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REVIEW PAPER

Groundwater arsenic contamination: impacts on human health and agriculture, ex situ treatment techniques and alleviation

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Abstract Groundwater is consumed by a large number of people as their primary source of drinking water globally. Among all the countries worldwide, nations in South Asia, particularly India and Bangladesh, have severe problem of groundwater arsenic (As) contamination so are on our primary focus in this study. The objective of this review study is to provide a viewpoint about the source of As, the effect of As on human health and agriculture, and available treatment technologies for the removal of As from water. The source of As can be either natural or anthropogenic and exposure mediums can either be air, drinking water, or food. As-polluted groundwater may lead to a reduction in crop yield and quality as As enters the food chain and disrupts it. Chronic As exposure through drinking water is highly associated with the disruption of many internal systems and organs in the human body including cardiovascular, respiratory, nervous, and endocrine systems, soft organs, and skin. We have critically reviewed a complete spectrum of the available ex situ technologies for As removal including oxidation, coagulation–flocculation, adsorption, ion exchange, and membrane process. Along with that, pros and cons of different techniques have also been scrutinized on the basis of past literatures reported. Among all the conventional techniques, coagulation is the most efficient technique, and considering the advanced and emerging techniques, electrocoagulation is the most prominent option to be adopted. At last, we have proposed some mitigation strategies to be followed with few long and short-term ideas which can be adopted to overcome this epidemic.

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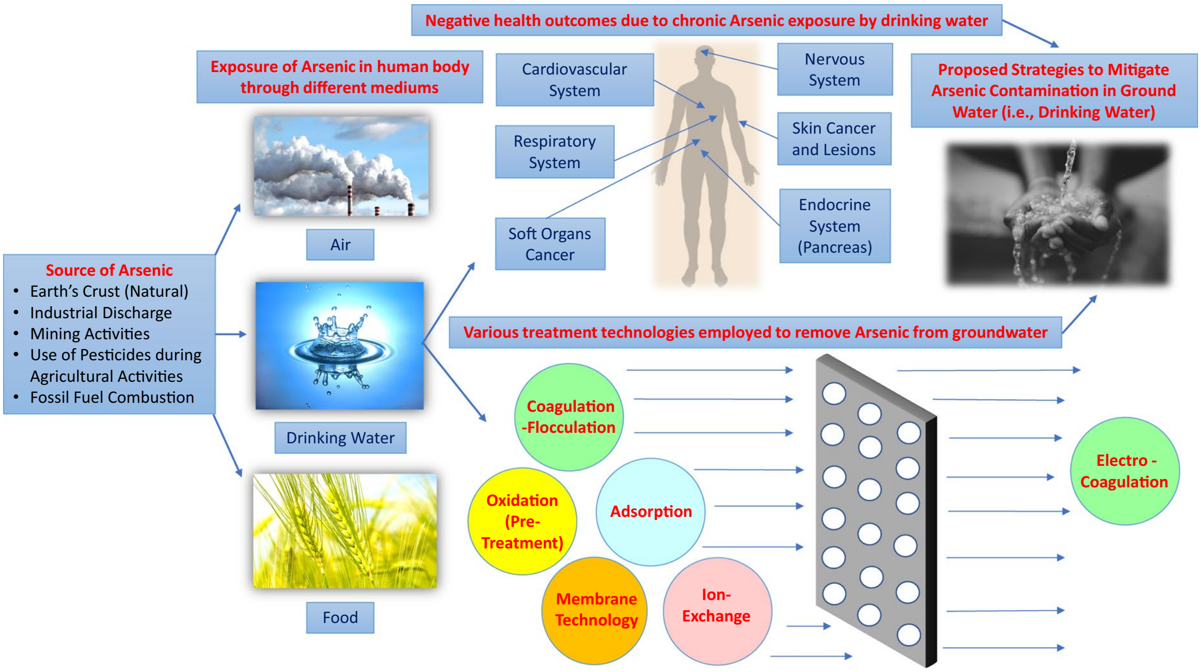
Graphical abstract



Keywords Drinking water · Water pollution · Sources of pollution · Decontamination · Mitigation strategies

Abbreviations

AOPs	Advanced oxidation process
As	Arsenic
BET	Brunauer–Emmett–Teller
COD	Chemical oxygen demand
DC	Direct current
EC	Electrocoagulations
GIS	Geographic information system
IS	Indian standard
IARC	International agency of research on cancer
PRB	Permeable reactive barrier
PDS	Peroxy-disulphate
PMS	Peroxy-monosulfate
UV	Ultraviolet
WHO	World Health Organization
ZVI	Zero valent iron

Introduction

Approximately 3% of the water available on earth is freshwater, of which 83% is locked in icecaps and glaciers, leaving just 17% accessible in the form of utilizable freshwater, and only 30% of this utilizable freshwater is groundwater. Deforestation, mining, industrialization, urbanization, over-exploitation and direct anthropogenic contamination are significant modifications of the surface land cover which has set groundwater under threat. The statistics show that about 14% of the world's population is unable to obtain clean drinking water and 32% is unable to use a suitable sanitation facility. Around 5 million people die due to water-borne diseases (i.e., typhoid, cholera, hepatitis A, etc.) in a year, and it is projected that the majority of the global citizens by 2025 are to be expected to reside in nations where water scarcity is mild to high (Azizullah et al., 2011).

The existence of heavy metals in groundwater has been a growing matter of concern as it causes a deleterious impact on human health. Numerous metals are vital for individuals in small amounts, but in excess, they can pollute water and lead to serious health complications for human beings (Azizullah et al., 2011; Raza et al., 2017). Heavy metals are chemical elements having a specific density greater than 5 g/ml, toxic at low concentrations, found in the earth's crust as naturally occurring elements that cannot be degraded in any way and include cadmium (Cd), mercury (Hg), thallium (Tl), chromium (Cr), arsenic (As), and lead (Pb) (Sharma et al., 2020). Arsenic has been classified as a category I cancer-causing substance among human beings by the International Agency of Research on Cancer (IARC), indicating that there is ample evidence of carcinogenicity in humans (Martinez et al., 2011). Arsenic is not present as a free element in nature but found as a compound of different elements like hydrogen, carbon, oxygen, chlorine, sulfur, lead, gold, mercury, and iron (Fazal et al., 2001). In numerous nations throughout the world, and especially in Asia, arsenic is considered a hazard to public health (Azizullah et al., 2011; Haque et al., 2008). Currently, 107 nations have recorded medium to excessive levels of As in groundwater, putting 296 million people at possible risk from As ingestion through various sources of exposure (Chakrabarti et al., 2019; Ghosh et al., 2019). In India, groundwater of a total of 718 districts is contaminated with one or more than one contaminant including nitrate, fluoride, Fe, As, Pb, Cr, and Cd across different states and the states like Punjab, Tamil Nadu, Telangana, Uttar Pradesh, Haryana, and West Bengal have most polluted groundwater across the country. To assess the level and severity of As contamination in groundwater, a thorough analysis performed in Nadia, West Bengal (India), found that all blocks under investigation contained As concentration more than 50 µg/l with the maximum of 3200 µg/l and they discovered As concentrations of more than 10 and 50 µg/l in 51.4% and 17.3% of tube wells, respectively, after analyzing the samples collected from different tube wells (Rahman et al., 2014).

