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Aquatic Arsenic: Phytoremediation Using Floating Macrophytes

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Abstract:

Phytoremediation, a plant based green technology, has received increasing attention after the discovery of hyperaccumulating plants which are able to accumulate, translocate, and concentrate high amount of certain toxic elements in their above-ground/harvestable parts. Phytoremediation includes several processes namely, phytoextraction, phytodegradation, rhizofiltration, phytostabilization and phytovolatilization. Both terrestrial and aquatic plants have been tested to remediate contaminated soils and waters, respectively. A number of aquatic plant species have been investigated for the remediation of toxic contaminants such as As, Zn, Cd, Cu, Pb, Cr, Hg, etc. Arsenic, one of the deadly toxic elements, is widely distributed in the aquatic systems as a result of mineral dissolution from volcanic or sedimentary rocks as well as from the dilution of geothermal waters. In addition, the agricultural and industrial effluent discharges are also considered for arsenic contamination in natural waters. Some aquatic plants have been reported to accumulate high level of arsenic from contaminated water. Water hyacinth (*Eichhornia crassipes*), duckweeds (*Lemna gibba*, *L. minor*, *Spirodela polyrhiza*), water spinach (*Ipomoea aquatica*), water ferns (*Azolla caroliniana*, *Azolla filiculoides*, and *Azolla pinnata*), water cabbage (*Pistia stratiotes*), hydrilla (*Hydrilla verticillata*) and watercress (*Lepidium sativum*) have been studied to investigate their arsenic uptake ability and mechanisms, and to evaluate their potential in phytoremediation technology. It has been suggested that the aquatic macrophytes would be potential for arsenic phytoremediation, and this paper reviews up to date knowledge on arsenic phytoremediation by common aquatic macrophytes.

Keywords: Arsenic, Aquatic macrophyte, Phytoremediation

1. Introduction:

Arsenic is a member of the nitrogen family with both metallic and nonmetallic properties, and is ubiquitous in the environment (soil, water, air and also in living matters) (Tamaki and Frankenberger, 1992). It is the 20th abundant element in earth crust (Woolson, 1975), 22nd in seawater (Brown et al., 1991) and 12th in the human body (Mandal and Suzuki, 2002). It has been used in various fields such as medicine, electronics, agriculture (pesticides, herbicide, insecticides, fertilizer, etc.), livestock (cattle and sheep dips), and as wood preservatives since its isolation in 1250 A.D. (Mandal and Suzuki, 2002; Sharma and Sohn, 2009). It is now well recognized that consumption of arsenic, even at low levels, leads to carcinogenesis (Mandal and Suzuki, 2002). The toxicity and redistribution of arsenic in the environment make it evoking public concern.

Although the occurrence of arsenic in the environment is mainly from minerals and geogenic sources, human activities such as mining, burning of fossil fuels, use of arsenic containing chemicals in agriculture also cause arsenic distribution in the environment (Bissen and Frimmel, 2003). Indiscriminate use of arsenical pesticides during the early to mid-1900s led to an extensive contamination of soil (Smith et al., 1998).

Arsenic is rarely found as its native form because of its bonding affinity with other elements. In the environment, arsenic exists in four oxidation states- arsenate (As(V)), arsenite (As(III)), arsenic (As(0)), and arsine (As(-III)) (Sharma and Sohn, 2009). As(V) is the thermodynamically stable state of arsenic in oxic waters, while As(III) predominates in reduced redox conditions (Cullen and Reimer, 1989). In natural waters, the dominant inorganic arsenicals are incorporated into microorganisms such as phytoplankton, and are converted to methylarsenicals and/or more high order organoarsenicals such as arsenosugar (Maher, 1984). In addition, the organoarsenicals are mineralized to inorganic- and methylarsenicals by bacteria (Hanaoka et al., 1995; Francesconi and Edmonds, 1996). Arsenic compounds found in the aquatic and terrestrial environments are listed in Table 1.

A large number of sites worldwide have been contaminated by arsenic from natural and anthropogenic sources (Mandal and Suzuki, 2002). Many countries, especially Taiwan, Argentina, India, Bangladesh, Mexico, Hungary, and Chile, have reported extensive arsenic contamination of their groundwater supplies (Smedley et al., 2002; Nikolaidis et al., 2004). The range of arsenic in uncontaminated soil ranges from < 1 to 95 mg kg^{-1}. The grand mean of soil arsenic is calculated to be 8.7 mg kg^{-1}. Elevated levels of arsenic in agricultural soil could pose a serious threat to plants and human health and the environment through the food chain pathways (Bruce et al., 2003; Duxbury et al., 2003). Arsenic is toxic to living organisms at high concentration, and inorganic arsenicals are proven carcinogens to human (Ng, 2005). Arsenic toxicity depends on its speciation, and generally inorganic arsenic species are more toxic than those of organic species (Meharg and Hartley-Whitaker, 2002; Ng, 2005). As(III) is more toxic than As(V), and dimethylarsinic acid (DMAA) and monomethylarsonic acid (MMAA) are more toxic than their parent compounds (Petrick et al., 2000).

Remediation of arsenic-contaminated soil and water is important concern. Various technologies are in place to clean up arsenic or to reduce arsenic exposure from contact with arsenic-contaminated soil and water. Technologies for remediation of arsenic-contaminated soil include excavation, immobilization, vetrification, soil washing/flushing and phytoremediation. Phytoremediation process includes phytoextraction, phytostabilization, phytovolatilization, phytotransformation, and rhizofiltration (Table 2). Treatment technologies, applicable for arsenic-contaminated water, consist of precipitation, membrane filtration, adsorption, ion exchange, permeable reactive barriers and biological treatment. Researchers working with phytoremediation have come to realize that the development of this technology needs a thorough understanding of the underlying processes at the genetic, molecular, biochemical, physiological and agronomic levels. Therefore, some intensive researches have been done on arsenic uptake mechanisms in plants. Plants initially accumulate arsenic into their roots through phosphate uptake pathway, i.e., active apoplastic or symplastic mechanisms and translocate to the above ground parts (shoots and leaves). The amount of arsenic translocated from roots to shoots indicates the phytoremediation efficiency of that plant. However, more then 90% of total arsenic accumulated into the plant is stored in roots. Few plants have the ability to translocate high amount of arsenic from roots to shoots. Chinese brake fern (*Pteris vittata* L.) has shown the highest ability to accumulate and translocate arsenic from roots to shoots (Ma et al., 2001). The discovery of *P. vittata* L. was a milestone in arsenic phytoremediation research.

2. Phytoremediation of toxic elements by aquatic macrophytes

Freshwater as well as seawater resources are being contaminated by various toxic elements through anthropogenic activities and from natural sources. Therefore, remediation of contaminated aquatic environment is important as it is for terrestrial environment. Phytoremediation of the toxic contaminants can be readily achieved by aquatic macrophytes or by other floating plants since the process involves biosorption and bioaccumulation of the soluble and bioavailable contaminants from water (Brooks and Robinson, 1998). In aquatic phytoremediation systems, aquatic plants can be either floating on the water surface or submerged into the water. The floating aquatic hyperaccumulating plants absorb or accumulate contaminants by its roots while the submerged plants accumulate metals by their whole body.

Many years ago, Hutchinson (1975) reviewed the ability of aquatic macrophytes to concentrate elements from the aquatic environment and reported that the levels of potentially toxic elements in the plants were at least an order of magnitude higher than in the supporting aqueous medium. Later on, Outridge and Noller (1991) reviewed the hyperaccumulation of toxic trace elements by aquatic vascular plants and discussed about the pathways and rates of elemental uptake and excretion, environmental factors that control uptake of elements, and the significance of trace elements uptake for the field of wastewater treatment and biomonitoring of pollutants, which is of great interest for bioremediation of aquatic systems. By this time, considerable number of literatures have been published which described different aspects of biogeochemistry, mechanisms and uptake of toxic elements by a large number of aquatic macrophytes (Table 3) to develop effective phytoremediation technology. Several aquatic macrophytes and some other small aquatic floating plants have been investigated for the remediation of natural and wastewater contaminated with Cu(II), Cd(II) and Hg(II) (Sen and Mondal, 1987; Selvapathy and Sreedhar, 1991; Alam et al., 1995). *Microspora* and *Lemna minor* were studied for Pb and Ni remediation (Axtell et al., 2003). Five common aquatic plant species (*Typha latifolia*, *Myriophyllum exalbescens*, *Potamogeton epihydrus*, *Sparganium angustifolium* and *Sparganium multipedunculatum*) were tested for Al phytoremediation (Gallon et al., 2004). Parrot feather (*Myriophyllum aquaticum*), creeping primrose (*Ludwigina palustris*), and water mint (*Mentha aquatic*) have been reported to remove Fe, Zn, Cu, and Hg from contaminated water effectively (Kamal et al., 2004). The *L. minor* was reported to accumulate Cu and Cd from contaminated wastewater (Kara, 2004; Hou et al., 2007). The submerged aquatic plant *Myriophyllum spicatum* L. has been reported as an efficient plant species for the metal-contaminated industrial wastewater treatment (Lesage et al., 2007). The aquatic plants *Rorippa* *nasturtium-aquaticum* (L.) and *Mentha* spp. accumulate arsenic from contaminated freshwater (Robinson et al., 2006). The encouraging results of previous studies regarding phytoremediation using aquatic plants gained the attention of researchers and scientists to continue studies in this field. In this review, previous works on phytoremediation of arsenic, one of the major and widespread contaminants, using aquatic macrophytes has been discussed.

