Full Length Research Paper

Kinetic and equilibrium studies of the adsorption of lead (II) ions from aqueous solution onto two Cameroon clays: Kaolinite and smectite

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The efficiency of the adsorbents; kaolinite (MY22s) and smectite (Sa01) for the removal of lead (II) ions from aqueous solution was investigated. Parameters such as contact time (t) and initial concentration (C0) with a particle size of 80 µm were studied by using a batch scale adsorption technique. This is with the intension of optimising appropriate conditions to be utilised on a commercial scale to decontaminate industrial effluents. Adsorption equilibrium was reached within 60 min for the 50, 70, 90, 110 and 130 ppm initial concentration (C0), of lead (II) ions with a 500 mg weight adsorbent. The adsorption capacity (Qt) of lead ion removal increases with increasing initial concentration, C0 of lead (II) ions in solution. The adsorption efficiency or percentage of removal reached 92% for 130 ppm initial concentration, C0, of lead (II) ions with Sa01. Kinetic modelling analysis using the linear correlation coefficient (R2) values showed that the adsorption mechanism follows the pseudo-second order model for the adsorption of lead (II) ions on Kaolinite (MY22s) and smectite (Sa01). The adsorption data were also modelled by using both the Langmuir and Freundlich classical adsorption isotherms. These experimental data fitted the Langmuir isotherm for kaolinite (MY22s) and Freundlich isotherm for smectite (Sa01).

Key words: Adsorption, smectite (Sa01), kaolinite (MY22s), lead (II) ions, kinetics, equilibrium adsorption, wastewater treatment.

INTRODUCTION

The progressive increase of industrial technology results in the continuous increase in environmental pollution. Industries discharge different types of heavy metals waste into the environment at an unprecedented and at a constant increasing rate. These heavy metals waste may be discharged into streams, rivers and lakes and the continuous enrichment of these waters with these metals waste beyond the healthy level may cause poisoning, leading to various sicknesses.

Consequently the treatment of polluted industrial wastewater before they are released into the environment remains a topic of global concern. Moreover, contamination of ground water is today a major concern in the management of water resources (Waid and Hossam, 2007; Ishizaki and Marti, 1983). The majority of organic pollutants are susceptible to biological degradation while heavy metal ions can not be degraded into harmless end-product (Riaz and Sohail, 2005). The toxicity of metals depends especially on their chemical forms rather than on their total elemental contents and therefore, speciation studies increasingly gain importance (Orumwense, 1996; Norotry et al., 2000). Lead is one of the heavy metals, which is highly toxic to human, plant and animal. The

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ABBREVIATIONS: MY22s, Kaolinite from Mayouom; West Region of Cameroon; Sa01, Smectite from Sabga; North West Region of Cameroon; CEC, Cationic Exchange Capacity; PSD, Pore Size Distribution; PZC, Point of Zero Charge.
metal is of special concern because of its environmental importance related to its well known toxicity and intensive use in industries (Pratik and Choksi, 2008).

The implications of lead pollution in animal and human beings are their interference with the activities of intracellular enzymes (Nagarethinam and Ananthakrishnam, 2002), causing for example anaemia, kidney failure, impairment of central nervous system and damage to DNA and RNA. According to the World Health Organization (WHO, 1971), the accepted range of lead (II) ions in water is 0.01 ppm (Waid and Hossam, 2007).

Some of the major sources of lead release into the environment are metallurgical industry, electroplating, metal finishing industries, paint manufacture, storage battery manufacture, petroleum refining, fuel combustion, photo graphic materials and drainage from ore and mines (Zamzow et al., 1990).

Developing countries suffer from water pollution; the expensive method of treatment is the main problems in these countries. This is greatly reduced by the utilization of natural material and processes (Adesola et al., 2006). Science provides many practical solutions to minimizing the present level at which these metals are introduced into the environment and for remediating (cleaning up) past problems; therefore avoiding their dangerous effect (Adesola et al., 2006; Reed and Matsumoto, 1993).

Some of the methods that are available to reduce heavy metal concentrations from wastewater are chemical precipitation, filtration, ion exchange, reverse osmosis, ultra-filtration, electrochemical deposition, coagulation and adsorption. However, the above mentioned methods are not economically feasible for small and medium size industries. It is therefore necessary to search for low cost techniques that may be effective, less environmentally degrading and economical (Orumwense, 1996).

For low concentration of metal ions in wastewater, the adsorption process is recommended for their removal because they are cheap, simple, sludge free and involve small initial cost and land investment (Norotry et al., 2000).