According to a study (Arain et al., 2009; Azizullah et al., 2011), one-third of people in an investigated area had rugged skin with black spots and arsenical skin lesions on some of the body parts, due to exposure to greater As levels and other conditions such as starvation, poverty, a lack of good medical services, and the prevalence of other pollutants functioning in coordination with As surplus. One more related research was carried out in which 285 subjects were sampled, off these two-third population suffered from long-term As toxicity. They found a close connection among drinking water As concentration and blood and hair samples of diseased patients with exposed skin. They suffered from several conditions such as respiratory disorders, stomach disorders, and fatigue. There were no such issues with people who drank municipal water having low As concentration (Azizullah et al., 2011; Kazi et al., 2009). A study carried out by Rahman et al. (2014) investigated several samples from different villagers and concluded that many villagers had arsenical skin lesions, and majority of the sampled concentration exceeded the toxic level.

The method of treating contaminated groundwater by eliminating toxic chemicals or turning them into non-dangerous chemicals is known as groundwater remediation (Concetta Tomei & Daugulis, 2013; Sharma et al., 2020). There are various traditional physical and chemical methods available for As elimination from groundwater, both in the laboratory and in the field such as adsorption, membrane-based technology, chemical precipitation, and ion exchange (Pal et al., 2009). Other non-traditional and alternative treatment technologies includes electrocoagulation (EC), advanced oxidation processes (AOPs), ultraviolet (UV) oxidation, microbial oxidation, bioelectrochemical systems, and many more. Due to high technological performance, low operating cost, low maintenance, and less energy demand, permeable reactive barrier (PRB) is considered as most favorable in situ remediation technique and is one of the revolutionary methods. This is an environment-friendly remediation technology for contaminated groundwater, which works by deployment of a porous barricade that carries reactive chemicals to capture and treat pollutants as the groundwater moves through it (Luo et al., 2016; Obiri-Nyarko et al., 2014; Park et al., 2002; Robertson et al., 2008). An experiment was carried out by Lee et al. (2009) to assess long-term degradation of As compounds with the help of Zero Valent Ion (ZVI) using the PRB technique in which they took samples at different intervals of time for 360 days. They showed that the rate of total As removal reduced by 45% and 39% under aerobic and anoxic conditions, respectively, due to saturation and concluded their experiment by suggesting that for effective As elimination, the reactive substance should be changed regularly.

Humans are exposed to As throughout their entire life span which is carcinogenic. Arsenic can get into the human body through various ways, typically, by drinking water or via dietary sources. Arsenic can act as a poisonous agent and therapeutic for human beings (Bhardwaj et al., 2018). Long-term exposure to As can be fatal to the human body, so studying the source of As and treating it appropriately can cure long-term exposure to the human body. The study provides complete review of As pollution and its negative health effects in the human body, treatment facilities, and mitigation strategies.

According to the authors' best knowledge, there is presently no literature available that contains comprehensive information on As, starting with the source of pollution, human health and agricultural implications, remediation techniques, and recommended mitigation measures all in one article. The study's main goal is to present a complete overview of work on As pollution, from its sources to its solutions, in a structured manner. One of our main focus is to aware researchers worldwide about the As contamination and also to emphasize solutions at the same time and that's why mitigation strategies are one of the key focuses in this article that has not been addressed on such scale yet in addition with its origin to its end.

This investigation concentrates on India as a geography, with very slight focus on few neighboring nations. This article provides an intriguing overview of As-polluted groundwater, its impact on healthcare and agriculture, and highlights a good number of investigations on treatment processes that have been carried out in the field as well as countermeasures, helping readers to build a solid understanding.

Sources of arsenic contamination

Arsenic is a trace metalloid substance found in the earth's crust. Arsenic-contaminated groundwater is attributed to one or more natural and anthropogenic causes. In nature, the presence of As in water bodies was reported as low as a few mg/l, and levels in soils can range from 0.01 mg/kg to some hundred mg/kg and it is found in nature in various oxidation states like +5, +3, 0 and −3 which are known as arsenate, arsenite, arsenic, and arsine, respectively (Bissen & Frimmel, 2003; Mitra et al., 2017). In addition, As can be seen in the form of sulfides and oxides in nature. Anthropogenic source of water and soil arsenic pollution includes the inappropriate production, utilization, and discard of As-containing industrial chemicals, widespread use of As-containing pesticides and phosphate fertilizers, emissions and drainage from smelting and mining of As-containing minerals, fossil fuel combustion, bacterial decomposition of natural or synthetic organic chemicals, etc. (Bissen & Frimmel, 2003; Henke, 2009). In the environment, there are more than 150 different As-containing minerals in which As is mostly found in sulfide minerals, among them three composites including Realgar or arsenic disulfide (As_2S_2), orpiment or arsenic trisulfide (As_2S_3), and arsenopyrite or ferrous arsenic sulfide (FeAsS) are known as the arsenic ore because they are readily available compared to other As composites with higher As concentration (Fazal et al., 2001). Arsenopyrite is the most common As ore mineral, and it is commonly accepted that Arsenopyrite, along with the other major As sulfide minerals realgar and orpiment, is only produced under elevated temperature conditions on the earth's surface. According to the chemical constituents, As is divided into two types—organic (less toxic) and inorganic (more toxic), in which inorganic As is further distinguished by two types—trivalent and pentavalent, and among them trivalent As is considered to be more toxic and more difficult to remove from water than pentavalent As (Fazal et al., 2001). Because of the greater retention stability of As species in the corresponding ore, there is less likelihood of As contamination in groundwater and surface water (Pal et al., 2009). Kumar et al. (2016) calculated the safe level of As in drinking water and different food ingredients to assess the possible health risk in As-polluted areas in Bihar, India. This study discovered that drinking water contributed to two-third of the overall estimated daily dietary consumption of 169 µg As per day, with the rest coming from the food intake. Since crude oil is extracted from the underground reservoirs, their end products (i.e., diesel, gasoline, kerosene, jet fuel, etc.) also contain small amounts of As. An experimental study carried out by De Jesus et al. (2012) showed that the presence of As in crude oil samples was about 0.214 mg/kg (maximum out of 7 samples). So, combustion products of crude oil may lead to emission of As in the atmosphere. Coal also contains a small quantity of arsenic as it is a fossil resource and can be mined either on the surface or underground due to which the combustion of coal can result in the release of As into the atmosphere. Ordinary bituminous and lignite coals have mean As concentrations of 9.0 and 7.4 ppm at the global level, respectively, and the global highest As concentration is 50 and 49 ppm, respectively (Pandey et al., 2011; Yudovich & Ketris, 2005). Moreover, if we consider this in the Indian scenario, the As content of Indian bituminous coal ranges from 22.3 to 62.5 ppm (Pandey et al., 2011). Fly ash is the by-product of the coal-based thermal power plant, generated by the combustion of coal, which represents a possible anthropogenic source of As threats. Arsenic levels in fly ash usually vary from 2 to 440 mg/kg, but they might reach 1000 mg/kg based on the composition of the coal used and the method of combustion (Pandey et al., 2011).

