Arsenic and antimony removal from drinking water by adsorption on granular ferric oxide

Eleni Sazakli, Stavroula V. Zouvelou, Ioannis Kalavrouziotis and Michalis Leotsinidis

ABSTRACT

Arsenic and antimony occur in drinking water due to natural weathering or anthropogenic activities. There has been growing concern about their impact on health. The aim of this study was to assess the efficiency of a granular ferric oxide adsorbent medium to remove arsenic and antimony from drinking water via rapid small-scale column tests (RSSCTs). Three different water matrices – deionized, raw water treated with a reverse osmosis domestic device and raw water – were spiked with arsenic and/or antimony to a concentration of 100 μg L⁻¹. Both elements were successfully adsorbed onto the medium. The loadings until the guideline value was exceeded in the effluent were found to be 0.35–1.63 mg g⁻¹ for arsenic and 0.12–2.11 mg g⁻¹ for antimony, depending on the water matrix. Adsorption of one element was not substantially affected by the presence of the other. Aeration did not affect significantly the adsorption capacity. Granular ferric oxide could be employed for the simultaneous removal of arsenic and antimony from drinking water, whereas full-scale systems should be assessed via laboratory tests before their implementation.

Key words | adsorption, antimony, arsenic, granular ferric oxide, rapid small-scale column test

INTRODUCTION

Arsenic and antimony are naturally occurring elements. The abundance of arsenic in the earth’s crust is about 0.5–2.5 mg kg⁻¹ (Mohan & Pittman 2007) and that of antimony is about 0.2–0.5 mg kg⁻¹ (Filella et al. 2002). Both are present in the aquatic environment as a result of rock weathering reactions, biological activity, geochemical reactions, volcanic emissions, and anthropogenic activities (Filella et al. 2002; Mohan & Pittman 2007).

Arsenic and antimony are metalloids and can exist in inorganic, organic, or mineral form. In the aquatic environment their speciation is governed by the pH and redox potential. Thermodynamic equilibrium considerations predict that pentavalent species, arsenates and antimonates, predominate and are stable in oxygen-rich aerobic environments. Trivalent forms, arsenites and antimonites, predominate in moderately reducing anaerobic environments (Mohan & Pittman 2007; Kolbe et al. 2011). In oxidizing conditions, the non-ionic form of arsenate, H₂AsO₄⁻, dominates at pH < 2.3, whereas H₂AsO₄⁻ is formed at pH 2.3–6.9 and at even higher pH, HAsO₄²⁻ is dominant. Under mildly reducing conditions, arsenite is thermodynamically stable and exists as a non-ionized arsenuous acid, H₃AsO₃, at pH below 9 (Mohan & Pittman 2007). As for antimony, its speciation in natural waters is more complicated than predicted in equilibrium thermodynamic calculations. The presence of significant amounts of Sb(III) in oxic waters and Sb(V) in oxygen-depleted systems has been reported. In the pH range of 2–11, Sb(III) should form a neutral complex, Sb(OH)₃, whereas Sb(V) should exist as a negatively charged complex, Sb(OH)₆⁻ (Guo et al. 2009).

Both arsenic and antimony are considered as priority pollutants by the United States Environmental Protection Agency (USEPA) and the European Union. The main health effects attributed to long-term exposure to arsenic involve skin, lung, bladder and kidney cancers, skin thickening and pigmentation, neurological and cardiovascular problems, and muscular weakness (Ansone et al. 2014). Antimony has no known biological function and, like arsenic, it is toxic. In general, trivalent species are considered far more toxic than pentavalent ones (Filella et al. 2002).
Both the World Health Organization and the European Union have set a provisional arsenic guideline value in drinking water to 10 μg L$^{-1}$ (WHO 2011). However, most of the developing countries in Asia have retained the earlier WHO guideline of 50 μg L$^{-1}$ as the maximum concentration limit of arsenic in drinking water (Sharma et al. 2014). The guideline value for antimony in drinking water is 5 μg L$^{-1}$ according to the European Union and 6 μg L$^{-1}$ according to USEPA (European Union 1998; USEPA 2009).

