Efficiency of eight modified materials for As(V) removal from synthetic and real mine effluents

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Abstract

Arsenic (As) contamination is a major problem especially for active gold mine operations. In the present study, eight low-cost materials including biochar (B), Fe-loaded biochar (BF), activated biochar (BC), Fe-loaded activated biochar (BCF and BFC), thermally modified dolomite (MD), wood ash (WA), and modified wood ash (MWA) were comparatively used for the efficiency in As(V) removal from synthetic and real mine effluents, through batch and column testing. Batch adsorption tests were conducted in beakers with a ratio adsorbent material and As(V) synthetic and real solutions of 0.1g: 10 mL at concentrations of 850 and 300 µg/L As, respectively. Column adsorption tests were performed in 3 reactors with As(V) concentration of up to 900 µg/L in contaminated neutral drainage (CND) collected from a local gold mine. Results from batch testing with synthetic effluents showed the best performance for As(V) removal in the following order: MD > WA > BCF > BF > BFC > MWA > BC > B. Consistent findings were obtained in batch and column testing with the real mine effluent. Although iron grafted biochars are good adsorbents, their performance for As(V) removal was limited probably because of the very low As concentration in this study. In the same time, MD was found to be the most efficient material for As(V) removal but the final pH must be monitored and eventually adjusted. As(V) was completely removed by MD in batch testing (99.9%) and column testing (99.6%) after more than 112 days to bellow the authorized monthly mean allowed by Canadian discharge criteria. Thus, MD seems to be the most efficient material among the tested ones for the removal of As(V) in batch and column testing from synthetic and mine effluents.

Keywords: Low-cost sorbents, adsorption, arsenic, batch and column tests, mine effluent

1. Introduction

Arsenic (As) is a “traditional contaminant” in the environment due to its high toxicity, carcinogenicity (group 1 carcinogens) and wide occurrence (Hu et al., 2015). Typically, As is found in water as arsenate As(V) (AsO_4^{3-}) and arsenite As(III) (AsO_3^{3-}), both species being present in non-ferrous ores such as copper, lead, zinc, gold and uranium (Lorenzen et al., 1995). In regions with active mining activities, As can be found in either acid mine drainage (AMD) or contaminated neutral drainage (CND). In CND, the common inorganic arsenic species are arsenate (H_2AsO_4) and arsenite (H_3AsO_3) (Hu et al., 2015). Other anthropogenic sources of As are discharges from various industries, including fertilizers, insecticides and herbicides production, glass manufacturing, and petroleum refinery (Ansone et al., 2013). The dissolution of As-containing minerals and ores, as well as industrial discharges may ultimately entail drinking water contamination. To protect the population and the environment, Canada legislation enforces As discharge criteria of 10 and 500 µg/L (Health Canada, 2006), in drinking water and wastewater, respectively. In Québec’s province of Canada, the monthly mean concentration allowed for As from mine effluents is 200 µg/L (MDDELCC, 2012).

Several methods are available for As removal from contaminated water, including coagulation, filtration, oxidation, precipitation, reverse-osmosis and ion-exchange resin (Kowalski, 2014). Most of these methods are costly, difficult to be employed at large scale and produce large amounts of unstable sludge. Adsorption is one common method generally used due to the availability of low-cost precursors for the production of adsorbents, easiness of operation in batch and column reactors, no sludge disposal and simple material’s regeneration (Mohan and Pittman, 2007). Various adsorbents have been tested for As removal from water including dolomite, biochar, fly ash and activated carbon (Calugaru et al., 2018). Activated carbons were found, despite their high advantages (specific surface area, abundant surface functional groups and well-developed porosity) not suitable for anionic contaminants because of their negative-charged surface (Hu et al., 2015).

The method used to improve the performance of conventional adsorbents for the removal of As is their impregnation with a cation, e.g., Fe, or the use of low-cost materials, e.g., fly ash, dolomite, biochar, containing an important amount of inorganics that could have the ability of interacting with anionic contaminants. The mechanism of Fe grafting or inorganics (e.g., Fe, Ca, Na, Mg) presented on the composition of carbon-derived materials includes: physical adsorption, reduction-oxidation, ion exchange, and complexation (Calugaru et al., 2018). In the case of Fe grafting, the role of the material is to fix Fe on accessible sites for subsequent sorption of contaminants and to provide a large surface area for interacting with them (Gu et al., 2005; Muñiz et al., 2009).