3. Phytoremediation of arsenic using aquatic macrophytes

As(V) is the predominant species in the oxic water, and As(V) and As(III) are bioavailable forms for the aquatic plants (Sizova et al., 2002). The dynamics of As(V) exchange between water and adsorbing colloids are analogous to those of phosphate, though the competition for exchange sites favors phosphate over As(V) (Mkandawire et al., 2004a). Thus, aquatic macrophytes can be a good tool for the remediation of arsenic-contaminated aquatic systems, and some species of aquatic macrophytes have already been reported to accumulate a formidable amount of arsenic from water (Lee et al., 1991; Elayan, 1999; Robinson et al., 2003; Mkandawire et al., 2004a; Mkandawire et al., 2004b; Mkandawire and Dudel, 2005; Robinson et al., 2006; Rahman et al., 2007; Alvarado et al., 2008; Mishra et al., 2008; Tripathi et al., 2008; Zhang et al., 2008; Mirza et al., 2010). *Lemna gibba* L. and *L. minor* L. are the most studied species of Lemnaceae family in phytoremediation and ecotoxicology (Mkandawire et al., 2004a; Mkandawire et al., 2004b; Mkandawire and Dudel, 2005). Some other aquatic macrophytes, such as water hyacinth (*Eichhornia crassipes*) (Alvarado et al., 2008; Mishra et al., 2008), butterfly fern (*Salvinia natans* and *Salvinia minima*) (Hoffmann et al., 2004; Rahman et al., 2008c), lesser duckweed (*L. minor* L.) (Robinson et al., 2005; Alvarado et al., 2008; Mishra et al., 2008), greater duckweed (*Spirodela polyrhiza* L. and *Spirodela intermedia*) (Rahman et al., 2007; Mishra et al., 2008; Rahman et al., 2008b;

Rahman et al., 2008d), water spinach (*Ipomoea aquatica*) (Lee et al., 1991), esthwaite waterweed/hydrilla (*Hydrilla verticillata*) (Lee et al., 1991), mosquito fern/water velvet (*Azolla caroliniana*) (Zhang et al., 2008), water fern (*A. filiculoides* and *A. pinnata*) (Rahman et al., 2008a; Zhang et al., 2008), alligator weed (*Althernanthera philoxeroides*) (Elayan, 1999), water cabbage/water lettuce (*Pistia stratiotes*) (Lee et al., 1991; Odjegba and Fasidi, 2004), needle spikerush (*Eleocharis acicularis*) (Ha et al., 2009a), watercress (*Nasturtium officinale* (formerly *Rorippa nasturtium-aquaticum*), *Lepidium sativum* L.) (Robinson et al., 2003; Robinson et al., 2005), rigid hornwort (*Ceratophyllum demersum*) (Robinson et al., 2005), petries starwort (*Callitriche petriei*) (Robinson et al., 2005), waterweed/pondweed (*Elodea canadensis*) (Robinson et al., 2005), miriophyllum (*Myriophyllum propinquum*), Brazilian waterweed (*Veronica aquatica*) (Robinson et al., 2005), curly waterweed (*Lagarosiphon major*) (Robinson et al., 2005), water pepper (*Polygonum hydropiper*), have also been reported to uptake arsenic from water.

4. Potential aquatic macrophytes for arsenic phytoremediation

4.1. Water hyacinth

Water hyacinth is a free-floating perennial aquatic plant native to tropical and sub-tropical South America, and is now widespread in all tropic climates. The genus *Eichhornia* comprises seven species of water hyacinth among which *E. crassipes* is the most common and have been reported to grow very first. However, its enormous biomass production rate, high tolerance to pollution (Ebel et al., 2007), and absorption capacity of heavy-metal and nutrient qualify it for use in wastewater treatment (Ebel et al., 2007; Fang et al., 2007).

A comprehensive study on the arsenic removal from water by *E. crassipes* was performed by Alvarado et al. (2008), and the results showed that it had a removal rate of 600 mg arsenic ha^{-1} d^{-1} under field condition and a removal recovery of 18% under laboratory conditions. The removal efficiency of water hyacinth was higher due to its high biomass production and favorable climatic conditions. Mishra et al. (2008) compared arsenic removal efficiency of *E. crassipes*, *L. minor* and *S. polyrhiza* from tropical opencast coalmine effluent and observed that *E. crassipes* had the highest removal efficiency (80%) compared to other aquatic macrophytes over a 25 d course. This was supposed to be due to faster growth rate (Muramoto and Oki, 1983; Kelley et al., 1999), greater biomass production, and higher uptake ability of arsenic. Water hyacinth represents a reliable alternative for arsenic bioremediation in aquatic system even though the plant may cause severe water management problems because of its huge vegetative reproduction and high growth rate (Giraldo and Garzon, 2002). So, the use of water hyacinth in phytoremediation technology should be considered carefully.

4.2. Duckweed

Duckweeds are small free floating aquatic angiosperm plants which do not have distinct stems and leaves. The whole plant body is reduced to form a flat small leaf-like structure called frond. Duckweed family comprises of 4 genera, *Lemna*, *Spirodela*, *Wolffia*, and *Wolfiella*, and of 34 species. Among these 4 genera, *Lemna*, *Spirodela*, and *Wolffia* have been reported to accumulate arsenic from water (Mkandawire et al., 2004a; Mkandawire et al., 2004b; Mkandawire and Dudel, 2005; Rahman et al., 2007; Alvarado et al., 2008; Rahman et al., 2008b; Zhang et al., 2009). Mkandawire and Dudel (2005) studied the bioaccumulation of arsenic in *L. gibba* L. both in field (tailing water) and at laboratory conditions. Results showed that samples from tailing water were approximately two-fold less than similar arsenic concentrations in laboratory solutions. This was because *L. gibba* in the surface mine water was exposed to arsenic for an undefined period, whereas samples used in laboratory trials were certified to contain a below-detection amount of arsenic and were exposed to arsenic only for 21 d. However, the mean bioaccumulation coefficients for *L. gibba* were two-folds higher in the laboratory than in the field. In an arsenic removal efficiency study with *L. minor* L., Alvarado et al. (2008) found that the removal rate for *L. minor* was 140 mg arsenic ha^{-1} d^{-1} with a removal recovery of 5%. Mkandawire and Dudel (2005) also reported that arsenic accumulation in *L. gibba* L. was correlated negatively with phosphate concentrations (Fig. 1) because of the fact that As(V) is the dominant form of arsenic under oxic conditions and is an analogue of phosphate; they compete for the same uptake carriers in the plasmalemma.

Greater duckweed (*S. polyrhiza* L.) was tested for arsenic accumulation under laboratory conditions by Rahman et al. (2007; 2008b) to investigate arsenic uptake efficiency and mechanisms. Results showed that when *S. polyrhiza* L. accumulated higher amount of arsenic from As(V) solution compared to that from DMAA solution (Fig. 2). They also observed that As(V) uptake into *S. polyrhiza* L. was negatively correlated with phosphate uptake and positively correlated with iron uptake. The facts were explained by the competitive uptake inhibition of As(V) by phosphate and adsorptive affinity of As(V) on iron oxides of root surfaces. In contrast, DMAA uptake in *S. polyrhiza* L. was neither affected by phosphate nor correlated with iron (Rahman et al., 2007). Thus, it has been proposed that *S. polyrhiza* L. might use different mechanisms for As(V) and DMAA uptake. However, arsenic uptake ability of *S. polyrhiza* L. suggests that this macrophyte would be a good option for phytoremediation of contaminated water.

Arsenic accumulation, speciation and tolerance of the rootless duckweed *Wolffia globosa* and its potential for arsenic phytofiltration was investigated by Zhang et al. (2009). It was reported that the frond of *W. globosa* can accumulate >1000 mg arsenic kg^{-1} dry weight (dw), and can tolerate up to 400 mg arsenic kg^{-1} dw (Zhang et al., 2009). The higher uptake efficiency (2 to 10 times higher than other duckweed or *Azolla* species) (Fig. 3) and higher tolerance competence of *W. globosa* would place this aquatic plant potential for arsenic phytoremediation.

