Adsorption Studies on the Removal of Hexavalent Chromium-Contaminated Wastewater using Activated Carbon and Bentonite

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Abstract

In this study, the adsorption potential of activated carbon and bentonite for removal of Cr(VI) ions from wastewater has been investigated. The study involves batch type experiments to investigate the effects of initial concentration, adsorbent dose, agitation speed, contact time, temperature and pH of solution on adsorption process and the evaluation of optimum conditions. The adsorption process has fit pseudo-second order kinetic models. Langmuir and Freundlich adsorption isotherm models were applied to analyze adsorption data, and both were found to be applicable to this adsorption process. Thermodynamic parameters, e.g., \( \Delta G^\circ \), \( \Delta S^\circ \) and \( \Delta H^\circ \) of the on-going adsorption process have also been calculated and found the sorption process as endothermic. Finally, it can be seen that bentonite was found to be more effective for the removal of Cr(VI) than activated carbon, under the same experimental conditions.

Keywords: Wastewater, Chromium, Activated Carbon, Bentonite, Removal

1. Introduction

The pollution of heavy metal ions in the environment is a critical problem because of their toxicity and other adverse effects on the receiving waters and/or soils. They must be removed from wastewater before discharging them into water bodies. Water pollution by chromium is due to both, natural sources and manmade activities. Chromium is found in rocks, animals, plants, soils and in volcanic dusts and gases. Various industrial processes which involve the use of Cr(VI), such as steel production, electroplating, leather tanning, nuclear power plants, textile industries, wood preservation, anodizing of aluminum, water-cooling and chromate preparation. Discharge of wastes containing chromium occurs into environment, with its varied concentration between 5 and 220 mg/L. Such discharge ultimately contaminates soil and water (Okuda et al, 1975; Ouki & Neufeld, 1997 and Altundogan, 2005). Chromite world mine production was estimated at a gross mass of 13x10^6 metric tons in 2002 (Papp, 2003).

In nature chromium exists in two most stable oxidation states, i.e., trivalent and hexavalent forms in aqueous systems. Although, at trace level, the trivalent form is considered as an essential nutrient (Alloway, 1995 and Rojas et al, 2005) whereas hexavalent form of chromium is toxic, carcinogenic and mutagenic in nature (Norseth, 1981; USDHHS, 1991; Cieslak-Golonka, 1995; Myers et al, 2000; Zhitkovich et al, 2002 and Dupont & Guillou, 2003). Kowalski (1994) revealed that the hexavalent form is about 500 times more toxic than trivalent form. Furthermore, Cr(VI) is highly mobile in soil and aquatic system, and also is a strong oxidant capable of being adsorbed by skin (Singh, & Singh, 2002). Considering its
toxicity and carcinogenic nature, the maximum levels permitted for trivalent chromium in wastewater is 5 mg/L and for hexavalent chromium as 0.05 mg/L (BIS, 1991 and Acar & Malkoc, 2004).

The most common methods of Cr(VI) ions removal from aqueous system are chemical precipitation, ion exchange, membrane processes, electrodialysis and adsorption (Patterson, 1977; George, 1985; Tiravanti et al, 1997 and Dahbi et al, 1999). Hexavalent chromium usually exists in wastewater as oxyanions such as chromate \( \text{Cr}_2\text{O}_7^{2-} \) and dichromate \( \text{Cr}_2\text{O}_4^{2-} \) and does not precipitate easily using conventional precipitation methods. Ion exchange, reverse osmosis and electrodialysis are efficient for Cr(VI) ions removal but the cost is relatively high (George, 1985).

Furthermore, chromium removal techniques such as coprecipitation, membrane techniques and solvent extraction are challenged by the removal of lower concentrations of metals from solution (Valix et al, 2006). On the other hand, adsorption has been proved to be one of the respective methods, which is simple, selective and economical process for the removal of heavy metal ions from aqueous solution.

In this work two different adsorbents (activated carbon and bentonite) were used for the removal of hexavalent chromium (Cr\(^{VI}\)) from wastewater. Bentonite is considered by many authors as main candidate in the decontamination and treatment of detrimental metal ions, because of its large specific surface area and high adsorption capacity (Tsai et al, 2001; Lin & Juang, 2002 and Al-Qunaibit et al, 2005).