Several studies have shown the potential of low-cost, both raw and modified materials for the removal of As(III) and As(V) from synthetic effluents. For example, Salameh et al. (2015) increased the adsorption capacity of raw dolomite from 0.65 to 2.16 mg/g of As(V) from synthetic effluent ($C_0 = 0.05-2$ mg/L) by charring. It was to note that dolomite charred at 800°C contains mainly $CaCO_3$ and MgO. Moreover, Sasaki et al. (2014) used thermally treated (24h at 105°C) waste cement and concrete sludge, composed mainly of $CaCO_3$, $Ca(OH)_2$, and SiO_2, which removed up to 92 and 100 % of As(V) (1.5 and 1.9 mg/g As(V) sorption capacity, respectively), when the initial synthetic solution contained 10 mg/L of As(V). Wood ash also contains significant amounts of $CaCO_3$ and CaO, which were reported effective for As removal (Calugaru et al., 2018; Genty et al., 2012; Girón et al., 2013), therefore wood ash was investigated in the present study.

Municipal biochar, rice husk and their modified form using $FeCl_3 \cdot 6\,H_2O$ impregnation removed up to 55 % ($C_0 = 90$ µg/L), 25% ($C_0 = 90$ µg/L) and > 95 % ($C_0 = 800$ µg/L) of As(V), respectively (Agrafioti et al., 2014). Sorption capacity at neutral pH was about 1 mg/g As(V) for empty fruit bunch biochar and for rice husk biochar, whereas iron loading increased it to 1.5–2.0 mg/g (Samsuri et al., 2013). Activated carbon and its impregnated product with Fe(III)-oxide adsorbed 0.09 and 4.5 mg/g of As(V) (C_0 = 1 mg/L), respectively (Reed et al., 2000). Other Fe-loaded activated carbons presented high adsorption capacity (51.3 mg/g) for the removal of As(V) from an As-rich synthetic effluent (C_0 = 20−22 mg/L) (Chen et al., 2007). Similarly, the maximum adsorption capacity of As(V) using pine biochar supported by zero-valent Fe improved from 0.2 mg/g to 124.5 mg/g (C_0 = up to 400 mg/L) (Wang et al., 2017). Ansone et al. (2013) developed new sorbents by impregnation with iron oxyhydroxides of peat, shingles, straw, sands, cane and moss. The most performant proved to be the iron-modified peat, which sorbed 98 % of As(V) when initial concentration was 100 mg/L. As for the iron-modified shingles, moss and straw, their As(V) removal percentage was of 95, 97 and 99, respectively, when initial concentration was 45 mg/L. Iron-modified cane and sand sorbed 97 and 94% of As(V) when its initial concentration was of 40 and 10 mg/L respectively. Biomass of *Staphylococcus xylosus* was loaded with Fe from $FeCl_3$ and the new material showed 4.23 mg/g sorption capacity when initial As(V) concentration was 10 mg/L. However, at neutral pH, the As(V) removal was of only 30% (Aryal et al., 2010). Zeolitic imidazolate framework-8 (ZIF-8) showed 90 mg/g As(V) removal capacity from synthetic solution initially containing 20 mg/L As(V) at pH = 8 (Liu et al., 2018). Moreover, AMD sludge alginate beads were synthetized and employed to remove As(V) from synthetic effluents. At pH 7, 90% of the As(V) was removed from the synthetic solution of initial concentration of 10 mg/L, which means that residual As(V) was still 1 mg/L (Lee et al., 2015). Thus, biochar and iron seems promising for As(V) removal in contaminated waters.

However, many of the available studies for As removal using raw and modified materials were conducted only with synthetic effluent whereas only very few studies have proposed the use of low-cost materials for the polishing treatment of As(V) from neutral mine effluents, at concentrations lower than 1 mg/L but higher than the threshold fixed by law. For instance, zerovalent iron (waste) fillings were employed for the passive field treatment of As in circumneutral (pH between 5.80 and 8.83) mine drainage in a former Sb mine in Slovakia. The system, containing 150 kg of iron fillings treated 360 L of mine drainage/h during 2.3 years, while average As concentration decreased from 452 to 50.6 μg/L. However, Fe concentration varied from 10.8 to 36.5 mg/L in the influent (mine drainage) and from 11.0 to 60.0 mg/L in the treated effluent (Sekula et al., 2018). In Québec's province of Canada, a second step should be added to this As treatment system, in order to decrease also monthly average Fe concentration below 3 mg/L.