4.3. Water fern (*Azolla*)

Species of mosquito/water fern (*Azolla*) is a small, free-floating fern, and is widely distributed in paddy fields, rivers, ponds and lakes. It can fix nitrogen by its symbiotic partnership with *Anabaena* which resides in the dorsal cavity of *Azolla* fronds; for this reason *Azolla* has been used as a green manure to improve soil fertility and rice production (Wagner, 1997). It has been reported that *Azolla* has a high capacity to accumulate toxic elements such as mercury, cadmium, chromium, copper, nickel and zinc (Sela et al., 1989; Rai, 2008; Rai and Tripathi, 2009), and can be used to remove contaminants from wastewater (Bennicelli et al., 2004; Arora and Saxena, 2005; Rakhshaee et al., 2006).

Three species of water fern (*A. caroliniana*, *A. filiculoides* and *A. pinnata*) have been studied for arsenic uptake from water. Zhang et al. (2008) investigated arsenic uptake and tolerance of fifty strains of *Azolla* and found that arsenic concentrations in the fronds varied by 13.5-fold (ranged between 29 and 397 mg kg^{-1} dw) upon exposure to 50 mM As(V) solution for 10 d. *A. caroliniana* showed the highest arsenic uptake potential in the frond (284 mg kg^{-1} dw) followed by *A. microphylla*, *A. pinnata*, *A. maxicana*, *and A. filiculoides* (Fig. 3). Upon exposure to As(V), As(III), DMAA and MMAA, methylated species uptake was accounted for only 1–5%, while As(V) and As(III) were for about 50-60 and 25-40% of the total arsenic, respectively (Zhang et al., 2008). Rahman et al. (2008a) also reported higher arsenic concentrations in fronds of *A. pinnata* when As(V) and As(III) were the sources of arsenic in the culture solution. Results revealed that the trends of arsenic uptake in *A. pinnata* was As(V) > As(III) > DMAA > MMAA (Rahman et al., 2008a; Zhang et al., 2008), which is consistent with uptake behavior of arsenic species by other aquatic macrophytes.

4.4. Butterfly fern (*Salvinia*)

Salvinia is a free-floating aquatic macrophyte of Salviniaceae. Its wide distribution, faster growth rate and close relation with other water ferns, including *Azolla* and *Lemna*, make it potential for phytoremediation. A number of species of this aquatic plant have been studied for phytoremediation of toxic elements (Banerjee and Sarker, 1997; Espinoza-Quinones et al., 2005; Sanchez-Galvan et al., 2008; Dhir, 2009; Wolff et al., 2009). Only two species, *S. natans* and *S. minima*, have been investigated for arsenic remediation (Hoffmann et al., 2004; Rahman et al., 2008c). Arsenic uptake kinetics of *S. minima* reveal that arsenic uptake was increased with increasing exposure time and arsenic concentration in the growth solution (Hoffmann et al., 2004). The increasing phosphate concentration decreased arsenic uptake in *S. minima* (Hoffmann et al., 2004), which is similar to that of other aquatic floating macrophytes.

Influence of phosphate and iron ions in selective uptake of arsenic species by *S. natans* L. was studied by Rahman et al. (2008c). Results showed that increasing phosphate concentration in culture solution decreased the arsenic uptake into the plant significantly, when the source of arsenic was As(V). On the other hand, arsenic uptake was not influenced by phosphate when the source was DMAA. Significant correlation was reported between arsenic and iron content in tissues of *S. natans* L. when the source was As(V), while the correlation was not significant when the source was DMAA (Rahman et al., 2008c). The results indicate that DMAA uptake mechanisms into *S. natans* L. are deferent from those of As(V), which is also same for *S. polyrhiza* (Rahman et al., 2007).

4.5. Water lettuce (*P. stratiotes*)

Pistia is a genus of aquatic plant in the family Araceae, comprising a single species, *P. stratiotes*, often called water cabbage or water lettuce. It floats on the surface of the water, and its roots hanging submersed beneath floating leaves. It is a common aquatic weed in the United States, particularly in Florida where it may clog waterways.

Although *P. stratiotes* mats degrade water quality by blocking the air-water interface, reducing oxygen levels in the water, and thus threatening aquatic life, it has been tested for metal remediation (Odjegba and Fasidi, 2004; Skinner et al., 2007), metal detoxification (Tewari et al., 2008), and treatment of urban sewage (Zimmels et al., 2006). Arsenic uptake by *P. stratiotes* has not been studied extensively. A field study carried out by Lee et al. (1991) reported that the average arsenic enrichment (bioaccumulation) factor of *P. stratiotes* was 8632 in roots and 2342 in leaf. It appears that arsenic translocation in *P. stratiotes* was slow and most of the arsenic was strongly adsorbed onto root surfaces from solution. This agrees with the earlier findings that arsenic compounds are less readily translocated through the root system of aquatic plants.

4.6. Watercress

Watercresses (*N. officinale*, *Nasturtium microphyllum* (formerly *R. nasturtium-aquaticum*, *R. microphylla*)) are fast-growing, aquatic/semi-aquatic, perennial plants native from Europe to central Asia, and one of the oldest known leaf vegetables consumed by human. Even though watercresses are popular vegetable in many countries, they have been reported to accumulate toxic elements from water and soil.

Geothermal activity in the Taupo Volcanic Zone (TVZ), New Zealand, has resulted increase of arsenic levels in soil, and lake and river water of the regions. Some aquatic plants in the TVZ have been reported to accumulate inordinate amounts of arsenic (Robinson et al., 2003; Robinson et al., 2006). One of the species of watercress *L. sativum* is consumed as a vegetable, especially by the indigenous community in New Zealand, (Robinson et al., 2003). An average of 29 and 16 mg kg^{-1} of arsenic have been found in leaf and stem of *L. sativum* grown in the Waikato river, New Zealand, respectively (Robinson et al., 2003). High concentrations of arsenic have also been reported in *N. microphyllum* (up to 138 mg kg^{-1} fresh wt) grown in TVZ, New Zealand (Robinson et al., 2006). Because of these higher levels arsenic in naturally grown *L. sativum* and *N. microphyllum* of Waikato River, New Zealand, it has been recommended that watercress from the Waikato River or other areas with elevated water arsenic concentrations might pose human health risk and should not consume (Robinson et al., 2003).

Greenhouse experiments also revealed that watercress reduced arsenic concentration in the solutions by 7.3% on average (Robinson et al., 2003). High arsenic uptake ability of *L. sativum* and *N. microphyllum* from water containing relatively low concentration of this element indicated their high bioaccumulation ability and therefore these plants may have potential for arsenic phytoremediation. Since watercresses are consumed as vegetable, these aquatic plants should carefully be used in arsenic phytoremediation.

4.7. Waterweed (*E. canadensis*) and Needle spikerush (*E. acicularis*)

E. canadensis is a perennial aquatic submerge macrophyte native to North America. It grows rapidly in favorable conditions and can choke shallow ponds, canals, and the margins of some slow-flowing rivers. *E. canadensis* is also known as Canadian waterweed and have been studied for toxic element uptake. Another aquatic macrophyte, *E. acicularis* have been tested in laboratory and field conditions to assess its accumulation capability of antimony, arsenic, copper, and zinc, and thereby it's potential in phytoremediation (Ha et al., 2009a; Ha et al., 2009b; Ha et al., 2011). Results show that *E. acicularis* accumulates 6.4 mg arsenic kg^{-1} dw after 10 d exposure to arsenic solution in laboratory conditions, and the removal rate of arsenic was 2.8 µg L^{-1} d^{-1} (Ha et al., 2009a). Ha et al. (2009b) also investigated the uptake ability of multiple heavy metals by aquatic macrophyte *E. acicularis* and its potential application for phytoremediation at an abandoned mining area in Hokkaido, Japan. The results indicated that *E. acicularis* showed great potential for the phytoremediation of mine tailing and drainage rich in heavy metals.

4.8. Esthwaite waterweed (*H. verticillata*)

Hydrilla (esthwaite waterweed or hydrilla), which was first discovered in the United States in 1960, is an aquatic plant genus which consist of one species; *H. verticillata*, however, some botanists divide it into several species. It is native to the cool and warm waters in Asia, Europe, Africa and Australia, with a sparse, scattered distribution; in Europe, it is reported from Ireland, Great Britain, Germany, and the Baltic States, and in Australia from Northern Territory, Queensland, and New South Wales (Langeland, 1996; Thorne and Reveal, 2007). A highly specialized growth habit, physiological characteristics, and reproduction make this plant well adapted to life in submersed freshwater environments (Langeland, 1996).

