
Safe disposal of solid wastes generated during arsenic removal in drinking water

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Abstract: Wastes generated during the treatment of water containing arsenic were mixed with Portland cement in 3 : 1 volume ratio, respectively, to produce mortars that were then used to manufacture monolithic bricks. Two different wastes, containing 1.0×10^3 and 2.0×10^3 mg As per kg of dried waste, were generated in experiments of aqueous trivalent arsenic ($[\text{As(III)}] = 50 \text{ mg L}^{-1}$) removal in columns filled with a mixture of zero-valent iron and sand (1%, w/w of ZVI). The mechanical tests indicated that the waste-containing bricks showed a decrease in the compression tests, while no significant differences were found in the flexural tests. Studies on arsenic leaching indicated that, in normal conditions, the amount of released arsenic is not significant, as extreme conditions are required to exceed the maximum allowable limit for non-hazardous waste. Even though the quality of the resulting mortar is lower, it is still well suited to make bricks for use in the construction of foundations or for final disposal in landfills.

Keywords: safe disposal; solid wastes; ZVI; zero-valent iron; arsenic removal; drinking water treatment; stabilisation/solidification; Portland cements mortars; arsenic leachates.

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1 Introduction

The consumption of arsenic-containing water is related with the occurrence of arsenicosis, a pathology that involves skin disorders such as dermatosis and keratosis, among others, ending sometimes in skin, lung, bladder, kidney and/or liver cancer, among others (Bardach et al., 2015). For this reason, the World Health Organization (WHO, 2008) has established a guideline concentration of $10 \mu\text{g L}^{-1}$ for total arsenic in drinking water; with this value, an estimated 226 million person are exposed around the world (Smedley and Kinniburgh, 2013). In Argentina, around 4 million inhabitants (Bardach et al., 2015), accounting almost 10% of the total population, are at risk for drinking groundwater naturally containing arsenic in amounts exceeding the $10 \mu\text{g L}^{-1}$ guideline value set by the Argentine Food Code in 2007 (CAA, 2007). This stringent

limit implies that more water treatment plants, with more efficient processes, are required, causing an increase in the generation of arsenic-containing wastes.

The abatement of arsenic in groundwater generates effluents and wastes that depend on the type of technology used, the concentration and speciation of arsenic (as As(III) is usually more difficult to remove than As(V)) and the quality of water to be treated (Fu et al., 2014; Litter et al., 2010). The effluents, such as those generated by reverse osmosis, have an increased concentration of salts and, in particular, of arsenic (Geucke et al., 2009; Litter et al., 2010), so this rejection water may require further treatment or special disposal. On the other hand, treatment processes that transfer arsenic from water to a solid phase, such as coagulation-flocculation and/or adsorption, generate sludge and solid wastes (Bordoloi et al., 2013; Litter et al., 2010; Sarkar and Paul, 2016; Sullivan et al., 2010). One practice of particular concern is the deposit of these wastes in dehydration lakes, which is a common process in drinking water treatment (Litter et al., 2010). In the long term, these dehydrated wastes may release arsenic into the soil and groundwater, thus polluting them (Sarkar and Paul, 2016); i.e., under reducing conditions arsenic is released from ferric hydroxide used in the treatment of drinking water (Clancy et al., 2013; Ghosh et al., 2006).

Arsenic removal with zero-valent iron (ZVI) also involves the retention of As in a solid phase and has attracted a lot of attention as ZVI can efficiently remove both As(III) and As(V) from groundwater (Litter et al., 2010; Su and Puls, 2001). The mechanism of arsenic removal by ZVI involves As(III) and/or As(V) adsorption and coprecipitation by the different iron (oxy)hydroxides formed after ZVI corrosion (Fu et al., 2014; Mak et al., 2009); under anoxic conditions, ZVI can even remove As by generation of insoluble As(0) (Fu et al., 2014; Yan et al., 2012). Although particulated iron (oxy)hydroxides can be used instead of ZVI, their removal capacity is usually lower (Mamindy-Pajany et al., 2011). Arsenic removal with ZVI has been largely studied for the development of household and small-scale filters, which combines the advantages of simple installation and use, easy maintenance and low cost (Litter et al., 2010; Noubactep et al., 2010). However, ZVI has to be combined with an inert material in these filters, as otherwise the expansion of the materials when ZVI is transformed into iron (oxy)hydroxides could cause the clogging of the filter; sand is one of the most used material as sand filters is a well known, low cost technology (Noubactep et al., 2010).

