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Adsorption of Pb^{2+} and Cd^{2+} ions from water by sugarcane bagasse-based adsorbents

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Abstract
Sugarcane bagasse, a fibrous residue from Sugar mills, is not easily biodegradable posing disposal challenges. Given its abundance in the sugar producing regions of Kenya, there is continued need to find other alternative uses. The objective of this study was to investigate the potential use of sugarcane bagasse-based adsorbents for removing Cd^{2+} *and* Pb^{2+} *from water. Four bagasse-based adsorbents namely activated bagasse (AB), non-activated bagasse (NAB), its activated charcoal (AC) and ash were prepared and used in batch experiments at different temperatures, pH, contact times, initial concentrations, adsorbent doses and agitation speeds. Adsorption data obtained was fitted against Freundlich and Langmuir isotherm models. It was found that the adsorption data for* Pb^{2+} *on ash, non-activated bagasse, activated bagasse and activated charcoal best fitted Freundlich isotherm model with* R^2 *values of 0.736, 0.958, 0.978 and 0.951, respectively with adsorption capacities of 62.7±0.01, 6.6±0.09, 18.4±0.01 and 3.7±0.01 mg/g in the same order. Ash was the best in removal of* Pb^{2+} *with capacity of 62.7±0.01 mg/g. Adsorption data for* Cd^{2+} *on ash, non-activated bagasse, activated bagasse and activated charcoal best fitted Langmuir isotherm model with* R^2 *values of 0.914, 0.995, 0.975 and 0.989, respectively and adsorption capacities of 40.9±0.1, 43.9±0.01, 52.9±0.01 and 45.4±0.03 mg/g in that order. Activated bagasse was the best in removal of* Cd^{2+} *with capacity of 52.9±0.01 mg/g.*

Keywords: *Adsorption, Sugarcane bagasse, Langmuir, Freundlich isotherm, water remediation.*

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I. Introduction

The sugar industry produces about 800 million tons of sugar annually on global scale which results in about 240 million tons of bagasse, some of which is used as fuel for mill boilers and electricity generation (Kalderis *et al*., 2008). The unused bagasse is not readily biodegradable for composting for application in the farms and therefore, poses disposal problems. Many efforts have been directed towards finding other applications of sugarcane bagasse (Osinubi and Alhassan, 2007; Aigbodion, 2008). Some reports show that it has potential in making of pulp and paper products, charcoal as well as building materials (Botha and Blottnitz, 2006; Kalderis *et al*., 2008).

Bagasse is rich in lignocellulose and has abundant pores giving it a large internal surface area (Mahamadi, 2011). Due to this property, sugarcane bagasse has also been applied in removing some heavy metal ions from contaminated water. The adsorptive capacity and efficiency of removing these ions from water differs from one metal to another, and depends on prevailing experimental conditions such as temperature, concentration and contact time (Gupta and Sharma, 2003; Ibrahim *et al*., 2006; Batnagar and Mika, 2010). Granular activated carbon which has been the most popular adsorbent in wastewater treatment is relatively expensive especially for small scale use (Wasewar, 2010). There is need to develop cheaper water treatment adsorbents from locally available sources and wastes while at the same time curbing disposal challenges. Milling in the sugar industry produces large quantities of bagasse that could be used to develop relatively cheaper adsorbents for water remediation. Therefore, this study set out to prepare sugarcane bagasse-based adsorbents namely, non-activated bagasse, bagasse activated with sodium hydroxide, its charcoal activated with phosphoric acid and its ash, and determine their adsorption capacities for the removal of Cd^{2+} and Pb^{2+} ions from ion spiked water under varying experimental conditions of pH, contact time, rate of shake, initial concentration, adsorbent dosage and temperature.

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II. Materials and methods

2.1 Sugarcane bagasse collection and preparation

The study used sugar cane bagasse produced by sugarcane chewing and not machine-crushed by sugar millers. Sugarcane bagasse was obtained from local markets around Kisii town, packed in polythene bags and transported to Kenyatta University chemistry laboratory where it was washed with hot water (80°C) until the washings became clear.. The clean bagasse was allowed to drip in air and dried in an oven at 105°C for 24 hours before use in the preparation of adsorbents.

2.2 Instrumentation

The concentrations of lead and cadmium ions in the various solutions were determined using atomic absorption spectrophotometer (model Buck 210, VGP, US); in flame mode using air-acetylene flame at a wavelength of 228.8nm for cadmium and 283.3nm for lead. The pH of all solutions at a temperature of 25°C were determined using a pH meter (model ST10, China).. All the agitation of mixtures was done using an electrical water bath shaker (model KOTTERMANN D3162, type 3047, West Germany). A series of standards of 2, 4, 6, 8 and 10mg/L were prepared for calibration of the instruments by serial dilution of a working solution (100mg/L) which was prepared from stock solutions (1000mg/L). All stock solutions were prepared from analytical grade salts of the target metals purchased from Fisher Scientific limited, Edmonton, Canada. A standard and blank sample was run after every seven samples to check instrumental drift. Calibration curve method was used to quantify the heavy metal concentration.

2.3 Preparation of Stock Solutions

The 1000mg/L stock solutions of lead (II) and cadmium (II) ions were prepared by separately dissolving 1.600g lead nitrate and 2.745g cadmium nitrate in 1000mL of distilled water in volumetric flasks, respectively. The solutions were then diluted to obtain standard solutions of 2, 4, 6, 8, and 10mg/L using equation

$C_1V_1 = C_2V_2$ Eqn 1Where C_1 is concentration of stock solution, V_1 is volume of stock solution to be diluted upto a volume V_2 to obtain desired concentration C_2. Other working solutions were likewise obtained by serial dilution of stock solutions. Nitric acid and sodium hydroxide solutions were used to adjust pH of the solutions., Concentrated phosphoric acid (80%) was used in the activation of bagasse charcoal while 1M NaOH was used to activate the bagasse.