High arsenic concentrations in drinking water have been recognized in many countries, especially in the South-East Asia Region (Bangladesh, India, Taiwan, Vietnam and China), where it is estimated that more than 45 million people are at risk of being exposed to more than 50 μg L$^{-1}$ of arsenic (Sharma et al. 2014). Arsenic occurrence is also evident in the ground water in other parts of the world, such as South America, the USA, and Europe (Nordstrom 2002). With the adoption of the most stringent guideline for arsenic, even more water sources have become non-compliant globally. Elevated antimony concentrations have been reported in the surface and well water in China as varying from 100 to 7000 μg L$^{-1}$ (Guo et al. 2009). Furthermore, it is known that in many aquatic environments, arsenic and antimony co-exist either due to natural weathering (Varnavas & Cronan 1988; Migon & Mori 1999) or originating from mining and industrial emission sources (Wilson et al. 2010).

Conventional and advanced treatment methods have been developed for the removal of arsenic from water (Mondal et al. 2015). Since antimony is chemically similar to arsenic, the same technologies are also expected to be effective for antimony removal. However, compared to arsenic, few studies have been conducted regarding the removal of antimony from water (Ilavsky et al. 2014; Kolbe et al. 2011). Removal technologies include oxidation, coagulation–flocculation, adsorption, ion-exchange, and membrane processes (Guo et al. 2009; Mondal et al. 2015). It is suggested that adsorption is one of the best methods, not only due to its efficiency, but also because of its simplicity and the availability of many different sorbents (Ansone et al. 2014). Nevertheless, constraints arise in their utilization, especially in low-income countries due to unsatisfactory efficiency and the high costs of some sorbents, which drives scientific attention to sorbents based on natural materials (Sharma et al. 2014). The main adsorbents of choice have been iron oxides, modified iron oxides and hydroxides, aluminum oxide and hydroxide and also silica, carbon, and organic polymers (Giles et al. 2011). Iron-based adsorbents (goethite, hydrous ferric oxide, granular ferric hydroxide (GFH), etc.) have been extensively used as sorbents. Arsenic has a high affinity for iron-(hydr)oxide surfaces, forming inner-sphere surface complexes through a ligand exchange mechanism with surface hydroxyl functional groups (Mondal et al. 2015).

Applied full-scale systems have revealed that no single option can be universally applied due to economic and cultural factors. Important aspects for the successful implementation of such technologies include their acceptability to users, the sense of ownership and expectations of women’s role in society (Jones-Hughes et al. 2013). In several rural areas or developing countries, point-of-use treatment systems have been developed, because of the lack of centralized water supplies and reliance on household or community-based technologies (Mondal et al. 2013). One major technical drawback is the high dependence of the removal system on the composition of the water matrix. For example, removal technologies based on naturally occurring iron are more appropriate in Vietnam than the Bengal Delta (India), because the groundwater of the latter contains low concentrations of iron and high concentrations of competitive phosphate (Sharma et al. 2014).

Rapid small-scale column tests (RSSCTs) are employed to aid in the selection and design of arsenic removal media for full-scale facilities, due to the similarity of mass transfer processes and hydrodynamic characteristics (Westerhoff et al. 2005; Poddar et al. 2013). The RSSCT technique is useful because it significantly reduces the time and the quantities of medium and water required to obtain typical performance data in comparison to pilot-scale columns. Two different scaling approaches are used. The constant diffusivity (CD) approach assumes that the effective surface diffusivity is independent of particle size, and hence identical for full-scale and RSSCT columns. The proportional diffusivity (PD) approach assumes that the effective surface diffusivity is linearly proportional to the particle radius and that surface diffusion is therefore the controlling mechanism (Sperlich et al. 2005). Both proportional and CD scaling relationships have recently been applied to RSSCTs for arsenic removal by porous metal (hydr)oxide adsorbents (Metcalf & Eddy 2005). According to Westerhoff et al. (2005), the PD approach appears to be valid for arsenate adsorption onto granular ferric hydroxide under the conditions presented in their study. However, other researchers have proved that the CD approach captures the behaviour of the laboratory column better than the PD approach, especially concerning the section up to 5,000 BV. Above 5,000 BV, both approaches approximate the curve sufficiently (Sperlich et al. 2005). The same observation is derived from the
recent study of Uwamariya et al. (2014), who conclude that the main controlling process for As(III) adsorption is intraparticle diffusion, whereas the surface diffusion contributes mainly to As(V) adsorption.

The aim of this study was to investigate the removal of arsenic and antimony, apart or in conjunction, from drinking water via adsorption onto an iron-based adsorbent medium (Bayoxide E33). To our knowledge, while there are numerous studies examining the removal of arsenic and some studies regarding the removal of antimony, only one previous study has been conducted concerning the simultaneous removal of the two metalloids by sorption onto granular iron hydroxide (akaganeite) via batch adsorption experiments (Kolbe et al. 2014). Given that As and Sb co-exist in many regions due to natural weathering, their simultaneous removal is of increasing importance.