Impacts of As on human health

Many citizens around the globe use groundwater as a means of drinking water. According to IS 10500 (2012), the acceptable limit for the total As content in the drinking water is set as a maximum of 0.01 mg/l, and some relaxation has been given in the case of the absence of an alternative source of drinking water in which the permissible limit for the total As content is set as a maximum of 0.05 mg/l. Additionally, the provisional guideline value given by World Health Organization (WHO) for As in drinking water is also 0.01 mg/l (or 10 μg/l). Toxicity and carcinogenic effects of As are determined by the oxidation states and chemical forms in which it exists (Hong et al., 2014; Rasheed et al., 2016). Arsenic can enter the body via several pathways, including inhalation, ingestion, and dermal interaction. Depending on the time of exposure, the effect of As is classified into two categories, acute and chronic effects. Humans who are exposed to As for extended periods face chronic health issues (Hong et al., 2014; Rasheed et al., 2016). While acute toxicity is usually reversible and happens when more than a sufficient amount of As is consumed in a short period, chronic toxicity occurs when As is ingested by polluted drinking water for a prolonged period. The most usual health concern correlated with As toxicity is arsenical keratosis and skin cancer (Jha et al., 2017). Acute exposure to As has no consequences if the dosage is smaller than 0.03 mg/kg of body weight/day, little consequence if the dosage is in between 0.03 and 0.3 mg/kg of body weight/day, produces consequences of illness if the dosage is in between 0.3 and 0.5 mg/kg of body weight/day, and fatal if the dosage is more than 1 mg/kg of body weight/day (Jha et al., 2017). According to Farmer and Johnson (1990), the human body holds between 40 and 60% of the As consumed in drinking water, so the more As-containing water we ingest, the higher is the consequent risk. In areas where people used to eat fish as a source of food, they could have been exposed to As from the fish, and in that situation, the magnitude of the issue and possible health effects are determined by measuring the bioaccumulation factor of an analyte, which correlates the concentration of the analyte under examination in fish and water bodies. An experiment conducted by Kar et al. (2011) investigated three major As species namely arsenite, arsenate, and dimethyl arsenic in a total of 9 samples. The samples were collected from the three different body parts (tissue, bone, and head) of two types of freshwater fish (shrimp and tilapia) from three distinct ponds of Taiwan. The results showed maximum bioaccumulation factor of 22, and out of three As species, dimethyl arsenic was found to be present in a higher amount compared to the inorganic As species (arsenite and arsenate). This may be dangerous to people of the surrounding areas who eat fish regularly and are more likely exposed to As by ingestion. A review conducted by Bhowmick et al. (2018) and Guha Mazumder and Dasgupta (2011) concluded the fact that high levels of As in the environment can have both carcinogenic and non-carcinogenic effects. Respiratory illness, gastrointestinal affliction, renal failure, cardiovascular disorder, neurotoxicity, negative hematological consequences, diabetes mellitus, adverse birth complications, and neonate mortality are examples of non-carcinogenic impacts. An experimental study carried out by Mazumder et al. (2012) shows that in respondents residing in As-contaminated areas, there is a coordination between chronic As toxicity and the probability of development of hypertension. Chronic As toxicity has been directly linked to the occurrence of cardiovascular diseases (Henke, 2009; Navas-Acien et al., 2005; Wang et al., 2007). A correlation between As exposure and skin cancer has been studied by many researchers. One of the study shows that UV-caused skin cancers have been directly linked with exposure to trivalent arsenite (Hong et al., 2014; Rossman et al., 2004). Furthermore, several researchers have examined the dose-dependent correlation between As ingestion and the prevalence of lung cancer (Ferreccio et al., 2000; Smith et al., 2006). In a research carried out in Taiwan, individuals who had drunk high-arsenic-polluted drinking water for the past 5 decades had an increased death rate; lower As levels of drinking water resulted in lower lung cancer death rates (Chiu et al., 2004; Hong et al., 2014). A study carried out by Chen et al. (2004) found better correlation between tobacco smoking and exposure to As, which shows that tobacco smoking coupled with As toxicity raises the probability of lung cancer, and vice versa. A new investigation in the Yatenga province, Burkina Faso, tested 240 people from 20 different villages for arsenical skin lesions in which As levels of tube-well water ranged from 1 to 124 µg/l. The concentration of As was higher than the WHO recommendation value of 10 µg/l in more than 50% of the tube wells. Among the tested people, melanosis and keratosis were found in 29.3% and 46.3% of the population respectively during clinical tests; prevalence of skin lesions was linked to As concentrations in tube-well water (Chakrabarti et al., 2019). Long-term exposure to As may have negative impacts on the result of pregnancy such as premature abortion, neonatal death, miscarriage or stillbirth, and lower birth weight (Bhowmick et al., 2018; Quansah et al., 2015). "Arsenicosis" is a disease that occurs as a result of prolonged As toxicity which appears in the form of skin lesions including melanosis, keratosis, and leucomelanosis (Shaji et al., 2021). Arsenicosis can be divided into four phases in which the first phase is called the preclinical phase because it occurs when the patient has no visible signs but As is found in urine or body tissues (Choong et al., 2007). The second phase usually needs nearly about half to 1 decade of As exposure and is referred to as a clinical phase. At this phase, the skin has a variety of consequences among which melanosis (skin pigmentation) is perhaps the most usual symptom and keratosis (skin hardening) is a more severe symptom. The third phase is known by complications that include damage to inner organs, and the last phase is known as a malignancy that comprises cancers or tumors that damages different organs or skin. An investigation found that As exposed people had a higher chance of evolving diabetes in their preliminary phase of young age, and the condition is much worse for young adults having arsenicosis, with a frequency of more than double in comparison with the young adults having no exposure (Khan et al., 2016).