The solidification/stabilisation (S/S) of wastes by their combination with building materials, such as Portland cement, allows the immobilisation of pollutants, making them less soluble and reducing their toxicity. Previous studies have demonstrated that S/S technology is a good method for minimising the risk of arsenic release into the environment (Clancy et al., 2013; Sullivan et al., 2010; Mahzuz et al., 2009; Minocha and Bhatnagar, 2007). The effectiveness of S/S depends on the nature of the residues (liquid, solid or sludge), the contaminants' interaction with other components of the residues (i.e., in the case of As, with Fe and Al oxides) or with the binder (i.e., calcium), the geological conditions of the deposition/placement site (transport properties, hydraulic gradient, gas infiltration, salinity, etc.), as well as micro- and macro-biological activity and freeze-thaw resistance. All these variables will affect the behaviour of waste in time. There are studies that shows that there have been no major failures (Bates and Hills, 2015; PASSiFy, 2010) on pilot plant scale S/S treatments conducted in the USA, UK and France. However, the structural properties of these mixtures may be altered, and the components of the building materials may modify the chemical as well as adsorption/desorption equilibrium of the retained arsenic (Litter et al., 2010; Sarkar and

Paul, 2016). Therefore, leaching tests are required to verify the proper immobilisation of the pollutant in the long term (Wickramanayake et al., 2003); besides, leaching tests should be part of the economic analysis when different As removal technologies are compared, as depending on the results the wastes may require special final disposition as dangerous wastes (Chen et al., 1999).

In this work, the final disposal by S/S of wastes generated in the treatment of aqueous As(III) with columns filled with sand and ZVI is analysed. Two wastes with different As concentrations were tested, and Portland cement was used as binder for the sand/iron/arsenic solid wastes. Bricks were built with the so-obtained mortar, and mechanical tests were performed on these bricks; leaching tests were then performed on the granular material obtained by grounding the bricks. The results were evaluated in accordance with the potential use of the bricks, and with the Argentinian legislation regarding the disposal of dangerous wastes.

2 Materials and methods

2.1 Materials

High purity zero-valent iron (ZVI) (>99.8%), with a particle diameter between 0.84 and 0.074 mm (>95%), and oriental-type silica sand ($[\text{SiO}_2] > 96\%$) were used for As(III) removal experiments. As(III) (as As_2O_3) was Merck (99.9%). Ascorbic acid (99%, Sigma-Aldrich), ammonium molybdate (98%, Mallinckrodt), antimonium tartrate (99%, Sigma-Aldrich), sulphuric acid (98%, Biopack) and potassium permanganate (99%, Merck) were used for As measurements. For pH adjustment, HCl (37%, Merck) and solid KOH (85%, Mallinckrodt) were used. Regular Portland cement (CP40, 90–100% clinker-calcium sulphate, 0–10% ashes, Loma Negra) was used as binding material for the preparation of mortars (a more precise description of the chemical composition of Portland cements can be found in Paria and Yuet, 2006). The remaining reagents used were of analytical quality; low conductivity distilled water ($1 \mu\text{S cm}^{-1}$) was used in all the experiments.

2.2 Wastes generation

The sand used was previously sifted by vibration through the sieves of Abrams series, and a particle size module of 3.5 was determined. It was separated into three fractions, and iron was incorporated in the fine fraction, module 1.6 (particle diameter < 1.5 mm). The wastes were generated by gravity circulation of a 50 mg L^{-1} As(III) aqueous solution at pH 7 through 3 cm diameter and 20 cm height columns filled with a mixture of zero-valent iron and sand (1%, w/w of ZVI); seven columns were used in parallel. The system was open to air, in order to have oxic conditions, and no Fe(II) nor Fe(III) were detected in the solution leaving the columns, as reported in previous experiments (Meichtry et al., 2015).

