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Evaluation of Aluminum-Coated Pumice as a Potential Arsenic(V) Adsorbent from Water Resources

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ABSTRACT: In this study, the removal of As (V) from water resources by using aluminum-coated pumice as a new adsorbent was assessed. The features of the adsorbent coating layer were observed by X-ray diffraction (XRD) and Scanning Electron Microscopy (SEM).The effects of various parameters such as adsorbent doses, pH, contact time, arsenate initial concentration and interfering ions in arsenic adsorption and achieving high removal efficiency were studied. The results showed that the adsorption of As (V) was extremely influenced by the phosphate interfering ions .It was also defined that more than 98% of As (V) was removed by 10 g/L of the adsorbent with initial As (V) concentration of 250 µg/L at pH=7 and in 160 minutes. The adsorption equilibriums were analyzed by Langmuir and Freundlich isotherm models. Such equilibriums showed that the adsorption data was well fitted with Freundlich isotherm model (R^2>0.99). The data achieved from the kinetic studies were processed by kinetic models of pseudo-first-order and pseudo-second-order .The results indicated that the pseudo-second-order model could describe the adsorption of As (V) by pumice coated with alum (R^2>0.92). According to achieved results, it was defined that aluminum-coated pumice not only was an inexpensive absorbent, but also a quite effective factor in removal of As (V) from water resources.

Key word: Arsenic, Water resources, Aluminum-coated pumice, Isotherms, Kinetics

INTRODUCTION

Arsenic is one of the most toxic and natural contaminants found in water resources. Arsenic mainly originates from natural geological sources such as arsenic-containing rocks and soils, and from some anthropogenic sources such as mining, insecticides and landfill leaching (USEPA, 1999). This contaminant has become a worldwide health concern. There have been several reports of arsenic poisoning in groundwater of Bangladesh, India, Vietnam, Argentina, Mexico (Maiti *et al.*, 2007) and Iran (Mosaferi, 2005), where millions of people depend on groundwater for drinking. Chronic exposure to arsenic contaminated drinking water may result in adverse health problems such as skin lesions and bladder, kidney, liver, nasal passage and prostate cancers (Jeong *et al.*, 2007; Hudak, 2010). Due to high toxicity and carcinogenic effect of arsenic, World Health Organization (WHO), European Commission (EU) and the United States Environmental Protection Agency (USEPA) have recommended a Maximum Contaminant Level (MCL) of 10µg/L for arsenic in drinking water (Haque*et al.*, 2008). In Iran, a maximum arsenic level of 50µg/L is permitted in drinking water by the Institute of Standard and Industrial Research of Iran (ISIRI).

Arsenic occurs in natural waters in both inorganic and organic forms. The inorganic form of arsenic is more toxic to human and usually occurs in two valence states: arsenate As (V) and arsenite As (III) (Dutta *et al.*, 2005; Pena *et al.*, 2005). Redox potential and pH

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are the most important parameters in domination of As (V) and As (III). As (V) is dominant in an oxidizing condition, whereas As (III) is dominant in reduced medium (Pokhrel and Viraraghavan, 2007).

Previous studies have shown that, As (V) in aerobic surface water is removed better than As (III) (Penrose, 1974). Moreover, As (III) is easily converted to As (V) by various oxidizing agents such as oxygen, ozone, free chlorine and hydrogen peroxide (Jeong *et al.*, 2007). Therefore, they are only As (V) compounds that should be removed in drinking water treatment. Several technologies may be applied for arsenic removal such as oxidation/precipitation (Leupin and Hug, 2005), Fe-electrocoagulation /co-precipitation (Hansen *et al.*, 2006; Nouri *et al.*, 2010), alum coagulation/precipitation (Wickramasinghe *et al.*, 2004), reverse osmosis and nanofiltration (Kosutic *et al.*, 2005) and metal-oxide adsorption (Hlavay and Polyak, 2005). However, according to the recommendation of USEPA, the best available technology (BAT) to remove arsenic is a technology that provides high removal efficiency and has a complete operation scale, logical and proper service life, and cost-effectiveness (Jeong *et al.*, 2007). Thus, the BAT for arsenic removal has led researchers to study and find new and improved treatment techniques for more effective arsenic removal from water. Adsorption is a separation or a treatment process in which organic or inorganic compounds are adsorbed from the solution to porous solid media with a large surface area (Do, 1998). Adsorption has a comparatively low cost, availability and easy operation and it easily separates a small amount of toxic elements from large volumes of solutions (Dhir and Kumar, 2010). These advantages of adsorption have motivated several researchers to use this process for arsenic removal from water. Some of the common adsorbents used for arsenic removal include activated alumina, activated carbon composites, granular ferric hydroxide, manganese green sand, natural laterite, rare earth oxides, surface-modified zeolit with cationic surfactant and mud (Raichur and Panvekar, 2002; Pokhrel and Viraraghavan, 2007; Zhang *et al.*, 2007; Tashauoei *et al.*, 2010). Recently, several studies have been carried out on impregnation or coating of various oxides, sand, carbon, spent catalyst and particles of natural stone. These adsorbents have been shown to be very effective for arsenic removal from drinking water. Impregnation or coating with chemicals enhances the sorption capacity of adsorbents (Tripathy and Rrachur, 2007). For example Liu and Huang (1997) have studied the As (V) adsorption in which the adsorption capacity of spent catalyst can be increased by coating with iron. Tripathy and Rrachur (2007) have studied the adsorption of As (V) in which the adsorption capacity of activated alumina can be significantly enhance by impregnating with alum.