The objectives of the present study are: 1) to study the effect of contact time of Cr\(^{VI}\) metal ions adsorption on activated carbon and bentonite; 2) to investigate the influence of pH on the adsorption of Cr\(^{VI}\); 3) to study the adsorption of Cr\(^{VI}\) at different temperatures and to calculate the adsorption thermodynamic parameters (i.e., \(\Delta H\), \(\Delta S\) and \(\Delta G\)); 4) to find the effect of adsorbent dose and initial concentration on the adsorption of Cr\(^{VI}\); 5) to describe the experimental data of adsorption isotherms through Langmuir and Freundlich models; 6) to investigate the adsorption properties of Cr\(^{VI}\) and to discuss its adsorption mechanism.

2. Materials and Methods

2.1. Materials and Reagents

All chemicals used in present work were either of analytical reagent (AR) or laboratory reagent (LR) grade. \(\text{K}_2\text{Cr}_2\text{O}_7\) (99%) and \(\text{H}_2\text{SO}_4\) (98% w/w) supplied by BDH chemicals Ltd. Distilled water was used in all preparations. Potassium dichromate \(\text{K}_2\text{Cr}_2\text{O}_7\) and deionized water were used to prepare synthetic chromium containing wastewater. Powder Activated Carbon (PAC) was produced by ADWIC whereas bentonite by Alexandria Company for Refractories.

2.2 Adsorbents Analysis

The surface area and particle size analyses for the two adsorbents i.e. PAC and Bentonite, were checked and recorded (Table 1). Elemental analysis for bentonite was evaluated using x-ray fluorescence (Philips PW 1390) as shown in (Table 2).

2.3. Apparatus and Instrumentation

Magnetic hot plate stirrer was used to stir the heavy metal ions solutions with adsorbents (activated carbon or bentonite). A known volume of heavy metal ions solution of a known initial ions concentration was mixed and stirred with a specific amount of adsorbent for certain period of time at fixed temperature and agitation rate. The pH values of the solutions were measured by digital pH meter (Model µ pH system-361, India). The metal ion concentration was measured using atomic-absorption spectrophotometer, AAS, (Model, AA55; Varian Inc., USA).

2.4. Experimental Procedures

Potassium dichromate \(\text{K}_2\text{Cr}_2\text{O}_7\) and redistilled water were used to prepare a stock solution of 1000 ppm Cr(VI) concentration, which was further diluted for the preparation of test solutions. Several solutions with different initial concentrations of potassium dichromate (50, 100, 200, 300 and 400 ppm) were prepared. The required pH was adjusted by drop wise addition of 0.1N \(\text{H}_2\text{SO}_4\), depending on the acidity of the sample. All experiments were carried out at \(25\) °C by adding different amounts of adsorbents (0.1, 0.3, 0.5, 0.7 and 1.0 g) to different concentrations of 250 ml of heavy metal ions solution. The agitation rate for all experiments was \(200\) rpm and the residence time was \(0, 15, 30, 45, 60, 75, 90\) and 120 minutes).

1 ml of sample was taken from reaction solution and diluted to 10 ml by redistilled water, adsorbents were then separated from the solution by using filter paper (Watman No. 40) and the residual Cr\(^{VI}\) ions concentration in the solution was then determined by atomic-absorption spectrophotometer (AAS).

The effects of several parameters, such as contact time, initial concentration, adsorbent dose, pH and temperature on the adsorption of Cr(VI) ions onto activated carbon and bentonite were studied.
2.5. Data Analysis

The uptake of Cr(VI) ions was calculated from the Mass Balance, define as the amount of solute adsorbed onto the solid. It equals the amount of solute removed from the solution. Mathematically, it can be expressed by Equation 1:

\[ q_e = \frac{(C_i - C_e)}{S} \]  

Where:
- \( q_e \): The heavy metal ions concentration adsorbed by an adsorbent at equilibrium (mg of metal ions/g of adsorbent)
- \( C_i \): The initial concentration of metal ions in the solution (mg/L)
- \( C_e \): The equilibrium concentration or final concentration of metal ions in the solution (mg/L)
- \( S \): The dosage concentration.