**METHODS**

**Adsorbent medium**

The adsorbent medium, called Bayoxide E33, was kindly provided by Severn Trent Services, Birmingham, UK. It is a synthetic granulated ferric oxide medium, consisting of 90.1% α-FeOOH (goethite) with a 120–200 m² g⁻¹ specific surface area. It is commercially available and, according to the supplier, it can efficiently remove both As(III) and As(V) from drinking water. Before use, Bayoxide was ground, sieved in the diameter range of 0.297–0.354 mm (45–50 mesh size) and rinsed with deionized water.

**Rapid small-scale column test experiments**

The laboratory columns were designed by scaling down a hypothetical pilot column using the CD assumption according to Equation (1) (Table 1).

\[
\frac{EBCT_{SC}}{EBCT_{LC}} = \left(\frac{d_{SC}}{d_{LC}}\right)^2 \times X
\]

where EBCT = empty bed contact time; \(d\) = particle diameter; \(X = 0\) for CD assumption (Metcalfe & Eddy 2003).

The subscripts SC and LC indicate a small-scale column (i.e., RSSCT column) and a large-scale column (i.e., pilot- or full-scale columns), respectively.

The EBCT assumed for the pilot scale was 5 minutes, considered to be representative of an industry standard.

Four glass columns of 1.5 cm diameter and 1 m height were used (Figure 1). Each RSSCT column was prepared by placing 9.8 g of the washed and sieved Bayoxide into the column. Layers (3 cm) of glass beads and glass wool were placed in the bottom of each column to retain the medium. The columns were backwashed with deionized water before the start of the experiment until the effluent water ran visually clear in order to enable full hydration and cleaning of the medium. The flow rate for a typical backwashing was 10–20 mL min⁻¹.

**Feeding solutions**

Stock As(III) and Sb(III) solutions of 500 mg L⁻¹ were prepared from their salts, NaAsO₂ and KSB₃C₄H₄O₆, respectively. Four solutions in three types of water were placed in 50 L polyethylene terephthalate containers and used in the column studies. The solutions were (i) 100 µg L⁻¹

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Pilot</th>
<th>Rapid small scale</th>
</tr>
</thead>
<tbody>
<tr>
<td>EBCT (min)</td>
<td>5</td>
<td>0.37</td>
</tr>
<tr>
<td>Particles diameter (mm)</td>
<td>1.2</td>
<td>0.325</td>
</tr>
<tr>
<td>Density (g L⁻¹)</td>
<td>490</td>
<td>490</td>
</tr>
<tr>
<td>Loading rate (m h⁻¹)</td>
<td>5</td>
<td>18.5</td>
</tr>
<tr>
<td>Flow rate (Q) (mL min⁻¹)</td>
<td>15,124</td>
<td>54.4</td>
</tr>
<tr>
<td>Column diameter (cm)</td>
<td>34</td>
<td>1.5</td>
</tr>
<tr>
<td>Water volume (m³)</td>
<td>1,089</td>
<td>0.287</td>
</tr>
<tr>
<td>Column length (mm)</td>
<td>417</td>
<td>113</td>
</tr>
<tr>
<td>BV (L)</td>
<td>38</td>
<td>0.020</td>
</tr>
<tr>
<td>Mass of adsorbent (g)</td>
<td>18,522</td>
<td>9.8</td>
</tr>
<tr>
<td>Experiment time (d)</td>
<td>50</td>
<td>3.67</td>
</tr>
</tbody>
</table>

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**Figure 1 | Experimental apparatus for rapid small-scale column tests.**
arsenic solution, (ii) 100 μg L⁻¹ antimony solution, (iii) mixed solution of 100 μg L⁻¹ arsenic and 100 μg L⁻¹ antimony, and (iv) mixed solution of 100 μg L⁻¹ arsenic and 100 μg L⁻¹ antimony in aerated condition. The types of water matrix were (a) deionized water (DW), (b) raw water treated with a reverse osmosis domestic device (TW), and (c) raw water (RW). The chemical analyses of TW and RW are presented in Table 2.

The feeding solutions were supplied to the columns at a flow rate of 54.4 mL min⁻¹ via dosing pumps. The flow direction was downward. To simulate as close as possible full-scale operational conditions, the columns operated for 16 hours followed by an 8-hour break. Each experiment was terminated when a plateau in the As or Sb concentration of the effluent was reached, i.e., the ratio of C_e/C_0 remained almost stable.