In this context, the objective of this study is to assess the comparative efficiency of several materials (wood ash, dolomite and biochar and their modified products) for As(V) removal from synthetic effluents and real CND, in batch and column reactors.

2. Materials and methods

2.1 Synthesis of adsorbents

2.1.1 Raw and modified biochar

Three types of materials were produced: 1) Biochar; 2) Activated biochar in presence of CO_2; and 3) Iron impregnation on 1) and 2). Firstly, residues of white birch (WB) from Abitibi-Témiscamingue, Québec, were milled (< 6 mm) and converted into biochar (B) through a fast pyrolysis CarbonFX technology. The biochar was then activated (BC) in a homemade prototype pilot furnace at 900 °C in presence of CO_2. More details on furnaces and conditions applied were presented elsewhere (Braghiroli et al., 2018). The B and BC were then impregnated with Fe by mixing with $FeCl_3$ solutions containing 2.5% Fe^{3+}, at pH 12, which was adjusted with NaOH 10 N (Sigma Aldrich). The mixture was soaked for 4 h and then dried at 70°C for 12 h. The dried material was afterwards washed with distilled water to reach pH 7, and dried again at

110 °C for 24 h. Finally, three materials were obtained: 1) Biochar impregnated with Fe (BF), 2) Biochar impregnated with Fe and further activated (BFC), and 3) Biochar activated and impregnated with Fe (BCF).

2.1.2 Modified dolomite

Dolomite mineral ($CaCO_3 \cdot MgCO_3$) was provided by Temiska Silice (Saint-Bruno-de-Guigues, QC, Canada) and thermally modified (MD) to produce a mixture of $CaCO_3$ (calcite) and MgO (periclase). MD was prepared by thermally activation of dolomite in an oven at 750°C for 1 h. More details on the preparation and initial/final characterization of MD are available elsewhere (Calugaru et al., 2016).

2.1.3 Raw and modified wood ash

Wood ash (WA) was provided by Boralex (Sanneterre, QC, Canada). Modified wood ash (MWA) was prepared by heating WA in an oven at 375 °C for the purpose of mineralizing its organic content and increasing of its mineral composition. The calcinated material was mixed with solid NaOH, and heated in an oven at 600 °C for 2 h. Then, the new material was hydrothermally treated at 95 °C until the evaporation of the liquid phase. The MWA prepared was then washed and dried at 140 °C for 24 h. More details on the preparation and characterization of MWA were provided elsewhere (Calugaru et al., 2017).

2.2 Physical and chemical characterization of biochar and activated biochar

Raw and modified materials were characterized for the following physicochemical parameters: pH, pH_{PZC}, elemental composition (C, H, N, S, and O) and Fe concentration, specific surface area, and pore volume. The pH was determined according to a standard test method (ASTM D3838 - 05(2017)) using a SevenMulti, Mettler Toledo (Greifensee, Switzerland) equipped with Inlab Routine Pro electrode. The pH_{PZC} (i.e., a pH value at which the sorbent surface has zero electrical charge density) was determined using the salt / solid addition method (Belviso et al., 2014; Bakatula et al., 2018). Elemental composition was determined in a CHNS elemental analyzer, Perkin Elmer 2400 CHNS/O Analyzer (Waltham, MA, USA), by combustion of the samples in a stream of pure O_2. Fe concentration was analyzed by X-ray fluorescence (XRF) (Axios mAX, PANalytical). Pore texture parameters were obtained by N_2 adsorption at -196 °C, and CO_2 at 0 °C, using a Micromeritics ASAP 2460 automatic apparatus (Norcross, GA, USA). Ultramicroporosity was analyzed by CO_2 adsorption (V_{μ, CO_2} (cm^3/g)), whereas micro- and mesoporosity were analyzed by N_2 adsorption. The N_2 adsorption isotherms obtained were used to evaluate: i) surface area, S_{BET} (m^2/g), calculated by the Brunauer-Emmett-Teller (BET); ii) micropore volume, V_{μ, N_2} (cm^3/g), determined by the Dubinin–Radushkevich (DR) equation (Dubinin, 1989); iii) total pore volume, V_t (cm^3/g), calculated from the amount of nitrogen adsorbed at the relative pressure of 0.97 (Gregg and Sing, 1991).