Dosage concentration, \( S \), is expressed by Equation 2:

\[ S = \frac{m}{V} \]  

Where:
- \( V \) is the initial volume of metal ions solution used (L) and \( m \) is the weight of dried used adsorbent (g)

The percentage of adsorption (%) is calculated using Equation 3:

\[ \% \text{ adsorption} = \left( \frac{(C_i - C_e)}{C_i} \right) \times 100 \]  

3. Results and Discussion

3.1. Effect of Contact Time

The effect of contact time on Cr(VI) adsorption on both activated carbon and bentonite was investigated to study the rate of Cr(VI) ions removal (Figure 1). It is easily seen from Figure 1 that the percentage removal of Cr(VI) metal ions increased with increasing the contact time. The % removal was 73% for activated carbon and 77% for bentonite after 120 min. It is clear that, at the beginning % removal increased rapidly in few minutes, by increasing contact time, % removal increased lightly and slowly till reach maximum value and this can be explained on the basis that as initially a large number of vacant surface sites are available for adsorption of metal ions but with passage of time the surface sites become exhausted (Zhan et al., 2000). These results indicate that the activated carbon and bentonite have a very strong capacity for adsorption of Cr(VI) ions in solutions.

3.2. Effect of Adsorbent Dose

The percentage adsorption of Cr\(^{6+}\) ions onto activated carbon and bentonite was studied at different adsorbent doses (0.4, 1.2, 2.0, 2.8 and 4.0 g/L), keeping pH (2.0), temperature (25°C), contact time (120 minutes) and initial Cr\(^{6+}\) ions concentration of (400 mg/L) constant (Figure 2). The results showed that with increase in adsorbent dose,
the percentage adsorption of Cr\(^{6+}\) was increased. Maximum removal was observed with adsorbent dose of 4 mg/L for both activated carbon and bentonite. Increasing the percentage of adsorption with adsorbent dose may be due to the increase in adsorbent surface area and availability of more adsorption sites (Namasivayam et al., 1998 and Garg et al., 2007). Unit adsorption was, however, decreased with increasing in adsorbent dose (Figure 3). This is may be due to overlapping of adsorption sites as a result of overcrowding of adsorbent particles (Garg et al., 2007).

The percentage removal of Cr\(^{6+}\) was studied by varying Cr\(^{6+}\) concentrations from 50 to 300 mg/L with 4 g/L adsorbent dose of activated carbon and bentonite with the same standard conditions as illustrated in Figure 4.

The results represented in Figure 4, showed that the percentage Cr\(^{6+}\) ions adsorption was decreased with increasing in initial concentration. But the actual amount of Cr\(^{6+}\) ions adsorbed per unit mass of the adsorbent was increased with increasing in Cr\(^{6+}\) ions concentration in the test solution as illustrated in Figure 5. At low concentration the ratio of surface active sites to the total metal ions in the solution is high and hence all metal ions interact with the adsorbent and are removed quickly from the solution. However, the amount of metal ions adsorbed per unit weight of adsorbent, \( q \), is higher at high concentration. According to these results, the initial Cr\(^{6+}\)
ions concentration plays an important role in the adsorption capacities. Higher concentrations of metal ions were used to study the maximum adsorption capacity of adsorbent (Mohanty et al, 2006 and Karthikeyan et al, 2005).

![Figure 5. The Effect Of Initial Concentration (Namely 50, 100, 200 and 300 mg/L) on Quantity Adsorbed of Cr\(^{+6}\) Ions (Conditions: Adsorbents Dose: 4 g/L; Agitation Speed: 200 rpm; Temperature: 25 ˚C)](image)

### 3.4. Effect of pH on the Uptake of Cr\(^{+6}\)

The pH of the solution is an important variable which controls the adsorption of the metal ions at the solid-water interface. Hence, the influence of pH on the adsorption of Cr\(^{+6}\) ions onto activated carbon and bentonite was examined in the pH range of 1-4. These results were represented in Figure 6, which showed that the adsorption capacities of Cr\(^{+6}\) ions onto both adsorbents increased significantly, with decreasing pH value and the maximum removals of Cr\(^{+6}\) ions by both adsorbents for contact time (120 min) were carried out at pH (2.0). The improved removal of Cr(VI) at low pH is probably due to reduction of hexavalent chromium to trivalent chromium ions (Anon, 1995), Equation 4.

\[
\text{Cr}_2\text{O}_7^{2-} + 14\text{H}^+ + 6\text{e}^- \rightarrow 2\text{Cr}^{3+} + 7\text{H}_2\text{O}
\]

Also at low pH, there is presence of a large number of H\(^+\) ions, which in turn neutralize the negatively charged adsorbent surface, thereby reducing hindrance to the diffusion of dichromate ions (Sharma, & Forster, 1994).