It is worthwhile to mention that simplicity in practice and low cost as well are often main factors in successful application of adsorbents. Due to the benefits of the adsorption process in purification as well as arsenic removal, the potential of the pumice stone particles was assessed in arsenate adsorption. Also, in order to increase the adsorption capacity, the adsorbent was coated by alum.

Pumice is a porous igneous rock. It is formed during explosive volcanic eruptions, when liquid lava is emitted into the air as a froth containing mass of gas bubbles. As the lava solidifies, the bubbles are frozen in to the rock. Pumice can form any kinds of magmas, including basalt, andesite, dacite and rhyolite (Akbal, 2005).

In this study, As (V) removal from aqueous solutions using aluminum-coated pumice (ACP) as a new adsorbent was investigated. Also affected parameters including adsorbent doses, pH, contact time, initial As (V) concentration and interfering ions concentration were studied.

MATERIALS & METHODS

Study area

This study was conducted in Qorveh region, which is located in the east of the Kurdistan province in west of Iran. In water resources of this region, level of arsenate is more than drinking water MCL. In other words, since there are many pumice mines in this region, obtaining this adsorbent is almost easy and arsenic adsorption by modifying pumice may be a suitable and effective as well as inexpensive method for arsenic removal from natural water resources. The pumice was obtained from a pumice mine in the region that is abundantly available.

Preparation of solutions

Stock solutions of As (V) were prepared by dissolving $Na_2HAsO_4.7H_2O$ in double distilled water. Aluminum solution for coating natural particles of pumice stones were prepared by dissolving $Al_2(SO_4)_3.16H_2O$ and all glassware and bottles were washed by 1 N HNO_3 and rinsed with double distilled water before usage. All chemicals purchased from Sigma Aldrich (Spain).

Preparation of aluminum-coated pumice (ACP)

Prior to the coating of alum on the surface of the pumice stone, pumice stone was crushed by a jaw crusher and was screened by a sieve (Mesh No. 50). Sieved particles were kept in 37% HCl for 24 h and were washed several times with double distilled water.

Then, the particles were immersed in double distilled water for 24 h and were dried at 105°C in the oven for 14 h. In order to coat the particles with alum, a solution of 0.5 M $Al_2(SO_4)_3.18H_2O$ was prepared. Afterwards, 50 g of pumice particles with 150 mL of 0.5 M alum solution were added into a beaker and pH was adjusted to 11 by adding 10 M NaOH solution drop by drop while stirring for 2 min. Thereafter, the beaker was placed in a static and stable state in laboratory temperature (25±1°C) for 72 h, and was dried at 110°C in the oven for 14 h as well. In order to remove traces of uncoated alum from the particles, the dried particles were washed again with double distilled water and were dried in the oven at 105°C for 14 h. Then the chemical composition of the pumice was determined by X-Ray Fluorescence (XRF) spectrometry (Model: Thermo, ARL, ADVAN´X Series). Besides, the surface area was revealed using Quntasorb surface area measurement apparatus.

Batch experiments

All experiments were conducted in batch mode and in a series of 250 mL conical flasks. Parameters were studied were pH (3-11), adsorbent doses (2.5-60g/L) and initial As (V) concentrations (50 µg/L, 250 µg/L and 2000 µg/L) (these concentrations were selected based on the maximum, mean and minimum concentrations of arsenic in natural water resources of the study area) as well as contact time (0-200 min). The conical flasks containing As (V) solution and the various doses of adsorbent were mixed by orbital shaker at 200 rpm in constant temperature (25±1°C). At the end of the adsorption process, the samples were filtered through 0.45 µm membrane filter, centrifuged at 3000 rpm and analyzed by atomic adsorption spectrophotometer (Model: 220 Varian, Australia). Afterwards, residual As (V) concentration was calculated by the equation (1):

$$q_e = C_o - C_e \times \frac{V}{M} \qquad (1)$$

Where q_e is the amount of the adsorbate (mg/g), C_0 is initial As (V) concentration (mg/L), C_e is residual As (V) concentration (mg/L), V is the volume of the solution (L) and M is adsorbent dose (g).

The pH was adjusted by pH meter (Model: Suntex sp-701, Taiwan) with diluted 0.1 M HCl and 0.1 M NaOH solution. After this stage, the optimum results were applied to natural drinking water of the water resources present in the study area. All experiments were duplicated and the means were reported.

Kinetic experiments

Batch experiments were carried out to determine the time profiles of arsenic adsorption to ACP. The samples were collected from the conical flask in 0, 1, 5, 10, 15, 20, 30, 40, 55, 70, 100, 130, 160 and 200 min., filtered, centrifuged and analyzed for arsenic concentrations.

