Arsenic waste from water treatment systems: characteristics, treatments and its disposal

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ABSTRACT

As with other water treatment systems, arsenic treatment creates not only quality water but arsenic waste as well. Management of arsenic waste is now becoming a major public concern due to its harmful effects on the surrounding environment, including serious health problems such as skin cancers and various internal carcinomas. The main aim of this paper is to review: (i) the characteristics of arsenic waste produced by arsenic treatment systems; and (ii) the treatment and disposal methods of this waste. Arsenic waste type or its characteristics play an important role in choosing the best method of treatment and disposal. Currently, encapsulation of arsenic waste through solidification/stabilization (S/S) techniques is considered to be the most attractive solution and this method is the focus of this review. A number of studies have used cement by itself and in combination with additives such as lime, iron, silicates, or fly ash in the S/S process. Although there is a lack of systematic investigations and differing procedures for testing the effectiveness of the treatment methods, it was agreed that incorporating additives could increase the effectiveness of the S/S process depending on the type and dose of additives.

Key words: arsenic disposal, arsenic leaching, arsenic waste, solidification/stabilization, water treatment

INTRODUCTION

Arsenic chemistry

Arsenic is a chemical element in the nitrogen family. However, it acts as a heavy metalloid, existing in both yellow and grey crystalline forms. Although arsenic can exist in several oxidation states (−III, 0, +III and +V) in the environment, such as natural water, arsenic is mostly found in two inorganic forms: trivalent arsenite [As(III)] or pentavalent arsenate [As(V)]; both exist as anions AsO$_3^{3−}$ and AsO$_4^{3−}$ (Thirunavukkarasu et al. 2001). Arsenic speciation in water is strongly affected by pH and pE and the distribution of its species as a function of pH and pE is presented in Figure 1.

The occurrence of arsenite over the arsenate is found to increase when the pE of water decreases. However, an increase in pH leads to elevated ionization of arsenic acid (H$_3$AsO$_4$) and arsenious acid (H$_3$AsO$_3$) of the As(V) and As(III) forms, respectively. For arsenate, HAsO$_4^{2−}$ predominates at low pH (pH < 6.9) and at higher pH, HAsO$_4^{2−}$ becomes dominant. In extremely acidic and alkaline conditions H$_3$AsO$_4$ and AsO$_4^{3−}$, respectively become dominant. In the case of arsenite, at low pH, H$_3$AsO$_3$ dominates and at high pH, HAsO$_3^{2−}$ dominates. The conversion of one arsenic species into another is governed by the two acids’ pK$_a$ values (H$_3$AsO$_4$: pK$_{a1}$, pK$_{a2}$, and pK$_{a3}$ are 2.20, 6.97 and 11.53, respectively; H$_3$AsO$_3$: pK$_{a1}$, pK$_{a2}$, and
Organic arsenic species may be produced by biological activity. However, their concentration in natural water is not significant except in highly polluted water. Other elements present in water such as Fe, Al and S may alter the speciation of As by forming precipitation, complexation or adsorption on their compounds. These species are also influenced by pH and Eh. For example, iron arsenate and arsenite are reported to be soluble in basic solutions. Aluminium arsenate is also soluble in strong bases (Taylor & Fuessle 1994).

**Arsenic toxicity**

The toxicity of arsenic depends on its speciation and concentration. The toxicity of arsenite is 25–60 times higher than that of arsenate (Subramanian 1988). Arsenic ranks 1,014 out of the 1,598 National Priorities List of hazardous waste sites in the USA (ATSDR 2000). Ingestion of inorganic arsenic can result in both cancer (skin, lung and urinary bladder) and non-cancer diseases that can be fatal to humans (Tseng 1977; Smith et al. 1992; ATSDR 2000). In addition, inorganic arsenic in drinking water may affect many organs such as the central and peripheral nervous systems, dermal, cardiovascular, gastrointestinal and respiratory systems. The most common ailments, such as keratoses and hyperpigmentation, may occur after 5–15 years of exposure to arsenic equivalent to 700 μg/day for a 70 kg adult (WHO 2001). The World Health Organization (WHO) estimated that a lifetime exposure to arsenic in drinking water at a concentration of 200 μg/L may constitute a 5% risk of getting skin cancer (Chen et al. 2006). The National Research Council (NRC 1999) of the USA informs that total cancer risk due to the consumption of drinking water with 50 μg As/L is 1/100. Due to the high toxicity of arsenic, regulations governing its management are stringent. This standard established by different health and quality controlling authorities varies, from 7 to 50 μg As/L (WHO 2001).