### 3.5. Effect of Temperature

Figure 7 indicates the effect of temperature on the removal efficiency of Cr\(^{+6}\) ions from wastewater using activated carbon and bentonite. Four different temperatures were considered in this study (25, 30, 35 and 40° C). The figure showed that the removal efficiency increased by increasing the temperature, where the maximum adsorption (76% approx.) is achieved at 40 ˚C in case of activated carbon. The adsorption process does not usually operate at high temperature (more than 40 ˚C) and this may be due to the damage of active binding sites and because of high operational cost, so experiments were performed at the temperature range (25-40 °C). Furthermore, in the case of activated carbon, the enhancement in adsorption capacity may be due to the chemical interaction between adsorbates and adsorbent, creation of some new adsorption sites or increased the rate of intraparticle diffusion of Cr(VI) ions into the pores of the adsorbent at higher temperatures (40 ˚C). Also, it is mentioned that, the metal cations become more faster with increasing temperature. While, in the adsorption of Cr(IV) onto bentonite at 40 °C, probably the available binding sites have been saturated and the adsorbent no longer can bind further ions as the number of sites available on the adsorbent is constant.

![Figure 6. Effect of pH for the Adsorption of Cr(VI) Ions onto Activated Carbon and Bentonite at 25 °C](image)

The augmentation of the removal efficiency by stepping up the temperature are due to: Firstly, the higher temperatures activate the metal ions for enhancing adsorption at the coordinating sites of the adsorbent, and the metal cations became more faster (Babel & Kurniawan, 2003 and Inglezakis et al, 2004). Secondly, acceleration of some originally slow step(s) and creation of some new activation sites on the adsorbent surface (Domenico & Schwartz, 1990). Herein, the phenomenon can be observed in both adsorbents, but activated carbon is affected much higher than in case bentonite Figure 7.

### 6. Adsorption Kinetics

Adsorption kinetics describe the solute uptake rate which intern control the residence time and hence the size of adsorption equipment. The experimental results for the adsorption of Cr\(^{+6}\) ions onto activated carbon and bentonite showed rapid initial adsorption rate followed by a slower rate. Initially, the adsorption sites are open and the metal ions interact easily with the sites and hence a higher rate of...
adsorption is observed. Further, the driving force for adsorption (the concentration difference between the bulk solution and the solid-liquid interface) is higher initially and this leads to a higher adsorption rate. However, after the initial period, slow adsorption may be due to slower diffusion of solute into the interior of the adsorbent. Several adsorption kinetics models have been developed to understand the adsorption kinetics and rate limiting step (Lagergren, 1898; Ho & Mckay, 1999; Uzun & Guzel, 2000 and Ricordel, 2001).

The following models have been widely used (Lagergren, 1898; Ho & Mckay, 1999; Uzun & Guzel, 2000 and Ricordel, 2001).

Equations 7 and 8 give the pseudo-second order model as:

\[
d\frac{dq}{dt} = k_2(q_e - q) \quad \text{Equation 7}
\]
\[
t = t/q_e + 1/k_2 q_e^2 \quad \text{Equation 8}
\]

Where:

- \( q \) and \( q_e \) are the amount of Cr\(^{6+}\) metal adsorbed per unit weight of adsorbent (mg/g) at time \( t \) and at equilibrium, respectively, and \( k_1 \) and \( k_2 \) are the adsorption rate constants. The initial adsorption rate \( (h) \) is equal to \( k_1 q_e \) and \( k_2 q_e^2 \) (mg/g min\(^{-1}\)) for first and second order models, respectively.

The applicability of the above two models can be examined by each linear plot of \( \ln(q_e - q) \) vs. \( t \), and \( t/q_e \) vs. \( t \), respectively and are represented in Figures 8 (a,c) and 8 (b,d) respectively. To quantitify the applicability of each model, the correlation coefficient, \( R^2 \), was calculated from these plots. The linearity of these plots indicates the applicability of the two models. However, the correlation, \( R^2 \), showed that the pseudo-second order model, fits better the experimental data (\( R^2 > 0.990 \)) than the pseudo-first order model (\( R^2 \) in the range of 0.888-0.964).

The kinetic parameters calculated are shown in Table 3. It has been found that the equilibrium sorption capacities determined using second-order model were in agreement with the experimentally determined equilibrium sorption capacities. Also, the rate constant decreased with the solution concentration increasing.