2.4 Preparation of adsorbents

2.4.1 Bagasse ash

About 300g of clean dry bagasse was burn heated at 600°C for 6 hours to produce 30g of ash, following the procedure described by Oliveira, 2010.

2.4.2 Bagasse charcoal

Charcoal was made by calcining 75g of washed, dry bagasse at 800°C carbonization temperature in a low oxygen environment for two days according to a literature method (Kameyama *et al.*, 2012). This produced 30g of charcoal which was ground in a mortar and activated by soaking in a solution of 80% phosphoric acid for 24 hours, filtered using Whatman filter paper No. 540 and rinsed with distilled water until a pH of 6 was obtained in the rinsing water. It was then dried in an oven at 100°C for 6 hours. The activated charcoal (AC) obtained was used in the adsorption experiments without any further modification.

2.4.3 Activated and non-activated bagasse

About 60g of dry clean bagasse was milled to fine particles using an electric grinder. Half of it was packed in plastic bottles and used for adsorption experiments without any further modification as non-activated bagasse (NAB). The remaining half was activated by soaking in 1M NaOH solution for 18 hours. It was then thoroughly washed in distilled water and dried in an oven at 80°C for 12 hours. The material obtained was used in the adsorption experiments as activated bagasse (AB).

2.5 Preparation of ion spiked water

Analytical grade cadmium nitrate and lead nitrate salts were used to prepare stock solutions of 1000mg/L which were then diluted with distilled water to desired concentrations of metal ions that include 2, 4, 6, 8 and 10mg/L for standards and 80mg/L for test solutions. Other concentrations prepared for testing the effect of initial metal ion concentration are 20, 40, 60, 80, 100, 200, 400, 600 and 800mg/L. The pH of spiked water was adjusted by adding drops of 0.1M nitric acid or 0.1M sodium hydroxide solution to attain pH of 6 for all experiments except the one investigating on effect of pH where adjustments were ranging from pH values of 2-8.

3 Batch adsorption experiments

For each experiment, 50mL aqueous solution of lead (II) ions and cadmium (II) ions were taken in 120mL plastic bottles containing a 0.1g of the adsorbent and agitated at a speed of 7rps, 25°C and pH of 6 for one hour, and then filtered through a Whatman filter paper No. 540. Adjustments were done on each parameter to determine their effects on adsorption efficiency. The concentrations of the residual Pb^{2+} and Cd^{2+} ions in the filtrates were determined using flame atomic absorption spectrometry. The percentage removal and amount of Pb^{2+} ions adsorbed were calculated using the equation 2 and 3, respectively.

$$\% removal = \frac{c_i - c_f}{c_i} \times 100 \qquad \text{Eqn 2}$$

Where c_i is the initial concentration (mg/L) and c_f is the final concentration (mg/L).

$$q = \left(\frac{c_o - c_t}{m} \right) v \qquad \text{Eqn 3}$$

Where q is the quantity of metal adsorbed per specific amount of adsorbent (mg/g) (Reza *et al.*, 2011); c_o and c_t are concentrations of metal ions in mg/L before and after adsorption time t ; v is volume of the solution used in litres, m is weight in grams of adsorbent used.

These equations were similarly used for calculating percentage removal and amount of Cd^{2+} ions adsorbed.

The adsorption data for Pb^{2+} and Cd^{2+} ions were fitted into Langmuir and Freundlich adsorption isotherm models as shown in equations 4 and 5, respectively.

$$\frac{1}{q_e} = \frac{1}{q_o} + \frac{1}{q_o k_l} \times \frac{1}{c_e} \qquad \text{Eqn 4}$$

Where q_o and k_l are Langmuir parameters related to maximum adsorption capacity and free energy of adsorption respectively; c_e is the equilibrium concentration in the aqueous solution and q_e is the equilibrium adsorption capacity of adsorbent.

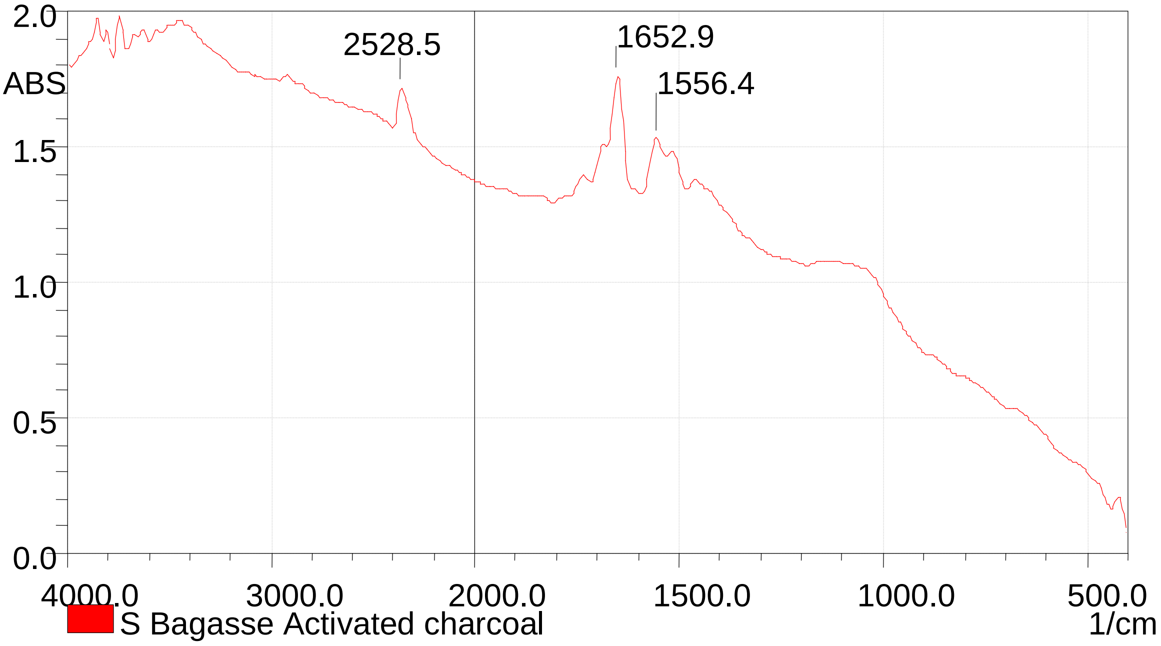
$$\log q_e = \log k_f + \frac{1}{n} \log c_e \qquad \text{Eqn 5}$$