H. verticillata has been tested for the uptake and remediation of arsenic, copper, lead, zinc and chromium from water (Lee et al., 1991; Elankumaran et al., 2003; Dixit and Dhote, 2010). Elankumaran et al. (2003) reported bioabsorption of copper from wastewater by *H. verticillata*. Dixit and Dhote (2010) studied chromium, lead and zinc uptake in and morphological changes of *H. verticillata* and observed that this macrophyte uptake the metals on a dose response basis. They also showed that *H. verticillata* decreased chromium, lead and zinc concentrations by 72-80, 68-86 and 60-80% after 4 wk, respectively. But the plants die at higher concentrations (> 20 mg kg^{-1}) and longer period of exposure. Srivastava et al. (2006) investigated the responses of *H. verticillata* (L.f.) Royle to different concentrations of copper (0.1–25 M) upon a 1–7 d exposure. Results showed that the plant accumulated high amount of copper with a maximum of 770 mg kg^{-1} dw when exposed to 25 M copper for 7 d. Thus, *H. verticillata* is would be a promising candidate for the phytoremediation of contaminated water. Lee et al. (1991) performed a systematic study on arsenic uptake by *H. verticillata* at laboratory condition. They also measured arsenic concentrations in field samples and found enrichment factor (EF) (also know as bioaccumulation factor) of 7106 (ranging between 1286 and 11141) for *H. verticillata*. The relatively high EF, reasonable size, ease identification and sampling, hardiness, relative abundance and common occurrence in the aquatic systems fulfill the criteria for *H. verticillata* to be an bioindicator of arsenic pollution in the aqueous ecosystem (Lee et al., 1991). Recently, Srivastava et al. (2010) investigated arsenic uptake and metabolism in *H. verticillata* and the results revealed that the accumulation of arsenic by the plant was dependant on both concentration of the metalloid in water and duration of exposure. It is evident that *H. verticillata* accumulates higher arsenic upon exposure to As(III) than to As(V) (Srivastava et al., 2010). Maximum arsenic accumulation of 231 mg kg^{-1} dw was observed when the plants were exposed to 10 μM As(III) for 7 d and the accumulation was 121 mg kg^{-1} dw when exposed to same concentration of As(V) for the same duration. Because of high arsenic uptake capability Srivastava et al. (2010) proposed *H. verticillata* as a promising aquatic plant for the phytoremediation of arsenic-contaminated water.

Laboratory studies reveal that arsenic uptake by *H. verticillata* is a function of the initial concentration of the element, and is inhibited at high phosphate concentration (Lee et al., 1991). Results showed that upon exposure to 0.4 mg kg^{-1} As(V), 12 to 160 mg kg^{-1} phosphate inhibited arsenic uptake in *H. verticillata* whereas 5 mg kg^{-1} phosphate did not influence the uptake significantly. Arsenic uptake was almost completely inhibited by 160 mg kg^{-1} phosphate. This might be due to the competitive uptake of phosphate and its chemical analogue As(V) in *H. verticillata*, and the same phenomenon has also been observed for other aquatic plants (Robinson et al., 2003; Mkandawire et al., 2004a; Rahman et al., 2008c, b). Arsenic accumulation in *H. verticillata* and the response of this plant to different concentrations of As(V) and As(III) were investigated by Srivastava et al. (2007; 2010). Results showed that arsenic accumulation was higher in plants exposed to As(III) compared to those plants exposed to As(V). *H. verticillata* was able to tolerate low concentrations of As(III) and As(V) by detoxifying the metalloid. In another study, Srivastava and D'Souza (2009) investigated the effect of sulfur (S) on arsenic accumulation in *H. verticillata*. Results showed that arsenic accumulation was about 2-fold higher upon exposure to either As(V) or As(III) in S-excess plants compared to that in S-sufficient and S-deficient plants.

5. Mechanisms of arsenic uptake in aquatic macrophytes

The inorganic forms (As(V) and As(III)) and the methylated forms (MMAA(V)) and DMAA(V)) are the main species of arsenic in natural water (Cullen and Reimer, 1989). Aquatic organisms, such as algae, reduce As(V) to As(III) and further biotransform to methylated arsenicals which results in the occurrence of thermodynamically unstable As(III) and methylarsenic compounds in natural water (Meharg and Hartley-Whitaker, 2002; Hellweger et al., 2003; Hellweger and Lall, 2004). The bulk of the total dissolved arsenic is inorganic arsenic in seawater (Peterson and Carpenter, 1983) and in freshwater (Seyler and Martin, 1989; Kuhn and Sigg, 1993). Although the predominant form of methylarsenicals is consistently DMAA(V) followed by MMAA(V), the existence of trivalent methylarsenic species in the environment has also been reported (Hasegawa et al., 1994; Hasegawa, 1996; Sohrin et al., 1997).

Arsenic uptake mechanisms in terrestrial hyperaccumulating plants have been studied and reported in a number of literatures (Asher and Reay, 1979; Meharg and Hartley-Whitaker, 2002; Tripathi et al., 2007; Zhang et al., 2009; Zhao et al., 2009). However, three mechanisms have been proposed for the uptake of arsenic species in aquatic macrophytes- i) active uptake through phosphate uptake transporters, ii) passive uptake through aquaglyceroporins, and iii) physicochemical adsorption on root surfaces. Plants mainly uptake As(V) through phosphate uptake transporters (Tripathi et al., 2007; Zhao et al., 2009), however, physicochemical adsorption on root surfaces has also been supposed to be an alternative uptake pathway for this arsenic species (Robinson et al., 2006; Rahman et al., 2008b). As(III), DMAA and MMAA gets into the plants by passive mechanism through the aquaglyceroporin channels (Zhao et al., 2009; Rahman et al., 2011).

5.1. Active uptake through phosphate uptake transporters

As(V) and phosphate are chemical analogs, and compete for uptake carriers in the plasmalemma (Mkandawire et al., 2004a). Therefore, more As(V) is expected to be desorbed in the solution with the increase phosphate concentration (Smith and Read, 2008). Plants uptake As(V) through phosphate transporters (Tripathi et al., 2007; Zhao et al., 2009) (Fig. 4). The aquatic plants also use the same mechanisms for the uptake of arsenic species. Mkandawire and Dudel (2005) and Rahman et al. (2007) showed that *L. gibba* L. and *S. polyrhiza* L. uptake As(V) through the phosphate uptake pathway.

5.2. Passive uptake through aquaporins/aquaglyceroporins

Although the exact uptake mechanisms for As(III) and organoarsenicals (DMAA and MMAA) in higher plants have not been identified, physiological studies suggest that these arsenic species are transported in rice by passive uptake mechanisms through aquaporins/aquaglyceroporins (Abedin et al., 2002; Meharg and Jardine, 2003; Rahman et al., 2011). A recent molecular study explained more clearly that As(III) is transported into rice roots by nodulin26-like intrinsic membrane proteins (NIPs), one of the major subfamilies of aquaporins transporter that facilitates the transport of neutral molecules such as water, glycerol, and urea (Ma et al., 2008). Aquaporins and aquaglyceroporins are two of three subfamilies of water channel proteins (WCPs), the transmembrane proteins that have a specific three-dimensional structure with a pore that permeates water molecules (Benga, 2009), which are permeable to water, glycerol, and/or other small, neutral molecules. The competition between glycerol and As(III) for uptake into rice (*Oryza sativa* L.) reveal that this arsenic species is transported across the plasma membrane through aquaporins/ aquaglyceroporins (Meharg and Jardine, 2003). A recent study also showed that glycerol inhibited DMAA and MMAA uptake in rice roots significantly ($p \leq 0.0001$ and 0.0077 for DMAA and MMAA, respectively) in a concentration-dependant manner (Rahman et al., 2011). Since rice is an aquatic/wetland plant, it can be elucidated from the above discussion that aquatic macrophytes may also uptake As(III), DMAA, and MMAA by passive uptake mechanisms through aquaporins/aquaglyceroporins (Fig. 4).

5.3. Physicochemical adsorption on root surfaces

Physicochemical adsorption, an alternative mechanism for As(V) accumulation into aquatic plants, has been proposed by Robinson et al. (2006). In this mechanism, suspended oxides of iron (Fe-plaque) on the aquatic plant surfaces adsorb and accumulate arsenic. Robinson et al. (2006) observed positive correlation between arsenic and iron concentrations in aquatic plants, which is supposed to be for the adsorption of arsenic on iron oxides on plant surfaces. But, which species of arsenic was mostly adsorbed by iron oxides was not clear from their studies. Rahman et al. (2008b) studied the adsorption of arsenic species on precipitated iron oxides on roots/fronds of *S. polyrhiza* L. and found significant correlation between arsenic and iron concentrations in tissues when the plant was exposed to As(V). There was no correlation between arsenic and iron in plant tissue when *S. polyrhiza* L. was exposed to As(III), DMAA, and MMAA. Results reveal that As(V) is predominantly adsorbed on precipitated iron oxides on the roots of aquatic plants and accumulated by physicochemical adsorption mechanism. Rahman et al. (2008b) also observed that upon exposure to As(V), arsenic and iron concentrations in *S. polyrhiza* L. were highly correlated in phosphate-deficient condition, while they were not significantly correlated in phosphate-sufficient condition. This can be attributed to the high adsorption of arsenic on iron plaque of the plant surfaces in phosphate-deficient solution, which was blocked by phosphate in phosphate-sufficient solution.

