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Adsorption of Lead (II) Ions by Activated Carbon Prepared From Different Parts of *Jatropha Curcas* Plant

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Abstract

Activated carbon samples prepared from the stem, root and seed coat of *Jatropha curcas* plant by activation using orthophosphoric acid was used for the removal of Pb (II) from aqueous solution. The samples were labeled ACS, ACR and ACC for activated stem, root and seed coat respectively. The effects of initial Pb(II) ion concentration, adsorbent dose, contact time, pH and ionic strength were studied to optimize the conditions required for maximum adsorption. At suitable condition (pH 8, contact time of 180, adsorbent dose of 1.4g, initial Pb(II) concentration of 40gL^{-1}), the adsorption capacities observed were 72.80% for ACR, 75.50% for ACS and 74.96% for ACC respectively. The mechanisms of the rate of adsorption were analyzed using pseudo 1st and 2nd order models. It was found that the adsorption mechanisms in the lead/adsorbents system follow the pseudo first order kinetics. The adsorption data were modeled by using both Langmuir and Freundlich isotherms. Freundlich isotherm was found to fit better fit in the experimental data than Langmuir isotherm. The result shows that ACS, ACR and ACC could be an effective adsorbents for Pb(II) removal.

Keywords: *Jatropha curcas*, Adsorbents, Adsorption, Lead (II) ions, Langmuir and Freundlich isotherm

1. Introduction

Heavy metal containing wastes are considered to be hazardous to both human life and the environment due to their acute toxicity and non-biodegradability, even when the heavy metals are present at trace concentrations (Zhang et al., 2005). The increase in the use of the major twenty heavy metals (i.e Lead, Cadmium, Cobolt, Nickel e.t.c) for over a few decades has inevitably resulted in increasing flux of metallic substances into natural sources of water, being polluted by many industries such as tannery, mining, alloying and battery; producing the (Kadirvelu et al., 2008). All lead compounds are considered cumulative poisons and acute lead

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poisoning can affect nervous system and gastrointestinal track (Zhang et al., 2005). Several conventional techniques used as effective methods for removal of metal ions from wastewaters have been developed, including electroplating, evaporation, oxidation, reduction, membrane separation, ion exchange and adsorption but most have disadvantages, such as continuous input of chemicals, high cost and even incomplete metal removal (Brown et al., 2000 and Park et al., 2007).

Adsorption is a worthy economical and very effective process for a variety of applications, and now it is considered an economical and efficient method for metal ions removal from wastewaters (McKay, 1996). The most generally used solid adsorbent is activated carbon which is used as a very efficient solid adsorbent in many different applications (Cooney, 1998). However, activated carbon is expensive and for effluents containing metal ions activated carbon requires chelating agents to enhance its performance, thus increasing treatment. Therefore, the need for alternative low-cost adsorbents has prompted the search for new and cheap sorption processes for aqueous effluent treatment (Meshko et al., 2006; Yavuz et al., 2003 and Erdem et al., 2004).

Many researchers have investigated low-cost materials as a viable adsorbents for heavy metal removal, namely, rice husk (Wong et al., 2003) barks (Reddy, 1997) teak leaves powder (King, et al., 2006) tree fern (Ho et al., 2002) groundnut shells (Shukla and Pai 2005), macadamia nutshell (Ahmed Pour and Do, 1997) holly oak (Presas and Freitas 2000) paper mill sludge (Nastin, 2000) among others. Although activated carbon has been extensively used as adsorbents but their adsorption capacities are shown to be largely controlled by their surface characteristics. The type of precursor and the method employed for activation are key factors that determine the quality of the carbon produced. The combination of the chemical and physical activation processes lead to production of activated carbon with specific surface properties. Chemical activation involves impregnation of the raw material with chemicals such as phosphoric acid, potassium hydroxide, and zinc chloride (Molina – Sabio et al., 1995; Molina – Sabio et al., 2004; Mohammed, 2006). The common features of activating agents is their effects during carbonization and therefore in development of desired pore structure, H_3PO_4 acid acts as a dehydrating agent that promotes the decomposition of carbonaceous materials during the carbonization process, which restricts the formation of tar, and increases the carbon yield. Vacant interstices in the carbon matrix are formed upon extensive post pyrolysis washing of the pore.

The extent of chemical activation can significantly alter the characteristic of the produced carbons.

Research interest is now focused on plant biomass adsorbent; this work involves the use of activated carbon from different part of *Jatropha curcas* as adsorbent for the removal of Pb(II) ions from simulated aqueous solution as this is not available in literature yet. The results of this study will ascertain the suitability of different parts of *Jatropha curcas* plant as raw material for the production of activated carbon, and its effectiveness in the removal of lead ions (Pb^{2+}) from simulated aqueous solutions.

2. Materials and Methods

The *Jatropha curcas* plant was obtained from Agbo-oba area in Ilorin Kwara State Nigeria. The root, stem and seed coat were separated out, sun dried and pulverized. The samples were used for the preparation of activated carbons using the following procedures.

2.1 Preparation of Activated Carbon

The dried roots, stem and seed coat of the plant (500g) were each carbonized separately in a muffle furnace at temperature of 450^{o}C for 4h. The carbonized samples were blended using a mortar and pestle and then sieved using a 150µm sieve. Each sieved charcoal was treated with 75% orthophosphoric acid (H_3PO_4) at a ratio of 1:1 by weight of charcoal/H_3PO_4. The mixture of H_3PO_4 and charcoal were dried in an oven for 24hr at 60^{0}C. Each of the samples was then washed with de-ionized water several times until the pH of washing solution was 7.0. The samples were filtered and the residue (charcoal) were mixed with 0.1M HCl in a beaker and stirred for 1 h on a magnetic stirrer. Each of the stirred mixture was finally washed with hot de-ionized water until the pH of washing solution reached 7.0. The washed activated carbon was dried and kept in a close container for the adsorption experiment.

2.2 Equilibrium Adsorption Experiment

The adsorption of Pb(II) ion on ACS, ACR and ACC was investigated by batch adsorption. The effect of solution pH (range 2 – 9), contact time, (15 – 300min), adsorbent dose (0.2 – 1.6g) and initial concentration (5-50mg/L) on the adsorption was studied at room temperature.