2.3 Batch testing

Batch testing was conducted in plastic tubes containing 0.4 g of each adsorbent material together with 40 mL of 1 mg/L As(V) synthetic CND using $HAsNa_2O_4 \cdot 7H_2O$ (Sigma Aldrich), at pH 7. The same procedure was repeated with real CND (< 1 mg/L) sampled from a local gold mine. The adsorbent and As solution were left mixing for 1, 4, 8, 10, 18, 24, and 48 h on a multi-position stirring plate, at 500 rpm, and at room temperature (20 °C). Then, the adsorbents and supernatants were separated by filtration. The pH, redox potential (Eh), and conductivity (µS/cm) of the supernatant were measured after each contact time. The Eh was measured using LDO Hatch meter (London, ON, Canada) with a double junction Ag/AgCl reference electrode. The concentrations of As(V) were determined by ICP-AES (Inductively Coupled Plasma - Atomic Emission Spectrometry; Varian, Vista-AX CCO, Palto Alto, California, USA).

Sorption capacity at any time q_t (mg/g) and the As(V) adsorption (%) were calculated following Eqs. 1 and 2, respectively:

$$q_t = [C_0 - C_t] \cdot \frac{V}{m} \quad (1)$$

$$Adsorption\ (\%) = [C_0 - C_e] \cdot \frac{100}{C_0} \qquad (2)$$

where C_0, C_t, V, and m represent, respectively, the concentrations of As(V) in effluent (mg/L) initially and at t moment, total volume of solution (L), and amount of the material used (g).

2.4 Column testing

Column testing was conducted only for the most performants among the studied adsorbents (MD, BF, BCF and BFC) in batch testing. The plastic column height and the mass of sorbents were around 200 mm and between 52 and 75 g (for biochar-derived materials), and 80 mm and 120 g (for MD), respectively. The volume of liquid in column was different according to the hydraulic residence time (HRT) as materials properties were different (porosity and density). This requested different column lengths and masses of materials to get 2 h of HRT, according to Eq. 3:

$$HRT = \frac{n\,.V}{Q} \qquad (3)$$

where n is the porosity, V_T is the total volume of the column and Q is the effluent flow rate.

Three different layers composed the columns: a permeable membrane (to prevent losses of fine sorbent material), a layer of glass beads (15 mm diameter) and a layer of adsorbent material. The influent was fed in a vertical upward flow to the columns, using a peristaltic pump. The average flow was of 0.5 mL/min for the MD column and of 1.5 mL/min for the BF, BCF and BFC columns. The sampling of treated effluent was carried out on a weekly basis. On the collected samples, pH, Eh, the conductivity and the concentrations of As(V) were measured as previously described (section 2.3). The CND was provided by a local site mine and according to Eh-pH diagram from Takeno (2005), the pH and Eh of the CND were found in the region of As(V) species.

3. Results and discussion

3.1 Characterization of materials

Elemental composition and porous structure of the materials: biochar, dolomite and wood ash are presented in Table 1. Biochar-derived materials presented high content of organic compounds (carbon, hydrogen, and oxygen) whereas modified dolomite and wood ash-derived materials had high content of Ca in a form of calcite, Mg, Si, Fe, and Na.

After thermal treatment, i.e., fast pyrolysis followed by activation, activated biochars showed higher carbon content (89.9 %) compared to B (75.4 %). As expected, after Fe impregnation, materials had increased Fe content: 3, 7.5, and 12.6 % for BF, BCF, and BFC, respectively, according to XRF analysis. Only the main inorganics presented in WA and MWA are shown in Table 1. Other chemical elements including Ba, Cr, Cu, Sr, Ti and Zn were also presented in WA and MWA composition and varied between 0.1 and 0.9 %.

Among the studied materials, BC showed the most developed porous structure, whereas MD had the lowest surface area followed by MWA, WA and BF. After Fe impregnation, a steep decrease in surface areas was noticed, as materials were loaded with Fe blocking most of their porous structure. This is consistent with previous studies (Samsuri et al., 2013; Wang et al., 2017).

All pHs are above pH_{PZC} so sorbent surface is negatively charged and potentially attracting cations, except in the case of B in which its pH was lower than it pH_{PZC} ($5 < 6.6$). With Fe addition, pH was still higher than pH_{PZC}, but oxyanions fixation (e.g. As), at somehow, was improved (Reed et al., 2000). High chemical affinity and complex bound between Fe and oxyanionic As species are also extensively reported (Gu et al., 2005; Chen et al., 2007; Hu et al., 2015; Wang et al., 2017). This might explain the behaviour of Fe grafted materials synthesized in the present study that displayed a pH higher than 10. Other similar studies found that As sorption was improved by increasing the pH ($pH > 8$; $pH_{PZC} = 7.2$) due to As speciation in solution (Zhang et al., 2015).