In Bangladesh, 1.2 million tube wells (29%) were found to be contaminated with arsenic. It was estimated that 85 million people were at risk from arsenic (Hossain 2006). In West Bengal, India, more than 6.5 million people use As-contaminated water as a drinking water source (Chakrabarti et al. 2004). An investigation of groundwater samples from private small scale tube-wells in the Red River Delta, Vietnam showed arsenic levels varying from 1 to 3,050 μg/L (Berg et al. 2001). Arsenic concentration was above 50 μg/L in 48% of samples and more than 150 μg/L in 20% of samples examined. The average concentration of arsenic was 430 μg/L in highly affected rural areas in Vietnam. The United Nations Children’s Fund estimated that 17% of Vietnam’s population is using groundwater for drinking purposes from private tube-wells (UNICEF 2002). The high arsenic levels found in the tube-wells indicate that several million people using untreated groundwater may be at significant risk of chronic arsenic poisoning. In the USA, about half of the country’s drinking water comes from groundwater. Smith et al. (1992) reported that 350,000 people were supplied with water containing more than 50 μg As/L, the previous US limit. Generally, in Australia, arsenic contamination of drinking water is not considered a common problem; however, this kind of water pollution has occurred in isolated areas. Arsenic concentration in

![Figure 1](https://iwaponline.com/ws/article-pdf/14/6/939/415859/939.pdf)
groundwater used as a source of drinking water in a small coastal community of Stuarts Point, NSW, was found to have up to 300 \(\mu\)g As/L, much higher than Australian standards of 7 \(\mu\)g/L (Smith & Jankowski 2003). The maximum As concentrations in some Victorian surface waters was found to be up to 2,830 \(\mu\)g/L due to effect of mining practices (DMID 1991).

**Arsenic removal from water and arsenic waste**

There are a number of studies using different treatment technologies such as coagulation, ion exchange, adsorption, and membrane processes to remove arsenic from water and wastewater. However, these technologies can only remove arsenic from water and wastewater but cannot destroy it completely (WHO 2000; Nguyen et al. 2004). The arsenic waste generated from treatment systems becomes a major public concern due to the harm it causes to the surrounding environment and people’s health (Ali et al. 2003; Huq et al. 2011; Rahman et al. 2014). This paper reviews the characteristics of arsenic wastes from the water treatment system and the current methods available for treating and disposing of these wastes.

**ARSENIC WASTE TYPES AND CHARACTERISTICS**

**Arsenic waste types**

Through the water treatment processes, arsenic waste has been created in solid and liquid forms.

**Solid form.** (i) Spent media: exhausted media/materials (adsorbents, ion exchange resins) which are not able to be regenerated or are no longer effective after regeneration. (ii) Sludge: from coagulation process and from activated alumina (AA) process when brine streams are precipitated (USEPA 2000; MacPhee et al. 2001).

**Liquid form.** Backwash, regeneration and rinse steps are usually applied in arsenic treatment when adsorption, ion exchange, iron/manganese and AA processes are employed. Liquid wastes created using these steps contain soluble arsenic which is detached from solid material as well as some very fine colloidal materials.

The treatment of water by nanofilter (NF) and reverse osmosis (RO) also generates concentrates which contain high arsenic concentrations (USEPA 2000; MacPhee et al. 2001).