2.3 Effect of initial concentration

The equilibrium adsorption of Pb (II) ion by ACS, ACR and ACC was carried out by shaking 0.5g each of the root, stem and seed coat samples with 25ml of Pb (II) ion solution (prepared form simulated aqueous solution) of initial concentration (range btw 5.0 – 50.0mg/L) on an orbital shaker for 4h at room temperature. Each of the mixture was filtered after shaking and the filtrate analyzed for Pb (II) ion concentration. Blank solutions without adsorbents were also treated the same and unabsorbed Pb (II) ion was determined, this served as controlled and initial concentrations. The adsorbed Pb ion at equilibrium Qe(mg/l) was calculated using the adsorption system mass balance (Elaigwu, et al 2009).

$$Q_e = \frac{V(c_i - c_f)}{M} \qquad (1)$$

Q_e = Equilibrium adsorption (mg/L)

V = Volume of Solution in (ml)

C_i = Initial Concentration (mg/l)

C_f = Final Concentration (mg/l)

2.4 Effect of Contact Time

The experiment was conducted using 25ml of 40mg/l prepared Pb(II) ion solution which was added to 0.5g each of ACS, ACR and ACC in beakers. The mixtures were shook for the different contact time (15, 30, 45, 60, 90, 120, 180, 240, 270 and 300min) and then filtered. The Pb(II) ion concentrations in the filtrate were analyzed using AAS. The quantity of Pb (II) ion adsorbed (Qe) was then calculated using the equation (2) (Badmus et al., 2007).

$$Q_e = \frac{c_i - c_f}{c_i} \times 100 \qquad (2)$$

2.5 Effect of pH

The effect of pH was carried by varying the pH (2 – 8) of the 25ml 40mg/LPb(II) solution from 2-8 using 0.1M KOH or 0.1MKNO_3. A 0.5g sample of dose was mixed with the solution in a 100ml flask. After shaking for 4h, the reaction mixture was filtered. The residual Pb^{2+} ion in the filtrate was determined by atomic absorption spectroscopy (Elaigwu et al., 2009).

2.6 Effect of Adsorbent Dosage

The experiments was carried out by using constant adsorbate of 25ml of 40mg/L Pb(II) solution with adjusted pH to the optimal pH of 6. Each of ACS, ACR and ACC of dosage (0.2, 0.4, 0.6, 0.8, 1.0, 1.2, 1.4 or 1.6g) was mixed with the solutions in a 100ml flask and the mixture was shaken for 4h. The mixture was filtered and analyzed for residual Pb(II) ion in the filtrate by atomic absorption spectroscopy.

3. Results and Discussion

3.1 Sorption Capacity of Jatropha Samples

The result of the sorption capacities of ACS, ACR and ACC shows that increase in adsorbate concentration results in increase in adsorption process. This trend suggested that there was increase in number of available molecule per binding site on the adsorbent as the metal concentration increases thus bringing about higher probability of binding the molecules to the adsorbent. The interaction between the adsorbent and the adsorbate was enhanced by the high availability of molecules of adsorbate in solution. In a similar research, it was reported that the increase in adsorption is consistent with the increase in adsorbate concentration (Elaigwu et al., 2009).

In the case of low concentration the ratio of the initial number of moles of Pb(II) ions to the available binding sites is higher and subsequently the fractional adsorption becomes independent of initial concentration. At higher concentrations the available sites for adsorption per molecule become decreased and hence the percentage adsorption of Pb(II) ion decreases since the number of available adsorption sites were constant even when the adsorbate concentration was increased (Jureeporn and Panjal, 2010).

The initial adsorbate concentration provides an important driving force to overcome mass transfer resistance of metal ions between the aqueous and solid phases (Donmez and Aksu, 2002). When equilibrium is established the percentage adsorption decrease with increase in concentration of Pb(II) ions, because the accessibility to the few unfilled sites was restricted partly due to repulsion by neighbouring metal ions and partly due to saturation of the surface site of samples. This can be explained by the limited number of active sites on ACS, ACR and ACC (Omar et. al., 2010).

3.2 Effect of Contact Time

The result of the effect of contact time on the adsorption of Pb(II) from aqueous solution is shown in Fig.1 This is made up of three segments that represents fast, slow and equilibrium. The adsorption of Pb(II) by the samples was rapid for the first 15min as a result of large and unrestricted available binding sites on the samples, thereafter the adsorption slowed down (43-55mg/l)as more sites were filled up within 20-120min. The slowest adsorption occurred between 120-180min (56-63mg/l) at which equilibrium was attained. The equilibrium position was maintained till 300mins. This is similar to the adsorption of metal ions on waste Chinese herbs reported to be biphasic (Liu et al., 2006), which explains that the initial fast phase occurs due to surface adsorption on the samples. The subsequent slow phase occurs due to diffusion of metal ions into the inner part of the biomass. It was observed that the adsorption rate was high at the beginning but plateau values were obtained at 180min. similar to what was reported by Liu et al., 2006. In addition, the slight variation in the amount of Pb (II) ion removal by the various samples could be related to the nature and concentration of the surface groups on the active sites. The active sites are responsible for interaction with the lead ions (Augustine et al., 2007).

The kinetic of metal ion sorption governs the rate which determines the residence time and is one of the important characteristics defining the efficiency of an adsorption (Krishnan and Anirudhan, 2003). The rate at which the adsorption takes place is most important, especially when designing batch sorption system.