Figure 1 shows the destructive human health outcomes because of the presence of As in drinking water for a prolonged period, whereas no medicines have been discovered which can treat chronic As poisoning. Only unpolluted drinking water, healthy food, and physical activities (i.e., exercise) can be utilized

Fig. 1 Negative health outcomes on different organs due to chronic As exposure

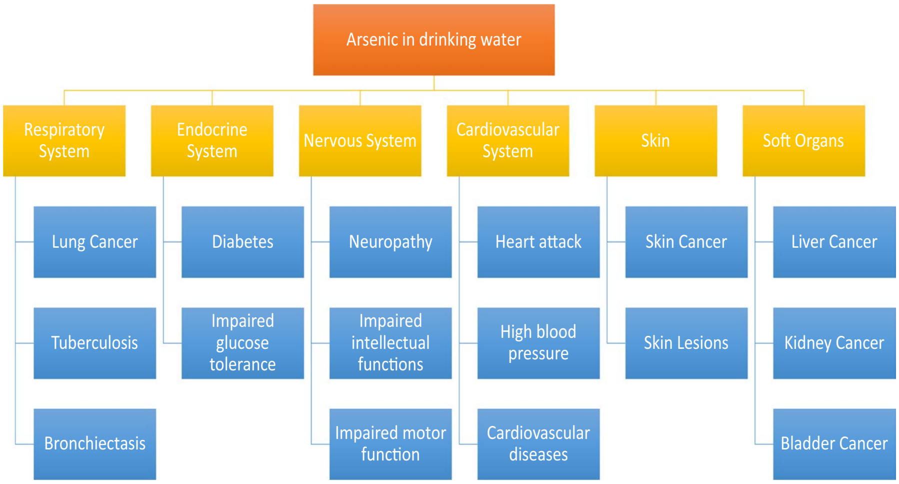


Fig. 2 The organs and systems of human body vulnerable by long-term As exposure

to combat this epidemic (Chakraborti et al., 2013). Figure 2 shows long-term exposure of As on different organs and system of human body.

Impact of As on agriculture sector

The majority of the crops (i.e., wheat, rice, pulses, sugarcane, etc.) and different types of vegetation grown in developed countries are highly vulnerable to As-contaminated water (Gong et al., 2020). The region close to the Indo-Gangetic delta and West Bengal (India) are having a major impact on As enrichment where As is washed up and carried throughout the Ganga river and finally ends up deposited in the sedimentary basin (Sharma et al., 2020). It further settles down to the groundwater bed and contaminates groundwater, which is further used for irrigation and drinking purposes (Pal et al., 2009). The contaminated groundwater leads to a decrease in crop productivity and quality (food chain contamination) especially of rice, as it is grown with irrigation water, thus resulting in accumulation by ten times more than other cereals. Some of the literature shows that As accumulation is highest in roots followed by stem, leaf and at last grains of the crops (Kundu et al., 2013; Senanayake & Mukherji, 2014). Crops that grow below the surface (i.e., carrot, potatoes) are highly exposed to As-contaminated groundwater, and the crops like rice have a high tendency to absorb As and transport through the food chain. Arsenic concentration in plants varied from 0.01 to 5.0 mg/kg worldwide (Mandal & Suzuki, 2002; Pal et al., 2009). Consumption of milled rice (staple food) is estimated at around 480 million tons annually (Muthayya et al., 2014). Water consumed by rice crops is around 750 mm which mostly comes from groundwater, rain and surface water (Aryal, 2013). Arsenic contaminated water gets accumulated on the outer bran layer surrounding the endosperm of the rice. Although cooking rice at boiling temperature can also reduce As content absorbed by rice grain (Menon et al., 2021). 75–90% of nutrient depletion is caused due to accumulation of As in bran (Upadhyay et al., 2019). WHO's permissible limit for As concentration in rice is 0.3 mg/kg (Gu et al., 2020). The inflated toxicity of the soil not only result in As enrichment of the outer layer of the soil but also reflected in the roots and stem of the plant (Abedin et al., 2002). Wheat crops exposed to As-contaminated water also behaved

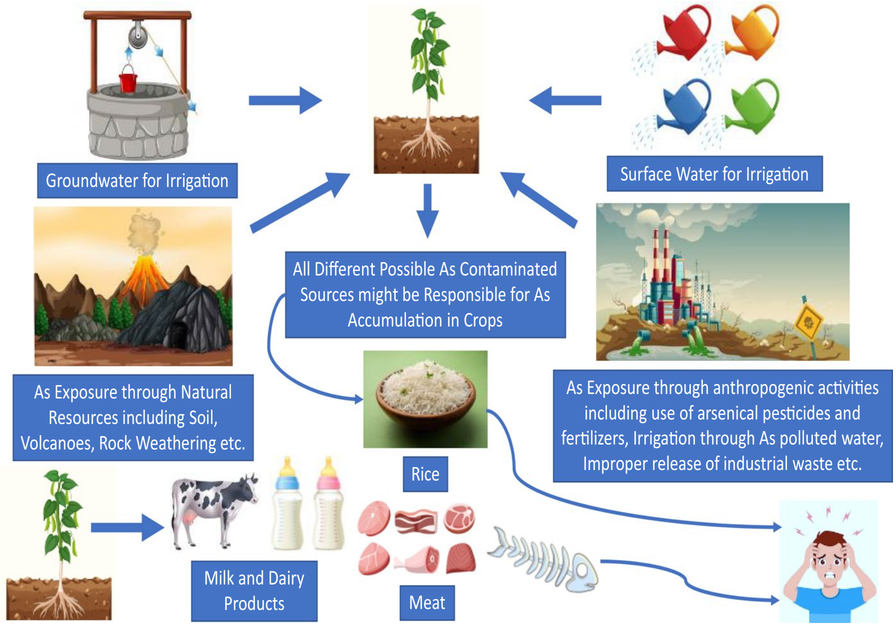


Fig. 3 Fate and transport of As in the food chain

similar to rice accumulation. Root development, biomass, germination and cell expansion, nutrients and uptake of water, respiration process, and photosynthesis showed significant variation under As stress which may lead to lower yield and productivity (Saeed et al., 2021). Its concentration on the wheat grain was found between 0.01 and 0.19 mg/kg (Bhattacharya et al., 2010). As can easily enter our food chain through our food and can lead to various diseases and health problems (Rai et al., 2019). Figure 3 shows fate and transport of arsenic in human food chain.