**Arsenic waste characteristics**

The treatment and handling cost of this waste depends on the characteristics of the waste. Frey et al. (1998) estimated that handling wastes generated by As removal processes (coagulation, lime softening, AA, anion exchange, and RO) account for 12–34% of the total water treatment costs.

Not many studies have been published on the characteristics of arsenic wastes derived from water treatment processes. The first report on arsenic waste characteristics was done by Clifford & Lin (1996) on sludge generated by treating spent ion-exchange regenerant using ferric or aluminium salts or lime in Hanford, California. They found that an arsenic concentration of 1.5 mg/L in the extract from the dried arsenic sludge. The extraction was carried out using the extraction procedure (EP) toxicity test. This test is an older regulatory test set by the US EPA to generate a liquid extract from solid wastes. In this test, the solid waste is crushed so that it can pass through a 9.5 mm sieve and arsenic is extracted by a 0.04 M acetic acid (pH = 5) leaching solution at a liquid/solid ratio of 16:1. The extraction is carried out via a 24 h agitation and the extract is analysed for a number of organic and inorganic constituents, including arsenic (LaGrega et al. 2010). Hathaway & Rubel (1987) used AA in a pilot study for removing arsenic from drinking water and they reported that arsenic contained in the dried sludge was 1,627 mg/kg. The extract determined by the EP toxicity test of this sludge was only 0.056 mg/L.

In 1990, the US EPA replaced the EP toxicity test with the toxicity characteristic leaching procedure (TCLP). Although the extraction time of TCLP is faster (only 18 h) than the EP toxicity test, it uses more acidic leaching fluid (Bricka et al. 1992). Clifford & Lin (1991) also report that the arsenic concentration in a leachate produced by TCLP of spent alumina regenerant was 0.6 mg/L, which is lower than the US EPA’s recommended current limit (5 mg/L). Low values of TCLP (0.05 mg/L to 0.066 mg/L) for spent alumina from two AA plants were also found by
Wang et al. (2000). The arsenic concentration in the influent water of the two plants ranged from 0.053 to 0.087 mg/L while the arsenic concentration in the effluent water was consistently below 0.005 mg/L. In another study, arsenic concentration in the extract of spent iron after arsenic was removed from water in two drinking water treatment plants in the United States was as low as that of spent alumina (Fields et al. 2000a). Arsenic concentrations in the influent and effluent of these two plants were 20.3–48.5 and 3.0–11.9 μg/L, respectively. Fields et al. (2000b) also reported that arsenic concentrations from TCLP were below 0.30 mg/L for samples collected from dewatered sludge lagoons at two coagulation plants in the United States. A smaller arsenic TCLP concentration (below 0.05 mg/L) was also discovered in another lime softening plant in the United States. Results from the study by MacPhee et al. (2001) with solid samples collected at eight different water treatment plants in the United States also showed that arsenic concentrations from all TCLP varied from 0.004 mg/L to 0.203 mg/L. These values were lower than the TCLP threshold limit. The authors concluded that disposal of the solid medias would not be TCLP-limited with respect to arsenic.

Regarding the ion exchange process, the amount of liquid (backwash, regeneration and rinse liquid) is 1.5–10% of the treated water (USEPA 2000). However, with highly efficient ion exchange resins and regeneration, the amount of ion exchange brine could be less than 1% of the treated water (An et al. 2005). The arsenic concentration of this liquid is influenced by the amount of arsenic removed and the quantity of water used.

Reviewing the previous data and calculations from raw water arsenic content and arsenic removal by different treatment technologies, Chwirka (1999) summarized the volumes and arsenic concentrations of wastes generated in the different treatment processes. His results are summarized in Table 1. According to his report, the arsenic concentrations in wastes varied from 0.098 mg/L for membrane technologies to approximately 10 mg/L for AA and ion exchange.