The pseudo–first–order and pseudo–second–order models are the most popular kinetic models to study the adsorption equilibrium (Acemioglu, 2005).

Pseudo– first–order model

The pseudo–first–order model is as equation 2 (Acemioglu, 2005):

$$\frac{dq_t}{d_t} = k_1(q_e - q_t) \qquad (2)$$

Where, q_e and q_t are the values of adsorbent (mg/g) at equilibrium and time (min), respectively. K_1 is the constancy of the adsorption rate (1/min).

Integration of equation (2) at the boundary of $q_t = 0$ at t = 0 and $q_t = q_t$ at t = t leads to equation (3):

$$\log\left(1 - \frac{q_t}{q_e}\right) = -\frac{k_1}{2.302}t \qquad (3)$$

Pseudo–second–order model

The pseudo–second–order model is as Equation 4 (Azizian, 2004):

$$\frac{dq_t}{d_t} = k_2(q_e - q_t)^2 \qquad (4)$$

Where K_2 is the constancy rate (mg/g).The linear form of equation (4) at the boundary of qt = 0 at t = 0 and $q_t = q_t$ at t = t can be described as equation (5):

$$\frac{t}{q_t} = \frac{1}{k_2 q_e^{\,2}} + \frac{1}{q_e t} \qquad (5)$$

Adsorption isotherm

The sorption isotherm experiments were carried out for several adsorbent doses ranging from 1.2–40 g/L at pH= 7 and a constant initial As (V) concentration of 250 µg/L. Thereafter, equilibrium times were deducted from the kinetic experiments and fixed in 24h and the reaction mixtures were filtered, centrifuged and analyzed for arsenic concentrations. Finally, the equilibrium data were analyzed in accordance with the Freundlich and Langmuir sorption isotherm models.

Freundlich isotherm model

The non–linear equation of Freundlich isotherm model is as Equation 6:

$$q_e = K_f C_0^{1/n} \qquad (6)$$

Where, q_e is the amount of adsorbate (mg/g), C_e is the equilibrium concentration of the adsorbate (mg/L), K_f and n are Freundlich isotherm constants, which respectively show the adsorption capacity and intensity of adsorbent. The linear form of Freundlich equation is as equation (7):

$$\log(q_e) - \log(K_f) + \left[\frac{1}{n}\right]\log(C_e) \qquad (7)$$

Values of K_f and n are obtained from the slope and interception of a plot resulted from log q_e versus log C_e.

Langmuir isotherm model

The non-linear form of Langmuir isotherm model is as equation (8):

$$q_e = \frac{q_m b c_e}{1 + b c_e} \qquad (8)$$

Where, q_e is the adsorbate value (mg/g), C_e is the equilibrium concentration of adsorbate (mg/L) in aqueous solution, q_m is the maximum capacity of the adsorbent and b is the Langmuir constancy. The linear form of Langmuir equation is as Equation 9:

$$\frac{c_e}{q_e} = \frac{c_e}{q_m} + \frac{1}{b q_e} \qquad (9)$$

Values of q_m and b can be obtained from the slope and interception of a plot of C_e versus C_e/q_e.

RESULTS & DISCUSSION

Characterizations of the adsorbent

The solid structure and photomicrography of the exterior surface of the natural pumice and ACP analyzed by using SEM showed in Fig. 1a and Fig. 1b. The SEM images of acid-washed natural pumice in Fig. 1a shown ordered silica crystals at its surface and, micro pores or roughness with small cracks found on the natural pumice surface. The ACP illustrated in Fig. 1b has had a significant rougher surface than the natural pumice. The results of the solid structure of the adsorbent which has been analyzed by XRD showed in Fig. 1c. The aforementioned figure indicates that the major constituents of the adsorbent include hematite, quartz, ilit and mica. The results of chemical composition of the pumice are presented in Table 1. It is worthwhile to mention that the particles size was 0.297 mm. It suggests that there be two major elements including SiO_2 and Al_2O_3 with 51.45% and 17.08% amounts of the adsorbent, respectively.

The surface area is an effective factor in arsenic adsorption capacity by the adsorbent. The BET surface area of the ACP was measured 27 m^2/g. While the specific surface area for some of the adsorbents has been reported by several investigations including 840 m^2/g for iron oxide impregnated activated carbon (FeAC) (Vaughan and Reed, 2005) and 300 m^2/g for highly porous activated alumina as well as 15 m^2/g for aluminum-loaded Shirasu-Zeolite (Kim *et al.*, 2004). The results showed that ACP specific surface had been less than aforementioned specific surface.