In general, the adsorption reaction is known to proceed through the following three steps:

1) Transfer of adsorbate from bulk solution to adsorbent surface, which is usually mentioned as diffusion

2) Migration of adsorbate into pores

3) Interaction of adsorbate with available sites on the interior surface of pores.

3.7. Adsorption Isotherms

Several models have been used in the literature to describe the experimental data of adsorption isotherm. The Langmuir and Freundlich models are the most frequently employed to describe equilibrium (Zeldowitsch, 1934). The Langmuir isotherm is based on the theoretical principle that only a single adsorption layer exists on an adsorbent and it represents the equilibrium distribution of metal ions between the solid and liquid phases. The basic assumption of the Langmuir adsorption process is the formation of a monolayer of adsorbate on the outer surface of the adsorbent and after that no further adsorption takes place. The Langmuir-type isotherm remains to be the most widely used for practical application. The Langmuir isotherm for pure component adsorption can be obtained from Equation 9:

\[
\frac{C}{q_e} = \frac{1}{q_{\text{max}} b} + \frac{1}{q_{\text{max}}} C_e \quad \text{Equation 9}
\]

Where:

- \( q_{\text{max}} \) is the Maximum Metal Ions uptake per unit mass of adsorbent (mg/g), which is related to the adsorption capacity and \( b \) is Langmuir Constant (L/mol) which is exponentially proportional to the heat of adsorption and related to the adsorption intensity.

Therefore, a plot of \( C/q_e \) versus \( C_e \) gives a straight line of the slope \( 1/q_{\text{max}} b \) and intercept \( 1/(q_{\text{max}} b) \) as in Figure 9.

Zeldowitsch (1934) assumed an exponentially decaying function of site density with respect to heat of adsorption and obtained the classical empirical isotherm, Equation 10:
\[ \ln q_e = \ln K_f + \frac{1}{n} \ln C_e \] .....................10

This is known as Freundlich isotherm. The Freundlich adsorption isotherm is an indicator of the extent of heterogeneity of the adsorbent surface, where the Freundlich constants \( K_f \) and \( n \), which respectively indicating the adsorption capacity and the adsorption intensity, were calculated from the intercept and slope of the plot of \( \ln q_e \) versus \( \ln C_e \) as shown in Figure 10.

This Freundlich type of behavior is indicative of surface heterogeneity of the adsorbents, i.e. the adsorbent sites (surface of activated carbon and bentonite) are made up of small heterogeneous adsorption patches that are homogeneous in themselves. The activation of adsorption sites takes place, leading to increased adsorption probably through the surface exchange mechanism.

<table>
<thead>
<tr>
<th>Adsorbent</th>
<th>Conc. (mg/L)</th>
<th>( q_e ) exp. (mg/g)</th>
<th>Lagergren First-Order</th>
<th>Pseudo-Second-Order</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>( K_1 ) (min(^{-1}))</td>
<td>( q_1 ) (mg/g)</td>
</tr>
<tr>
<td>Activated Carbon</td>
<td>50</td>
<td>10.5</td>
<td>2.7x10(^{-2})</td>
<td>6.95</td>
</tr>
<tr>
<td></td>
<td>100</td>
<td>17.25</td>
<td>2.6x10(^{-2})</td>
<td>15</td>
</tr>
<tr>
<td></td>
<td>200</td>
<td>29.25</td>
<td>2.3x10(^{-2})</td>
<td>27.7</td>
</tr>
<tr>
<td>Bentonite</td>
<td>50</td>
<td>10</td>
<td>2.8x10(^{-2})</td>
<td>5.6</td>
</tr>
<tr>
<td></td>
<td>100</td>
<td>18</td>
<td>2.7x10(^{-2})</td>
<td>13.2</td>
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<td></td>
<td>200</td>
<td>31.75</td>
<td>2.5x10(^{-2})</td>
<td>24.5</td>
</tr>
</tbody>
</table>

Figure 8. Pseudo-First Order Kinetic Plots for the Adsorption of Cr\(^{6+}\) Ions onto: a) Activated Carbon and c) Bentonite; Pseudo-Second Order Kinetic Plots for the Adsorption of Cr\(^{6+}\) Ions on: b) Activated Carbon and d) Bentonite. (Conditions: Concentrations: 50, 100 and 200 ppm; pH: 2.0; Adsorbents Dose: 4 g/L; Agitation Speed: 200 rpm; Temperature: 25 °C)
From the calculated results of Langmuir and Freundlich isotherm constants, given in Table 4, it can be shown that the Freundlich model yields a much better fit than the Langmuir model, when the $R^2$ values are compared in Table 4. This suggests that the boundary layer thickness is increased. The Freundlich constant $K_f$ indicates the sorption capacity of the sorbent. From Table 4, the values of $K_f$ are 1.84 and 1.58 for activated carbon and bentonite, respectively.