Arsenic concentration in liquid wastes is site-specific. Chen et al. (1999) surveyed arsenic concentration of arsenic wastes using the TCLP test in six different utilities in the United States and found that arsenic levels in wastes were well below the existing US EPA limit (5.0 mg/L; see Table 2). However, the arsenic levels increased up to ten times when the leaching test was carried out using the waste extraction test (WET). WET is another leaching test which is applied in California. Instead of using acetate as in TCLP, the WET uses citrate as the leaching agent. Results from WET are usually higher than that of TCLP as citrate can chelate several regulatory elements whereas acetate generally forms weaker complexes with these elements (Simmons 1996).

Another study by MacPhee et al. (2001) with eight different liquid waste samples collected at various locations across the United States indicated that arsenic concentrations in wastes varied widely from about 0.5 mg/L to 1.7 mg/L for most samples, except ion exchange and AA

<table>
<thead>
<tr>
<th>Treatment technology</th>
<th>Volume of wastes produced (L/ML)</th>
<th>As concentration in wastes volume (mg/L)</th>
<th>Quantity of solids produced (kg/ML)</th>
<th>As concentration in solids (mg/kg dry weight)</th>
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<tbody>
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<td>9.25</td>
<td>21.6</td>
<td>1,850</td>
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<td>4.2</td>
<td>239.7</td>
<td>165</td>
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<tr>
<td>Ion exchange</td>
<td>4,000</td>
<td>10</td>
<td>2.8</td>
<td>14,250</td>
</tr>
<tr>
<td>AA</td>
<td>4,200</td>
<td>9.52</td>
<td>2.8*</td>
<td>14,250*</td>
</tr>
<tr>
<td>Iron oxide coated sand</td>
<td>21,000</td>
<td>1.9</td>
<td>2.8*</td>
<td>14,250*</td>
</tr>
<tr>
<td>NF/RO</td>
<td>664,000</td>
<td>0.098</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>Coagulation/microfiltration</td>
<td>52,600</td>
<td>0.76</td>
<td>13.5</td>
<td>2,957</td>
</tr>
</tbody>
</table>

*based on calculation; NA – not applicable.
NF/RO – nanofiltration/RO.

Table 1 | Summary of arsenic waste characteristics (Chwirka 1999)
regenerants. The arsenic concentration of water subjected to plant treatment varied from 0.023 to 0.149 mg/L. Arsenic concentration of composite ion exchange regenerant and the AA regenerant samples were 10.5–24.8 mg/L and 2.6 mg/L, respectively. These samples' characteristics are presented in Table 3. The results show that ion exchange regenerant wastes had the highest alkalinity and pH (7,000 mg/L as CaCO₃ and 9.7, respectively). Membrane concentrate has the highest total hardness (around 1,560–1,750 mg/L as CaCO₃) and high total dissolved solids (TDS) and conductivity (exceeded 10,240 mg/L and 22,640 μS/cm).

More recent results by An et al. (2005), Steinwinder & Zhao (2007), and Pakzadeh & Batista (2011) demonstrate that ion-exchange brines can contain very high arsenic concentration (5–300 mg/L), alkalinity (1–10 g/L as CaCO₃), and ionic strength (0.6–2.1M).

ARSENIC WASTE TREATMENT AND DISPOSAL

Liquid form

Regenerant treatment

At this stage only a few studies have examined the removal of arsenic from liquid waste. Ferric or aluminium salts are usually used to treat concentrated spent regeneration brine and the arsenic removal depends on the ratio of Fe (or Al)/As (Clifford & Lin 1991). Clifford & Lin (1991)
reported that 90% of arsenic can be removed from wastewater containing 90 mg/L of arsenic and 50,000 mg/L TDS when the Fe/As mole ratio was 12. Removal efficiency can improve to more than 98% if the Fe/As mole ratio rises to 20. In order to reduce the regenarant needs and volume of processed wastes, Steinwinder & Zhao (2007) used FeCl₃ for recovering and reusing spent brine. They found that pH played an important role in arsenic removal efficiency; in fact approximately 98% efficiency was achieved at pH of 9 and Fe/As molar ratio of more than 10. At a lower Fe/As molar rate of 5, the removal efficiency increased to more than 99.9% when the pH was reduced to 6.0–7.0. After precipitation, brine can be reused for another regeneration run after adjusting its pH to 10. Pakzadeh & Batista (2011) also noted that a low pH of 4.5–6.5 was optimal for removing As(V) from ion exchange brine. Furthermore high ionic strength led to slightly improved arsenic removal efficiency. In their study, the Fe/As molar ratio needed to treat ion exchange brines was very low, ranging from 1.3 to 1.7.