Two different wastes were prepared:

- (1) containing 1.0×10^3 mg of As per kg of dried waste
- (2) containing 2.0×10^3 mg of As per kg of dried waste; these results are similar to those found in previous studies of As removal with ZVI (Meichtry et al., 2015).

Waste (1) was prepared by recirculating a total 4.7 L of As(III) solution through each column until no changes were observed in the As concentration, (final $[\text{As}]_{\text{Tot}} \approx 0.10 \text{ mg L}^{-1}$); for waste (2), 9.4 L of As(III) solution were recirculated (final $[\text{As}]_{\text{Tot}} \approx 0.25 \text{ mg L}^{-1}$). For wastes (1) and (2), a removal higher than 99% of the initial As was obtained. The wastes of the seven columns were mixed, and then were homogenised and thermally dried at 60°C before their solidification and stabilisation in cement mortars.

2.3 Bricks manufacturing

Cement-based stabilisation/solidification: The wastes were stabilised through solidification by mixing with cements in a 3 : 1 mass ratio; a total of 2.0 kg of cement mortars (on dried base) were obtained for each waste, enough to prepare three specimens as prismatic bricks of 4 cm × 4 cm × 16 cm. The bricks were then compacted, removed from their moulds after 72 h according to IRAM 1570 and IRAM 1622 standards (Argentinian Official Standards), and mechanically tested 28 days later. Two waste-containing materials ((1) and (2)), and two control materials ((C1) and (C2)) were obtained:

(1) 25% of Portland cement and 75% of waste (sand with 1% of As polluted ZVI).
Concentration of As total in the material: $0.75 \times 10^3 \text{ mg kg}^{-1}$.

(2) 25% of Portland cement and 75% of waste (sand with 1% of As polluted ZVI).
Concentration of As total in the material: $1.5 \times 10^3 \text{ mg kg}^{-1}$.

(C1) 25% of Portland cement and 75% of sand.

(C2) 25% of Portland cement and 75% of sand-ZVI mixture (1% of ZVI, no As).

It should be mentioned that As concentration in materials (1) and (2) were within the range of arsenic-bearing solid wastes (0.1–7500 mg kg⁻¹, Clancy et al., 2013).

2.4 Mechanical tests

Flexural strength tests were conducted with a ‘Soiltest’ load frame, and the flexural resistance reported was the pressure that splitted the bricks in two pieces: triplicate tests were done for each sample (all the prepared bricks were tested) and the results were averaged. This test measures concrete tensile strength according to the NRMCA – CIP-16 (2000) code. Compression tests were performed with an ‘Omnia’ load frame, and the compression resistance reported was the pressure that caused the collapse of the brick pieces obtained from the flexural tests; each samples was measured six times, and the results were averaged. Previous experiments (not shown) indicated that flexural tests had no results on the results of the compression tests.

2.5 Leaching tests

Various leaching experiments were carried out: with distilled water (solid–water ratio 1 : 20) and adaptations of the USEPA-TCLP, Method 1311 (USEPA-TCLP, 1992); USEPA-LSP, Method 1313: Liquid–Solid Partitioning as a Function of Extract pH Using a Parallel Batch Extraction Procedure (USEPA-LSP, 2012), and the CAL-WET: California Waste Extraction Test (CAL-WET, 1982).

The leaching tests were performed in 250 mL Erlenmeyer flasks at room temperature ($20 \pm 2^\circ\text{C}$). The cemented material was grounded in an agatha mortar, and only particles with diameter below 2 mm were used. The flasks containing the corresponding amount of material and 100 mL of the leaching solution were placed in a FERCA orbital shaker at 200 rpm. At least duplicate tests were done for each sample; when the standard deviation for total As was either higher than 10% of the average value or three times higher than $50 \mu\text{g L}^{-1}$ (As quantification limit), triplicate or quadruplicate experiments were performed, and the two closer results were used. The following leaching tests were performed:

- Water*: 5 g of the material were shaken for 18 h with 100 mL of pure water.
- Method 1311-USEPA-TCLP (USEPA-TCLP, 1992)*: 5 g of the material were shaken for 18 h with 100 mL of an aqueous solution of acetic acid at pH 2.9.
- CAL-WET Method (CAL-WET, 1982)*: 10 g of the material were shaken for 48 h with 100 mL of an aqueous solution of citric acid at pH 5.0.
- Method 1313-USEPA-LSP (USEPA-LSP, 2012)*: 10 g of the material were shaken for 24 h with 100 mL of an aqueous solution of hydrochloric acid at the concentrations indicated in Table 1.