### Analysis

Effluents were collected every 3–5 hours when 10–15 L of feeding solutions had passed through the columns. All samples collected for arsenic and antimony analysis were acidified and stored at 4 °C until analysis. The concentrations of As and Sb in the effluents were determined by graphite furnace atomic absorption spectroscopy (APHA 2012 method 3113B). It was particularly important to apply the hydride generation technique (APHA 2012 method 3114B) to the first effluents, since the concentration of the two elements was very low.

### Removal efficiency

The ability of the Bayoxide medium to remove the two elements was assessed in terms of loadings (x/m), defined as the mass of the adsorbate adsorbed per unit mass of adsorbent (mg g⁻¹). The number of bed volumes (BV), up to either the corresponding guideline value of the element (As or Sb) or until the end of the experiment (as described above) was also calculated.

### RESULTS AND DISCUSSION

Within the frame of the experiments, the effectiveness of arsenic and antimony adsorption onto the Bayoxide medium was studied and the influence of water matrix and aeration was evaluated.

### Physicochemical parameters

In the experiments, with the non-aerated feeding solutions, the pH was 6.0 in deionized water, 6.6 in treated water and 7.3 in raw water. The redox potential was up to 150 mV. Under these conditions, the two elements are in the non-ionized trivalent form and adsorption takes place only through a Lewis acid–base (ligand exchange) reaction onto the non-ionized surface functional group (Banerjee et al. 2008).

After aeration, the pH increased to 7.1 in deionized water, 7.0 in treated water, and 8.1 in raw water. Given that the redox potential in the aerated solutions was, in all cases, above 450 mV, it is expected that the trivalent forms of the elements have been oxidized. In that case, the two elements are in the ionic pentavalent form and adsorption onto the hydrous iron oxide medium is known to take place via Coulombic as well as Lewis acid–base interactions (Banerjee et al. 2008).

### Adsorption onto the medium

For all experiments, breakthrough curves were constructed from the column data by plotting the ratio of C_e/C_0 for As or Sb with the volume of water processed. The Thomas model was used to study the dynamic behaviour of the columns. The model is described by Equation (2) (Reynolds & Richards 1996)

$$C_e = \frac{1}{1 + e^{k(Q/(x/m - C_0)V)}}$$  

(2)
where $C_e$: As or Sb concentration in the effluent (μg L$^{-1}$); $C_0$: As or Sb concentration in the influent (μg L$^{-1}$); $k$: Thomas rate constant (mL min$^{-1}$ μg$^{-1}$); $Q$: flow rate (mL min$^{-1}$); $q_o$: loading of As or Sb onto the medium (μg g$^{-1}$); $m$: mass of adsorbent (g); $V$: throughput volume (L).

The $R$ values ranged from 0.996 to 0.999 for all curves derived from the experimental data. Figure 2 depicts two characteristic curves created according to the analyses results from the effluent of the feeding solution containing 100 μg L$^{-1}$ As and 100 μg L$^{-1}$ Sb in deionized water.

Finally, 18 curves were constructed and after integration the adsorbed mass of each element onto the Bayoxide medium was calculated for two points, i.e., at the point where the effluent concentration reached the guideline value and at the end of each experiment (when a plateau was reached). The results of these calculations, along with the BVs per each matrix and element, are presented in Table 3.

The Bayoxide medium is able to adsorb arsenic or antimony when these elements are in separate solutions. The arsenic loadings, expressed by $x/m$, varied between 1.59 and 0.54 mg g$^{-1}$ until the guideline value was reached, while the corresponding values for antimony were between 0.49 and 0.33 mg g$^{-1}$. For both elements, the loadings depended on the water matrix, being higher in deionized water and lower in raw water (Figure 3). The drop in loadings was sharper for arsenic than for antimony.

Concerning the adsorption of both elements simultaneously, the adsorbed amount of arsenic was not substantially affected by the presence of antimony or vice versa. It seems like the adsorption sites of arsenic are different from those of antimony, or at least the two elements are not competitive. This is in agreement with Kolbe et al. (2011) who reported no competition between the sorption of the two elements when they are present in a ratio 1:1. However, these researchers noticed that in the presence of a 10-fold excess of arsenate, the loading of antimonate was decreased by about 31%. However, the sorption of arsenate was not disturbed by the presence of a 10-fold excess of antimonate, possibly due to the different size and geometry of the two ions (antimonate is hexahedral and larger than arsenate, which is tetrahedral) (Kolbe et al. 2011).