6. Arsenic metabolism and detoxification in aquatic macrophytes

Arsenic exists predominantly as As(V) in oxic condition and as As(III) in reduced condition (Zhao et al., 2009). As(V) shares the same transporter with phosphate in plants while As(III) gets into the plant cells through aquaporins of NIPs (Zhao et al., 2009; Srivastava et al., 2010). These two forms of arsenic induce differential biochemical responses in aquatic plants because of their dissimilar chemical properties (Srivastava et al., 2007). As(V) does not bind to thiol ligands whereas As(III) has high binding affinity to sulphhydryl (-SH) groups of peptides such as glutathione (GSH) and phytochelatins (PCs) (Raab et al., 2007; Zhao et al., 2009). Arsenic speciation in plant tissues shows that arsenic is predominantly present in the As(III) oxidation state, even though plants had been exposed to As(V). This means that, following uptake, As(V) is reduced to As(III) in plant cells efficiently, and that most plants have high competence for As(V) reduction (Zhao et al., 2009). The reduction of As(V) to As(III) is mediated by GSH (Delnomdedieu et al., 1994) and by enzyme (Bleeker et al., 2006), which is thought to be a detoxification mechanism of the plants. Upon uptake, As(V) and As(III) have been reported to produce reactive oxygen species (ROS) inside the cells (Meharg and Hartley-Whitaker, 2002), and plants control the production of ROS by various enzymes and cellular compounds (Mittler, 2002). The GSH can act as an antioxidant and is required for the synthesis of metalloid chelating ligands, the PCs (Srivastava et al., 2007). Schmoger et al. (2000) reported rapid induction of PCs biosynthesis in cell suspension cultures of *Rauvolfia serpentina*, in seedlings of *Arabidopsis*, and in enzyme preparations of *Silene vulgaris* upon exposure to As(III) and As(V). Others (Raab et al., 2005; Raab et al., 2007) also identified a number of As(III)-GSH and As(III)-PCs complexes in sunflower (*Helianthus annuus*), which are subsequently sequestered in the vacuoles and excreted from the cell.

Recently, arsenic accumulation and detoxification in aquatic plant *H. verticillata* have been investigated by Srivastava and D'Souza (2009) and by others (Srivastava et al., 2007; Srivastava et al., 2010). Arsenic detoxification mechanisms in aquatic plants seems to be as same as in the terrestrial plants. Upon exposure to either As(III) or As(V), *H. verticillata* increased the synthesis of thiols such as PCs, and increased antioxidant enzyme activity (Srivastava et al., 2007). Although the levels of thiolic componds (such as NP-SH, cysteine, GSH, and oxidized glutathione (GSSG)) in *H. verticillata* were increased significanlty upon exposure to both As(III) and As(V) (Srivastava et al., 2007), As(III) was found to induce the activities of cysteine synthase and γ-glutamylcysteine synthetase and the amount of cysteine and GSH to higher levels than that of As(V) (Srivastava et al., 2010). The analysis of PCs revealed that the accumulation of PC_1 and PC_2 in *H. verticillata* was increased with the increase of both As(III) and As(V) concentrations (Srivastava et al., 2007). Thus, phytochelatins and antioxidant systems in *H. verticillata* respond differentially during As(III) and As(V) stress which is thought to be the detoxification mechanism of the plant.

7. Management and treatment of phytoremediating aquatic macrophytes

With increasing interest about the phytoremediation of arsenic contaminated water by hyperaccummulating aquatic plants and the promising economic and environmental prospects of this technology, the eventual fate of those plants with high arsenic burden will be a vital issue in the future. Although a number of field studies as well as laboratory experiments have shown that some aquatic plants are potential for arsenic accumulation and would be used for phytoremediation of arsenic contaminated water, large scale implementation of this technology has not been reported yet. With large scale implementation of this technology, management and disposal of the huge amount of phytoremediating plants with high arsenic content will be an important concern. If the phytoremediating plants will not be disposed properly, those plants might be another source of arsenic contamination in the environment. Up to date, studies about the management and disposal of arsenic phytoremediating aquatic plants are inadequate. Without proper management and disposal of the phytoremediating plants with high arsenic content, only the accumulation and removal of the metalloid from water by aquatic plants would not be enough for the successful implementation of this emerging technology. There may be some processes for the disposal of high arsenic burden aquatic plants, but it is difficult to elucidate if this would be economically and environmentally feasible or not.

7.1. Carbonization and incineration

The high arsenic content aquatic plants may be used for the making charcoal and the bye-product gas can be used as fuel. Previously water hyacinth has been used in this purpose (Thomas and Eden, 1990). There are two problems in using the aquatic phytoremediating plants for making charcoal. First, there is a need to reduce the water content, and second the ash content of air-dried water is too high (for example, 40% for water hyacinth (Thomas and Eden, 1990)) to get a good fuel as an end product. The high investments and technological level necessary also make carbonization.

Incineration of the hyperaccumulating plants would be one option. Sun drying and direct burning of water hyacinth is used on a small scale in certain parts of the world to use the ash as fertilizer (Gunnarsson and Petersen, 2007). Fresh aquatic plants have high moisture content and therefore, it may take longer time for drying. In addition, there is no evidence whether arsenic is completely vanished after burning the plants. Incineration of the plants with high arsenic content may also be a source of arsenic emission in the air. It has been reported that burning high arsenic-containing coal is one of the major sources of arsenic exposure (10-20% of the total arsenic exposure) for the population of Guizhou, China (Liu et al., 2002). Another study also revealed that burning coal with high arsenic content increased arsenic content in hair, urine, and blood in children residing in polluted area (Bencko and Symon, 1977). So, burning arsenic hyperaccumulating aquatic plants would not be environmentally safe, and would be hazardous for human health.

7.2. Hydrolysis and fermentation

Liquid fuel, such as ethanol, may be produced from phytoremediating aquatic plants by hydrolysis together with fermentation which would make aquatic plants a good substrate. Hydrolysis and fermentation also require yeast fermentable sugars that may available only to a very low extent in aquatic phytoremediating plants. Some kind of pre-treatment is, therefore, needed to make the sugar more easily available for chemical hydrolysis (Gunnarsson and Petersen, 2007). The pre-treatment requires a relatively high temperature, strong acids and pressurized reactors. Thomas and Eden (1990) conclude that hydrolysis of water hyacinths to produce fuel is only feasible in situations where there is a high need for ethanol as a liquid fuel because of the negative energy balance. Even it is economically feasible to produce fuel from phytoremediating aquatic plants, arsenic content in bye-product sludge and its recontamination possibility should be tested.

7.3. Briquetting

Briquettes have been widely sold commercially for cooking food. Briquetting would be a good option for the treatment of the phytoremediating aquatic plants. Thomas and Eden (1990) reported briquetting as a possible treatment of water hyacinth. The briquettes are made by sun-drying the water hyacinth for a few days, disintegrating, screening and chopping the dried water hyacinths to pieces about 6 mm long. The shredded water hyacinth can then be compressed into briquettes or pellets. The material resulting after briquetting water hyacinth has an energy density of 8.3 GJ m^{-3}, which is comparable to charcoal which has 9.6 GJ/m^3 (Thomas and Eden, 1990; Gunnarsson and Petersen, 2007).

7.4. Anaerobic digestion and production of biogas

Anaerobic digestion is a biological process by which organic matter is degraded in the absence of oxygen and biogas is produced as a bye-product. The gas can be used directly for cooking, heating or production of electricity. Biogas has very useful by-products and positive impacts on public health and pollution. This, together with the growing shortage of firewood and rising cost of fossil fuels, has made anaerobic digestion increasingly demanding. These advantages of the process might make it well suited for use in developing countries (Gunnarsson and Petersen, 2007). Aquatic plants, such as water hyacinths, can be degraded easily, and give quite a high gas yield (Gunnerson and Stuckey, 1986).

Biogas production from phytoremediation biomass would be a viable, interesting and environmentally sound idea for phytoremediating aquatic plant management. Chanakya et al. (1993) found that water hyacinth has a high content of fermentable matter and therefore a high potential for biogas production. However, there are some limitations in biogas production form water hyacinth. The high lignin content can reduce the actual production and the low bulk density could result in large voids with poor compaction and low feed rates (Chanakya et al., 1993). The use of crop residue and aquatic weeds has been tested for biogas production by El-Shinnawi et al. (1989). They mixed rice straw, maize stalks, cotton stalks and water hyacinths with cow dung and were digested in different containers. The mixture of water hyacinth and cow dung was found to produce more biogas per kg organic matter than that of maize and cotton stalks, but the total biogas production per kilogram dry matter added was lower for the water hyacinths. The low values for total gas production was probably mostly due to the high lignin content and low percentage of volatile solids in the water hyacinths (Chanakya et al., 1993; Gunnarsson and Petersen, 2007). Gunnarsson and Petersen (2007) showed that water hyacinths can compete well with any kind of animal manure as a substrate for biogas production.