Table 1. Chemical and physical composition of the pumice

Constituent	Percentage (%)
SiO_2	51.45
Al_2O_3	17.08
CaO	6.44
Fe_2O_3	6.32
MgO	6.17
Na_2O	5.67
K_2O	3.26
TiO_2	1.54
P_2O_5	0.66
SO_3	0.52
SrO	0.22
MnO	0.09
Other materials	0.56

The effect of the adsorbent doses and the initial As (V) concentration

The effect of adsorbent at different doses on removal percentage of As (V) is shown in Fig. 2. The figure reveals that As (V) uptake has been increased rapidly from 2.5 g/L to 10 g/L, and marginally thereafter. The reason why the adsorption efficiency increases by the increase of the adsorption dose is that when the adsorption dose increases, there is more specific surface for arsenate adsorption. In addition, when the adsorption dose increases more, the amount of removal doesn't increase, and it is because there is no adsorbent. Also, the amount of As (V) adsorption increases when the initial concentration decreases and the adsorbent dose stays fixed. It is because there is more specific surface for As (V) adsorption. In most of the previous studies, such as arsenic adsorption onto iron oxide and aluminum (Jeong *et al.*, 2007) and natural laterite (Matti *et al.*, 2007) as well as activated alumina impregnated with alum (Tripathy and Raichur, 2007), the arsenic removal increases by increasing the adsorption dose and decreasing arsenic initial concentration. According to the results of the previous studies, the adsorbent dose chosen for the experiments had been 10 g/L.

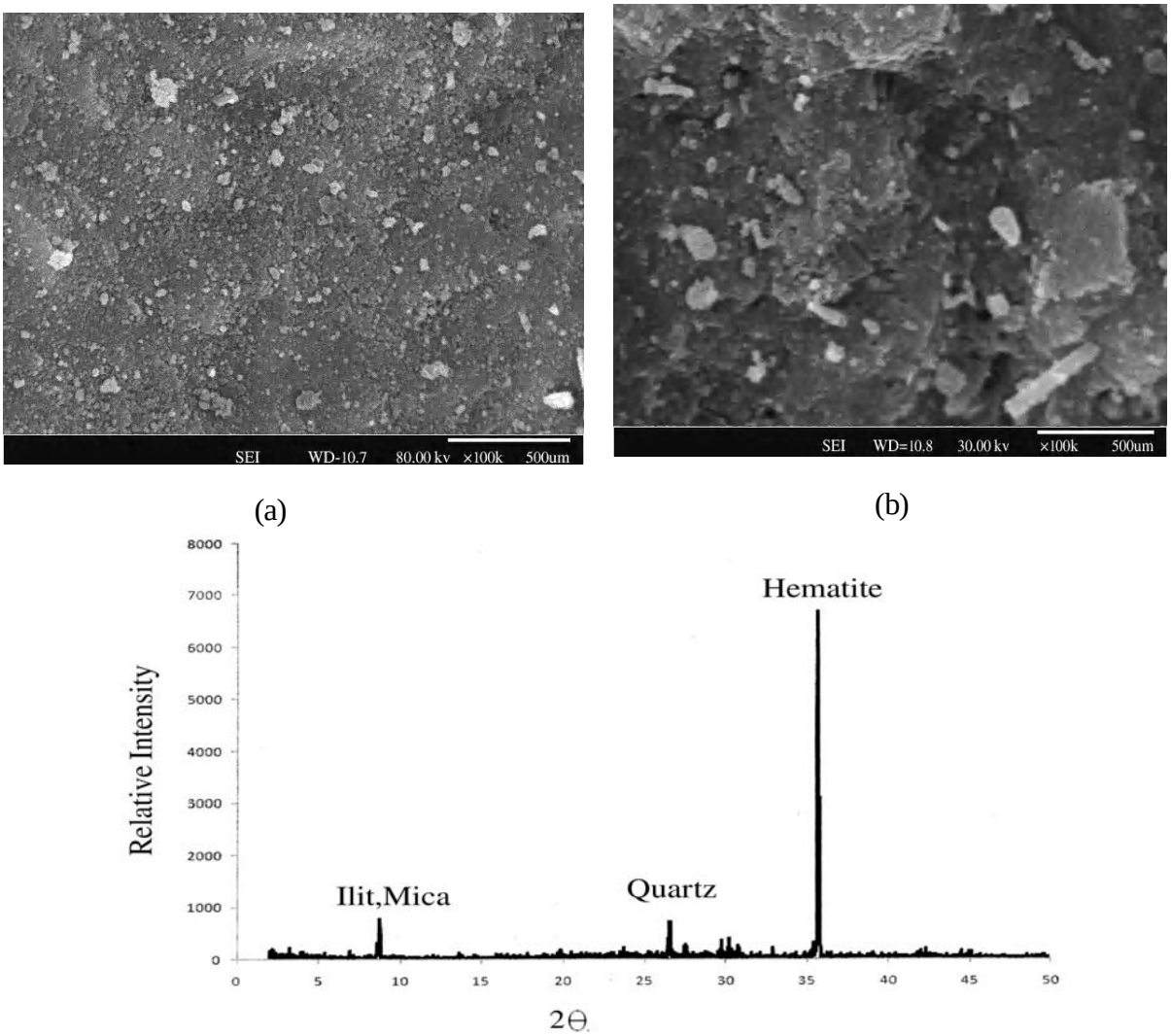

Fig. 1. SEM image of the pumice (a), SEM image of the aluminum –coated pumice (b), X-ray diffraction spectrum (XRD) of the aluminum -coated pumice (c)