**Effect of water matrix**

In the experiments concerning the simultaneous removal of the two elements, the loadings were found in the highest level in deionized water, followed by treated water and then by raw water (Figure 3). This can be explained by the presence of competitive ions such as silicates, phosphates, sulphates, etc. (Nguyen et al. 2011). Silica has been reported to be the main competitor of arsenic adsorption by occupying the adsorptive surface; however, Bayoxide E33 has been shown to have the lowest capacity for silica compared with other adsorptive media. Phosphates compete with arsenates’ adsorption onto iron hydroxides because they follow similar adsorption kinetics (Nguyen et al. 2011).

**Effect of aeration**

Aeration in the feeding solutions with both elements did not affect substantially the $x/m$ values for arsenic. In accordance with this, Thirunavukkarasu et al. (2003) found that the adsorptive capacities of granular ferric hydroxide for As(III) and As(V) were similar, 0.88 and 0.82 mg g$^{-1}$, respectively. However, other researchers claim that oxidation of arsenite to arsenate is a standard means for increasing sorption performance (Kolbe et al. 2011).
For antimony, an increase in the x/m values is observed only in the aerated deionized water and not in the other two matrices. The different behaviour of Sb compared to As needs further investigation through the construction of isotherms, considering that pentavalent forms are generally expected to show greater adsorptive efficiency than trivalent ones (Kolbe et al. 2011).

**Comparison of adsorptive loadings**

In this study, we evaluated the loadings of As or Sb onto the adsorptive medium Bayoxide E33 until a plateau was reached, i.e., the ratio of $C_e/C_0$ remained almost stable. Through the breakthrough curves, it is possible to calculate the As or Sb loadings when the concentrations in the effluent reached the corresponding guideline values (As: $10 \mu g/L$, Sb: $5 \mu g/L$). Arsenic loadings were determined to vary between 0.35 and 0.87 mg g$^{-1}$ in treated or raw water (Table 3), until the concentration in the effluent reached $10 \mu g/L$. In deionized water, the corresponding loadings varied between 1.53 and 1.63 mg g$^{-1}$.

Using the same adsorbent medium, Amy et al. (2005) found comparable loadings at the point of $C_e$ 10 µg L$^{-1}$. For instance, at pH 7 and an initial concentration of 100 µg L$^{-1}$ As, they found 1.13 mg g$^{-1}$ for As(V) and 0.87 mg g$^{-1}$ for As(III) in the absence of interfering ions, while we found 1.63 mg g$^{-1}$ and 1.59 mg g$^{-1}$, respectively. In the presence of interferences, the loadings for As(V) ranged from 0.24 to 0.82 mg g$^{-1}$, compared to 0.56 to

**Table 3 | Arsenic and antimony removal efficiency of the Bayoxide medium**

<table>
<thead>
<tr>
<th>Arsenic</th>
<th>Antimony</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>BV1</strong></td>
<td>x/m</td>
</tr>
<tr>
<td>Feeding solution: 100 µg L$^{-1}$ As</td>
<td></td>
</tr>
<tr>
<td>DW</td>
<td>7,827</td>
</tr>
<tr>
<td>TW</td>
<td>6,021</td>
</tr>
<tr>
<td>RW</td>
<td>2,710</td>
</tr>
<tr>
<td>Feeding solution: 100 µg L$^{-1}$ As + 100 µg L$^{-1}$ Sb</td>
<td></td>
</tr>
<tr>
<td>DW</td>
<td>7,544</td>
</tr>
<tr>
<td>TW</td>
<td>3,763</td>
</tr>
<tr>
<td>RW</td>
<td>1,776</td>
</tr>
<tr>
<td>Feeding solution: 100 µg L$^{-1}$ As + 100 µg L$^{-1}$ Sb (aerated)</td>
<td></td>
</tr>
<tr>
<td>DW</td>
<td>8,068</td>
</tr>
<tr>
<td>TW</td>
<td>4,415</td>
</tr>
<tr>
<td>RW</td>
<td>1,801</td>
</tr>
</tbody>
</table>

$x/m$: As or Sb loadings on Bayoxide (mg g$^{-1}$).
$C_e/C_0$: Ratio of effluent to influent concentration.
**BV1**: Bed volumes processed until the concentration of the corresponding element in the effluent reaches the guideline value (As: $10 \mu g/L$, Sb: $5 \mu g/L$).
**BV2**: Bed volumes processed until a plateau was reached.