Singhal and Rai (2003) studied the use of water hyacinth and channel grass, used for phytoremediation of industrial effluents, in biogas production. Although extensive studies have not been done on this subject, there is a good possibility to use phytoremediation plants in biogas production. But arsenic content and speciation in sludge from the anaerobic digestion should be investigated to prevent its redistribution in the environment.

8. Conclusions:

Increasing use of arsenic-contaminated underground water for agricultural purposes (irrigation in rice and other crop fields) in the arsenic affected areas, especially in Bangladesh and West Bengal (India), results in the increased concentration of arsenic in freshwater systems. Arsenic contamination in freshwater systems poses health threat not only to the aquatic organisms but also to the humans. High level of arsenic have been reported in a number of aquatic plants grown in contaminated water bodies (rivers and lakes) which are consumed as vegetable in many countries (Robinson et al., 2006). A number of studies revealed that humans are exposed to arsenic poisoning not only from contaminated drinking water but also from food items grown in and collected from arsenic contaminated sites. Therefore, remediation of arsenic contaminated aquatic systems is important for human health.

Phytoremediation of contaminated water by aquatic macrophytes would be a good option in long term. A large number of aquatic plant species have been tested for the remediation of toxic elements from fresh water systems. Few aquatic plants (mostly macrophytes) have shown the ability to accumulate high level of arsenic from water. Among those aquatic plants, water hyacinth (*E. crassipes*), duckweed (*Lemna*, *Spirodela*, and *Wolffia*), water fern (*Azolla* spp.), Hydrilla (*H. verticillata*), and watercresses (*N. officinale*, *N. microphyllum*) have been proposed to be potential for phytoremediation due to their arsenic hyperaccumulation ability and growth habit. A number of studies revealed that phytoremediation of arsenic using aquatic macrophytes would be a good option to clean polluted water. Even though a number of aquatic plants have been shown high arsenic uptake and suggested to be effective for arsenic phytoremediation, the management and disposal of those phytoremediating aquatic macrophytes is a major concern for the successful implementation f phytoremediation technology.

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Table 1: Important inorganic, organic, and biological forms of arsenic in the environments

Name	Abbreviation	Formula/Structure	Reference
Inorganic arsenicals			
Arsine	As(-III)	AsH_3	*
Arsenious acid or arsenite	As(III)	$As^{3+}(OH)_3$	*, **
Arsenic acid or arsenate	As(V)	$H_3As^{5+}O_4$	*, **
Methylarsenicals			
Methylarsine	-	AsH_2CH_3	*, **
Dimethylarsine	-	$AsH(CH_3)_2$	*, **
Trimethylarsine	-	$As(CH_3)_3$	*, **
Monomethylarsonous acid	MMAA(III)	$As(OH)_2CH_3$	*, **
Dimethylarsinous acid	DMAA(III)	$As(OH)(CH_3)_2$	*, **
Monomethylarsonic acid	MMAA(V)	$AsO(OH)_2CH_3$	*, **
Dimethylarsinic acid	DMAA(V)	$AsO(OH)(CH_3)_2$	*, **
Trimethylarsine oxide	TMAO	$AsO)(CH_3)_3$	*, **
Tetramethylarsonium ion	TMA^+	$As^+(CH_3)_4$	***
Organoarsenicals			
Arsenocholine	AsC	$(CH_3)_3As^+CH_2CH_2O$	**
Arsenobetaine	AsB	$(CH_3)_3As^+CH_2COO^-$	**
Rosarsone	-	$C_6H_6AsNO_6$	**
Arsenosugars			
Dimethylarsinoylribosides	AsS		****
Triaklylarsonioribosides	AsS		

Others			
Dimethylarsinoylethanol	DMAE		*****
Glycerophospho(arsenocholine)	GPAC		
Glycerophosphatidylarsenocholine	-		***

* Maher, 1984; Kaise et al., 1988; Francesconi and Edmonds, 1996; Craig, 2003; Sharma and Sohn, 2009.

** Craig, 2003; O'Day, 2006.

*** O'Day, 2006; Sharma and Sohn, 2009.

**** Francesconi and Edmonds, 1996; Sharma and Sohn, 2009.

***** Francesconi and Edmonds, 1996.

Table 2: Different phytoremediation processes (Vamerali et al., 2010)

Phytoextraction	In this process, plants uptake pollutants from soil and water, and translocate to and store in the harvestable biomass of the plants. Phytoextraction aims to remove pollutants form the contaminated sites. This process is usually observed in hyperaccumulating plants resistant to the pollutants.
Phytostabilization	Plants reduce mobility and phytoavailibility of contaminants in the environment. This process does not remove pollutants from contaminated sites but reduces mobility and excludes metals from plant uptake.
Phytovolatilization	Hyperaccumulating plants uptake pollutants from soil and water, and translocate to the aerial parts of the plants, and volatilize the pollutants in the air.
Phytotransformation	This process is one kind of plant’s defense mechanism to the environmental pollutants. The hyperaccumulating plants modify, inactivate, degrade (phytodegradation), or immobilize (phytostabilization) the pollutants through their metabolism.
Rhizofiltration	Usually aquatic plants perform this process. The hyperaccumulating aquatic plants adsorb and absorb pollutants from aquatic environments (water and wastewater).

Table 3: A number of aquatic plants have been tested for the remediation of trace elements from water. This table summarizes the aquatic plants which have been studied for phytoremediation of trace elements.

Common Name	**Scientific Name**	**Trace Elements**	**References**
Duckweed	*Lemna gibba* L.	As, U, Zn	Fritioff and Greger, 2003; Mkandawire et al., 2004a; Mkandawire et al., 2004b; Mkandawire and Dudel, 2005
Lesser duckweed	*Lemna minor* L.	As, Zn, Cu, Hg	Fritioff and Greger, 2003; Kara, 2004; Miretzky et al., 2004; Robinson et al., 2005; Alvarado et al., 2008; Mishra et al., 2008
Star duckweed	*Lemna trisulca* L.	Zn	Huebert and Shay, 1992
Water hyacinth	*Eichhornia crassipes*	As, Fe, Cu, Zn, Pb, Cd, Cr, Ni, Hg	Wolverton and McDonald, 1978; Muramoto and Oki, 1983; Delgado et al., 1993; Vesk et al., 1999; Cordes et al., 2000; Chandra and Kulshreshtha, 2004; Odjegba and Fasidi, 2007; Alvarado et al., 2008; Espinoza-Quiñones et al., 2008; Junior et al., 2008; Mishra et al., 2008; Dixit and Dhote, 2010
Water-starwort	*Callitriche cophocarpa*	Cr(V)	Augustynowicz et al., 2010
Petries starwort	*Callitriche petriei*	As	Robinson et al., 2005
Common reed	*Phragmites australis*	Cr, Cu, Ni, Pb, S, V, Zn, Cd	Deng et al., 2004; Ghassemzadeh et al., 2008; Baldantoni et al., 2009
Butterfly fern	*Salvinia rotundifolia*	Pb(II)	Banerjee and Sarker, 1997; Dhir, 2009
	Salvinia natans	As, Ni, Cu, Hg(II)	Sen and Mondal, 1987, 1990; Sen and Bhattacharyya, 1993; Rahman et al., 2008c
	Salvinia minima	As, Pb, Cd, Cr	Olguin et al., 2003; Hoffmann et al., 2004; Sanchez-Galvan et al., 2008

	Salvinia herzogii	Cd, Cr	Maine et al., 2004; Suñe et al., 2007
Eared watermoss	*Salvinia auriculata*	Zn, Hg, Cr	Molisani et al., 2006; Espinoza-Quiñones et al., 2008; Wolff et al., 2009
Greater duckweed	*Spirodela intermedia*	Cu, Zn, Mn,Cr, Pb	Miretzky et al., 2004
	Spirodela polyrhiza L.	As, Hg	Rahman et al., 2007; Mishra et al., 2008; Rahman et al., 2008b
Indian/Sacred lotus	*Nelymbium speciosum*	Cr, Cu, Ba, Ti, Co, Pb	Vardanyan and Ingole, 2006
	Ludwigia perennis L.		
Arrowhead	*Sagittaria sagittiflia* L.		
-	*Nymphoides ceristatum*		
Shoreline seapurslane	*Sasuvium portulacastrum* L.		
-	*Nymphae stellata*		
Water spinach	*Ipomoea aquatica*	As, Cd, Pb, Hg, Cu, Zn	Lee et al., 1991; Göthberg et al., 2002; Gothberg et al., 2004; Hu et al., 2008; Wang et al., 2008
Eelgrass/Eelweed	*Vallisneria spiralis L.*	Cu, Cd, Hg	Rai and Tripathi, 2009; Wang et al., 2010
Esthwaite waterweed	*Hydrilla verticillata*	As, Pb, Zn, Cr	Lee et al., 1991; Dixit and Dhote, 2010
Mosquito fern	*Azolla caroliniana*	As	Zhang et al., 2008
Water fern	*Azolla filiculoides* *Azolla pinnata*	As, Hg, Cd	Rahman et al., 2008a; Rai, 2008; Zhang et al., 2008; Rai and Tripathi, 2009
Elephant's ear	*Colocasia esculenta*	Cd, Cu, Pb, Zn	Cardwell et al., 2002
Umbrella sedge	*Cyperus eragrostis*		
Spike rush	*Eleocharis equisitina*		
Parrot's feather	*Myriophyllum aquaticum*		