3.2 Batch testing with synthetic and mine effluent

Comparative evolution of As(V) concentration from synthetic effluent in batch studies (Fig. 1 a) showed high removal with some materials (e.g., MD) but only slight variation with others (e.g., B, BC and MWA). After 48 h of testing, As(V) concentration varied from 850 to 780 µg/L for B and MWA, respectively. In the same time, BC decreased As concentration from 850 to 590 µg/L during the first hours; then, increased it back to 790 µg/L probably due to its negative charged-surface, in electrostatic repulsion with the anionic contaminant. Some expected variation of As(V) concentration was noticed but was probably due to the heterogeneity of materials, especially for BFC and BCF. The MD showed the highest adsorption capacity for the treatment of As(V) from synthetic effluent due to the presence of $CaCO_3$ and MgO from dolomite [$CaMg(CO_3)_2$] charring process.

As a general trend, biochar-derived materials enhanced the pH (from around 7 to 10) and decreased the Eh (from around 220 to 120 mV). For MD, there was higher increase of pH (from around 7 to 11) while the Eh was maintained around the initial value (90 mV). In both cases, at somehow, there is no connection between these parameters and the efficiency of As(V) treatment. Also, As(V) precipitation was not evidenced in the present study. However, it was reported in the literature the precipitation of arsenic oxide and arsenic carbonate with the use of modified dolomite for the sorption of As(V) (Salameh et al., 2015).

In the present study, only four materials adsorbed the most of As(V) and found bellow 200 µg/L, the threshold (T) according to Québec's guideline of As in mine effluents, in the following order: MD, WA, BCF and BF.

In batch testing carried out for As(V) removal from CND at approximately 300 µg/L (Fig. 1 b), all materials showed satisfactory performance in reducing the concentration to below the discharge criteria in the following order: MD > WA > BF > BCF > BFC ~ MWA > B > BC. Adsorption kinetics of As(V) on raw and modified materials with synthetic and real mine effluents are presented in Fig. 1 c) and d), respectively. The MD proved to be the best in As(V) removal by sorption from synthetic effluent (q_t = 820 µg/g), whereas BCF presented the highest adsorption capacity (Fig. 1 d)) from real mine effluent (q_t = 270 µg/g).

The pH of biochar-derived materials after adsorption testing with synthetic and CND effluents (Fig. 1 e) and Fig. 1 f), respectively) varied from 6 to 10, and from 7 to 9, respectively with respect to the contact time. However, MWA, MD and WA raised the pH of the effluent to around 12, 10−11, and 9−10, respectively, in both types of effluents to above discharge criteria (6 to 9.5).

3.3 Column testing with mine effluent

The column testing was carried out with MD and Fe-loaded biochars based on their better comparative performance and pH range allowed by Québec's requirements for CND discharge. Other contaminants (e.g., Mn, Cr, Co, Pb, Ni, Mn), in addition to As(V), were also present in the CND but at acceptable concentrations according to Québec's law.

The MD showed the best efficiency for As(V) removal with very low As concentrations (0.5─1.9 µg/L) in treated water for over 112 days of testing (Fig. 2). In the same time, BCF remained below 200 µg/L for 63 days, but after 70 days As(V) concentration exceeded the criteria at the same As(V) concentration as CND. The BF and BFC displayed saturation at about 20 days and were less effective for the treatment of As(V) in columns. These materials showed some toggling during batch testing. For this reason, the hydraulic residence time should be higher than 2 h and, consequently, the flow rate lower than 1.5 mL/min to favor the substrate – As(V) interactions and therefore the As(V) treatment.

Even though they showed the least performance for As(V) removal, Fe-loaded biochars maintained almost the same pH of the effluent, whereas MD raised the pH to between 8.75 and 10.12. After 84 days, the pH was raised to higher than 9.5 until 112 days. Thus, MD was found to be the most appropriate material for the treatment of As(V) but pH must be monitored and eventually adjusted following batch or column treatment.

3.4 Comparative performance of tested materials

Considering the As removal efficiency, from synthetic and real effluents in batch testing, and the number of days when As concentration was below 200 μg/L in the column testing, MD seems the most performant among all materials tested within this study (Table 1). BCF is another promising material, mainly for the treatment of the real mine effluent (bearing a lower As concentration). In the same time, BF and BFC showed an average performance for As removal through column tests in presence of the CND.