Figure 3 | Loadings (x/m) of arsenic and antimony until the guideline value. Feeding solutions A: 100 µg L$^{-1}$ As or Sb, B: 100 µg L$^{-1}$ As + 100 µg L$^{-1}$ Sb, C: 100 µg L$^{-1}$ As + 100 µg L$^{-1}$ Sb + aeration. Matrix DW – deionized water, TW – treated water, RW – raw water.

For antimony, an increase in the x/m values is observed only in the aerated deionized water and not in the other two matrices. The different behaviour of Sb compared to As
0.87 mg g\(^{-1}\) in the present work. A significantly lower loading was found for As(III) (0.03 mg g\(^{-1}\)) in comparison to this study (0.54 mg g\(^{-1}\)). Most of other previous studies have examined the behaviour of other iron adsorbents. Westerhoff et al. (2005) reported arsenic loadings from 0.02 to 0.84 mg g\(^{-1}\) onto the GFH medium before the RSSCT effluent arsenic concentrations exceeded 10 μg L\(^{-1}\). The experiments were conducted with groundwater containing initial arsenic concentrations of 14–51 μg L\(^{-1}\). Other researchers estimated the adsorption capacity of GFH to be 1.1 mg As(V) g\(^{-1}\) when the initial arsenic concentration was 100 μg L\(^{-1}\) (Banerjee et al. 2008). Thirunavukkarasu et al. (2003) calculated that 0.15 mg As(III) and 0.26 mg As(V) were adsorbed per gram of GFH before the arsenic effluent concentration reached 5 μg L\(^{-1}\).

In our study, the loadings for arsenic (expressed by x/m) when the effluent concentrations still met the requirements for drinking water quality, were higher than the corresponding ones for antimony in almost all cases, as depicted in Figure 3. Exceptions were observed in the non-aerated raw water and in the aerated deionized water containing both elements. Systematically lower antimony loading (0.091 mg g\(^{-1}\)) is reported by Ilavsky (2014) for the same medium and for an initial concentration of 58 μg L\(^{-1}\). In batch adsorption experiments, the loading of arsenate at the guideline value and neutral pH was calculated to be 2.05 mg g\(^{-1}\) GFH, 7 times higher than the values obtained for antimonate, but for its guideline value of 5 μg L\(^{-1}\) (Kolbe et al. 2011).

Comparison of loadings at the plateau was not possible, given that the \(C_e/C_0\) concentrations in the effluents differed when the plateau was reached and the experiments were terminated (Table 3). The maximum loading capacities for the Bayoxide medium are expected to be higher than those estimated at the plateau, because at the time the experiments were terminated, the \(C_e/C_0\) ratios were lower than 100%.

**Limitations of the study**

The major limitation of the study is the lack of experiment in pilot scale. However, similar studies have shown that RSSCT can successfully simulate the operation of pilot columns (Sperlich et al. 2005; Westerhoff et al. 2005). Another limitation could be the lack of development of adsorption isotherms that could predict the maximum adsorptive capacities of the medium for simultaneous removal of arsenic and antimony, but that was beyond the scope of this study.

**Practical issues**

Several technologies are successful in the removal of arsenic, including adsorption. From the present work, it can be derived that this process is also effective for antimony removal. The method is simple and acceptable to consumers (Thirunavukkarasu et al. 2003). Nevertheless, the effectiveness of this method is highly dependent on the quality of water that is to be treated. RSSCT’s, as well as pilot column tests, should be performed before employment of a method based on adsorption media so as to assess its viability and its successful implementation.

In all experiments, the ratio \(C_e/C_0\) was found to be below 80% at the plateau, meaning that the medium was not exhausted. Consequently, it is apparent that a full-scale treatment plant must include more than 1 unit in series in order to exploit the entire adsorption capacity of the medium, although in such a case the higher initial capital cost should be considered.

**CONCLUSION**

The present study evaluates the utilization of an adsorbent medium, Bayoxide E33, for arsenic and antimony removal from water, either separately or in conjunction.

The adsorption process for simultaneous removal of the two elements is quite effective. The presence of arsenic seems not to affect substantially the removal of antimony and vice versa. The composition of water matrix with respect to anions and cations affects the adsorption capacity of the medium for both elements, while their oxidation state has a negligible effect. The loadings, at the point where \(C_e\) reaches the corresponding guideline values, were found to be 0.35–1.63 mg g\(^{-1}\) for arsenic and 0.12–2.11 mg g\(^{-1}\) for antimony, depending on the concentrations of interfering ions.

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