The addition of calcium salts to produce a concentration of 90 mM and 210 mM calcium to the brine treatment process can reduce leachable arsenic by 80% and 60%, respectively (Steinwinder 2006). This addition can also decrease chemical costs by 18% and the mass of sludge produced by 20%. An et al. (2005) developed a new class of starch-bridged magnetite nanoparticles (0.57 g Fe/L) for removing As(V) from spent ion exchange brine. This material could remove nearly 100% of As(V) from a simulated spent ion exchange brine containing 300 mg/L As(V) and 6% (w/w) NaCl within 1 h. As(V) adsorption capacity was not affected very much by NaCl concentration in the brine.

The treatment of ion exchange brine by ferric chloride addition can generate sludge. However, Clifford et al. (2003) showed that this As-laden sludge can easily pass the current TCLP test (limit of 5 mg/L). In contrast An et al. (2005) in their recent findings showed that if the Fe/As molar ratio was less than 10, the arsenic concentration from TCLP of spent material could be higher than 5 mg/L. Failing the TCLP or WET test means that the wastes will be considered hazardous and the disposal costs will increase significantly, up to three times higher than the normal waste (Meng et al. 2001). MacPhee et al. (2001) conducted a number of experiments to determine the effectiveness of different treatment options for removing arsenic from wastes produced by: AA adsorption, Fe-Mn removal system, adsorption clarifier – Fe-Mn removal system, RO, NF and ion exchange. They found that ferric chloride performed better than alum in removing arsenic from wastes with a removal efficiency ranging from 88 to 99%.

Membrane concentrate treatment

The As removal efficiency of ion exchange is slightly less than that of RO membrane (about 90% for ion exchange and 95–99% for RO membrane) but the volume of ion exchange brine is much lower than that of membrane concentrate (0.3 and 15–25% of feed water for ion exchange and RO respectively) (Frey et al. 1998). Therefore the As concentration in nanofiltration (NF) or RO concentrates of plant treated arsenic contaminated water was much lower than those of ion exchange brines (0.1–0.5 mg/L compared to 10–30 mg/L for ion exchange brine). However, it is usually still higher than the United States’ discharge limit of 0.01 mg/L. The ionic strengths in membrane concentrate are similar to those of regenerants (Pakzadeh & Batista 2011; Xu et al. 2013). Mercer & Tobiason (2008) found that more than 90% of arsenic from membrane concentrate was removed during in situ iron precipitation at Fe/As molar ratios of more than 10. They also found that the high ionic strengths in membrane concentrate reduce the effect of pH on arsenic removal by the in situ formed hydrous ferric oxide (HFO). Arsenic removal by HFO increased considerably when pH fell from 8 to less than 6.5 at a low ionic strength of 0.2 M. However, there was not much improvement in arsenic removal when the ionic strength was increased to 0.7 M. MacPhee et al. (2001) showed that ferric chloride could remove 80–99% arsenic from desalting membranes. A recent analysis on the removal of arsenic from RO concentrate (TDS > 10,000 mg/L) has been done by Xu et al. (2015). In this research the arsenic level (as arsenate) in RO concentrate collected from a groundwater desalination plant was 0.029 mg/L, which exceeded the discharge permit. The arsenic in this RO concentrate can be successfully removed by coagulation with ferric salts (FeCl₃ or Fe₂(SO₄)₃) or even ferric sludge (taken from surface water treatment). This approach of ‘waste treat waste’ provides a promising low-cost solution. However, the
sludge from the treatment must go through TCLP testing to decide the appropriate disposal method.