Where k_f and n are Freundlich constants that indicate adsorption capacity and adsorption intensity respectively.

III. Results and discussion

Preparation of the adsorbents

Four adsorbents were prepared namely sugarcane bagasse: raw sugarcane bagasse (activated and non-activated), and sugarcane bagasse ash and sugarcane bagasse charcoal. The adsorptive capacities for Pb^{2+} and Cd^{2+} ions from water were determined for all four adsorbents under different experimental conditions and the results are herein reported and discussed. The following are FTIR results showing the functional groups of adsorbents.


2.0
ABS
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2528.5
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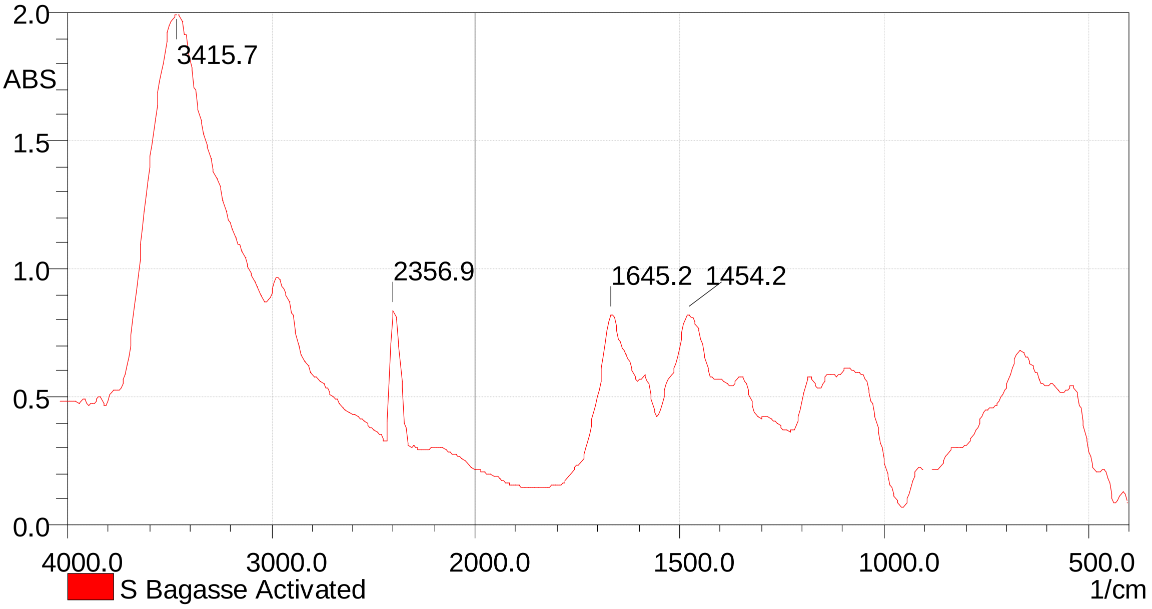
🟥 S Bagasse Activated charcoal