Treatment technologies for As remediation

In recent years, comprehensive research has been carried out on As elimination methodologies from

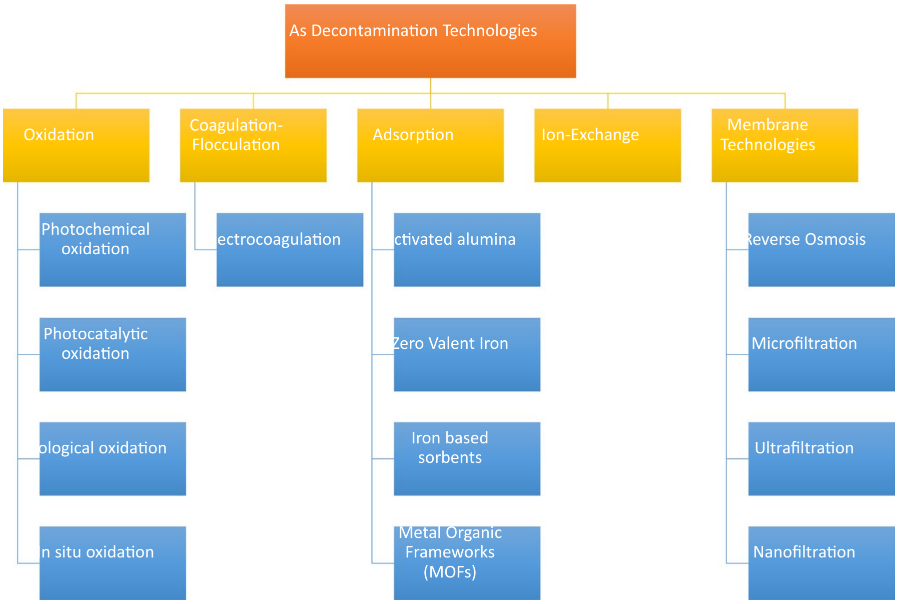


Fig. 4 Different technologies used to remove As from groundwater

groundwater through experiments in laboratory and field. Chemistry and presence of different constituents in the As-polluted water are the two most important considerations in deciding the removal of As (Nicomel et al., 2015; Singh et al., 2015). To remove the pollutant, scientists and researchers use a variety of approaches and processes with an emphasis on separation techniques including adsorption, membrane-based technology, coagulation–flocculation, and ion exchange (Chatterjee et al., 2017). Since As can't be degraded, it can either be transformed from an aqueous phase to a non-aqueous phase (i.e., solid phase as the most suitable phase) or removed from the water directly by other processes. Only after transforming arsenite to arsenate, it can be removed efficiently (Nidheesh & Singh, 2017). Technologies for As-contaminated groundwater treatment are considered more efficient with the use of a two-step process involving initial arsenite to arsenate oxidation, accompanied by an arsenate removal method (Nicomel et al., 2015; Pous et al., 2015). Additionally, toxicity of arsenite makes the pre-oxidation step essential, which is much greater than arsenate. The oxidation method involves different chemical oxidants including Ozone (O_3), Peroxy-monosulfate (PMS), hydrogen peroxide (H_2O_2), peroxy-disulfate (PDS), potassium permanganate ($KMnO_4$), etc. It is not mandatory that an approach or a process that works well in one location also works well in others and the selection of the technology is also an essential factor because every approach or process get their own advantages and disadvantages. Various treatment technologies for the removal of As from groundwater are depicted in Fig. 4.

Oxidation

A study conducted by Gude et al. (2018) showed that pre-oxidation dosage decreases arsenite and increases arsenate concentration before passing through the sand filter for further treatment. This method was used in the Netherlands to achieve 1 µg/l of arsenite in drinking water by pre-oxidation treatment followed by the sand filtration. In this experimental study, Natural Groundwater was used for treatment to remove arsenite. The flow rate used for all experiments was set to 1 ml/min. Manganese was used as an oxidizing agent at 8 mg/l of concentration. The concentration of arsenite was 135 µg/l which decreases to 9 µg/l after treatment. Complete removal of arsenite by oxidation was achieved within 22 days in the fresh sand filter.

As can also be removed by oxidation using solar energy. The experiment was executed by O'Farrell et al. (2016) with the aim to reduce the arsenate level to 10 µg/l by using solar energy. The sample used in the experiment was prepared in the laboratory. The initial concentration of arsenate in the sample was 1000 µg/l. The solar reactor (compound parabolic collector) consists of 10 parallel borosilicate glass tubes connected with opaque plastic fitting to form a continuous flow in the pipe. The detention time for oxidation was 8 h. Fe was also added at 10 mg/l to get optimum results. The use of UV radiation was found to be a minimum of 7.5 kJUV/l. The coagulants used in the experiment were alum and *Moringa oleifera* (drumstick tree). Activated *Moringa oleifera* for As removal through adsorption mechanism was studied by Sumathi and Alagumuthu (2014). The concentration of both coagulants was 20 mg/l. The flocculant added to the sample was potato starch, and the concentration for the same was 20 mg/l. After treatment, the concentration of arsenate was reduced to 10 µg/l.

An experiment was performed by Bora et al. (2016) to remove arsenite from the solution prepared in the laboratory having a concentration of 100 µg/l. During the experiments, the initial pH was adjusted to 8.3. The oxidant used for treatment was $KMnO_4$. The $KMnO_4$ solution was added in varying amount from 5 to 100% of Fe^{2+}. The following values: 80%, 85%, 90%, 93%, 95% of $KMnO_4$, are equivalent to 18.85 mg/l, 20 mg/l, 21 mg/l, 21.88 mg/l, 22.16 mg/l of initial $KMnO_4$ concentration, respectively, where initial concentration of Fe^{2+} was 25 mg/l. The source of Fe^{2+} was ferric sulfate, and $FeCl_3$ was added as a coagulant. The experiment showed that dose of $KMnO_4$ about 80–95% gives better results for removal. The study found that arsenite concentration reduces to 2 µg/l after the oxidation–coagulation process at optimized pH of 8.3. The decrement in Fe concentration was also noted after the oxidation–coagulation process.