Disposal

Depending on the concentration of arsenic in the liquid wastes and prevailing regulations, these liquid wastes can be discharged – with or without treatment – into the sanitary sewer system, deep well injection or sometimes into evaporation ponds/lagoons (USEPA 2000). The treatment of liquid waste will create another solid waste that needs to be properly handled.

Solid form

Landfill

Generally, there is no specific guideline for the safe disposal of arsenic waste into the environment (USEPA 2000). In the USA, depending on the results of the TCLP test, the material can be classified as hazardous waste if the arsenic in the leachate is higher than 5 mg/L and this waste needs to be disposed into hazardous waste landfill. WET with same limit threshold of 5.0 mg/L has been applied in California. The media which have low arsenic in the leachate can be disposed of into sanitary landfill or alternatively may be used for land application (USEPA 2000).

The most serious arsenic pollution in water has been found in Bangladesh and West Bengal, India. There are 18 large scale arsenic and iron treatment plants in Bangladesh and each plant produces about 170 m³ arsenic-laden sludge per year (Basak & Islam 2008). Arsenic sludge from the water treatment system in Bangladesh is usually disposed of in landfills (Hassan et al. 2014). However, in some areas, especially where a decentralized treatment system is employed, arsenic wastes are directly discharged into the nearby water body or low-laying areas (Sullivan et al. 2010; Hassan et al. 2014). This improper disposal method can lead to pollution of the surrounding environment, especially surface and groundwater sources. Huq et al. (2011) showed that the disposal of As-filter sludge (iron-based and the AA-based, 0.5–1.0 T/ha) on soil led to arsenic accumulation in plants, and they concluded that exhausted filters cannot be safely stored on soil, particularly agricultural soils.

Biological treatment – mixing with cow dung

There were a number of studies on transforming arsenic present in biosolids and contaminated soils into soluble and volatile species by microbes (Mukhopadhyay et al. 2002; Turpeinen et al. 2002; Islam et al. 2005; Li et al. 2011). Clostridium collagenovorans, Desulfovibrio gigas, Desulfovibrio vulgaris, Methanothermobacter thermautotrophicus, Bacillus idriensis and Sphingomonas desiccabilis have been found to be able to transfer arsenic to trimethylarsine and arsine respectively through biomethylation process (Michalke et al. 2000; Liu et al. 2011). However, as per our knowledge, the application of biological treatment on arsenic waste treatment and disposal is still limited. Banerjee (2010) found that anaerobic digestion for a period of 50 days could remove up to 99.7% arsenic from arsenic-laden sludge (average of 6.10 g As/dm³ of dry sludge) of conventional arsenic removal plant and stated that this method could be a feasible selection for the safe and cost-effective disposal of arsenic-laden sludge. The mixing of arsenic waste with cow dung beds in the backyards of Bangladesh and Indian household is also considered as an optimal disposal method, especially for rural areas in developing countries (Visoottiviseth & Ahmed 2008).

The results from the experiments of Mohapatra et al. (2008) have shown that cow dung can be used as the major substrate for the growth of microbes during arsenic detoxification and the maximum uptake was 1.08 mg of arsenate/g of substrate. Recently, Rahman et al. (2014) found that arsenic removal using dung beds increased when As/cow dung weight ratio decreased. In this context most of the arsenic was removed during the first few days of mixing. The removal of arsenic from both solid wastes (As bound to iron solids) and aqueous solution in the presence of fresh cow dung is explained by the bio-chemical (e.g., bio-methylation) processes. As reducing bacteria/microbes present in the cow dung transform arsenic into soluble and volatile species through the processes of oxidation, reduction and methylation (Anderson & Bruland 1991; Huang et al. 2012). The results from Rahman et al. (2014) also showed that removal of arsenic under aerobic
conditions was similar to that for anaerobic conditions. However, the conversion of a major portion of the soluble arsenic species into gaseous arsine (AsH₃, also the toxic form of arsenic) is the limiting factor of this method.