2.0
ABS
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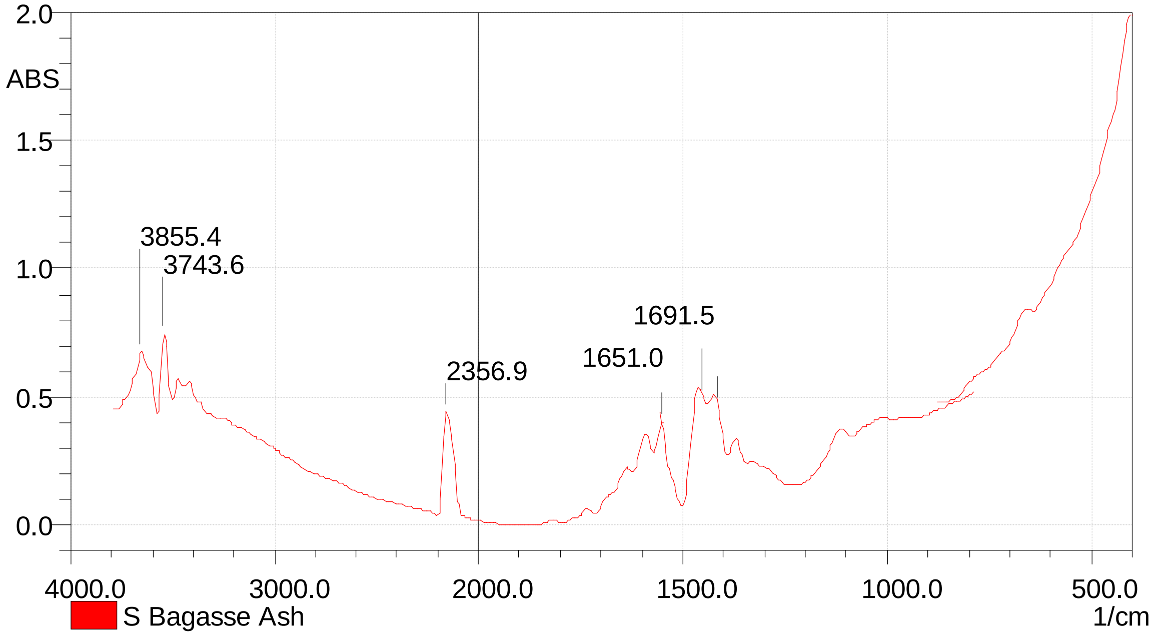

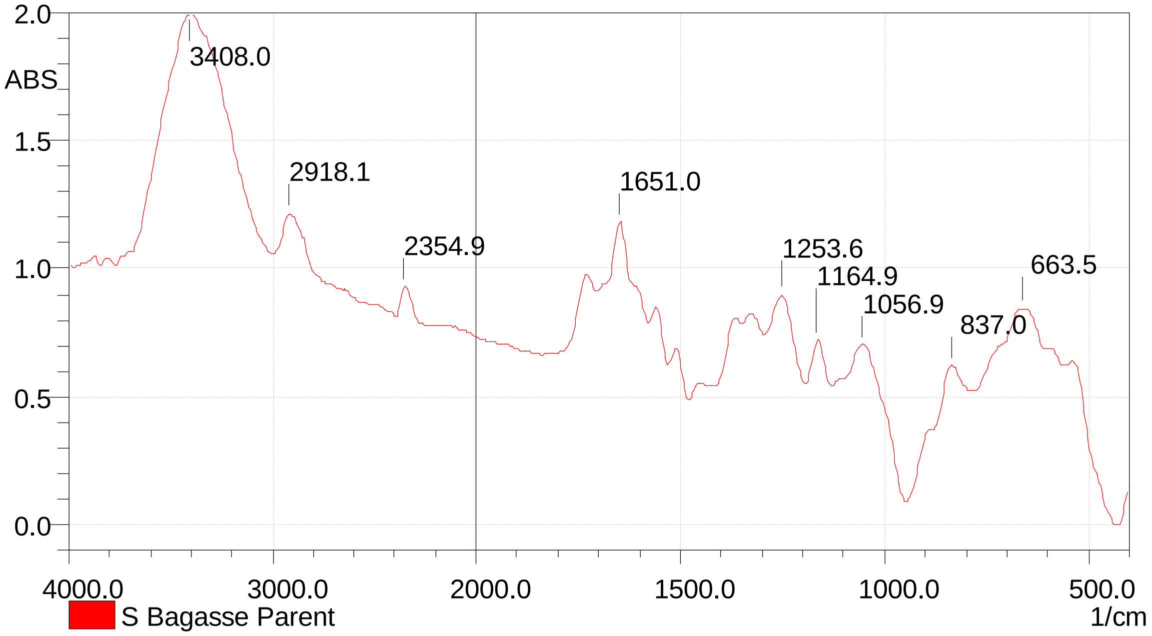

3.1 *Effect of Contact Time*

The amount of time required for optimum adsorption of the ions by the various bagasse based adsorbents were determined by varying the contact time for a constant amount of adsorbent at ambient temperature. The results are summarized in Figures 1 and 2 which show the trends in variation of percentage removal of lead and cadmium ions, respectively, as contact time was varied in intervals of 20 minutes from 20 to 100minutes.

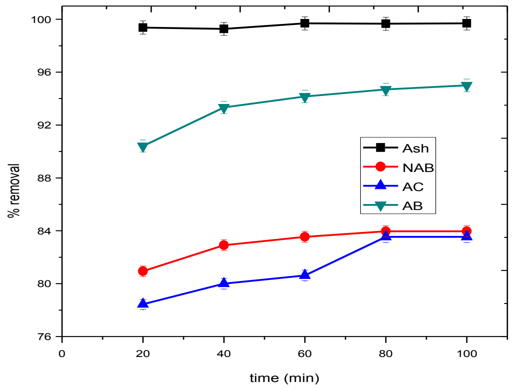

Figure 1: Effect of contact time on percentage removal of Pb^{2+}at 25°C, 80mg/L, 7rps, pH 6, adsorbent dose 0.1g

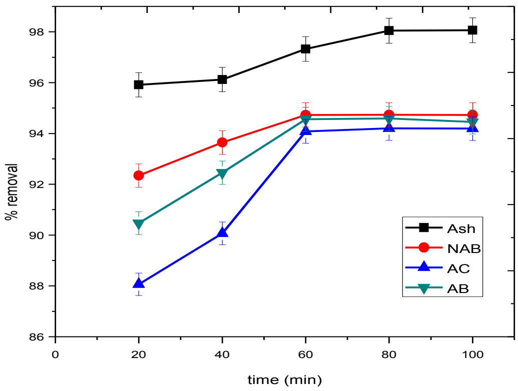

Figure 2: Effect of contact time on percentage removal of Cd^{2+}at 25°C, 80mg/L, 7rps, pH 6, adsorbent dose 0.1g

