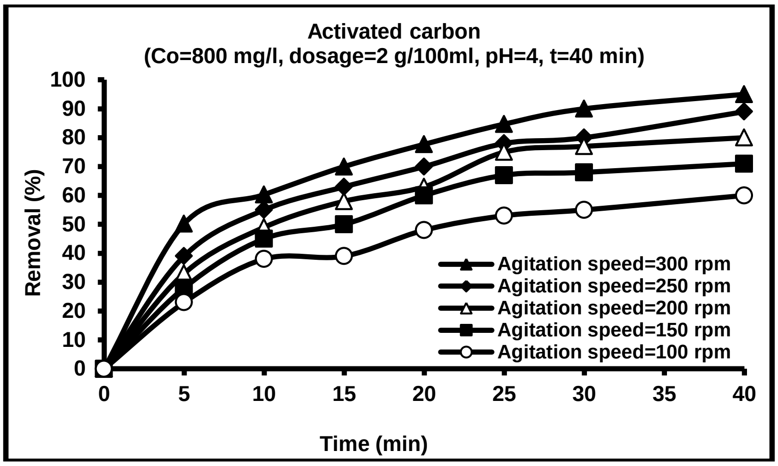

Figure 5. Removal efficiency of lead onto activated carbon at different agitation speeds.

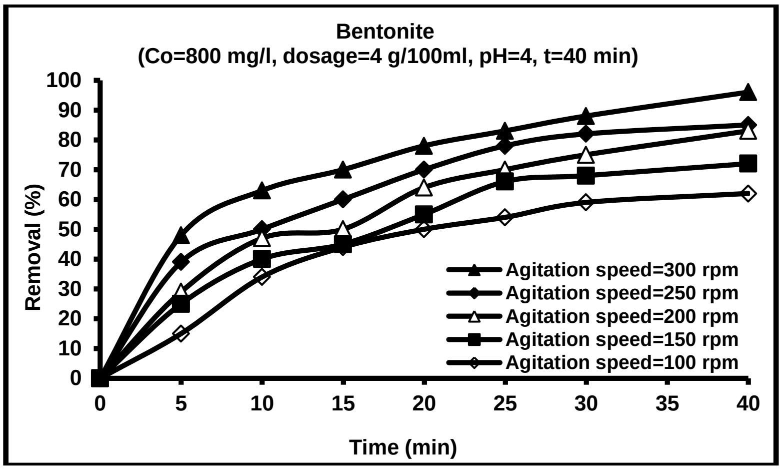

Figure 6. Removal efficiency of lead onto bentonite at different agitation speeds.

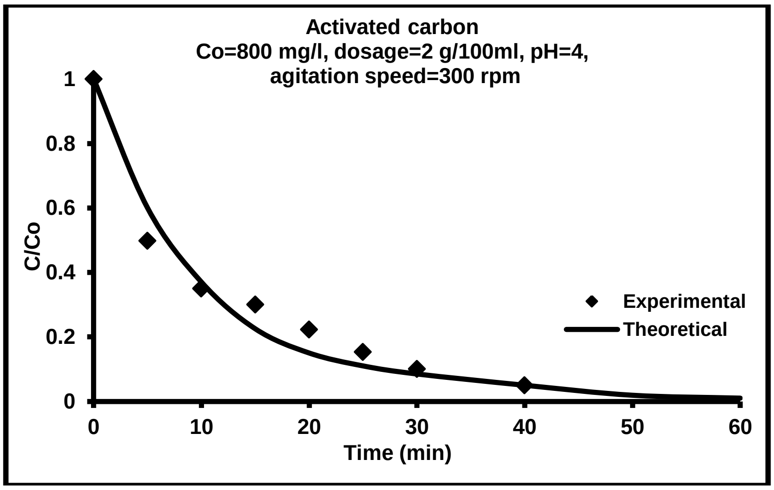

Figure 7. Comparison of measured concentration- time decay with that predicted by batch model for activated carbon system.

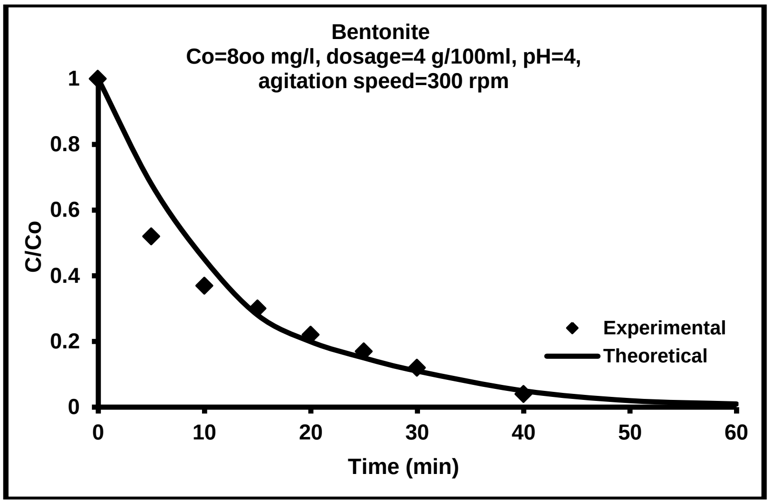

Figure 8. Comparison of measured concentration- time decay with that predictedby batch model for bentonite system.