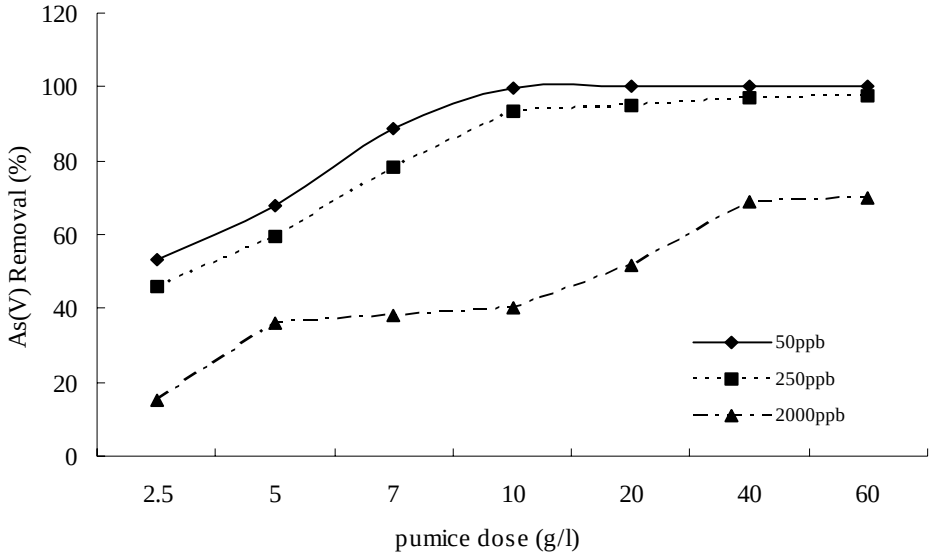

Fig. 2. The effect of the adsorbent doses and initial As (V) concentrations on arsenate removal (T: 24± 1^{oC}, pH= 7, confidence intervals = 95%)

The effect of the solution pH

In order to investigate the effect of the pH on As (V) adsorption into ACP, the experiments were conducted at an initial As (V) concentration of 250 µg/L with an adsorbent dose of 10g/L. As it is shown in Fig. 3, the adsorption of As (V) was very high at pH range of 4 to 8 and the maximum level was at PH=7 with the amount of 93.2% .Thereafter, the amount of adsorption decreased remarkably at higher pH values . Only 35% of the adsorption occurred at pH= 11. It is significant to mention that the same process has been seen in the previous studies such as arsenic adsorption to iron - modified high expanded clay aggregates (Huque *et al.*, 2008) , arsenate adsorption to iron and aluminum oxides (Pokhrel and Virarughavan, 2007), arsenic adsorption to rare earth oxides (Raichur and Panvekar, 2002) and arsenic adsorption to activated aluminum impregnated with alum (Tripathy and Rrachur, 2007) .When pH increases , the adsorption decreases because the adsorption surface is negatively charged and columbic repulsion increased (Matti *et al.*, 2007). In the pH range of 3–11, arsenate is predominantly presented in the species of $H_2AsO_4^-$ and $HAsO_4^{2-}$. Therefore, it can be concluded that those are the major species being adsorbed on the surface of ACP. The sorption process of arsenate by ACP is as equations 10-11:

$$Al(OH)_3 + H^+ + H_2AsO_4^- \longrightarrow Al(OH)_2"H_2AsO_4 + H_2O \quad (10)$$

$$Al(OH)_3 + 2H^+ + HAsO_4^{2-} \longrightarrow Al(OH)\ "H_2AsO_4 + 2H_2O \quad (11)$$

Based on the results of the investigation, pH=7±0.05 was chosen for other experiments .Such amount is somehow the same as what can be found in natural waters such as the drinking water of the study area. Fig. 3.

Kinetics of the adsorption

The rate of sorption is one of the most important parameters in evaluating the efficiency of sorption. In order to estimate the rate of adsorption and determine the behavior of the adsorptive, the adsorption kinetics of As (V) into ACP was studied at different intervals of time. Based on the results of the investigations, during the first 5 min, 40% of the adsorption took place rapidly; while 58% of the adsorption occurred during the next 155 min. Maximum adsorption (98%) was observed at 160 min within the equilibration time. The rate constants of As (V) adsorption were calculated by rate expression of pseudo–first–order and pseudo–second–order models which have been previously described. In pseudo–first–order model, the adsorption rate constancy (k_1) can be determined from the slope

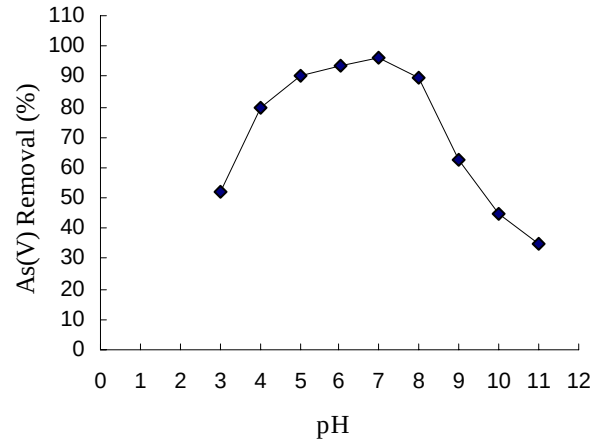

Fig. 3. The effect of pH variation on As (V) removal (T: 24± 1°C, initial As (V) concentration= 250µg/L, adsorbent dosage= 10g/L, confidence intervals =95%)

of linear plot log (1- q_t/q_e) versus t (Fig. 4a). K_1 and the correlation coefficient (R^2) were found to be 0.0009 and 0.6928, respectively, which were low indicating that the adsorption of As (V) into ACP does not follow pseudo–first– order model. In pseudo–second–order model the constancy rate (k_2) was calculated from the slope and interception of a plot of t/q_t versus t (Fig. 4b). The value of k_2 and R^2 for this model was found to be 0.0163 and 0.9977, respectively. The low k_2 and high R^2 values suggest that the adsorption be under control of pseudo–second–order model. The parameters relating to the two kinetic models are presented in Table 2.