There was a general increase in percentage removal of ions with increase of contact time till equilibrium was achieved at 60 to 80 minutes. However, the increase in the removal of Pb^{2+} ions by sugarcane bagasse ash from 40 to 100 minutes was not statistically significant at 95% confidence level. The increase in removal of lead (II) ions from 40 to 100 minutes by activated bagasse (AB) and non-activated bagasse (NAB) was also not statistically significant at 95% confidence level, p-value <0.001. This indicates that maximum removal can occur after 40 minutes. These results are consistent with those obtained by Wang and coworkers who investigated removal of cadmium ions from water using bamboo charcoal (Wang *et al*., 2010). The rapid adsorption at the beginning was possibly due to the high concentration gradient between the ions in solution and the large number of vacant sites on the surface of adsorbent. With time, the adsorption onto the remaining unoccupied sites becomes more difficult because of the repulsive forces between neighboring adsorbed solute particles in solid phase and those in the bulk liquid phase (Sahu *et al*., 2008). The highest percentage removal of 98.05±0.04 % for cadmium (II) ions and 99.38±0.00 % for lead (II) ions using Ash was realized at 80 and 60 minutes, respectively.

3.2 *Effect of pH*

The different adsorbents were exposed to the test solutions at different pH values of 2, 4, 6, 7 and 8 and their adsorption capacities determined. The results are summarized in Figure 3 and 4. . There was precipitation seen at pH beyond 8 due to formation of insoluble metal hydroxides; hence the study did not go beyond pH 8.

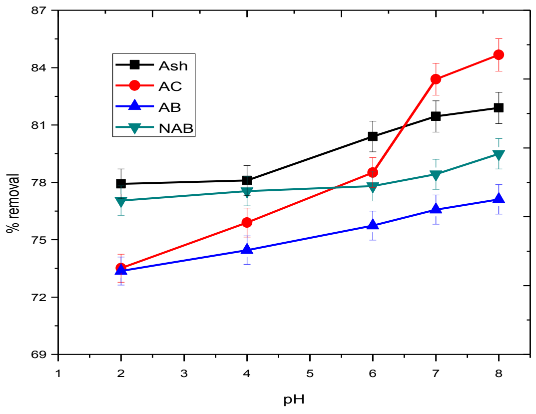

Figure 3: Effect of pH on percentage removal of Cd^{2+} at 25°C, 80mg/L, 7rps, contact time 1hour, adsorbent dose 0.1g

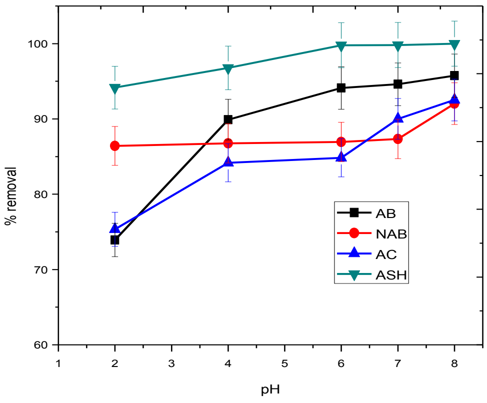

Figure 4: Effect of pH on percentage removal of Pb^{2+} at 25°C, 80mg/L, 7rps, contact time 1hour, adsorbent dose 0.1g

The percentage removal of metal ions increased with pH. Between pH 2 and 4 the observed increase in removal of Cd^{2+} ions by sugarcane bagasse ash was, however, not statistically significant at 95% confidence level, p-value <0.001. Similarly, between pH 2 and 6 the observed increase in percentage removal of Pb^{2+} ions by NAB was not statistically significant at 95% confidence level, p-value <0.001. The increase could be attributed to the reduced competition between H^{+} ions and metal cations for the same active site (Machida *et al.*, 2012; Al-Riyami *et al.*, 2014). At low pH values, there is excessive protonation of the active sites hindering the formation of bonds between metal ions and the adsorbent sites. As the pH increases, the adsorbent surface turns out to be less positive as linked H^{+} is discharged from the active sites. Thus, electrostatic attraction between the metal ions and adsorbent surface tends to increase. From the results, it is evident that removal of both lead and cadmium ions from water is pH dependent. At pH 8, highest percentage removal of 84.66±0.06 % for cadmium (II) ions and 100±0.00 % for lead (II) ions was attained using AC and Ash respectively. However, at pH 8, precipitation of some ions may have occurred hence pH 7 is taken to be the optimum pH for adsorption with highest removal of 99.82±0.00 % for Pb^{2+} using ash and 83.39±0.05 % for Cd^{2+} using AC.

3.3 *Effect of Temperature*

The percentage removal of Pb^{2+} and Cd^{2+} ions by the four adsorbents was investigated as a function of temperature and the observations summarized graphically in figures 5 and 6 respectively. The experiments were done at temperatures of 20, 40, 60, 80 and 100°C.