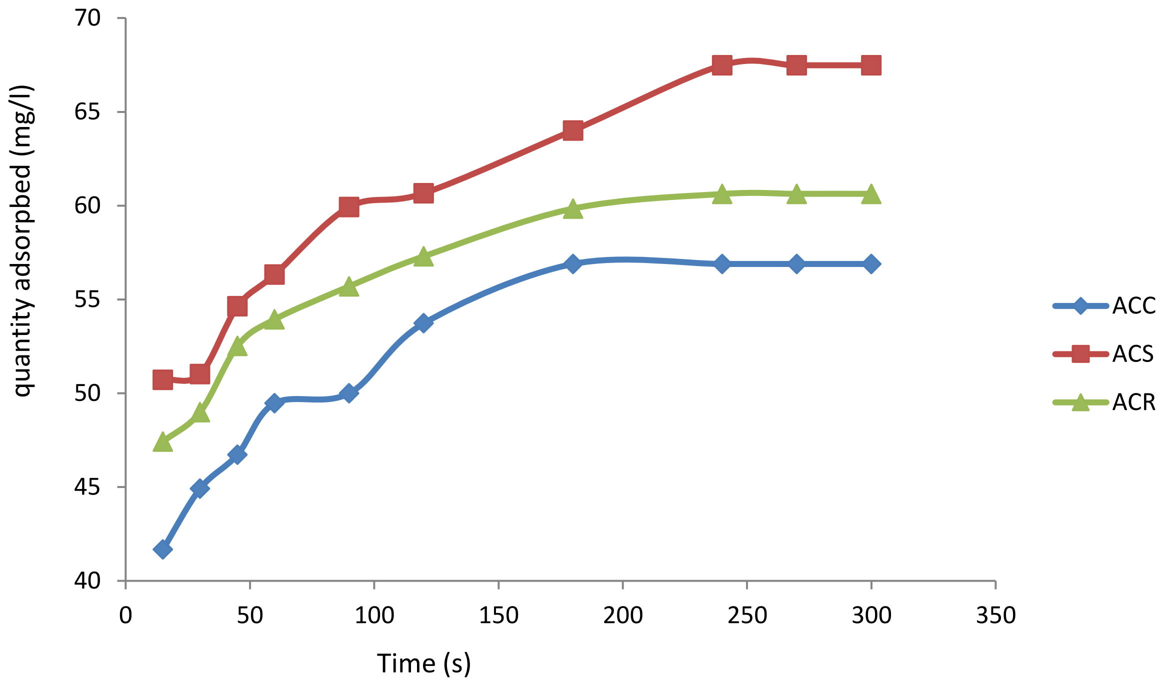

Fig1 Effect of contact time for Pb (II) ion

The kinetic of Pb(II) ion sorption onto the activated carbon samples and the variation of sorption capacity of Pb(II) with contact time for each adsorbent is shown in Fig.1

The initial faster rate may be due to the availability of the un-used sites on the surface of the adsorbent which depends on surface area has reported by (Namasivayam et al., 1996). In addition it was also stated that slight variation in the amount of Pb(II) ion removal by the various samples could be related to the nature and concentration of the surface groups (active sites) responsible for interaction with the lead ions (Augustine et al., 2007). The *Jatropha* samples (adsorbents) contains abundant cellulosic units including a matrix of COO^-, SH^-, OH^- which are incorporated during chemical modification of surface active sites on the adsorbent matrix and hence take part in metal ion binding (Horsfall et al., 2006).

Generally it was observed that characteristic of metal ion adsorption is that as the sites are progressively filled, the more difficult the sorption becomes, and adsorption process tends to be more unfavorable (Namasivayam et al., 1996; Horsfall et al., 2006; Johnson, 1990, Abdus-Salam, and Adekola, 2005).

3.3 Effect of pH for Pb (II) ion

It has been documented that pH is an important parameter in adsorption process because it affects the solubility of the metal ions, concentration of the counter ions on the functional groups of the adsorbent and the degree of ionization of the adsorbate during reaction (Badmus et al., 2007; Sheng et al., 2004; Bradl, 2004; Baes and Mesmer, 1976).

The amount of Pb(II) ions removed per unit mass of ACS, ACR and ACC in the solution increase up to pH 6, after which it became steady. The pH affects the surface charge of the activated carbon and the degree of ionization of the adsorbate species. As shown in Fig 2. The Pb(II) ion attaining maximum adsorption capacity value of around, 52.80% for ACR, 57.50% for ACS and 54.9 6% for ACC respectively. This result obtained is not far from what was reported by Kobya et al., 2005. However after pH 7 there was a decrease in adsorption, these decrease may be due to formation of insoluble hydroxyl complexes. Low et al. (1995) reported that at low pH values the surface of the adsorbent would be closely associated with hydroxonium ions (H_3O^+), by repulsive forces, to the surface functional groups, consequently decreasing the percentage removal of metal ions. As the solution pH increases, the onset of the metal hydrolysis and precipitation began usually at $pH > 7$ and the onset of adsorption therefore occurs before the beginning of hydrolysis as reported by Baes and Mesmer (1976). It was also reported by (Sheng et al., 2004; Bradl, 2004) that lead undergoes hydrolysis to $Pb(OH)^+$ at pH between 6-8. Above pH 9 solid lead hydroxides $Pb(OH)_2$ is thermodynamically the most stable, while $Pb(OH)_3^-$ is predominantly at pH above 11. In this study as the pH of the absorbing medium was increased from 2-8, there was a corresponding increase in deprotonation on the adsorbent surface leading to a decrease in H^+ ion on the adsorbent surface. This creates more negative charges on the adsorbent surface which favours adsorption of positively charge species as reported (Ghanem and Mikkelson, 1988; Kadirvelu and Namasivayam, 2003; Abdulsalam and Adekola, 2005).

Badmus et al. (2007) reported that the hydrolysis of cations occurs by the replacement of metal ligands in the inner coordination sphere with the hydroxyl groups. This replacement occurs after the removal of the outer hydration sphere of metal cations. In addition the results obtained in this study tallies with reports from other researchers, which show the dependence of metal ion adsorption on pH over a wide range (2-12) with S-like adsorption curve (Johnson, 1990; Bruce, 2000, Glover et al., 2003). The solubility of metals is known to be lowered at higher pH (at pH >

8) values as reported by (Inbaraj and Sulochana, 2002; Abdulsalam and Adekola 2005) which is due to be the likely hood of precipitation of the hydroxide forms of the adsorbate species. Badmus et al., (2007) also reported that the result that gave S-like curve was a reflection of adsorption and precipitation only.