Table 1 Leaching test on mortar specimens (1) ([As] total in the specimen $\approx 0.075\%$ w/w) and (2) ([As]total in the specimen $\approx 0.15\%$ w/w). pH final is the pH of the leachate. The Total As is the sum of the leached As(III) and As(V)

Leaching test (Material)	pH_{final}	Cond. (mS cm^{-1})	[As(V)] (mg L^{-1})	[As(III)] (mg L^{-1})	[As] _{tot} (mg L^{-1})	
Water (1)	9.84 ± 0.07	0.53 ± 0.01	≤ 0.05	≤ 0.05	≤ 0.1	
Water (2)	9.53 ± 0.13	0.53 ± 0.03	≤ 0.05	≤ 0.05	≤ 0.1	
TCLP (1)	8.87 ± 0.06	2.18 ± 0.06	0.08 ± 0.02	0.08 ± 0.02	0.16 ± 0.04	
TCLP (2)	7.85 ± 0.08	1.98 ± 0.04	0.08 ± 0.01	0.14 ± 0.06	0.22 ± 0.07	
CAL-WET (1)	7.63 ± 0.10	1.72 ± 0.06	≤ 0.05	0.12 ± 0.04	0.17 ± 0.04	
CAL-WET (2)	7.60 ± 0.04	1.74 ± 0.03	≤ 0.05	0.41 ± 0.06	0.46 ± 0.06	
LSP [HCl]	0.05 M (1)	7.20 ± 0.07	6.67 ± 0.08	0.08 ± 0.01	0.19 ± 0.03	0.27 ± 0.05
	0.05 M (2)	8.50 ± 0.08	6.67 ± 0.06	≤ 0.05	0.44 ± 0.06	0.49 ± 0.06
	0.10 M (1)	8.34 ± 0.04	10.9 ± 0.3	0.08 ± 0.03	0.10 ± 0.02	0.18 ± 0.05
	0.10 M (2)	7.43 ± 0.08	8.11 ± 0.10	0.07 ± 0.02	0.13 ± 0.04	0.20 ± 0.06
	0.20 M (1)	6.33 ± 0.07	13.2 ± 0.6	0.13 ± 0.04	0.28 ± 0.05	0.41 ± 0.10
	0.20 M (2)	6.28 ± 0.08	14.5 ± 0.4	0.15 ± 0.03	0.47 ± 0.06	0.62 ± 0.09
	0.30 M (1)	5.48 ± 0.07	28.9 ± 0.3	0.53 ± 0.05	0.34 ± 0.05	0.88 ± 0.10
	0.30 M (2)	5.44 ± 0.13	29.2 ± 0.7	0.73 ± 0.10	0.54 ± 0.06	1.27 ± 0.17
	0.50 M (1)	2.32 ± 0.07	41.9 ± 1.0	1.45 ± 0.07	11.3 ± 0.7	12.7 ± 0.8
	0.50 M (2)	1.76 ± 0.11	44.9 ± 0.7	1.55 ± 0.09	11.8 ± 1.2	13.4 ± 1.3
0.75 M (1)	0.46 ± 0.10	81.0 ± 1.0	0.18 ± 0.05	25.2 ± 1.5	25.4 ± 1.5	
0.75 M (2)	0.64 ± 0.07	120.4 ± 0.4	0.28 ± 0.03	40.1 ± 2.2	40.4 ± 2.2	