The process of adsorption implies the presence of a solid adsorbent that binds molecules by physical attractive forces, ion exchange, and chemical bonding; it is advisable that the adsorbent is available in large quantities, easily regenerable and cheap.

Activated carbon is a potential adsorbent for the removal of several organic and inorganic pollutants but due to it high cost and (10-15%) loss during regeneration; alternative low cost adsorbents have attracted the attention of several investigators to provide an alternate for the high cost activated carbon (Emmanuel and Olalekan, 2008).

In the present study, kaolinite (MY22s) and smectite (Sa01) clays which were obtained from Mayoum and Sabga, both from the West and North West regions of Cameroon respectively were investigated as a potential and low cost adsorbent for lead (II) ions removal from aqueous solutions.

MATERIALS AND METHODS

Two local clay materials; kaolinite (MY22s) and smectite (Sa01), lead nitrate salt Pb(NO$_3$)$_2$ (AR sample, S.D. Fine-Chem Ltd. India) were used in this study. Double distilled (DD) water was used for the preparation of all the reagents.

Adsorbent

These local materials were sun dried, converted into fine powder by the use of a mortar and screened through a sieve (Retsch) to get a geometrical size of 80 μm, the used beaker was kept into an oven (HEREAUS) at 110°C for a period of 24 h (Hanafiah and Ngah, 2006) in order to remove water content and volatile impurities, removed and cooled in a desiccator containing CaCl$_2$ (drying agent) for 30-60 min. These adsorbents were removed from the desiccator and the required mass 0.5 g (that was used all through), weighed and stored in an airtight plastic container for the experiment. The characterization of these local materials is tabulated in Tables 1 and 2 (Tonle et al., 2003; Njoya et al., 2007).

Adsorbate

1000 mg/L of lead (II) ions was prepared by dissolving 1.599 g of hydrous lead nitrate (Pb(NO$_3$)$_2$) in a double-distilled water and made up to 1000 ml. The stock solution was diluted to obtain required standard solutions.

Batch adsorption experiments

The batch kinetic experiments of the adsorption studies were conducted at room temperature (25°C) in a 250 ml screw-cap conical flask. For each run, 0.5 g of the adsorbent was weighed and placed in the flask containing 20 ml solution of lead (II) ions of a desired concentration ( ranging from 50 to 130 ppm) at pH - 4.9. The suspension was stirred for interval of time between 30-300 min, using a magnetic stirrer. After agitation, the suspensions were centrifuged at 5000 rpm for 10 min to separate the solid and liquid phases. The supernatant was capped into test tubes, labelled, and was followed by complex formation.

Formation of Lead dithizonate complex

Since lead compounds can form coloured complexes with dithizone solution together with the aid of a suitable buffer, the concentration of lead in solution could therefore be determine by colorimetric analysis (Diaper and Kuksis, 1957). The concentration of the lead dithizonate is sensitive to and readily affected by the concentration of the lead ion, pH, the character and concentration of the buffer (Diaper and Kuksis, 1957). 2 ml of the solution containing lead (II) ions was added to 2 ml buffer solution of ammonium chloride/ammonia and 2 ml solution of dithizone in ethanol. The chemical reaction of the complex formation and structure is as shown in Figure 1:

\[
Pb^{2+} + 2HD_{2} \rightarrow PbDZ_{2} + 2H^{+}
\]
Table 1. Chemical Composition of the clays (kaolinite and smectite).

<table>
<thead>
<tr>
<th>Mineral composition</th>
<th>SiO₂ (%)</th>
<th>Al₂O₃ (%)</th>
<th>Fe₂O₃ (%)</th>
<th>MnO (%)</th>
<th>MgO (%)</th>
<th>CaO (%)</th>
<th>Na₂O (%)</th>
<th>K₂O (%)</th>
<th>TiO₂ (%)</th>
<th>P₂O₅ (%)</th>
<th>l.i</th>
<th>Total (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>MY22s</td>
<td>48.4</td>
<td>33.0</td>
<td>2.3</td>
<td>&lt; d.1</td>
<td>&lt; d.1</td>
<td>0.1</td>
<td>&lt; d.1</td>
<td>3.1</td>
<td>0.3</td>
<td>&lt; d.1</td>
<td>11</td>
<td>98.2</td>
</tr>
<tr>
<td>Sa01</td>
<td>63.3</td>
<td>14.5</td>
<td>4.2</td>
<td>&lt; d.1</td>
<td>0.2</td>
<td>0.7</td>
<td>0.4</td>
<td>2.3</td>
<td>0.2</td>
<td>&lt; d.1</td>
<td>13</td>
<td>98.8</td>
</tr>
</tbody>
</table>

d.l = detection limit; l.i = lost at ignition.