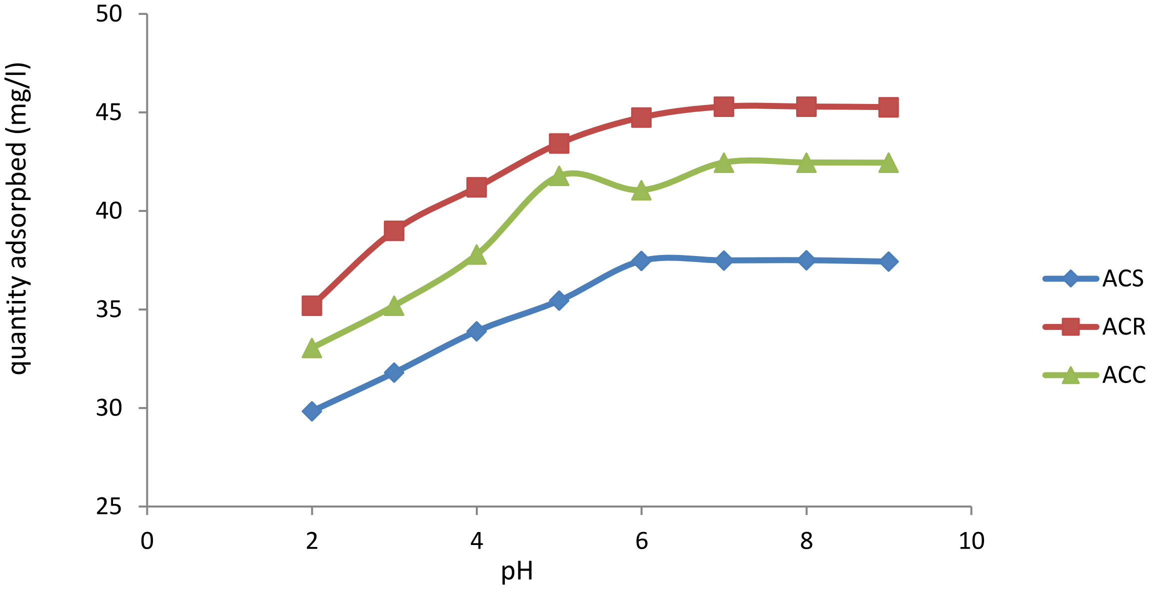

Fig. 2 Effect of pH for Pb (II) ion adsorption

The result thus indicates that binding follows an ion-exchange mechanism that involves electrostatic interaction between the negatively charged groups in the walls of the substrate and metallic cations which was in agreement with the result reported by Elaigwu et al., (2009).

3.4 Effect of Adsorbent Dose for Pb(II) ion

The effects of adsorbent doses on the removal of Pb(II) ion shown in Fig. 3. That the adsorption of Pb(II) ion is a function of sample dosage 47mg/l. It is apparent that increasing the adsorbent dose resulted to an increase in the amount of Pb(II) ions adsorbed but there was decrease in the amount adsorbed per unit mass. It has been observed that there is decrease in adsorption density with increase in the adsorption sites during the adsorption process (Yu et al., 2003; Pehliven et al., 2008).

Another reason may be due to the particle interaction, such as aggregation, resulting from high adsorbent dose. Such aggregation would lead to decrease in surface area of the adsorbent and an increase in diffusion path length as reported by Shukla et al. (2002).

In this study it can be concluded that the appropriate adsorbent dose is 1.4g and after this dose the rate of Pb(II) ion removal is almost constant as the dose increases which is in agreement with what Yu et al., (2003); Pehliven et al. (2008) reported.

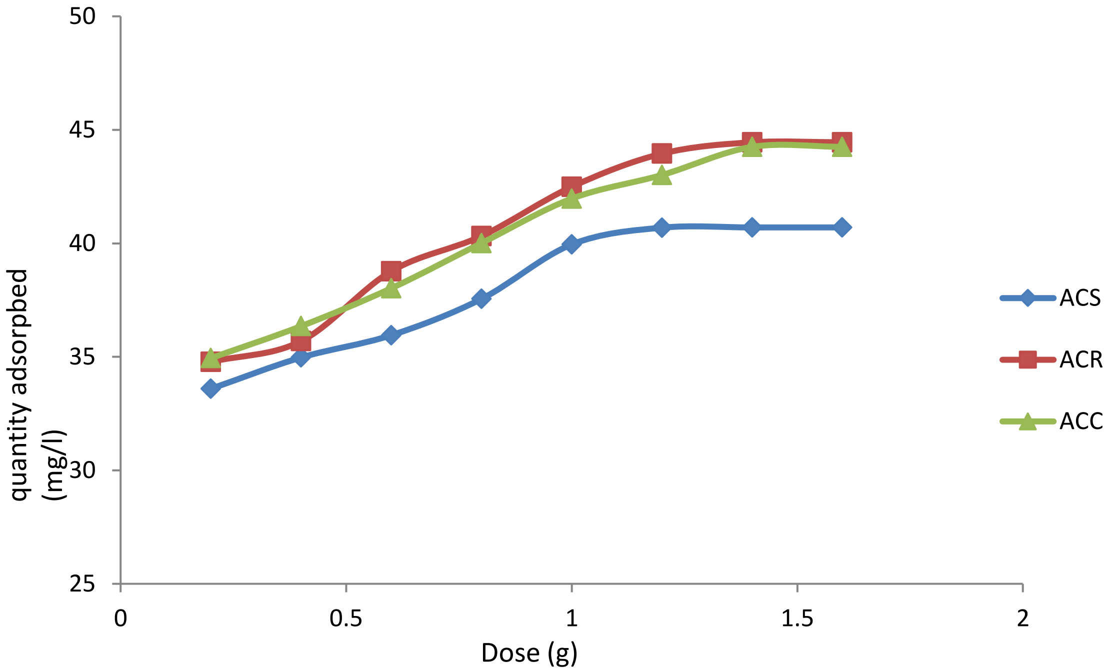

Fig. 3 Effect of adsorbent dose on Pb (II) ion

3.5 Adsorption isotherms

The results were subjected to Langmuir and Freundlich isotherm models.

3.5.1 Langmuir isotherm

Langmuir isotherm assumes monolayer adsorption onto a surface containing a finite number of adsorption sites of uniform strategies of adsorption with no transmigration of adsorbate in the plane of surface.

The linear form of Langmuir isotherm equation is given as

$$\frac{Ce}{qe} = \frac{1}{kqm} + \frac{Ce}{qm} \qquad (3)$$

Where q_e is the amount of adsorbate adsorbed at equilibrium (mg/g), k is the Langmuir constant related to the energy of adsorption (L/mg), q_m is the maximum sorption capacity corresponding to complete monolayer coverage (mg/g), and C_e is the equilibrium solute concentration (mg/L). When C_e/q_e was plotted against C_e, a straight line with slope of $1/q_m$ was obtained. The Langmuir curves pertaining to the Pb(II) ion adsorption by ACS, ACR and ACC were provided in Figure 4.