2.6 Analytical determinations

As(V) and As(III) were determined applying the spectrophotometric technique described in Lenoble et al. (2003) (limit of quantification for As: 0.050 mg L^{-1}). A double beam Shimadzu spectrophotometer, model UV-1700, equipped with 1 cm optical path quartz cells, was used for As measurements. Five millilitre samples were filtered through $0.45 \mu\text{m}$ Millipore cellulose acetate filters, 25 mm diameter, and poured into 10 mL volumetric flasks; then, the chemical reagents required for the colour development were added in the order indicated in Lenoble et al. (2003), and the flask was completed with distilled water and left in the dark for 2 h before measurement. When As concentration was higher than 5 mg L^{-1} , 1 mL sample were taken instead of 5 mL. Total arsenic content of selected samples was also measured by ICP-OES, following the procedure described in Meichtry et al. (2015). As the differences found in the total arsenic concentrations determined by both methods were within the experimental error ($\pm 10\%$), for most samples only the spectrophotometric technique was use, due to its lower analysis time and measurement costs, and because it allowed As speciation.

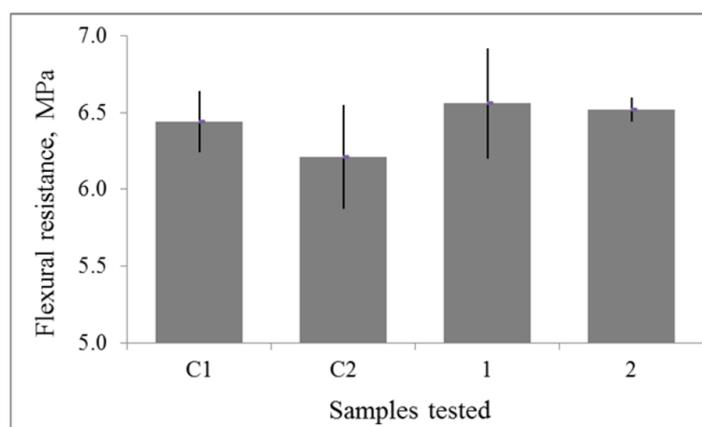
pH and conductivity determinations were performed with a Hanna meter equipped with a combined pH electrode and a conductivity sensor.

3 Results and discussion

3.1 Mechanical strength tests

The four different sample bricks obtained (waste-containing materials (1) and (2), and control materials (C1) and (C2)) were submitted to flexural tests in order to evaluate the tensional strength of the materials and the effect of As and Fe on this property; the results obtained after triplicate experiments, together with their standard error, can be observed in Figure 1.

Figure 1 Flexural test on prismatic specimens Portland cement. The error bars are the standard deviation of triplicate measurements

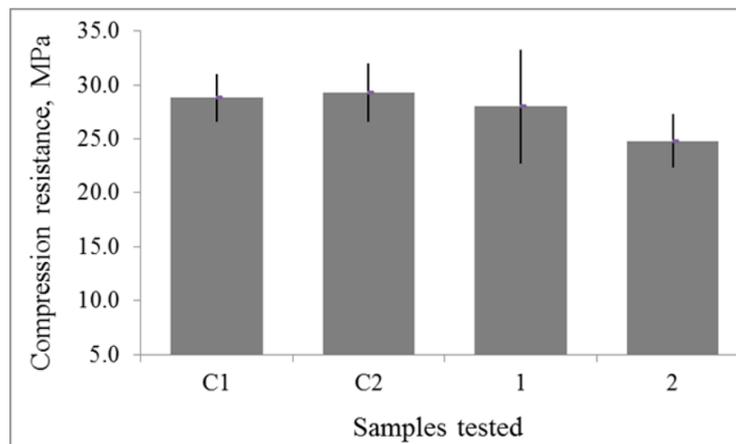


The results obtained indicate only small differences, within the measurement error, in flexural strength between the different samples; the standard errors are within the

0.7 MPa limit accepted for this test (NRMCA – CIP-16, 2000). Therefore, it can be concluded that neither Fe nor As, at the concentrations used in this work, influences the flexural resistance of the material. Fernández-Jiménez et al. (2005), using a Fe concentration of the same order of magnitude, reported a similar decrease in the flexural strength (around 15%) as that observed between (C1) and (C2) due to the presence of iron; the authors also found a decrease in the flexural strength when As alone and As combined with Fe were used, but their As concentration was much higher.