Furthermore, the values of $n$ are 2.26 and 1.78 for activated carbon and bentonite, respectively. It is noted that the values of $n$ are bigger than 1, reflecting the favorable adsorption. On the other hand, the $q_{max}$ and the adsorption intensity values of bentonite are higher than activated carbon. The calculated $b$ values indicate the interaction forces between bentonite surface and Cr$^{6+}$ ions are stronger than in case of activated carbon, this means that the bentonite is more powerful adsorbent than activated carbon. These results indicate that both adsorbents have a very strong adsorption capacity towards Cr$^{6+}$ ions.

### 3.8. Thermodynamic Parameters

Thermodynamic parameters such as free energy ($\Delta G$), enthalpy ($\Delta H$) and entropy ($\Delta S$) changes of adsorption can be evaluated from the following Equations 11 and 12:

\[ K_c = \frac{C_{Ae}}{C_e} \]  
\[ \Delta G^\circ = -RT \ln K_c \]

Where:

$K_c$ is the equilibrium constant and $C_{Ae}$ and $C_e$ (both in mg/L) are the equilibrium concentrations for solute on the sorbent and in the solution, respectively.

The $K_c$ values are used in Equations 11 and 12 to determine the $\Delta G^\circ$, $\Delta H^\circ$ and $\Delta S^\circ$; the $K_c$ may be expressed in terms of the $\Delta H^\circ$ (kJ mol$^{-1}$) and $\Delta S^\circ$ (kJ mol$^{-1}$K$^{-1}$) as a function of temperature, Equation 13:

\[ \ln K_c = -\frac{\Delta H^\circ}{RT} + \frac{\Delta S^\circ}{R} \]

Thermodynamic parameters such as free energy of sorption ($\Delta G^\circ$), the heat of sorption ($\Delta H^\circ$) and standard entropy ($\Delta S^\circ$) changes during the sorption process were calculated using Equations 11 and 12, temperature range of 25-40 °C , initial concentration 50 mg/L of Cr$^{6+}$ ions and dose 4 g/L for activated carbon and bentonite. ($\Delta H^\circ$) and ($\Delta S^\circ$) were obtained from the slope and intercept of a plot of $\ln K_c$ versus $1/T$ (Figure 11). The values of these parameters were recorded in Table 5.
The negative values of $\Delta G^\circ$ indicate the spontaneous nature of the process and more negative value with increase of temperature showed that an increased in temperature favours the sorption process. The positive values of $\Delta H^\circ$ indicated that the sorption process was endothermic in nature and the positive values of $\Delta S^\circ$ showed the increased randomness at solid/solution interfaces during the adsorption of metal ions onto both adsorbents and also reflected the affinity of activated carbon and bentonite toward Cr$^{+6}$ ions under consideration. It is also suggested that the positive values of entropy indicated some structural changes in the adsorbate and adsorbent (Altundogan et al, 2000; Ajmal et al, 2003 and Fuhrman et al, 2004).

4. Conclusion

Both adsorbents can be successfully used for removing of chromium (VI) ions from wastewater. The maximum adsorption percentages of Cr(VI) ions were achieved within 120 minutes. The adsorption percentages of Cr$^{+6}$ ions increased sharply by increasing adsorbent dose. As the initial concentration of ions increased, the percentage removal using activated carbon and bentonite decreased. The best temperature for the maximum adsorption is found to be 30-40 °C for activated carbon, while the adsorption using bentonite is less affected by increasing temperature. The maximum removals of Cr(VI) ions by both adsorbents were carried out at pH 2. The pseudo-second kinetic order model is suitable for describing the adsorption system. The obtained experimental data has been well described by Langmuir and Freundlich isotherm models into both activated carbon and bentonite.

Different thermodynamic parameters, viz, $\Delta H^\circ$, $\Delta S^\circ$ and $\Delta G^\circ$ have also been evaluated and it has been found that the sorption was feasible, spontaneous and endothermic in nature. The positive value of the entropy change, suggested the increased randomness.

References