However, Table 1 provides the experimental data for all the adsorbents studied. The Langmuir constants k and q_m were calculated from equation (3) above.

$$R_L = \frac{1}{1+KCo} \tag{4}$$

The values obtained are presented in Table 1 the essential characteristics of the Langmuir isotherm can be expressed in terms of a dimensionless equilibrium parameter (R_L) which is defined by equation (4) above. Where K is the Langmuir constant and C_o is the initial concentration (mg/L). The value of RL indicates the type of the isotherm to be either favorable ($0 < R_L < 1$), unfavorable ($R_L > 1$), linear ($R_L = 1$) or irreversible ($R_L = 0$).

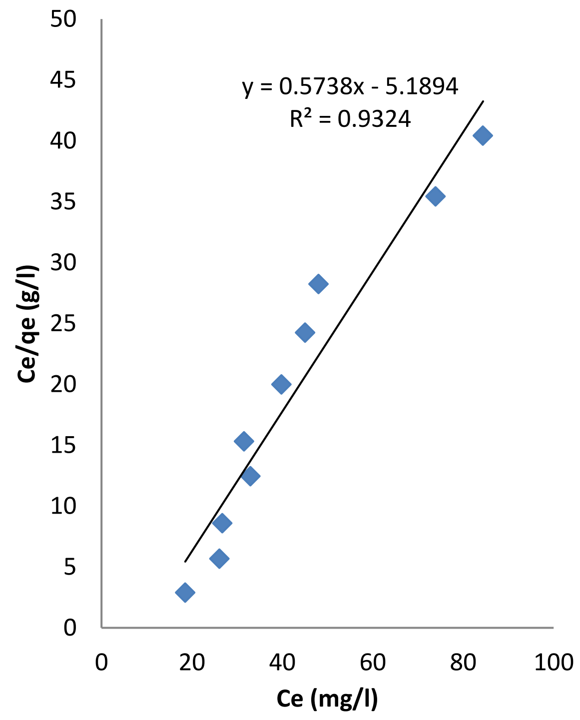

FIG. 4a Langmuir isotherm for Pb(ii) ion onto ACS

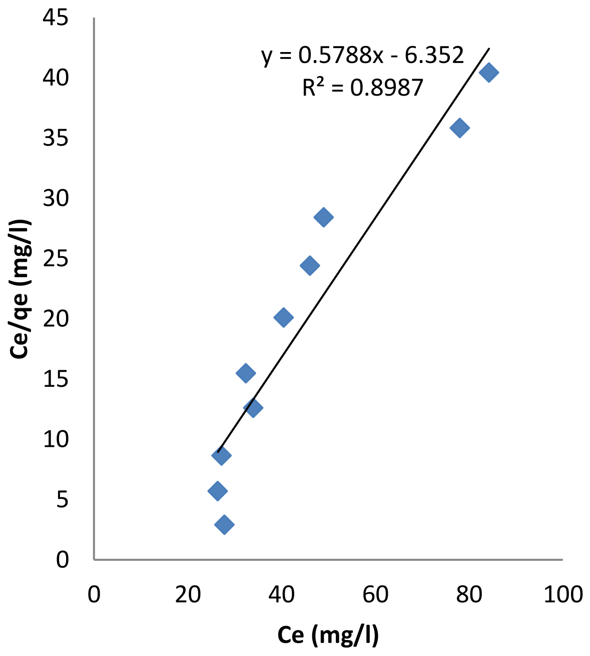

FIG. 4b Langmuir isoterm for Pb(ii) ion using ACR

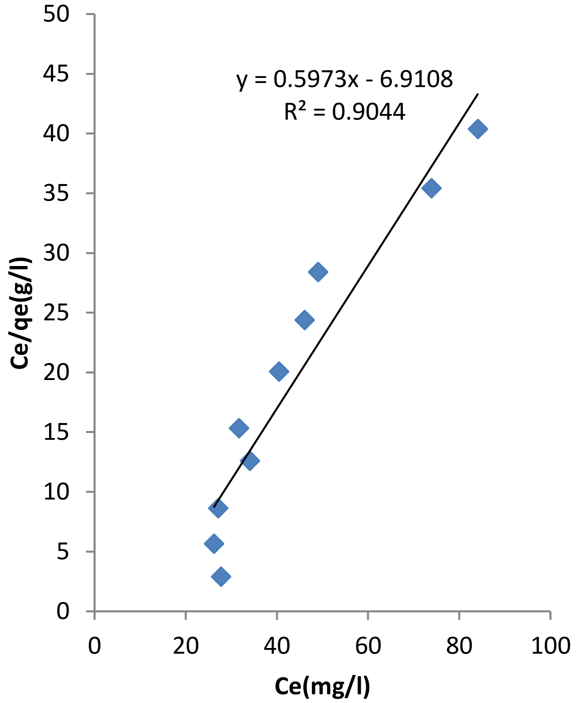

FIG. 4c Langmuir isotherm for Pb(ii) ion using ACC

The correlation coefficient, R^2, the maximum sorption capacity, q_m and the equilibrium constant K being calculated are presented in the Table.1 below

Table 1: Adsorption isotherm constants and their correlation coefficients for the adsorption of Pb(II) ion onto ACS, ACR and ACC

Adsorbent	Langmuir Constant				Freundlich Constant		
	q_m (mg/g)	K (l/mg)	R^2	R_L	1/n	K_f (mg/g)	R^2
ACS	0.573	0.337	0.932	0.069	0.476	9.3760	0.863
ACR	0.157	0.272	0.898	0.084	0.560	12.190	0.796
ACC	0.594	0.242	0.904	0.094	0.577	12.912	0.847

The sorption capacity of ACS, ACR and ACC was enhanced by 2-fold. Since R_L value indicates isotherm shape, if $R_L>1$ the isotherm is unfavorable, $R_L=1$ implies that the isotherm is linear, $0< R_L<1$ indicates that the isotherm is favorable and $R_L=0$, shows that the isotherm is irreversible as reported by Niveta et al., (2007); Karthikeyan and Siva (2008). However in the present study R_L was found between 0 and 1 for Pb(II) ion adsorption onto prepared activated carbon indicating that ACS, ACR and ACC are good adsorbents for lead ion in aqueous medium.