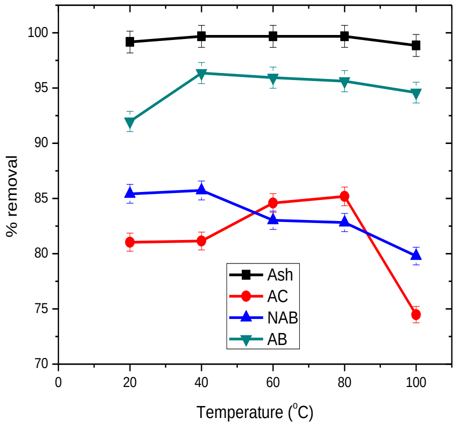

Figure 5: Effect of temperature on percentage removal of Pb^{2+} at pH 6, 80mg/L, 7rps, contact time 1hour, adsorbent dose 0.1g

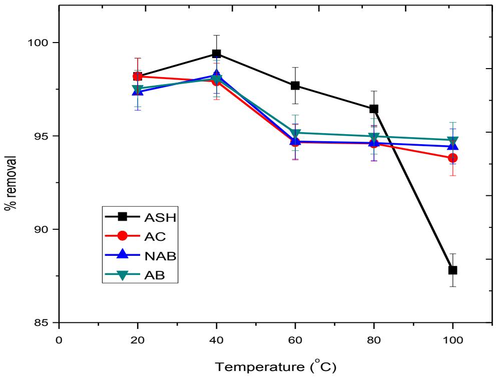

Figure 6: Effect of temperature on percentage removal of Cd^{2+} at pH 6, 80mg/L, 7rps, contact time 1hour, adsorbent dose 0.1g

Each adsorbent showed an increase in percentage removal of both ions up to an optimum temperature of 40°C after which there was a decreasing trend in adsorption. The increase in adsorption with increase in temperature may be attributed to increase in kinetic energies of ions which increase their diffusion speed. Furthermore, additional sorption sites also open up increasing the capacity of the adsorbent.

The decrease in adsorption beyond optimum temperature may be attributed to desorption which may have occurred since metal ions acquire sufficient energy to overcome the enthalpy of adsorption (Mataka *et al.*, 2010). Excess heat above optimum for each adsorbent also decomposes the metal-adsorbent surface adsorption complexes and alters adsorbent surface sites hence decreased ion removal (Muriithi *et al.*, 2014). These results were consistent with those reported by Adebowale and co-workers on adsorption of lead (II) and cadmium (II) ions onto tripolyphosphate-modified kaolinite clay (Adebowale *et al.*, 2008). Generally, removal of lead and cadmium ions is affected by temperature. The highest percentage removal of 99.39±0.00 % and 99.69±0.00 % for cadmium (II) and lead (II) ions respectively was realized at 40°C using Ash.

3.4 *Effect of Initial Metal Ion Concentration*

Figure 7 and 8, show how the percentage of $Pb^{2+}$and Cd^{2+} ions, removed varied as the initial concentrations were varied from 20mg/L to 1000mg/L.

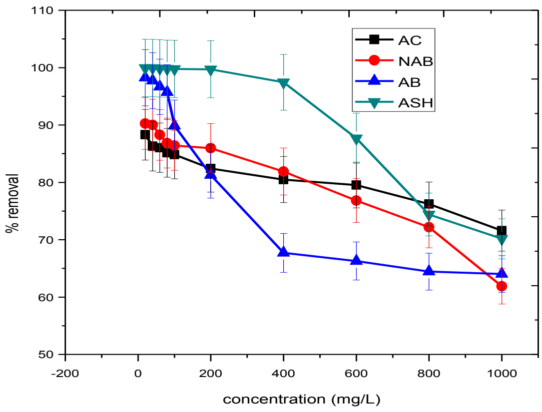

Figure 7: Effect of initial metal ion concentration on percentage removal of Pb^{2+} at pH 6, temperature 25°C, 7rps, 1hour contact time and adsorbent dose of 0.1g

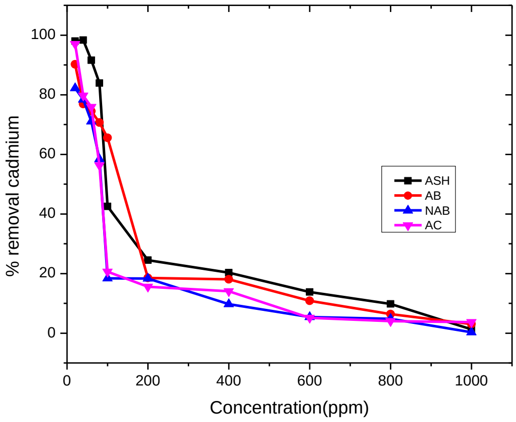

Figure 8: Effect of initial metal ion concentration on percentage removal of Cd^{2+} at pH 6, temperature 25°C, 7rps, 0.1g adsorbent and 1hr contact time

Generally, there was decline in percentage removal of both ions as initial concentration increased. The decline is accredited to saturation of active adsorption sites and probably the repulsive forces between adjacent adsorbed solute molecules in solid phase and in the bulk liquid phase. All adsorbents generally had lower percentage removal for cadmium (II) ions than lead (II) ions due to differences in some chemical properties. These include smaller hydrated radius of Pb^{2+} (0.401 nm) than Cd^{2+} (0.426 nm), higher affinity of Pb^{2+} for most functional groups in organic matter and also it has higher electronegativity than Cd^{2+} ion (Nightingale, 1959).

These properties make it better than Cd^{2+} for electrostatic attractions and inner surface reactions (Huheey, 1983; McBride, 1994). Other studies have also shown that there is preference for Pb^{2+} over Cd^{2+} as exhibited by some adsorbents such as soil (Appel and Lena, 2002). From these results, it can be deduced that efficiency of removal of lead (II) and cadmium (II) ions from water depends on initial ion concentration where lower concentration of ions in solution increase the removal efficiency of adsorbents. Highest percentage removal of 98.32±0.00 % and 99.95±0.00 % for cadmium (II) and lead (II) ions respectively was achieved at 20mg/L using Ash.

3.5 Effect of agitating speed

Figures 9 and 10 respectively show how the percentage of Cd^{2+} and Pb^{2+} ions adsorbed as the agitation speed was varied from 2rps to 10 rps.