After the flexural test, six bricks of each material were obtained, each with a length of around 8 cm (half the length of the original bricks). All these bricks were submitted to compression tests in order to evaluate the compression resistance of the materials and the effect of As and Fe; the averaged results obtained after the six times experiments and their corresponding standard error can be observed in Figure 2.

Figure 2 Compression test on prismatic specimens Portland cement. The error bars are the standard deviation of six measurements



Compression tests did not show significant differences between the mortar control specimens with and without ZVI. A 15% reduction in mechanical compression strength, only slightly higher than the measurement error, was observed in the case of the specimen with the highest As content (Figure 2). While this result indicates a lower quality mortar for material (2), it can still be considered well suited for use in construction.

Various studies have indicated that compression strength decreases as the percentage of contaminated sludge (As-containing Fe or Al (oxy)hydroxides) increases in cement mortars. Some authors recommend such percentage should be 4% (Mahzuz et al., 2009), while others estimates that up to 25% can be used (Chan and Azhar, 2015; Rouf and Hossain, 2003; Zografou et al., 2015). Furthermore, the studies conducted by Minocha and Bhatnagar (2007) showed that concentrations of up to 500 mg L⁻¹ of As(V) even increased the material resistance. Then, the observed decrease in the compression strength in the samples with the arsenic-containing wastes should be related to the presence of ZVI and/or Fe oxides, as reported by Fernández Olmo et al. (2001), regarding the effect of Fe₂O₃ in aged samples.

3.2 Leaching experiments

As(V) and As(III) concentration in the different leaching tests was analysed according to the procedure described in Section 2.6. It should be indicated that, if all As were leached from material (1), As concentration would be, for samples (1), 37.5 mg L^{-1} with the water and TCLP tests and 75 mg L^{-1} with the other two tests; for material (2), the values would be twice the corresponding values of material (1). Leaching tests were also performed in control materials (C1) and (C2), but no As was detected in the leachate; then the As contribution of the sand, ZVI and the cement to the As measured was neglected. The results of the different leaching procedures tested for the materials (1) and (2) are shown in Table 1.

The results show that As is leached largely as As(III), which is consistent with the higher mobility of this species in water (Litter et al., 2010), and indicates that As(III) oxidation to As(V) is not complete during the stabilisation/solidification process; also, it can be observed that only for the LSP test at $[\text{HCl}] \geq 0.3 \text{ M}$ a significant amount ($>1\%$) of the total As contained in the material was leached. For the two types of mortar specimens analysed (with 0.075 and 0.15% of total As concentration), the results of the leaching procedure show higher total As concentrations for sample (2), consistent with its higher percentage of As.

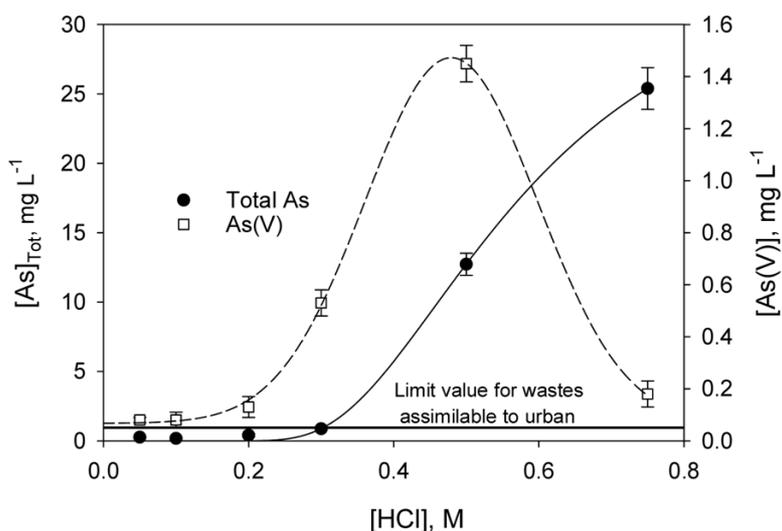
For landfill disposal, the Argentinian legislation, both Resolution 97/01, Annex III (Ministry of Social Development and Environment, 2001) and Decree 831/93, Annex V (Ministry of Social Development and Environment, 1993), require analytical determination of leachates and the use of the Toxicity Characteristic Leaching Procedure, Method 1311 (USEPA-TCLP, 1992) to predict the behaviour of arsenic waste stabilised in a landfill, with an arsenic limit value of 1 mg L^{-1} . However, it must be noted that even if these requirements are met, many researchers are raising concerns on the ability of this method to predict stability in the long term (Clancy et al., 2013; Wickramanayake et al., 2003); besides, differences up to 10 times in leached As between tests TCLP and CAL-WET have been reported (Chen et al., 1999), and thus the classification of a waste as hazardous may depend a lot on the leaching procedure used. The tests carried out with water and under the TCLP and CAL-WET methods indicate that the As leached values are within the limit allowed for non-hazardous waste, and only at $[\text{HCl}] \geq 0.3 \text{ M}$ this limit is surpassed.