3.5.2 Freundlich Isotherm

Freundlich isotherm assumes heterogeneous surface energies, in which the energy term in Langmuir equation varies as a function of the surface coverage. The well-known logarithmic form of Freundlich isotherm is given by the following equation,

$$\text{Log } q_e = \log K_f + \frac{1}{n}\log C_e \tag{5}$$

Where C_e is the equilibrium concentration of the adsorbate (mg/L) and q_e is the amount of adsorbate adsorbed per unit mass of adsorbent (mg/g). K_f and n are Freundlich constants representing the adsorption capacity and intensity of adsorption respectively. The values of K_f and 1/n were obtained from the slope and intercept of the plot of log q_e against log C_e. The slope l/n ranging between 0 and 1 measure the adsorption intensity or surface heterogeneity, becoming more heterogeneous as its value gets closer to zero. The calculated Freundlich constants K_f and n are presented in Table.1 The Freundlich plots for the adsorption of Pb(II) onto ACS, ACR and ACC are presented in Fig 5(a-c)

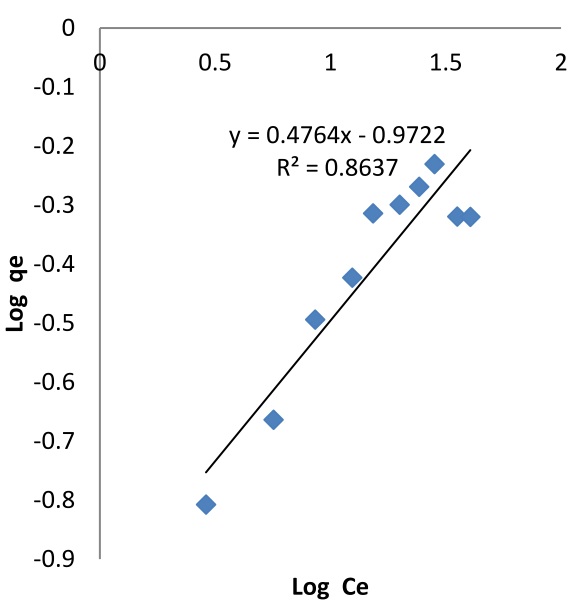

FIG. 5a Freudlich isotherm for adsorption of Pb(ii) ion onto ACS

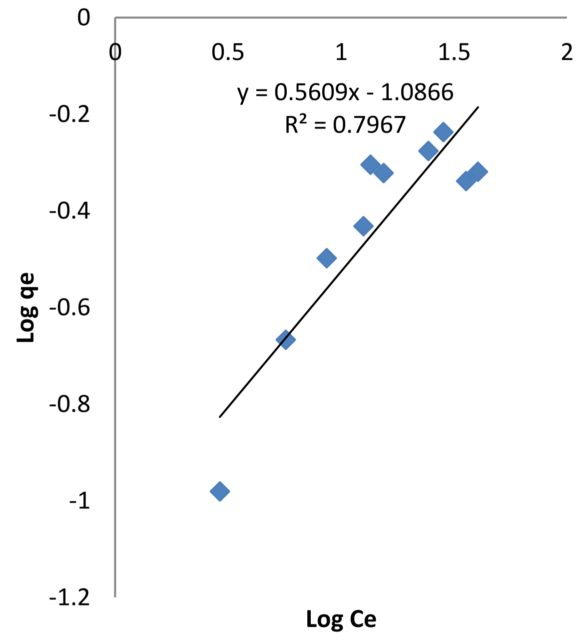

FIG. 5b Freudlich isotherm for adsorption of Pb(ii) ion onto ACR

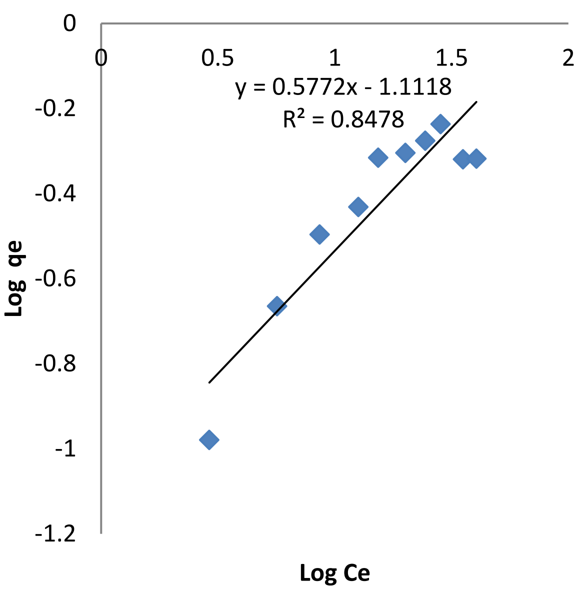

FIG. 5c Freudlich isotherm for adsorption of Pb(ii) ion using ACC

The values (0.796-0.863) of the correlation coefficients indicate that the data in this study favours Freundlich isotherm (Fig.5). The main characteristic of Freundlich equation is based on the assumption that all sites vary in their adsorption and most active energies sites are occupied first and the surface is continually occupied until the lowest energy sites are filled. The K_f is indicative of energy or intensity of the reaction and K_f values obtained range from 9.376-13.000 this is in agreement with the report of Krishnan et al. (2005) for Pb(II) uptake by montmorillonite. It was also reported that by definition the adsorption intensity is less than unity (Krishnan et al., 2005; Niveta et al., 2007), and the values (0.476-0.577) of 'n' for Pb(II) adsorption onto ACS, ACR and ACC in the present study agree with those obtained by (Krishnan et al., 2005; Niveta et al., 2007).

A plot of log q_e against log C_e from linearized Freundlich equation yielding a straight line confirmed the suitability of the model. The values of K_f and 1/n were obtained from the intercept and slope of the straight line Fig 5

3.6 Kinetic Modeling

The sorption kinetics is an important aspect in the control of pollutants removal process, once it provides valuable insights into the reaction pathways and into the mechanism of sorption reactions as stated by Ho and McKay (1999). The Lagergren's first-order kinetic model and the Ho's pseudo-second-order model are the most frequently used in the literature to predict the mechanism involved in the sorption process (Ho and McKay, 1999). The Lagergren's first order model is expressed by the equation:

$$\frac{dqt}{dt} = k_1(q_e\text{-}q_t), \tag{6}$$

where q_t and q_e (mg/g) are respectively the amounts of absorbed metal at time t and at equilibrium respectively and k_1 (min^{-1}) is the first order rate constant. The linear form of the equation is

$$\text{Log } (q_e\text{-}q_t) = \log q_e - \frac{k1}{2.303} t \tag{7}$$

Adsorption rate constants (k_1) and adsorption capacity (q_e) for the adsorption of lead (II) ions by ACS, ACR and ACC were calculated from the slope and intercept of the plots of log (q_e - q_t) against t as shown in Table 2.