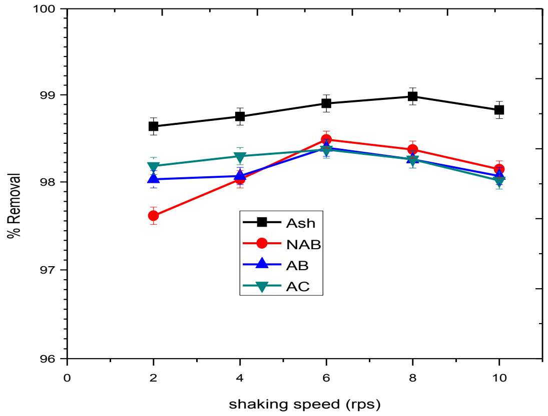

Figure 9: Effect of agitation speed on percentage removal of Cd^{2+} at pH 6, temperature 25°C, 80mg/L, contact time 1hour and adsorbent dose 0.1g

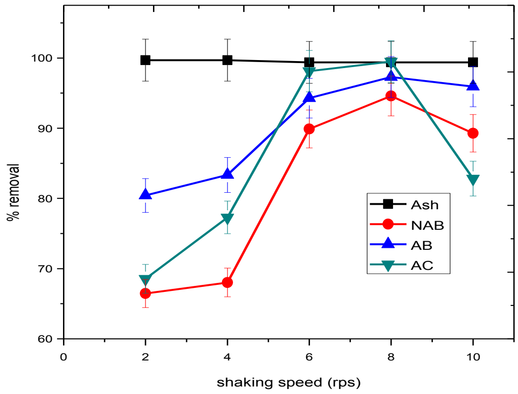

Figure 10: Effect of agitation speed on percentage removal of Pb^{2+} at pH 6, temperature 25°C, 80mg/L, contact time 1hour and 0.1g of adsorbent

The results show general increase in percentage removal of both ions as agitation speed increased up to 6rps for cadmium ions and 8rps for lead ions. Further increase in speed decreases the percentage removal. The increase was however not statistically significant between 2 to10rps for Ash in removal of lead (II) ions at 95% confidence level with p-value <0.001. Thus the efficiency of ash was not affected much by shaking speed and this may be due to other processes like precipitation since ash may have increased the solution pH (Noonpui *et al.*, 2010). The increase of adsorption with increase in agitation speed for other adsorbents may be due to the decrease in boundary layer thickness close to the adsorbent particles which is as a result of increased degree of mixing (Argun *et al.*, 2007). These results are comparable to those obtained by Bernard and coworkers who investigated the removal of heavy metals from industrial wastewaters using coconut shell (Bernard *et al*., 2013). Since the diffusion of heavy metal ions to the layer between adsorbent particles and the solution increases with increase in shaking speed, the external mass transfer rate of the metal ions may have also increased and the equilibrium would have been achieved more rapidly (Muriithi *et al.*, 2012). However, when the shaking speed rose above optimum for each adsorbent, percentage removal decreased. This may have occurred because the high agitation speed provided additional energy enough to break newly formed bonds between the adsorbent surface and the metal ions therefore desorption takes place. Thus, optimum agitation enhances proper contact between the metal ions in solution and the adsorbent binding sites therefore stimulates effective transfer of adsorbate ions to the adsorbent sites (Argun *et al.*, 2007). Highest percentage removal of 98.98±0.01 % for cadmium (II) ions was reached at 8rps using Ash. At the same speed of 8rps, highest removal of 99.48±0.01 % for lead (II) ions was achieved using AC.

3.6 Effect of Adsorbent Dosage

Figures 11 and 12 show graphically how the percentage removal of Cd^{2+} and Pb^{2+} ions respectively varied as adsorbent doses were increased from 0.1 to 0.5grams.

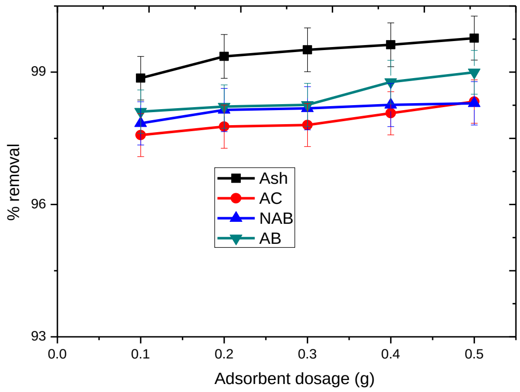

Figure 11: Effect of adsorbent dosage on percentage removal of Cd^{2+} at pH 6, temperature 25°C, 80mg/L, contact time 1hour and 7rps

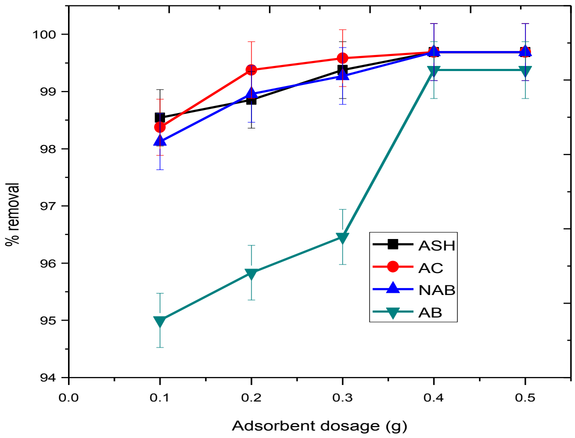

Figure 12: Effect of adsorbent dosage on percentage removal of Pb^{2+} at pH 6, temperature 25°C, 80mg/L, contact time 1hour and 7rps