For both materials, total arsenic leaching is negligible when water is used as leaching solution ($[\text{As}]_{\text{Tot}} < 0.1 \text{ mg L}^{-1}$); as this test represents the readily available fraction, that indicates the amounts of free arsenic (i.e., not bounded to any cation or metal hydroxide) and correspondingly, the most dangerous one (Yliniemi et al., 2015, and references therein), it can be concluded that there is no free arsenic neither in materials (1) nor (2). The TCLP and CAL-WET methods indicate the amount of weakly adsorbed arsenic, although it may include As(III) and/or As(V) leached when Fe hydroxides are dissolved by these organic acids (van Herck and Vandecasteele, 2001); the results of Table 1 indicate thus that As is strongly retained by the materials. The bricks were not directly tested with the leaching solutions, so As physical encapsulation in the materials (i.e., as solid As_2O_3), as previously reported (Fernández-Jiménez et al., 2005) cannot be totally discarded. However, the low $[\text{As}]_{\text{Tot}}$ measured for almost all experiments shown in Table 1 indicate that the main stabilisation mechanisms are:

- 1 strong As(III)/As(V) adsorption over Fe (oxy)hydroxides (Fu et al., 2014; Litter et al., 2010)
- 2 coprecipitation with Fe(II)/Fe(III) (Clancy et al., 2013; Fu et al., 2014; Litter et al., 2010) or with Ca(II) (present in high concentration in the materials as calcium sulphate) as calcium arsenate (Zhu et al., 2006) and/or
- 3 other mechanism of chemical retention (Yliniemi et al., 2015), i.e., sulphate replacement by As in ettringite (calcium–aluminium sulphates minerals present in cement) (Sullivan et al., 2010).

The results obtained with the LSP method (USEPA-LSP, 2012) indicate a strong dependence on pH, being required very extreme conditions ($\text{HCl} > 0.2 \text{ M}$ and $\text{pH}_{\text{final}} < 6$) to reach the maximum allowable limit of 1 mg L^{-1} for non-hazardous waste (Figures 3 and 4).

Figure 3 Leaching test on material (1) ($[\text{As}]_{\text{Tot}}$ in the specimen $\approx 0.075 \text{ \% m/m}$). The Total As is the sum of the leached As(III) and As(V). The horizontal solid line indicates the total arsenic limit value accepted for the disposal of solids as urban wastes