A direct comparison between the materials evaluated in the present study and other sorbents effective for the As treatment available in the literature is difficult as experimental conditions are different. However, at least MD and BCF showed satisfactory performance in As(V) removal from synthetic and real CND in batch and column reactors. The As removal by MD and BCF were 99.7% from average initial concentrations of 196 and 290 μ/L, respectively. The number of days when As concentration was below 200 μg/L in the column treated effluent was 112 and 63 for MD and BCF respectively. Briefly, MD and BCF were promising for As(V) removal from neutral mine effluents, at final concentrations lower than 1 mg/L As but higher than the discharge criteria.

4. Conclusion

The efficiency of several low-cost, raw and modified materials was compared for the treatment of As(V) from synthetic and real CND effluents, in batch and column testing. Among the Fe-loaded biochars, the activated biochar impregnated with Fe (BCF) was the most efficient material, whereas modified dolomite (MD) showed the best efficiency. Both materials reduced As(V) concentration to below discharge criteria according to the actual Québec's law. Therefore, they proved promising for reducing overall toxicity of real mine effluents. Moreover, both materials could be appropriate for passive treatment of mine waters. Further, the availability of the raw material (to reduce transport costs) and, the activation cost, whenever both raw materials are easily accessible, would eventually guide the employer. However, the use of MD requires pH control and final adjustment.

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Figure captions:

Fig. 1: a) and b) As(V) concentration; c) and d) adsorption capacity (q_t); and e), and f) the pH as function of time using a synthetic (C_0 ~ 850 μg/L) and real mine CND (C_0 ~ 300 μg/L) effluents, respectively, on raw and modified materials.

Fig. 2: Evolution of a) As(V) concentration and b) pH as function of time during the treatment of a real CND on raw and modified biochar, dolomite and wood ash

Table captions

Table 1. Textural and physicochemical properties of biochars and activated biochars

Table 2. Comparative performance of tested materials

Table 1. Textural and physicochemical properties of biochars and activated biochars

	B	*BC*	*BF*	*BFC*	*BCF*	*MD*	*WA*	*MWA*
Textural properties								
S_{BET} (m^2/g)	177**	881*	57**	311*	549*	4.2*	44*	23*
V_t (cm^3/g)	-	0.53	-	0.20	0.29	-	-	-
V_μ (cm^3/g)	0.11	0.33	0.06	0.12	0.22	-	-	-
V_m (cm^3/g)	-	0.20	-	0.08	0.27	-	-	-
Physicochemical properties								
pH	5.0	10.2	7.7	10.5	10.2	11.6	13.8	12.6
pH_{PZC}	6.6	9.5	6.4	8.7	9.9	11.1	11.4	12.7
C (%)	75.4	89.9	61.1	72.2	75.9	-	-	-
H (%)	3.5	0.9	3.6	1.3	1.5	-	-	-
N (%)	0.9	0.4	0.2	0.2	0.2	-	-	-
S (%)	0.5	0.0	0.4	0.0	0.2	-	-	-
O° (%)	19.7	8.8	31.7	18.8	9.6	-	-	-
Ca (%)	-	-	-	-	-	19.9	14.2	36.4
Si (%)	-	-	-	-	-	1.5	12.8	22.7
Fe (%)	-	-	3.0	7.5	12.6	0.7	5.3	16.9
Al (%)	-	-	-	-	-	0.2	2.6	4.7
K (%)	-	-	-	-	-	0.2	2.1	1.6
Mn (%)	-	-	-	-	-	0.1	1.1	2.8
Na (%)	-	-	-	-	-	-	0.7	10.5
Mg (%)	-	-	-	-	-	3.9	0.7	1.0

S_{BET} obtained from **CO_2 and *N_2 adsorption analysis; °O content was obtained by difference

Table 2. Comparative performance of tested materials

Adsorbent material	*gAC:mL*	*As removal (%) (C_0, µg/L)*		*Number of days < 200 µg/L*
		Batch (Synthetic effluent)	***Batch (Real effluent)***	***Column (Real effluent)***
B	1:100	1.9 (854)	63.2 (156)	-
BC	1:100	9.8 (886)	43.5 (156)	-
BF	1:100	75.2 (854)	97.2 (255)	28
BCF	1:100	80.1 (842)	95.4 (290)	63
BFC	1:100	32.5 (798)	89.3 (260)	21
MD	1:100	99.9 (836)	99.7 (196)	112
WA	1:100	86.9 (758)	96.3 (127)	-
MWA	1:100	8.9 (858)	68.1 (89)	-