Table 2: Kinetics and correlation coefficients based on pseudo-first and pseudo-second order kinetics for Pb (II) ion

Adsorbent	Kinetic constant for Pseudo-first order			Kinetic constant for Pseudo-second order			
	q_e (mg/g)	$K_1(min^{-1})$	R^2	q_e (mg/g)	K_2 (g/mg)	H	R^2
ACS	23.227	0.004	0.969	5.525	4.59×10^{-4}	1.4×10^{-2}	0.998
ACR	14.520	0.003	0.966	7.519	2.83×10^{-4}	1.6×10^{-2}	0.999
ACC	16.596	0.004	0.917	5.560	5.18×10^{-4}	1.6×10^{-2}	0.999

The pseudo-first order plot of Pb^{2+} on ACS, ACR and ACC are illustrated in Fig.6

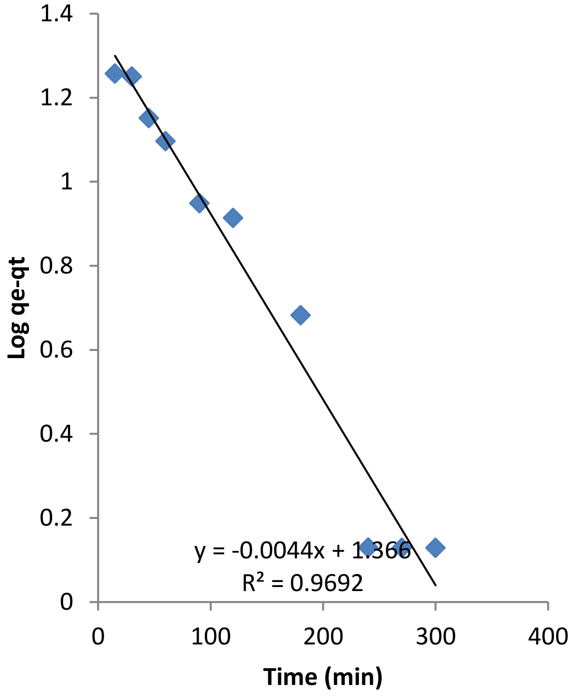

FIG.6a Pseudo-first order kinetics of Lead on ACS

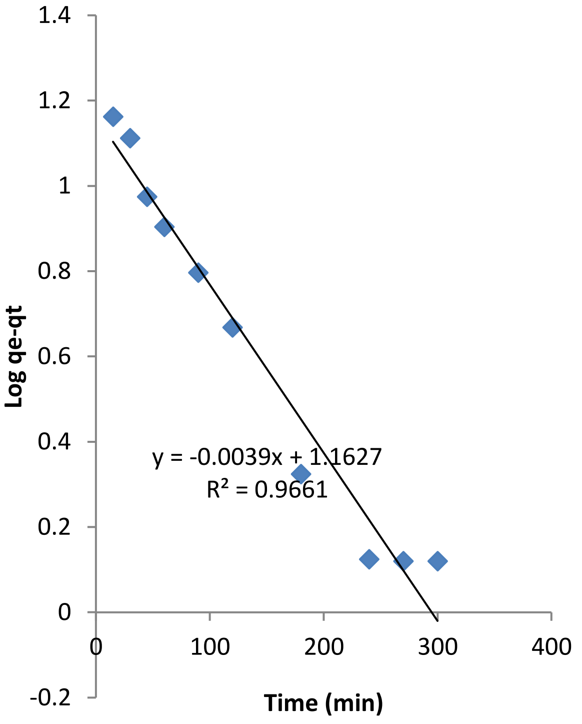

FIG.6b Pseudo-first order kinetics of Lead on ACR

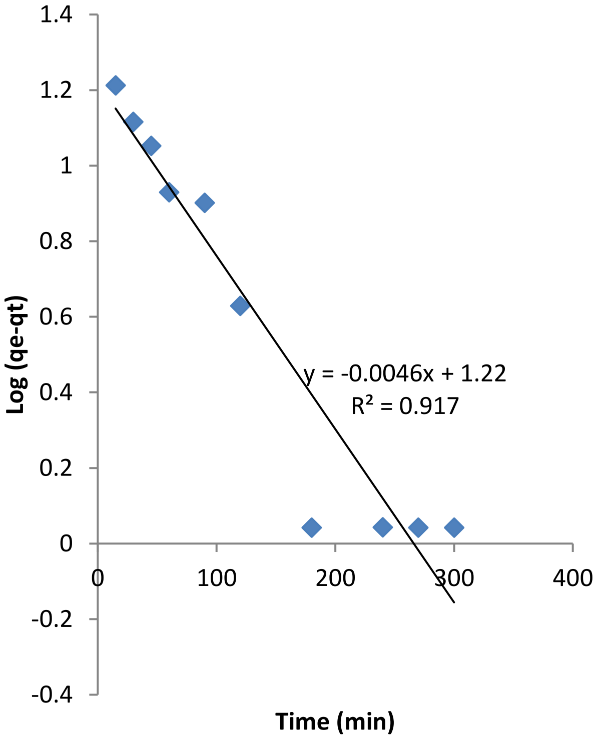

FIG.6c Pseudo-first order kinetics of Lead on ACC

The q_e value obtained as shown in Table.2 with range and mean indicating an agreement with q_e = 10.128 reported by Adesola et al. (2008), and q_e being 13.78 as reported by Ong et al. (2007). The more recent pseudo-second-order model expressed in equation 8 has been extensively used by several researchers (Ho and McKay, 1999). This model is expressed by the following equation:

$$\frac{dqt}{dt} = K_2(q_e\text{-}q_t)^2 \tag{8}$$

Where k_2 is the rate constant of pseudo-second-order model (g/mg min).