Miriophyllum	*Myriophyllum propinquum*	As	Robinson et al., 2005
Water lily	*Nymphaea violacea*	Cd, Cu, Pb, Zn	Cardwell et al., 2002
	Nymphaea aurora	Cd	Schor-Fumbarov et al., 2003
Marshwort	*Nymphoides germinata*	Cd, Cu, Pb, Zn	Cardwell et al., 2002
Knotweeds	*Persicaria attenuatum*		
-	*Persicaria orientalis*		
-	*Persicaria subsessilis*		
-	*Potamogeton orchreatus*	As	Robinson et al., 2005
Willow smartweed	*Persicaria lapathifolium*	Cd, Cu, Pb, Zn	Cardwell et al., 2002
-	*Potamogeton javanicus*		
Fennel pondweed	*Potamogeton pectinatus*	Cd, Pb, Cr, Ni, Zn, Cu	Demirezen and Aksoy, 2004
Curled dock	*Rumex crispus*	Cd, Cu, Pb, Zn	Cardwell et al., 2002
River clubrush	*Schoenoplectus validus*		
Cumbungi	*Typha domingensis*		
Cumbung	*Typha orientalis*		
Lesser Bulrush	*Typha angustifolia*	Cd, Pb, Cr, Ni, Zn, Cu	Chandra and Kulshreshtha, 2004; Demirezen and Aksoy, 2004
Bulrush	*Typha latifolia*	Cr, As, Zn, Pb, Cd, Cu. Ni	Ye et al., 1997; Ye et al., 1998; Hozhina et al., 2001; Blute et al., 2004; Chandra and Kulshreshtha, 2004; Deng et al., 2004; Pratas et al., 2007; Sasmaz et al., 2008
Waterweed/Pondweed	*Elodea canadensis*	As, Pb, Cr, Zn, Cu, Cd	Mayes et al., 1977; Mal et al., 2002; Fritioff and Greger, 2003; Chandra and Kulshreshtha, 2004; Robinson et al., 2005; Dogan et al., 2009

Brazilian Waterweed	*Veronica aquatica*	As	Robinson et al., 2005
Water-milfoil	*Myriophyllum spicatum*	Co, Cr, Cu, Pb, Zn, Ni	Keskinkan et al., 2003; Chandra and Kulshreshtha, 2004; Lesage et al., 2007
Fragrant water lily	*Nymphaea odorata*	Cr	Chandra and Kulshreshtha, 2004
Pickerelweed	*Pontederia cordata*		
Tape grass/Eel grass	*Vallisneria spiralis*	Hg	Gupta and Chandra, 1998
Wild celery	*Vallisneria americana*	Cr	Chandra and Kulshreshtha, 2004
-	*Nymphaea spontanea*	Cr(VI)	Choo et al., 2006
Shichito matgrass	*Cyperus malaccensis* Lam.	Pb, Zn, Cu, Cd	Deng et al., 2004
Swamp rice grass	*Leersia hexandra* Swartz.		
Burma reed	*Neyraudia reynaudiana*		
Flagroot	*Acorus calamus* L.		
-	*Eleocharis valleculosa*		
Water pepper	*Polygonum hydropiper*	As	Robinson et al., 2005
Reed canary grass	*Phalaris arundinacea* L.	Pb, Zn, Cu, Cd	Deng et al., 2004
-	*Equisetum ramosisti* Desf.		
Soft rush	*Juncus effusus* L.		
-	*Polypogon fugax* Steud.		
-	*Egeria densa*	As	Robinson et al., 2005
Alligatorweed	*Althernanthera philoxeroides*	As, Pb	Elayan, 1999
Water lettuce	*Pistia stratiotes*	As, Cr, Pb, Ag, Cd, Cu, Hg,	Lee et al., 1991; Maine et al., 2004; Miretzky et al., 2004; Espinoza-

		Ni, Zn	Quiñones et al., 2008; Espinoza-Quiñones et al., 2009
Floating pondweed	*Potamogeton natans*	Zn, Cu, Cd, Pb	Fritioff and Greger, 2003, 2006
Willow moss	*Fontinalis antipyretica*	Cu, Zn	Goncalves and Boaventura, 1998) (Martins and Boaventura, 2002
Needle spikerush	*Eleocharis acicularis*	As, In, Ag, Pb, Cu, Cd, Zn, Sb, Ni, Mg	Ha et al., 2009a; Ha et al., 2009b; Ha et al., 2011
Rigid hornwort	*Ceratophyllum demersum*	As, Pb, Zn, Cu	Keskinkan et al., 2004; Robinson et al., 2005
Watercress	*Lepidium sativum* L.	As	Robinson et al., 2003
-	*Najas indica*	Pb	Singh et al., 2010
Watercresses	*Nasturtium officinale*	Cu, Zn, Ni	Kara, 2005
Curly waterweed	*Lagarosiphon major*	As	Robinson et al., 2005

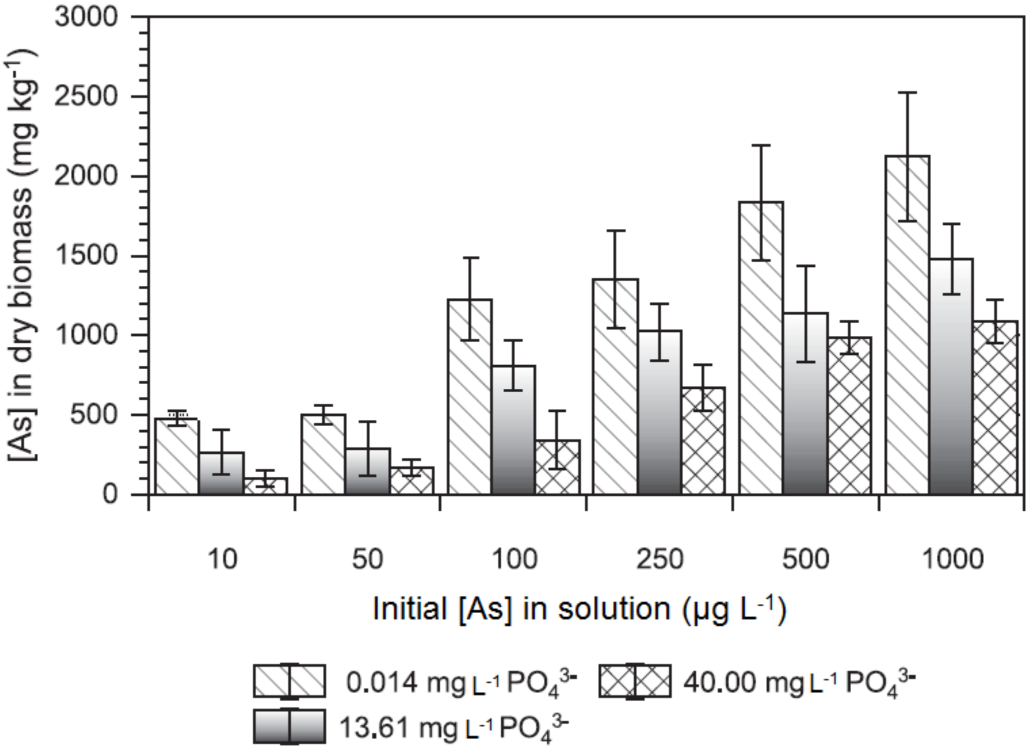

Fig. 1: Influence of phosphate concentrations on arsenic uptake in *L. gibba* L. (Mkandawire and Dudel, 2005).