Table 2. Physical characteristics of kaolinite (MY22s) and smectite (Sa01).

<table>
<thead>
<tr>
<th>Sample</th>
<th>Parameter</th>
<th>kaolinite (MY22s)</th>
<th>smectite (Sa01)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Colour</td>
<td>Gray</td>
<td>Dirty white</td>
</tr>
<tr>
<td></td>
<td>Mineral composition (%)</td>
<td>Kaolinite (75.8); quartz (9.4); illite (7.7); florencite (0.4); rutile (3.7); goethite (2.5)</td>
<td>Montmorillonite (79.2); quartz (10); feldspath (5); hematite (2.53); limonite (0.1)</td>
</tr>
<tr>
<td></td>
<td>External specific area</td>
<td>15 m²/g</td>
<td>50 m²/g</td>
</tr>
<tr>
<td></td>
<td>CEC</td>
<td>78.2 × 10² (m eq/100 g)</td>
<td>86 × 10² (m eq/100 g)</td>
</tr>
</tbody>
</table>

Figure 1. Structure of lead dithizonate complex.

The orange unstable complex formed was allowed for 24 h for complete formation of a stable light yellow complex.

Each experiment was duplicated under identical conditions. Lead (II) ions concentration was determined spectrophotometrically using a UV-Visible spectrophotometer set at 490 nm (Shanmugavalli, et al., 2006).

The quantity adsorbed by a unit mass of an adsorbent (Qₑ) and the adsorption percentage (%R) at an instant was calculated from the differences between the concentrations of lead (II) before and after adsorption.

Adsorption kinetics

The study of adsorption kinetics describes the resistance to solute transfer from the solution up to the boundary layer at the solid-liquid interface to the pore water and then to the solid.

It is well known that adsorption kinetics is mainly controlled by the following steps;

1. Solute molecules transfer from the solution to the boundary film;
2. Solute molecules transfer from the film to the surface of the sorbent (external diffusion);
3. Diffusion from the surface to intra-particle sites and
4. Interaction of solute molecules with the available sites on the internal surface (Demirbas et al., 2007).

The evolution of the adsorption process can be followed by measuring the number of particles adsorbed per unit time. Many kinetic models have been proposed for the adsorption of solutes on solids. They include amongst others the Lagergren’s pseudo-first order kinetic model, pseudo-second order model, the Elovich model, mass transfer and intraparticle diffusion models.

**Pseudo-first order kinetic model**

The pseudo first order equation was suggested by Lagergren, (Demirbas et al., 2007) for the adsorption of solid-liquid systems. It is generally expressed as follows:

\[
\frac{dQ}{dt} = k_1 (Q_e - Q_t)
\]

where, \(Q_e\) and \(Q_t\) are the adsorption capacity at equilibrium and at time \(t\) respectively (mg/g). \(K_1\) is the rate constant of pseudo-first order adsorption (min⁻¹). After integration and applying boundary conditions, \(t = 0\) to \(t = t\) and \(Q_r = 0\) to \(Q_r = Q_e\), the integrated form of equation (1) becomes:

\[
\ln (Q_e - Q_t) = \ln (Q_e) - K_1 t
\]
This equation is verified if the plot of \( \ln (Q_0 - Q_t) \) as a function of time gives a straight line. \( K_1 \) is deduced from the slope of the line and \( Q_e \) gives the vertical intercept.

**Pseudo-second-order model**

The pseudo second order adsorption kinetic rate equation is expressed as (Demirbas et al., 2007):

\[
\frac{dQ_t}{dt} = k_2 (Q_e - Q_t)^2
\]

Where \( k_2 \) (mg \( \text{g}^{-1} \text{min}^{-1} \)) is the rate constant of pseudo second order adsorption. From the boundary conditions \( t = 0 \) to \( t = t \) and \( Q_i = Q_0 \) to \( Q_t = Q_e \), the integrated form of equation (3) becomes:

\[
\frac{1}{Q_e - Q_t} = \frac{1}{Q_e} + k_2 t
\]

This is the integrated rate law for a pseudo second order reaction. Equation (4) can be rearranged to obtain equation (5) which has a linear form:

\[
\frac{1}{Q_e} = \frac{1}{k_2 Q_e^2} + \frac{t}{Q_e}
\]

If the initial adsorption rate \( h_0 \) (mg \( \text{g}^{-1} \text{min}^{-1} \)) is:

\[
h_0 = k_2 Q_e^2
\]

Then, equations (5) and (6) become:

\[
\frac{1}{Q_e} = \frac{1}{h_0} + \frac{t}{Q_e}
\]

Thus, from equation (7), plots of \( t/Q_e \) versus \( t \) give values of \( Q_e \) and \( k_2 \) from the slopes and intercepts respectively.