An investigation was carried out to oxidize As(III) by using H_2O_2 as a catalyst and adsorb As(V) in the hybrid oxidation and adsorption process (Su et al., 2021). The initial concentration of the synthetic As(III) sample was 20 mg/l and dose of MnL (Mn-doped lanthanum oxycarbonate with 5.26% Mn), and H_2O_2 were 200 mg/l and 0.534 mM correspondingly.

Experiments were carried out at room temperature (25 ± 2 °C) and different pH of 3, 5, 7, 9, and 11. In H_2O_2–MnL combined system, highest removal efficiency of 88% was observed at pH 7, and pH of 5 and 9, around 70% As(III) was removed within 6 h. As a result, Mn-doped $La_2O_2CO_3$ was proved to be efficient in combination with H_2O_2 and also can be used as a catalyst for oxidation of As(III) at a very wide range of pH (5–9), and As(III) can be reduced up to 10 µg/l. The wide working range of pH considerably expands the range of applications for this oxidant. In this pH range, concentration of As(V) increased for first 30 min and then reduced moderately confirms that As was removed by both the oxidation of As(III) and adsorption of As(V).

Coagulation

Due to the low-cost and high-efficiency treatment, removal of As from the contaminated groundwater using adsorption and coagulations is more promising. Coagulations treatment of contaminated water involves adding coagulant agents (Fe^{3+} and Al^{3+}) which helps in formation of positively charged flocs (Tolkou & Zouboulis, 2020). The coagulation process with neutral pH value in high As water is the best low costing and efficient treatment technique (Song et al., 2006). The study was performed using a few samples of the underground mine drainage system. Using ferric ions as a coagulant, authors were able to remove 99% of the As from the acidic water, whereas the removal of As was not so promising in alkaline water. 100 mg/l of ferric sulfate was used as a coagulant agent. Addition 2.5 gm/l coarse calcite as a coagulant increased the removing efficiency of As in alkaline water by 20–30% resulting in 99% removal from the residual As concentration of 5 mg/l at pH 6.

A study was conducted by Khan et al. (2002) in which they collected As-contaminated groundwater samples from 28 different tube wells. Samples were stirred at 50 rpm for 180 s before mixing with coagulant agents (i.e., ferric chloride and alum) under various pH conditions (pH range of 4–9). Addition of 200 ppm of ferric chloride at pH 7–8 led to 91% of As removal, whereas inclusion of 25–50 ppm alum to 81% As removal. The study concluded ferric chloride was a better coagulant agent than alum due to ion inclination in the ferrous compound (Bina et al., 2013).

Bina et al. (2013) focused on removal of arsenite and arsenate species in groundwater. Arsenite has more toxicity rate than arsenate due to a slower excretion rate (Kuivenhoven & Mason, 2019). During initial experiments, $FeCl_3$ (60 mg/l) at pH 7 was used as a coagulant for the coagulation process. The removal efficiency was found to be better for arsenate than arsenite. Addition of chitosan (0.2–2 mg/l) in the mixer as a coagulant agents resulted in more removal efficiency and less dosage of $FeCl_3$. Nearly 90–100% and 60–80% removal efficiency were found for Arsenate and arsenite, respectively. Initial concentration plays a significant role in the As removal. The less initial concentration tends toward better removal efficiency (Hao et al., 2018).

An experiment was carried out by Oza et al. (2020) to remove total As from the simulated As-containing water by combined EC and ultrasonic processes. The sequence is sonication treatment followed by EC process. In the EC method, highly pure iron electrodes were employed. Firstly, sonication process was carried out with initial concentration of 0.29 mg/l, initial pH of 7.8, ultrasound frequency of 40 kHz, and 2 h of treatment time. After that EC process was conducted at adjusted initial pH of 8.2, current density of 20.83 A/m^2, and 1 h of treatment time. This combination technique was effective in eliminating 98.44% of total As. While employing EC reactor alone with current density of 41.66 A/m^2, pH of 8.4, and initial As concentration of 0.5 mg/l, the optimum treatment time found was 15 min for removing 98.09% of total As.

Adsorption

An activated alumina-based As removal unit was used by Jalil and Ahmed (2001) for the removal of arsenate. Arsenate removal efficiency was found better than arsenite removal efficiency. The water samples were collected from the different tube wells in Bangladesh, and 65% of the sample collected were having 2–10 mg/l of iron depending upon the location. This iron plays a significant role in the performance of the alumina column. Iron oxidized at less pH producing insoluble iron resulting in $Fe(OH)_3$ through hydrolysis. The activated alumina adsorption was able to remove As concentration less than 50 ppb at a pH level of 5.5 with the removal efficiency of 97%. Dissolved iron in groundwater increased the performance efficiency of the treatment resulting in more removal efficiency of As (Bhattacharya et al., 2015).

A study conducted by Bang et al. (2005) shows the removal of As using ZVI and the effects of dissolved oxygen and pH in the process. Experiment results of arsenite and arsenate were obtained under oxic and anoxic conditions. An arsenate at 9 h of mixing in oxic condition at pH level 6 showed 86% removal efficiency, whereas removal efficiency drops down to only 4% at anoxic condition. Similar results were obtained in arsenite with the removal efficiency of 99.8% and $<9\%$ in oxic and anoxic conditions, respectively. The rate of iron corrosion and As removal can be increased by the presence of dissolved oxygen and low pH (Rezaei & Vione, 2018).

Another study was performed by Li et al. (2010) on a low arsenic sample water, using a ferrous-based red mud sludge as an adsorbent for removal of As from the water. Positively charged nanoparticles on the surface of Fe_3O_4 from red mud sludge act as an adsorbent agent for As removal (Wang & Liu, 2021). 2×10^{-4} mol/l ferrous and 7.7×10^{-2} g/l red mud were mixed with the initial concentration of 16–1.4 mg/l arsenate for 24 h—resulting in 91.2% As removal efficiency. The pH was maintained at 8 during the experiment. Increased $FeCl_2$ content in red mud from 10 to 30% increased removal efficiency while reducing the pH of the solution by 6.4.