**Solidification/stabilization**

Recovery and reuse of arsenic residue seems to be impractical due to the high treatment cost and very limited market for recovered materials. Also the dilution and dispersion of arsenic waste is not a safe solution due to the high toxicity of arsenic. A number of studies have recently focused on the encapsulation of arsenic waste through solidification/stabilization (S/S) techniques.

Cementitious solidification for land disposal of most toxic elements has been recognized by the US EPA as the best demonstrated available technology (Landrett 1986). In most of the research on S/S with arsenic, cement has been used as a sole solidifying material (Buchler et al. 1996) or in combination with iron (II and III) and aluminium compounds, fly ash, and silicates (Chu et al. 1991; Taylor & Fuessle 1994; Akhter et al. 1997; Hassan et al. 2014). Only a few studies have tried to incorporate arsenic waste into industrial slags and encapsulate them using polymers (De Villiers 1995; Carter et al. 1995).

The arsenic waste type or its characteristics play an important role in the S/S process. The results in the study by Buchler et al. (1996) showed that As concentration in the leachate varied from 1.7 to 2.1 mg/L when the S/S process was executed with waste containing sodium arsenate or sodium arsenite. However, the As concentration in the leachate increased significantly to 510 mg/L when waste contained arsanilic acid. To increase the effectiveness of S/S, a number of additives were used in the cementation process. Taylor & Fuessle (1994) added ferrous sulphate, ferric sulphate or aluminium sulphate to the S/S process. They found that S/S did improve due to the precipitation of iron-arsenic or aluminium-arsenic compounds and these precipitates had been encapsulated within or adsorbed onto the cement matrix. The oxidation states of both the iron and arsenic affect S/S efficacy in which ferrous sulphate performed better than ferric sulphate and arsenate S/S was more easily accomplished compared to arsenite S/S. The presence of excessive sulphate in the S/S process decreased cement hydration and in the long term this can lead to harmful effects as arsenic can be leached into the natural environment. The combination of fly ash with cement during the S/S process proved to be less effective than the other combinations (Akhter et al. 1997). Another disadvantage of using fly ash is the bulking problem. Bulk ing resulting from treatment with cement-fly ash is nearly 100%, which is five times higher than the bulking of treatment by silicates and metal hydroxides.

Nguyen et al. (2009) attempted to use S/S for safe disposal of exhausted iron oxide coated sponge (IOCSp) after using it to remove arsenic from water by mixing it with cement, sand, lime and fly ash. The leaching test was carried out by agitating 10 g of the concrete with 100 mL of acetic acid (10%) for 24 h. Their results showed that arsenic concentration in the leachate was very low, ranging from 1 to 35 μg/L and the concentration of arsenic in the leachate decreased when the ratios of cement/exhausted material increased. In their study, the ratio of cement:sand in the samples was based on the Vietnamese standard for concrete. Their results also showed that adding lime to the cement/arsenic waste could increase the capture of arsenic. After 1 month of using the S/S process, only 0.1–1.1% of arsenic was leached from the concrete samples. This indicated that the exhausted IOCSp can be treated properly using the S/S method.

Hassan et al. (2014) tried to use arsenic–iron sludge for making bricks. In their study, the sludge collected from an arsenic–iron removal plant in Bangladesh was mixed with clay at ratios from 3 to 12% of the total weight of sludge-clay mixture and fired at three stages: (i) 105 °C for 2 days; (ii) 500 °C for 12 h; and (iii) 1,000 °C for 12 h. Results showed that the quality of bricks was strongly affected by sludge proportion and firing temperature. The bricks' compressive strength was highest (15.1 MPa) at a sludge ratio of 6%. The leaching was done using three different conditions: acidic condition, alkaline condition and neutral condition. At a sludge ratio of 6%, the amount of arsenic leaching was very low (below 0.075 mg/L).