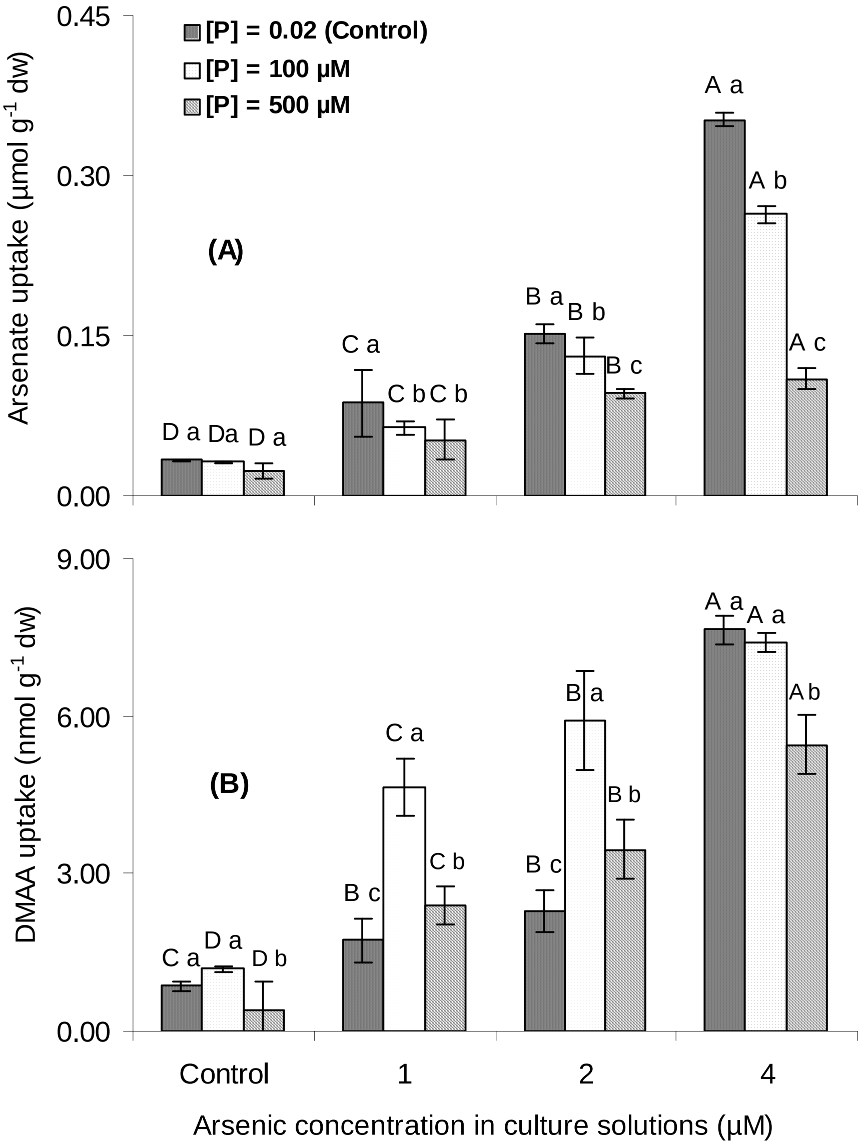

Fig. 2: Arsenic uptake in *S. polyrhiza* L. exposed to the different concentrations of As(V) (a) and DMAA (b). Error bars represent ±SD ($n = 3$). Different capital letters indicate significant differences between arsenic treatments and small letters indicate significant differences between phosphate treatments at 5% level (Rahman et al., 2007).

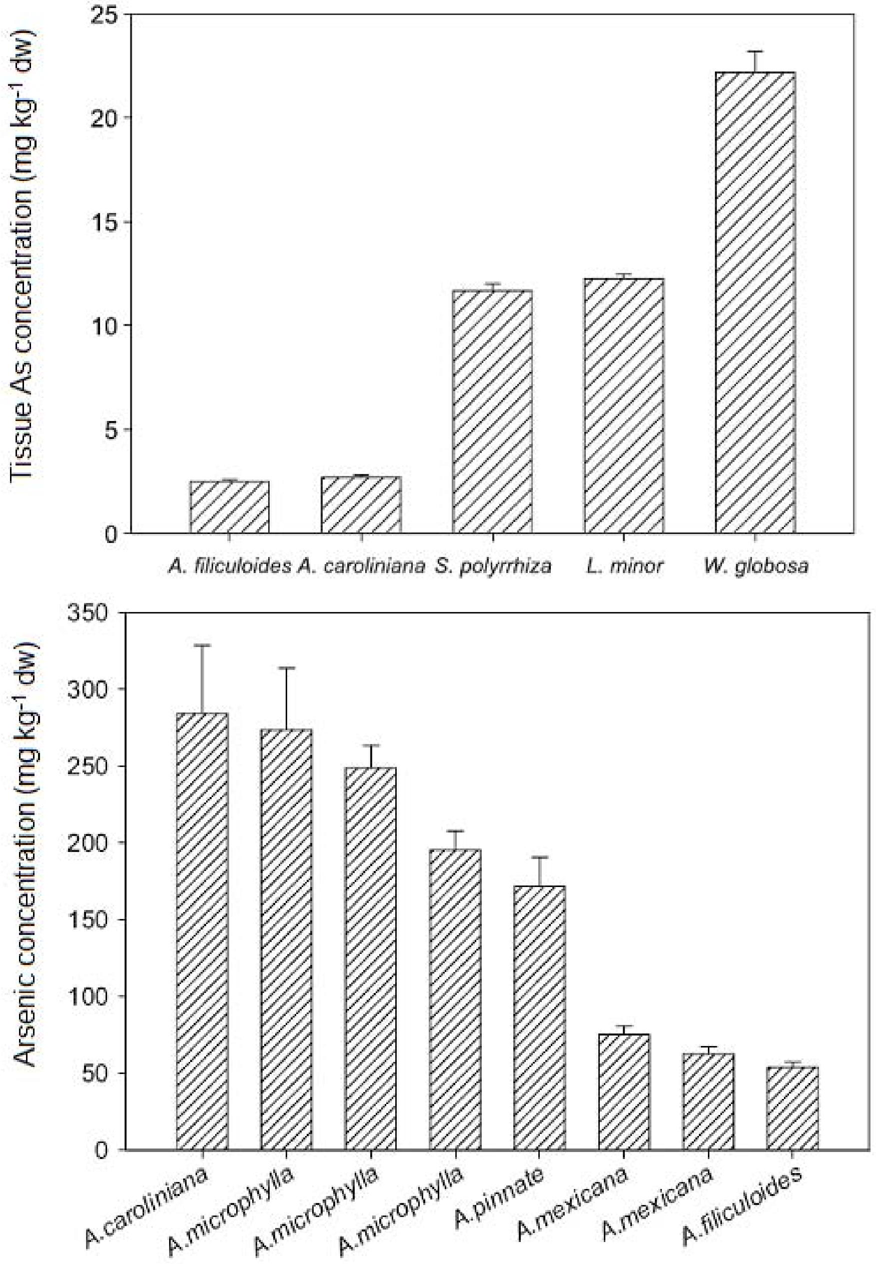

Fig. 3: Arsenic concentration in duckweed (*S. polyrhiza*, *L. minor* and *W. globosa*) and *Azolla* grown in nutrient solutions with 1 µM and 50 µM As(V) for 5 and 10 d, respectively. Data are means ± SE (*n* = 3) (Zhang et al., 2008; Zhang et al., 2009).

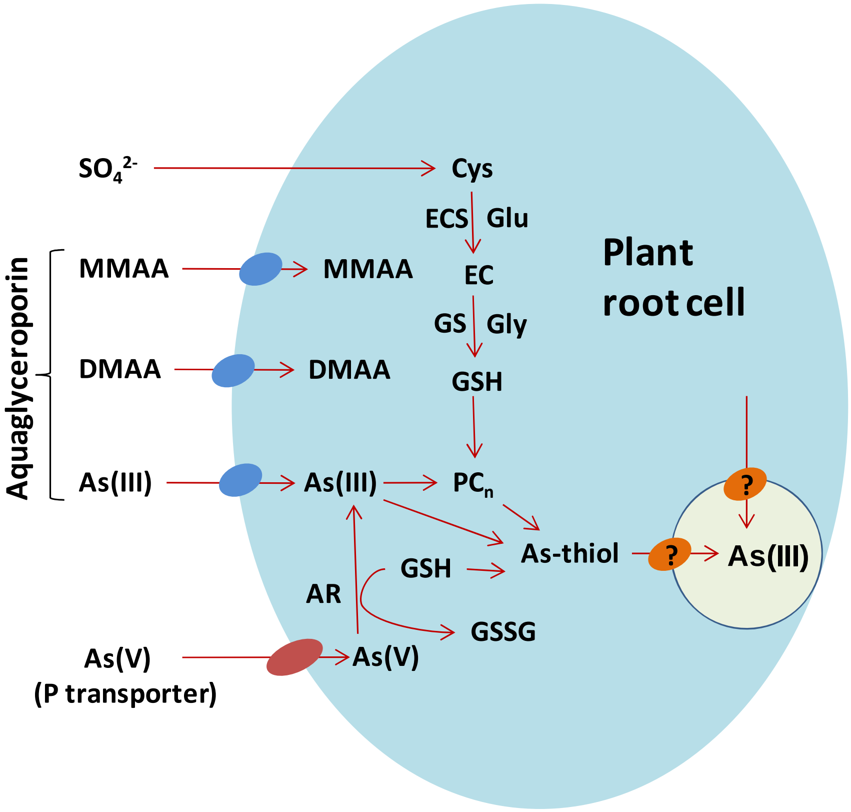

Fig. 4: Mechanisms of arsenic uptake into plant cells (Tripathi et al., 2007). As(V) is transported through phosphate transporters, and As(III) through aquaglyceroporins (Tripathi et al., 2007; Rahman et al., 2008b; Zhao et al., 2009). The organoarsenic species (MMAA and DMAA) might be taken up by aquaglyceroporins (Rahman et al., 2011).