Table 2. Parameters related to the kinetic models for the ACP adsorbent

	k_1	k_2	k_m	k_p	q_e	R^2
Pseudo–first order	0.0009	-	-	-	09.58	0.6928
Pseudo–second order	-	0.0163	-	-	00.57	0.9977

The adsorption isotherms

In order to design an appropriate sorption system to remove As (V) from drinking water, it is important to find the well–fitted isotherm curve of ACP. In this study, common isotherms (Langmuir and Freundlich) were employed at different doses of adsorbent (1.2 g/L–40 g/L) and at pH= 7. Equilibrium times were deducted from kinetic experiments and fixed at 24 hr. Fig. 5a and Fig. 5b showed the linear Freundlich and Langmuir isotherm forms, respectively. The correlation coefficients (R^2) calculated for these isotherms by using linear regression procedure for adsorption of As (V) are shown in Table 3. As it is evident from the R^2 values in Table 3, the Freundlich isotherm yielded best fits to the experimental data, probably due to the

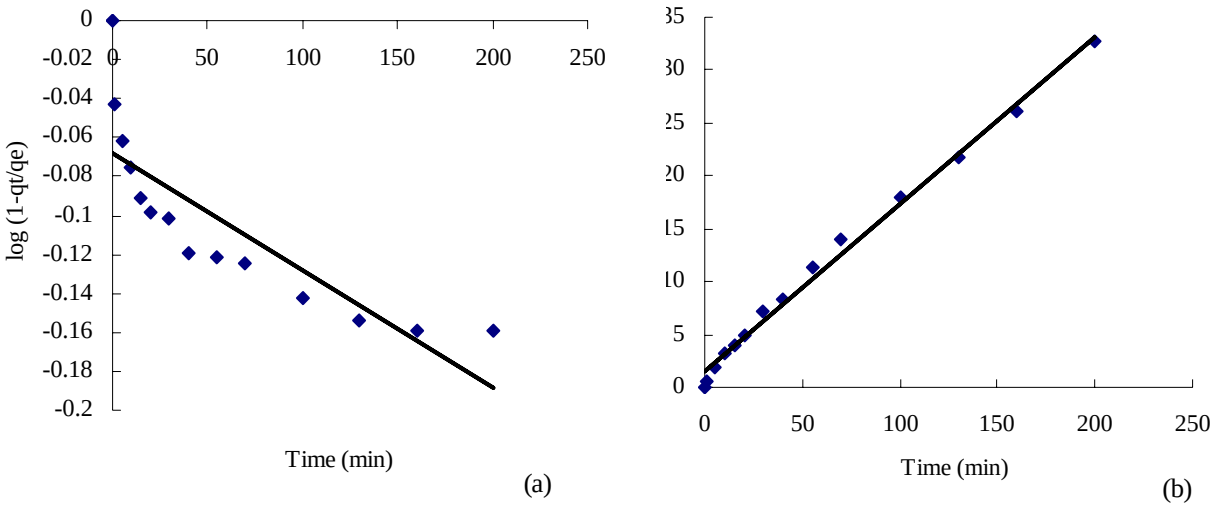

Fig. 4. Fitting of the obtained data onto pseudo-first-order model (a), pseudo-second-order model (b) (T: 24± 1^{oC}, pH= 7; initial As (V) concentration= 250 µg/L, adsorbent doses= 10g/L, confidence intervals = 95%)