**The Elovich model**

The Elovich equation is generally expressed as:

\[
\frac{dQ_t}{dt} = \alpha \exp(-\beta Q_t)
\]

Where, \( \alpha \) is the initial sorption rate (mg \( \text{g}^{-1} \text{min}^{-1} \)) and \( \beta \) is the desorption rate constant (g mg\(^{-1}\)h) during any one experiment. To simplify the Elovich equation, Chien and Clayton, (1980) and Elkhather et al., 2007) assumed \( \alpha \beta t \gg 1 \) and by applying the boundary conditions \( Q_t = 0 \) at \( t = 0 \) and \( Q_t = Q_e \) at \( t = t \), equation (8) becomes:

\[
Q_e = \frac{1}{\beta} \ln(\alpha \beta) + \frac{1}{\beta} \ln t
\]

Thus, a plot of \( Q_t \) versus \( \ln (t) \) should yield a linear relationship with a slope of \( (1/\beta) \) and an intercept of \( (1/\beta) \ln (\alpha \beta) \) if the sorption process fits the Elovich equation (Mirina and Mile, 2002).

**Mass transfer model**

The general equation for the mass transfer model is as follows:

\[
C_r = D_v k_v t
\]

Where, \( C_r \) and \( C_i \) are the initial concentration and concentration of the solute (mg/L) at an instant \( t \), while \( t \) is the agitation time (min). \( D \) is the mass transfer constant and \( k_v \) the adsorption constant.

The linearization of equation (10) permits us to obtain the following expression:

\[
\ln(C_0 - C_i) = \ln D + k_0 t
\]

This equation is verified if the plot of \( \ln (C_0 - C_i) \) as a function of time gives a straight line. \( k_0 \) is deduced from the slope of the line and \( D \) is obtained from the vertical intercept (Pratik and Choksi, 2008).

**The Intra-particle diffusion model**

When the intra-particle mass transfer resistance is the rate limiting step, then the sorption process is described as being particle diffusion controlled (Igwe and Abia, 2007). The intra-particle diffusion model is expressed as equation (12)

\[
R = k_d t^a
\]

Where \( R \) is the percentage of lead (II) ions adsorbed, \( t \) the contact time, \( a \), the adsorption mechanism, \( k_d \) the intra-particle diffusion rate constant (mg g\(^{-1}\) min\(^{-1.5}\)). A linear form of equation (12) is:

\[
\ln R = \ln K + a \ln t
\]

The plot of \( \ln R \) versus \( \ln t \) should give a linear relationship from where the constants \( a \), and \( K \) can be determined from the slope and intercept of the plot respectively.

Higher values of \( k_d \) illustrate an enhancement in the rate of adsorption whereas larger a values illustrate a better adsorption mechanism. This is related to an improved bonding between lead (II) ions and the adsorbent particles.

**Adsorption Isotherms**

The equilibrium relationship between adsorbent and adsorbate are described by adsorption isotherm, usually the ratio between the quantity adsorbed and that remaining in solution at a fixed temperature at equilibrium. In this paper, the Langmuir isotherm and the Freundlich isotherm were investigated.

**The Langmuir isotherm**

The Langmuir adsorption isotherm is often used for adsorption of a solute from a liquid solution. The Langmuir adsorption isotherm is perhaps the best known of all isotherms describing adsorption and is often expressed as:
\[ Q_e = \frac{Q_m K C_e}{1 + K C_e} \]  

(14)

Where, \( Q_e \) (mg of adsorbate per g of adsorbent) is the adsorption density at the equilibrium solute concentration. \( C_e \), \( C_0 \) is the equilibrium concentration of adsorbate in solution (mg/L). \( Q_m \) (mg of solute adsorbed per g of adsorbent) is the maximum adsorption capacity corresponding to complete monolayer coverage. \( K \) is the Langmuir constant related to energy of the adsorption (L of adsorbate per mg of adsorbent).