To remove higher concentrations of As, an Mg and Al layered double hydroxide (LDH) modified by sulfur species (LDH-S) was developed by Huang et al. (2022). Mg/Al molar ratio was set at 2:1. In a continuous flow configuration with initial As(V) concentration of 20 ppm, solution was passed along the column, which contained 300 mg of adsorbent. The LDH (2:1)-S adsorbent decreased As(V) concentrations to less than ppb levels within 120 min, which, when compared to the WHO recommendation, demonstrates the adsorbent's capabilities in real-world scenario. At the mentioned molar ratio, maximum adsorption of As(V) was observed as 40.8 mg/g at room temperature having BET surface area of 39.2 m^2/g, adsorbent dosage of 50 mg, initial As(V) concentration of 100 ppm (sample volume of 30 ml), pH of 9, and contact time of 16 h. Effect of presence of different anions and cations on As removal indicated that PO_4^{3-} significant effect on As(V) adsorption, and Cl^-, Br^-, and I^- had negligible effect on As(V) adsorption at the same conditions mentioned above except initial As(V) concentration of 50 ppm.

Membrane filtration

A study was performed by Jiang et al. (2016) for the removal of As from groundwater using three different nanofiltration membranes. All the membranes were made from polyamide having a film thickness of 0.14 mm, 0.158 mm, 0.151 mm, respectively. The size of the membrane used in the experiment was 15, 55, and 100 mm^2. The initial pH was maintained at 7.42. The initial concentration of arsenate in the sample was 30 µg/l. The flow rate maintained during the experiment was 1 l/min at a pressure of 10 bar. The arsenate concentration decreases to 10 µg/l using the NF-3A membrane after the treatment.

Four different membranes were used in the experiment carried out by Yu et al. (2013) to remove arsenate. In the laboratory, the sample was artificially developed. The effective membrane area was 139 cm^2. The water permeability of membrane ESNA-1LF was 2.4×10^{-8} m^3/s m^2 kPa and the $MgSO_4$ rejection capacity of membranes was 97%. The sample had 200 µg/l of arsenate initially. During the experiment, the pH and temperature were maintained at 7.5–8 and 21 °C, respectively. The flow rate was maintained at 2 l/min. This study found that membrane ESNA-1LF had the highest arsenate removal efficiency of 94%.

Recently, experiments by Boussouga et al. (2021) were performed in the laboratory to remove arsenate using nanofiltration. Two different membranes were used in the experiment namely semi-aromatic-based polyamide and fully aromatic polyamide. The pore radius and permeability of the membrane (NF-90) were 0.29 ± 0.02 nm and 8 ± 2 l/h/bar, respectively. The sample had an initial arsenate concentration of 250 µg/l. The temperature was set at 25 °C during the experiment. With pH of 2, the high rejection capacity of arsenate was observed at higher salinity. Additionally, $MgSO_4$ rejection was $>97\%$. This study shows that filtration with NF-90 can remove arsenate to achieve the WHO standard of 10 µg/l at pH of 2.

In a study, samples were taken from various groundwater sources of Sila Massif (Italy) for the experiment of arsenite removal by nanofiltration and reverse osmosis (Figoli et al., 2020). The membrane used in the experiment was made from polyamide-TFC. The pH, flow rate and temperature were maintained at 6.7, 9 l/min and 16 °C. The duration of the experiment was 1 h. The flux/psi ratio of both membranes was 22/100 and 39/100, respectively. The efficiency of the membranes was tested by passing the sample at different pressures (3, 7, 11 and 15 bars) to the membrane. After testing, samples having 59–118 µg/l arsenic concentration have reduced to 10 µg/l at 15 bars following the filtration process.

In an integrated system, using nanofiltration coagulation has been developed by Pal et al. (2014). The water sample had 180 mg/l arsenite at pH level 5. Firstly, pre-oxidation using a $KMnO_4$ oxidizing agent is used for the conversion of arsenite to arsenate. The coagulant agent used was ferric salt. The rate of coagulant added in treatment was 250 mg/l. The flux of the membrane was 144–145 l/m^2/h. After pre-oxidation, the nanofiltration process was conducted by membrane NF-1 (Sepro membrane). After various tests, the result concluded that 4 mg/l of $KMnO_4$ pre-oxidation agent is optimum to convert 98–99% of arsenite to arsenate from groundwater, and nanofiltration removed 93–95% of arsenic from groundwater.

Another study for As(III) removal was carried out by Ma et al. (2021), in which an ultrafiltration (UF) membrane consisting of hydrophilic nickel carbon nanotube layer as cathode was prepared. A 6-µm-thick CNT layer was deposited onto the UF membrane using a pressure deposition process. The concentration of As(III) in raw water was spiked to 750 µg/l, in both synthetic and real tap water. Membranes utilized in the investigation were compressed at 689 kPa for minimum of 12 h before operation. As a result, it can be concluded that while treating synthetic water at flux of 8.9 l/m^2 h, initial pH of 6.57, 7 kPa pressure, and 1.29 l/m^2 h/kPa of permeance, 93% of As(III) rejection was achieved. Consequently, while treating real tap water at flux of 8.9 and 9.1 l/m^2 h, initial pH of 7.69 and 7.69, 6.89 and 28 kPa pressure, and 1.29 and 0.32 l/m^2 h/kPa of permeance, 73% and 93% of As(III) rejection was obtained.

Ion exchange

Kinetic studies on arsenite and arsenate removal by Cl and OH-exchangeable form of anion exchange resins had been carried out by Rao et al. (2015). The sample used in the experiment was synthetically prepared in the laboratory. The initial concentration of arsenate in the sample was 500 µg/l. The pH and temperature of samples was set at 7.5–8 and 28 °C. About 94% of arsenate removal by the OH-exchangeable form of anion exchanger was noticed within the first 60 min of reaction, whereas the maximum removal was 97% after 270 min of reaction. The performance of the Cl^- exchangeable form of anion was noticed to be inferior as only 81% removal took place after 60 min. Although arsenate removal was good, and arsenite removal was less than 21% by either of the resins, even after 270 min of reaction. By using an OH-exchangeable form of anion exchanger, the arsenate can be efficiently removed up to standards as per WHO guidelines for As concentration in drinking water.