In some areas, a constructed wetland (CW) system is used to treat arsenic contaminated water. As a consequence, soil and sediments from the CW are contaminated by arsenic. Nakwanit et al. (2011) found that the level of arsenic
in soil and sediment of a CW in Nakorn Si Thammarat Province, Thailand was 75–90 mg/kg soil and 5,069–23,438 mg/kg sediment, respectively. These values were higher than Thailand’s soil standard of 3.9 mg/kg for agricultural land and 27 mg/kg for non-agricultural land. They tried to use S/S (cement:sand: sediment/soil = 19:56:25) for these soils and sediments and discovered that arsenic concentration in the leachate was less than 0.73 mg/L. This concentration was below Thailand’s maximum contaminant level of 5 mg/L and thus there is no need for a secure landfill to store these S/S blocks.

Possible disposal methods for As wastes generated from different water treatment technologies are presented in Table 4 (USEPA 2000). Solids from all types of wastes can be further solidified and stabilized (S/S) using cement, lime, fly ash, etc. before land disposal.

### CONCLUSIONS

The amount and characteristics of arsenic waste derived from a water treatment system depends very much on arsenic concentration in the water and efficiency of the treatment system. Arsenic concentration in liquid waste from spent filter backwash water of adsorption, membrane concentrate is usually low, whereas those from ion exchange

<table>
<thead>
<tr>
<th>Waste form</th>
<th>Waste source</th>
<th>Water treatment technology</th>
<th>Waste type</th>
<th>Possible methods</th>
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<tbody>
<tr>
<td>Liquid</td>
<td>Regeneration</td>
<td>Anion exchange</td>
<td>– Spent backwash</td>
<td>– Sanitary sewer</td>
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<td>– Spent regenerant</td>
<td>– Direct discharge</td>
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<td>– Spent rinse stream</td>
<td>– Evaporation ponds/lagoon</td>
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<td>AA</td>
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<td>– Spent neutralization (acid)</td>
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<td>– Spent rinse liquid filtrate (when brine streams are precipitated)</td>
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<td>Media adsorption</td>
<td>– Spent backwash</td>
<td>– Sanitary sewer</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>– Spent regenerant</td>
<td>– Direct discharge</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>– Spent rinse stream</td>
<td>– Evaporation ponds/lagoon</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Iron and Mn removal</td>
<td>Filter backwash</td>
<td>– Sanitary sewer</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Membrane concentrate</td>
<td>NF and RO</td>
<td>– Direct discharge</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Brine (reject and backwash streams)</td>
<td>– Sanitary sewer</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>– Deep well injection</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>– Evaporation ponds/lagoon</td>
</tr>
<tr>
<td>Solid</td>
<td>Anion exchange</td>
<td>Spent resin</td>
<td></td>
<td>– Landfill</td>
</tr>
<tr>
<td></td>
<td>AA</td>
<td>Spent alumina sludge (when brine streams are precipitated)</td>
<td></td>
<td>– Hazardous waste landfill</td>
</tr>
<tr>
<td></td>
<td>Media adsorption</td>
<td>Spent media</td>
<td></td>
<td>– Return to vendor</td>
</tr>
<tr>
<td></td>
<td>Iron and Mn removal</td>
<td>Sludge (if separated from backwash water)</td>
<td></td>
<td>– Landfill</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Spent media</td>
<td></td>
<td>– Hazardous waste landfill</td>
</tr>
</tbody>
</table>

Solids from all types of wastes can be further solidified and stabilized (S/S) using cement, lime, fly ash and other materials before land disposal.
and AA regenerants are high (up to 300 mg/L). Coagulation by ferric or aluminium salts is a common method for removing arsenic from liquid waste and ferric salts were found to be much better than aluminium salts in this treatment. The TCLP of most solid wastes was below the current TCLP threshold limit (5.0 mg/L) as recommended by the US EPA and therefore they can be disposed of into sanitary landfills. However, there is still a risk of pollution from this waste and further treatment will be needed if a higher stringent standard is applied. Currently, S/S is the most effective method for managing the waste.

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