The above equation can be rearranged to the following linear form:

\[ \frac{C_e}{Q_e} = \frac{1}{Q_m K} + \frac{C_e}{Q_m} \]  

(15)

The linear form can be used for linearization of experimental data by plotting \( C_e/Q_e \) against \( C_e \). The Langmuir constants \( Q_m \) and \( K \) can be evaluated from the slope and intercept of the linear equation.

This is the simplest physically possible isotherm. It is based on three assumptions:

1. Adsorption cannot proceed beyond monolayer coverage.
2. All surface sites are equivalent and can accommodate at most one adsorbed atom.
3. The ability of a molecule to adsorb at a given site is independent of the occupation of neighbouring sites (Lezech et al., 2000).

Freundlich adsorption Isotherm

The Freundlich isotherm is the earliest known relationship describing the adsorption equation and is often expressed as (Najua et al., 2008):

\[ Q_e = K C_0^{1/n} \]  

(16)

Where \( Q_e \) is the quantity of solute adsorbed at equilibrium (adsorption density: mg of adsorbate per g of adsorbent), \( C_0 \) is the concentration of adsorbate at equilibrium, \( K \) and \( n \) are the empirical constants dependent on several factors and \( n \) is greater than one.

This equation is conveniently used in linear form by taking the logarithmic of both sides as:

\[ \ln Q_e = \ln K + \frac{1}{n} \ln C_e \]  

(17)

A plot of \( \ln Q_e \) against \( \ln C_e \) yielding a straight line indicates the confirmation of the Freundlich isotherm for adsorption. The constants \( n \) and \( K \) can be determined from the slope and the intercept respectively.

These two models are widely used, the former assuming that maximum adsorption occurs when the surface is covered by one layer of adsorbate; and the latter being purely empirical.

RESULTS AND DISCUSSION

Effect of agitation time and initial lead (II) ions concentration on adsorption

The equilibrium adsorption capacity of kaolinite (MY22s) and smectite (Sa01) was found to increase with increased in initial lead concentration (Figure 2), it was found out that on increasing the concentration of the lead (II) ions in solution from 50 to 130 ppm, the amount removed increases from 0.68 to 3.40 mg/g (MY22s), 1.52 to 4.68 (Sa01) as shown in Table 3. This indicates that there are plenty of adsorption sites on Sa01 than MY22s available for the adsorption of lead. Similar trends were observed for the other adsorption cases indicating that the adsorption capacity increases with increasing initial lead ion concentration within the experimental operating conditions (Reed and Matsumoto, 1993).

This is in agreement with the results obtained by other investigators (Hanafiah and Ngah, 2006) for the adsorption of lead (II) on rubber leaf powder. The increase of loading capacities of the sorbent with increasing initial lead (II) ion concentrations may also be due to a higher interaction between substituted lead (II) and sorbent (Ömer et al., 2005). Moreover, this can be explained by the fact that more adsorption sites were being covered as the metal ions concentration increased (Najua et al., 2008).

Effect of contact time

The results on Figure 2 shows that the enhanced adsorption of lead (II) ions with increase in agitation time may be due to the decrease in boundary layer resistance to mass transfer in the bulk solution and an increase in the kinetic energy of hydrated ions. By increasing the agitation time, the boundary layer resistance will be reduced and there will be an increase in the mobility of ions in the solution (Hanafiah and Ngah, 2006). For 50, 70, 90 and 130 ppm lead (II) ions initial concentration, the adsorption equilibrium time was obtained after 60 minutes for both MY22s and Sa01.

This result is interesting because equilibrium time is one of the important considerations for economical wastewater treatment applications (Francis, 2006). The maximum amount of lead ions was adsorbed within the first 60 min (92% of total metal ions adsorbed) and thereafter the adsorption proceeded at a slower rate until equilibrium was reached.

Kinetic studies

Kinetic equations have been developed to explain the transport of metals onto various adsorbents. To analyze the sorption rates of lead (II) metal ions onto smectite and kaolinite, five simple kinetic models were tested.

From these kinetic models, only the pseudo-second order model agreed well with the experimental data with Sa01 being the most following their correlation coefficients; Figure 3. These kinetic models are only concerned with the effect of the observable parameters on the over all rate of sorption (Cassey, 1997).
**Figure 2.** The adsorption capacity $Q_t$ (mg/g) of lead (II) ion on MY22s and Sa01 at different initial concentrations $C_0$ (50-130 ppm) with time $t$ (min).