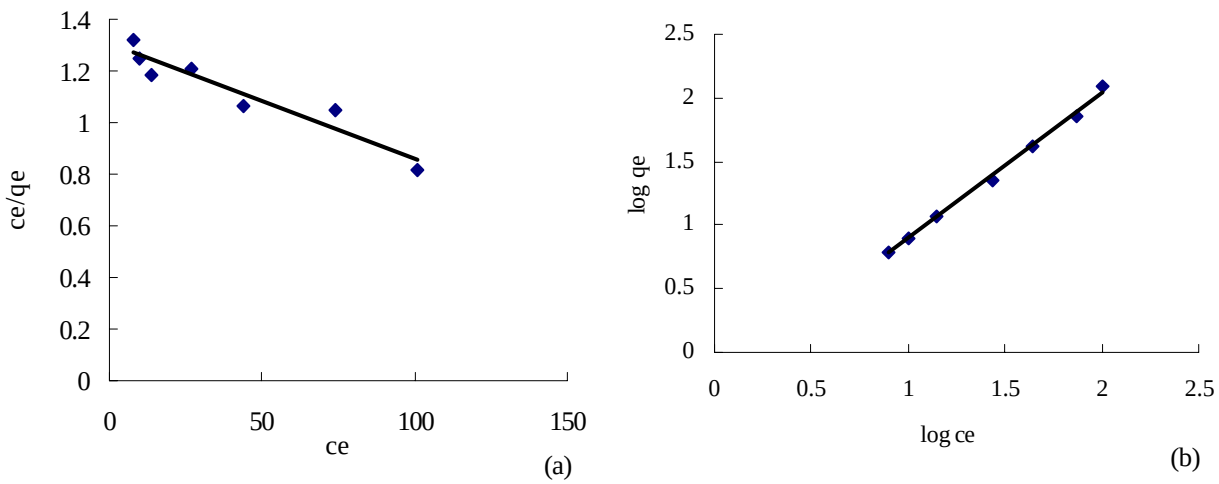

Fig. 5. Freundlich isotherm equation (a); Langmuir isotherm equation (b) plot for As (V) adsorption on ACP (T: 24±1^{oC}, pH= 7, contact time= 24 h, initial As (V) concentration=250 µg/L, adsorbent doses= 1.2-40g/L, confidence intervals = 95%)

Table 3. Freundlich and Langmuir isotherm equations parameters

	q_m (mg/g)	b (L/mg)	k_f	n	R^2
Freundlich	-	-	0.540	0.860	0.990
Langmuir	208.33	3.66×10^{-3}	-	-	0.954

heterogeneous nature of the surface sites involved in the arsenic uptake. This result also signifies that surface energy does not remain constant during the process of adsorption but various with the surface coverage (Gaur and Dhankhar, 2009). A similar trend is observed in the case of phenol and 4–chlorophenol adsorptions into pumice treated with cationic surfactant (Akbal, 2005). The k_f and n values in Freundlich isotherm model for As (V) adsorption to ACP at 20^{oC} temperature were found to be 0.54 and 0.86, respectively.

Evaluation of As (V) adsorption efficiency by ACP in natural drinking water of the study area

To find the best condition for As (V) removal from drinking water resources of the study area, the last part of this study considered the optimum condition of As (V) removal obtained from the batch experiment