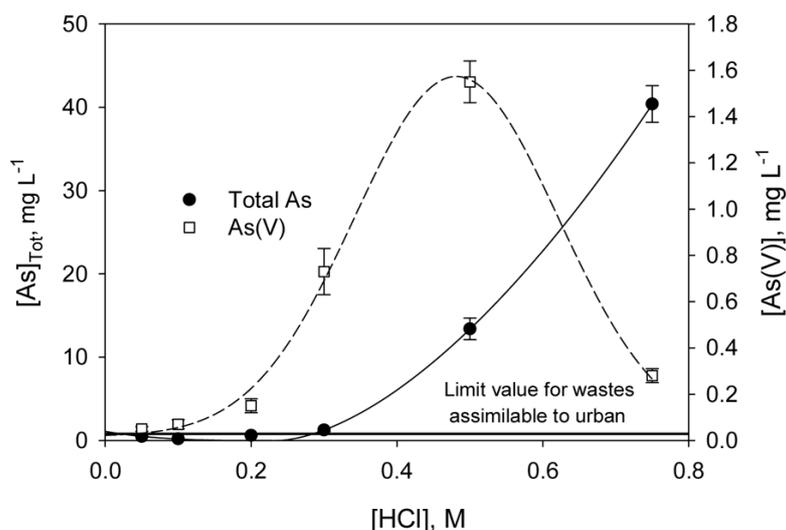


As can be clearly observed, a maximum in As(V) concentration at $[\text{HCl}] = 0.5 \text{ M}$ can be observed for both materials (1) and (2); this cannot be related with As(III) oxidation in solution by atmospheric O_2 during the tests, as under pH values direct As(III) oxidation by O_2 can be neglected (Bundschuh et al., 2012; Clancy et al., 2013; Litter et al., 2010). However, it should be noted that $[\text{As(V)}]$ is very similar to the error of $[\text{As}]_{\text{Tot}}$, and thus this As(V) maximum can be only an analytical error.

When Fe is present, the mobility of As is largely controlled by Fe speciation: under oxic conditions, Fe(III) (oxy)hydroxides are responsible for the scavenging of the stable As(V) and for the oxidation of As(III), while under anoxic conditions, As(V) is released during the reduction of Fe(III) to Fe(II), as Fe(II) has a higher solubility than Fe(III) and/or can form reduced Fe phases (magnetite, green rusts, mackinawite) that have lower

As affinity; besides, the more mobile As(III) can be formed when the redox potential of the media equals the As(V)/As(III) redox potential (Charlet et al., 2011; Ghosh et al., 2006; Hsu et al., 2016; Yunmei et al., 2004). Considering that As(V) leaching in the treated samples was almost negligible, it must be noted that the solidification/stabilisation treatment with Portland cement can be of particular interest for Argentina and other regions where As(V) is the main As species in groundwater (Bundschuh et al., 2012).

Figure 4 Leaching test on material (2) ([As] total in the specimen $\approx 0.15\%$ m/m). The Total As is the sum of the leached As(III) and As(V). The horizontal solid line indicates the total arsenic limit value accepted for the disposal of solids as urban wastes



Finally, it should be emphasised that, although ZVI can efficiently remove both As(III) and As(V) from water, there are no studies on the influence of contaminant speciation and the presence of carbonates, chloride, phosphate, sulphates and other anions when this technology is applied in sand filters (Noubactep, 2015); moreover, the incidence of these anions in the leaching process is almost unknown. One of these few studies was performed by Yliniemi et al. (2015), which determined that after alkali activation anionic species become leachable when the standard test EN 12457-3 (2003) is applied; in particular for As, the leaching increase was very significant, reaching concentrations that exceed 0.5 mg L^{-1} (Finland legal limits, FINLEX[®], 2006). Future works will be directed to study the effect of anions besides chloride and carboxylic acids, as well as the redox potential of the media, on As leaching; also, a detailed characterisation of the solid materials by techniques as X-ray diffraction, ATR-FTIR, SEM, Raman and Mossbauer spectroscopy, will be performed.

4 Conclusions

The mechanical strength of waste-containing mortars was only slightly affected by the presence of arsenic waste in concentrations up to 0.15% (w/w), maintaining a suitable quality as construction material.

Under normal leaching conditions, the limits to classify the waste stabilised and solidified with cement as hazardous waste were not reached.

Extreme conditions ($\text{HCl} > 0.2 \text{ M}$) are required to exceed the 1 mg L^{-1} limit allowed for hazardous materials according to Argentine law.

For these reasons, it can be concluded that the stabilisation/solidification methodology is a suitable option for the final disposal of arsenic waste. In addition, its characteristics enable the use of such waste as construction material.

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