**Table 3.** Langmuir and Freundlich equilibrium constants.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>MY22s 80 µm</th>
<th>Sa01 80 µm</th>
</tr>
</thead>
<tbody>
<tr>
<td>$Q_{(exp)}$ (mg/g)</td>
<td>3.4400</td>
<td>4.7400</td>
</tr>
<tr>
<td>$K_L$ (Lmg$^{-1}$)</td>
<td>0.4760</td>
<td>0.8170</td>
</tr>
<tr>
<td>$Q_m$ (mg/g)</td>
<td>0.9770</td>
<td>3.1370</td>
</tr>
<tr>
<td>$R^2$</td>
<td>0.9770</td>
<td>0.8190</td>
</tr>
<tr>
<td>$K_f$ (L g$^{-1}$)</td>
<td>24.7100</td>
<td>0.4900</td>
</tr>
<tr>
<td>$1/n$</td>
<td>-0.2030</td>
<td>0.8480</td>
</tr>
<tr>
<td>$R^2$</td>
<td>0.06220</td>
<td>0.9634</td>
</tr>
</tbody>
</table>

**Pseudo-second order**

The pseudo second order adsorption kinetics for the linearised rate equation is expressed as:

\[
\frac{1}{Q_t} = \frac{1}{Q_e} + \frac{t}{h_0} \frac{1}{Q_e} 
\]

(7)

The plots of $(t/Q_t)$ versus $t$ are as shown in Figure 3. Absorption data at 25°C best fitted with the linearised Langmuir isotherm for MY22s, with the correlation coefficient constant ($R^2$) of 0.9770. For the Freundlich adsorption isotherm, only the smectite (Sa01) experimental data fitted the model with $R^2$ equal 0.9634. Linear curves, obtained by plotting $C_e/Q_e$ versus $C_e$, indicate the validity of Langmuir isotherm and $\ln Q_e$ versus $\ln C_e$, indicate the validity of Freundlich isotherm (Mirina and Mile, 2002; Shanmuga et al., 2007).

The experimental data are better represented by the Langmuir isotherm as indicated by the high $R^2$ values in Table 3 and Table 4. The fact that the Langmuir isotherm fits the experimental data very well may be due to the homogenous distribution of active sites on the adsorbent surfaces; since the Langmuir equation assumes that the surface is homogenous. Thus the applicability of the Langmuir isotherm in the present system indicates the monolayer coverage of lead (II) ions on the outer surface of the adsorbents (Najua et al., 2008).

However, the Freundlich isotherm better describes
adsorption with the adsorbent Sa01. The values of 1/n, ranging from 0 to 1, is a measure of adsorption intensity or surface heterogeneity that becomes more heterogeneous as its value gets closer to zero (Ketcha et al., 2007). A value for 1/n below 1 indicates a normal Langmuir adsorption isotherm, while 1/n above 1 is indicative of cooperative adsorption (Abia and Asuquo, 2006).

The value of 1/n = 0.8480 obtained with the adsorbent Sa01, supports the fact that, adsorption takes place homogenously on the surface.

**Conclusion**

Equilibrium and kinetic studies were determined for the adsorption of lead (II) ions from aqueous solutions onto smectite (Sa01) and kaolinite (MY22s) in the concentration range from 50 to 130 ppm and the initial pH of 4.9 at an ambient temperature of 25°C.

Following the R² values, adsorption of lead (II) ions onto Sa01 followed both Langmuir and Freundlich isotherms models with that of Freundlich being higher than Langmuir. But with MY22s, the equilibrium data of adsorption are in good agreement only with Langmuir isotherms model at an ambient temperature (25°C).

The kinetic sorption model for pseudo-second other was in good agreement with the experimental equilibrium data. A contact time of one hour was enough for the system to reach equilibrium. Kinetic data would be useful for the fabrication and designing of wastewater treatment plants. The adsorption capacities of MY22s and Sa01
were compared for the uptake of lead (II) ions at optimum (maximum removal) conditions and the relative adsorption capacities are: MY22s, (3.36 mg/g) < Sa01, (4.73 mg/g).

The results of analysis of all the adsorbents clearly established that Sa01, is efficient in the adsorption of lead (II) ion from aqueous solution. Sa01 is not only economical, but easily available. Hence it is practically feasible for developing counties and will be useful for the economic treatment of wastewaters containing lead metal.

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REFERENCES