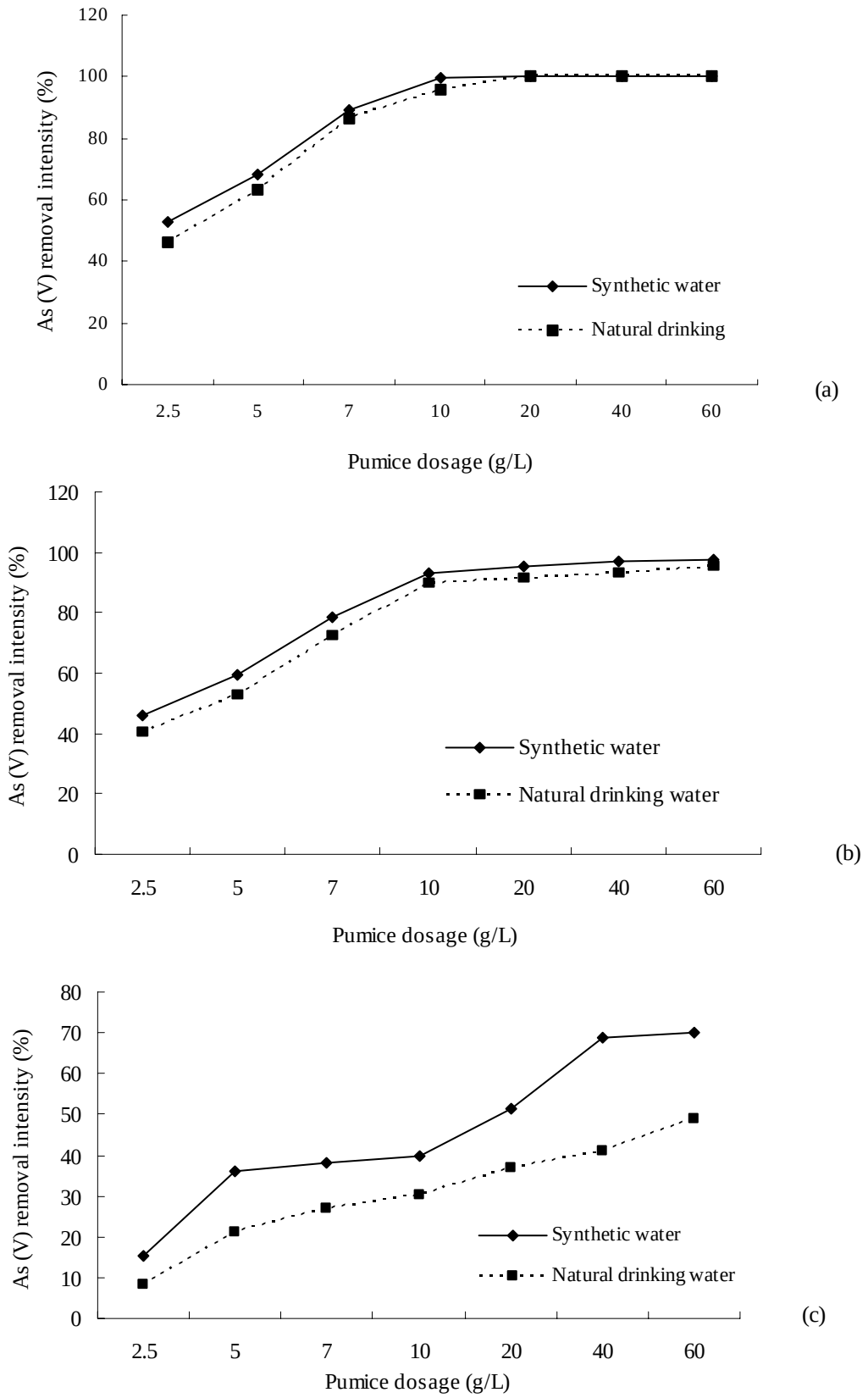

Fig. 6. Comparison of As (V) adsorption efficiency by ACP in natural drinking water of the study area with synthetic water (T: 24±1$^{\circ C}$; pH= 7; adsorbent doses= 2.5 g/L –60 g/L, initial As (V) concentration: 50 µg/L (a); initial As (V) concentration: 250 µg/L (b); initial As (V) concentration: 2000 µg/L (c); confidence intervals =95%)

applied to natural drinking water. The results summarized in Fig. 6. As it is shown in this figure, the optimum conditions for Arsenate removal from synthetic solutions with the initial concentrations of 50 and 250 µg/L are the same as what it is for drinking water of the study area with similar concentrations. The results achieved from the experiments carried out on As (V) synthetic solutions with 2000 µg/L for the drinking water having the same As (V) concentration were not optimum. In this study, it was achieved that amount of arsenate removal with the same adsorption dose, pH and contact time from the drinking water of the study area was less than the synthetic water. This could be related to the concentration of the anions such as phosphate, sulfate, nitrate and chloride in the drinking water of the study area. Such anions could have interfered with arsenate anions and competed with them for adsorption on the adsorbent sites. In two drinking water samples (the concentration of As (V) were 50 and 250 µg/L) ,the concentrations of anions weren't very high ,while in the third sample (the concentration of As (V) was 2000 µg/L) they were very high , especially the concentration of phosphate ion which was higher in the third sample than two other samples. Therefore, another experiment was planned to be carried out to identify the rules of the interfering ions in As (V) adsorption.

The effect of interfering ions

In order to assess the process of As (V) adsorption efficiency potential of ACP, the interference of some common anions including nitrates, sulfates, chlorides and phosphates were studied. The effect of those anions on As (V) removal in initial concentrations of 2000 µg/L was investigated by applying several concentrations of chloride from 0 mg/L–250 mg/L, phosphate from 0 mg/L–50 mg/L and nitrate from 0 mg/L–150 mg/L as well as sulfate from 0 mg/L –300 mg/L in the solutions. Those ranges were selected according to the anions concentrations in drinking water samples of the study area. It was found that the As (V) adsorption reduced slightly in presence of all the anions except phosphate.By increasing phosphate concentration, As (V) removal reduced highly. Thus, it is inferred that there exists a competition between arsenate and phosphate for occupying the binding sites of the adsorbent. Moreover, it have showed that in the presence of phosphate ions, As (V) removal from water by activated alumina impregnated with alum is significantly decreased (Tripathy and Raichur, 2007). As a result, the hypothesis is proved. That is to say, phosphate anions have interfered and have competed with arsenic anions for the binding sites of the adsorbent.

CONCLUSION

The aim of this study was to evaluate the potential of aluminum-coated pumice in removing As (V) from drinking water resources. Based on the results of the investigation, were found that Aluminum-coated pumice (ACP) was an effective and inexpensive adsorbent for As (V) removal from drinking water and more than 98% of As (V) at an initial concentration of 250 µg/L was adsorbed by 10 g/L of it after 160 min contact time. The maximum ACP As (V) adsorption efficiency was obtained at pH= 7.

One of the remarkable features of this adsorbent is that it performs very well in the pH range of the natural drinking water. Therefore, in order to use such adsorbents, it is not necessary to change the pH of the study area water.

The adsorption data of the adsorbent can be best fitted by Freundlich isotherm model which demonstrates the heterogeneous sites on the adsorbent surface. Based on the kinetic models, pseudo–second–order model was shown to explain the As (V) adsorption by ACP the best of all. Moreover the amount of the interfering ions was inconsiderable and a pretreatment was not mandatory except for the phosphate anions in the initial phase. Furthermore no significant water quality change occurs after As (V) removal by ACP. Therefore, the adsorbent may be applied easily at water treatment facilities even in home treatment systems.

Finally it is suggested that this As (V) removal adsorbent be almost useful and applicable in the study area thanks to simplicity in practice and low cost as well. However, further researches are needed regarding such an issue. Also, it is suggested that new policies be implemented to apply such a methodology widely in treatment industries due to the aforementioned advantages.

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