Definite integration of the above equation for boundary conditions $qt = 0$ when $t = 0$ and $q_t = q_t$ at $t = t$, give the following form of equation.

$$\frac{t}{\mathrm{qt}} = (1/k_2q_e^2) + \frac{1}{\mathrm{qt}}\,t \tag{9}$$

The initial biosorption rate, h (mg/g min) is defined as:

$$h = K_2q_e^2 \tag{10}$$

The equilibrium adsorption capacity (qe) and the pseudo-second order rate constants k_2 were obtained from the slope and intercept of the plots of t/q_t against t for various adsorbents as shown in figure 7 and are reported in Table 2.

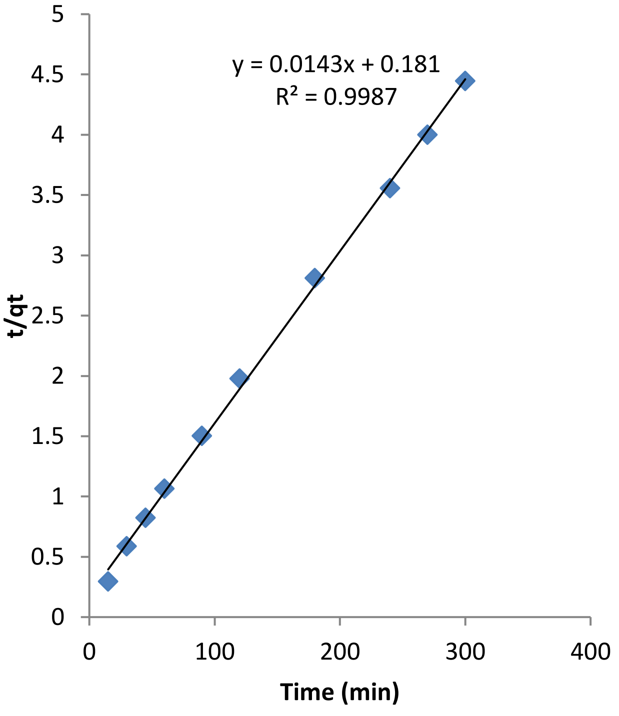

FIG.7a Pseudo-second order kinetics of Lead on ACS

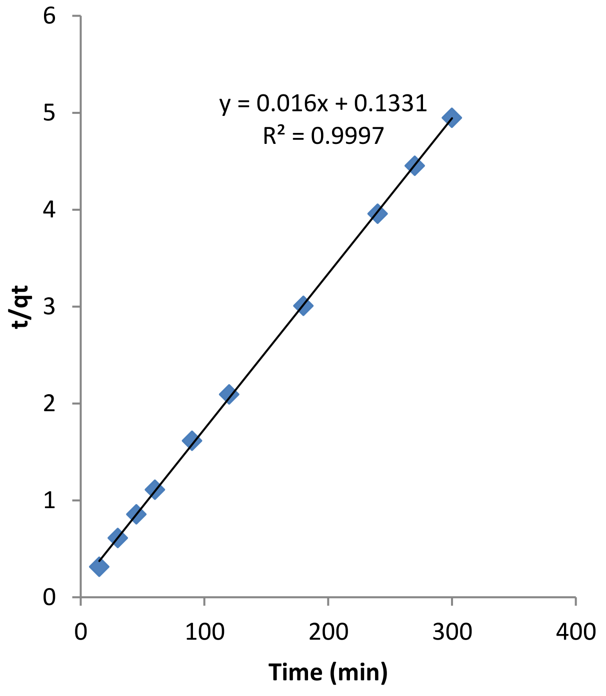

FIG. 7b Pseudo-second order kinetics of Lead on ACR

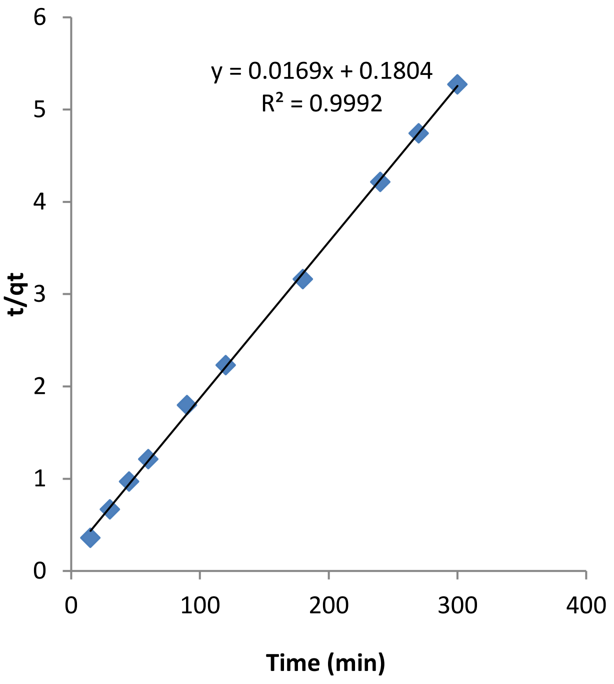

FIG. 7c Pseudo-second order kinetics of Lead on ACC

It is clear from table 2 that, the correlation coefficient of pseudo-first-order model for Pb (II) ion was higher in comparison to the pseudo-second order model. This suggests that the calculated correlation coefficients are consistent and closer to unity for pseudo-first-order kinetics model than the pseudo-second order kinetic model, therefore, the adsorption kinetics could well be explained and approximated more favourably by pseudo-first order kinetic model for ACS, ACR and ACC adsorbents, respectively. These results correlate with the assertion of (Ramamurthi and Jaikumar, 2009; Lima et al 2010) in the biosorption of acid green dye from aqueous solution by brewery waste and copper sorption by activated carbon of Ipomoea Carnea.

4. Conclusion

The present study evaluates the activated carbon ACS, ACR and ACC made from *Jatropha curcas* stem, root and seed coat as an effective adsorbent for the removal of lead from aqueous solution. The adsorption isotherms fit for both Langmuir and Freundlich isotherms but

Freundlich isotherm is found to be more favourable than Langmuir isotherm. It was concluded that ACS, ACR and ACC activated with H_3PO_4 has good adsorbing capacity for lead. The adsorption of Pb(II) was found to be maximum at pH 8, adsorbent dose of 1.4g and 120min contact time. The adsorption of lead by ACS, ACR and ACC all follows pseudo-first order rate kinetics.

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