The results show that as the adsorbent doses were increased, the percentage removal of both ions increased up to 0.4g of adsorbent for lead (II) ions and 0.5g for cadmium (II) ions. This is due to the fact that lead (II) ions have smaller hydrated radius (0.401 nm) compared to Cd^{2+} (0.426 nm) (Nightingale, 1959). This means that equilibrium is reached much earlier for Pb^{2+} than Cd^{2+}. The increase in percentage removal when the dosage was increased would be attributed to unsaturation of adsorption sites and increase in the number of available adsorption sites for metal ion attachment (Bernard *et al.*, 2013). The highest percentage removal achieved for cadmium (II) ions was 99.77±0.00 % at 0.5g dosage using Ash. For lead (II) ions, highest removal of 99.69±0.00 % was reached at 0.4g dosage using Ash, AC and NAB.

3.7 Adsorption Isotherms

Table 1: Langmuir and Freundlich constants for cadmium ions

	Langmuir				Freundlich	
Sample	q_o mg/g	k_l dm^3/g	R^2	1/n	k_f mg/g	R^2
ASH	40.816±0.10	0.887	0.914	0.222	15.740±0.01	0.695
NAB	43.860±0.01	0.003	0.995	-1.202	72.780±0.01	0.892
AB	52.910±0.01	0.104	0.975	0.324	5.970±0.05	0.746
AC	45.413±0.03	0.002	0.989	0.333	11.066±0.10	0.596

Table 1 shows that data for Ash, NAB, AB and AC on cadmium (II) ions best fit into Langmuir isotherm model owing to the higher correlation coefficients (R^2) of 0.914, 0.995, 0.975 and 0.989 respectively compared to the R^2 of 0.695, 0.892, 0.746 and 0.596 respectively for Freundlich isotherm model. This suggests monolayer adsorption onto homogenous surface and that the removal of cadmium ions was by chemisorption (Karthikeyan *et al.*, 2005). The adsorption affinities of the adsorbents were expressed by k_l values of 0.887, 0.003, 0.104 and 0.002 dm^3/g respectively for Ash, NAB, AB and AC. The adsorption capacities of the adsorbents represented by q_o values were 40.816, 43.860, 52.910 and 45.413 mg/g for Ash, NAB, AB and AC respectively. Despite the high capacity (k_f) of 72.780 mg/g for NAB on Freundlich model, 1/n value was less than 0.0 indicating that adsorption was not favourable for this model in addition to the lower R^2 value (Mohan and Karthikeyan, 1997).

Table 2: Langmuir and Freundlich constants for lead ions

	Langmuir			Freundlich		
Sample	q_o mg/g	k_l dm^3/g	R^2	1/n	k_f mg/g	R^2
ASH	79.302±0.01	1.001	0.730	0.262	62.661±0.01	0.736
NAB	48.733±0.05	1.001	0.932	0.704	6.622±0.09	0.958
AB	96.154±0.01	1.000	0.955	0.453	18.365±0.01	0.978
AC	500.000±0.01	1.000	0.899	0.867	3.681±0.01	0.951

Table 2 shows that data for Ash, NAB, AB, and AC best fit Freundlich isotherm model since the correlation coefficients (R^2) are higher with values of 0.736, 0.958, 0.978 and 0.951 respectively compared to R^2 values of 0.730, 0.932, 0.955 and 0.899 respectively in Langmuir model. This implies multilayer adsorption onto heterogeneous surface and that the removal of lead ions was by physisorption. Adsorption intensity (1/n) values were between 0 and 1, that is, 0.262, 0.704, 0.453 and 0.867 respectively for Ash, NAB, AB and AC. This is an indication that adsorption was favourable in Freundlich model. The low R^2 for ash may be an indication that lead ions were removed by both precipitation and adsorption as the ash could have increased the solution pH (Noonpui *et al.*, 2010). It is important to note that though the adsorbents show higher adsorption capacities (q_o) of 79.302, 48.733, 96.154 and 500.000 mg/g for Ash, NAB, AB, and AC respectively in Langmuir model compared to their corresponding k_f values of 62.661, 6.622, 18.365 and 3.681 mg/g respectively in Freundlich model, their R^2 values are lower in Langmuir model meaning that the data did not fit well in Langmuir model.

4 Conclusions

Various bagasse-based adsobents namely activated bagasse (AB), non-activated bagasse (NAB), its activated charcoal (AC) and ash were prepared and evaluated for efficacy and efficiency for removal of heavy metal ions from water. From the results, it was found that highest efficiency in terms of percentage removal was achieved at a temperature of 40°C, 8rps shaking speed, pH value of 8 and 20 mg/L initial metal ion concentration for both cadmium (II) and lead (II) ions. For cadmium ions, the highest % removal for AB, NAB, AC and Ash was 99.00, 98.37, 98.33 and 99.77% respectively at 95% confidence level with p-values <0.001. For lead ions, highest % removal for AB, NAB, AC and Ash was 99.38, 99.69, 99.69 and 99.95% respectively at 95% confidence level with p-values <0.001. Activated bagasse (AB) had the highest adsorption capacity (52.910 mg/g) for cadmium (II) ions while bagasse ash had the highest capacity (62.661 mg/g) for lead (II) ions. The adsorbent dosage that showed highest removal was 0.4g for lead (II) ions and 0.5g for cadmium (II) ions. Highest removal occurred at 60 minutes contact time for lead (II) ions and 80 minutes for cadmium (II) ions.

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