An examination by Issa et al. (2011) was conducted for the removal of arsenic in Serbia. The samples were prepared in the laboratory using deionized water. In this experiment, both organic (monomethyl arsenic acid and dimethyl arsenic acid) and inorganic As (as arsenate and arsenite) were separated. The resin used in the experiment was (1) strong base anion exchange, Lewatit Monoplus M 500 having an average bead size of 0.61 mm, (2) hybrid-iron resin having an average bead size of 0.35 mm, and (3) hybrid-silver chloride resin prepared in the laboratory having average bead size of 0.56 mm. The diameter and length of the column in which As was removed were 2 cm and 20 cm, respectively. In a batch experiment with hybrid AgCl resin, at pH 9, around 50 µg arsenate and arsenite adsorption per gram of resin were observed when the initial total As concentration, resin dosage, temperature, and time were 500 µg/l, 1 gm, 20 °C, and 2 h, respectively. Authors observed that this resin has a low adsorption capacity for both arsenate (80 µg/gm) and arsenite (85 µg/gm) in a continuous flow system having a flow rate of 1.25 ml/min compared to other resins, but when they tested real water sample, they found that resin was stable and had higher As separation efficiency. Optimum conditions of flowrate and resin dosage were observed at 1.25–1.66 ml/min and 6 gm, respectively, in a continuous experiment. The authors concluded that hybrid AgCl resin was efficient enough to separate organic (passed through the resin) and inorganic As (retained by the resin).

A polymer-bridged nanoparticle fiber was developed by An et al. (2011) for the removal of arsenate from water by ion exchange. The mean particle size

Table 1 Summary of various technologies used for the treatment of As-contaminated water

Technology used	Initial as concentration	Type of water used	Observations	References
Photocatalytic Oxidation (1.05 mol% Zn doped Fe_3O_4 microsphere as a catalyst)	1.48 mg/l as total As	Real groundwater	Catalyst dosage = 2 g/l pH = 7, t = 4 h Complete oxidation of As(III) to As(V)	Nguyen et al. (2020)
Photochemical oxidation (H_2O_2 + UV)	0.1 mg/l as As(III)	Real groundwater with further purification using $NaAsO_2$	[H_2O_2] = 5 mg/l, pH = 8, UV dose = 2000 mJ/cm^2, 85% oxidation efficiency	Sorlini et al. (2014)
Photocatalytic oxidation (UV + H_2O_2)	10 mg/l as As(III)	Ultrapure water	[TiO_2] = 0.05 g/l, pH = 9, [O_2] = 32 ppm (through bubbling), T = 30 °C, 90% oxidation efficiency in first 10 min	Molinari and Argurio (2017)
UV-assisted Catalytic Oxidation and Adsorption	10 mg/l as As(III)	Ultrapure water	UV-LED power = 19 W, λ = 395 nm, irradiance = 10 W/m^2, T = 25 °C, [H_2O_2] = 60 mg/l, [$FeTiO_3$] = 1000 mg/l, pH = 7, D_p of $FeTiO_3$ < 100 µm, BET surface area = 6 m^2/g, As(III) oxidation efficiency ≈ 100% at t = 75 min, total As removal efficiency ≈ 100% at t = 105 min, stirring speed = 500 rpm	Garcia-Costa et al. (2021)
	10 mg/l as As(III)	Real groundwater spiked with As(III)	UV-LED power = 19 W, λ = 395 nm, irradiance = 10 W/m^2, T = 25 °C, [H_2O_2] = 60 mg/l, [$FeTiO_3$] = 1000 mg/l, pH = 7, D_p of $FeTiO_3$ < 100 µm, BET surface area = 6 m^2/g, As(III) oxidation efficiency ≈ 100% at t = 90 min, total As removal efficiency ≈ 100% at t = 120 min, stirring speed = 500 rpm	
Electrochemical Oxidation(Pt/Ti as both anode and cathode)	1 mg/l as As(III)	De-ionized water	[Persulfate] = 500 mg/l, [Fe^{2+}] = 250 mg/l, pH = 3, applied voltage = 2 V, time = 60 min, As(III) oxidation efficiency with external aeration ≈ 100%, mixing speed = 700 rpm	Syam Babu and Nidheesh (2022)

Table 1 (continued)

Technology used	Initial as concentration	Type of water used	Observations	References
(Fe as both anode and cathode)			[Persulfate] = 500 mg/l, pH = 3, applied voltage = 2 V, time = 60 min, removal efficiency with external aeration ≈ 100% [for both As(III) and As(V)], average operating cost = \$ 0.33/m^3, mixing speed = 700 rpm	
(Fe as both anode and cathode)	1.25–1.65 mg/l as total As	Real groundwater	[Persulfate] = 500 mg/l, pH = 3, applied voltage = 2 V, time = 45 min, removal efficiency with external aeration ≈ 100% [for both As(III) and As(V)], mixing speed = 700 rpm	
Coagulation ($Ti(SO_4)_2$)	1 mg/l as As(III)	Double distilled water	Coagulant dose = 25 mg/l pH = 8, removal efficiency > 90%	Sun et al. (2013)
Electrocoagulation (Fe plate electrodes)	285 μg/l as total As	Real groundwater	pH = 7.6, removal efficiency = 96.9%, *t* = 6 min (to reach WHO standard value of < 10 μg/l), energy consumption = 0.338 kWh/m^3, operating cost = \$ 0.101/m^3, sludge produced = 0.072 kg/m^3	Kobya et al. (2015)
(Fe balls electrodes)			pH = 7.6, removal efficiency = 99.3%, *t* = 20 min (to reach WHO standard value of < 10 μg/l), energy consumption = 3.330 kWh/m^3, operating cost = \$ 1.185/m^3, sludge produced = 0.073 kg/m^3	
Electrocoagulation (Fe electrode)	50 mg/l as As(III)	De-ionized water	pH = 4, current density = 5.4 A/m^2, removal efficiency = 99.50%, *t* = 30 min, energy consumption = 0.33 kWh/m^3, stirring speed = 150 rpm	Can et al. (2014)

Table 1 (continued)

Technology used	Initial as concentration	Type of water used	Observations	References
Electrocoagulation (Fe-Al hybrid plate electrode)	0.5 mg/l as As(III)	Ultrapure water	pH = 7, current density = 47 A/m^2, aerationintensity = 0.32 l/min, removal efficiency > 99.94%, t = 20 min, operating cost = $ 0.0782/m^3	Song et al. (2016)
Electrocoagulation (Fe ball electrode)	50 µg/l as As(III)	Real groundwater	pH = 8.5, applied current = 0.05 A, removal efficiency = 99%, t = 4.94 min, diameter of Fe ball = 9.24 mm, column height in the reactor = 7.49 cm, air flow rate = 9.98 l/min, operating cost = $ 0.01/m^3	Demirbas et al. (2019)
Electrocoagulation (Fe scrap electrode)	10 µg/l as total As	Real groundwater	pH = 7.42, removal efficiency = 93.5%, t = 8 min, applied current = 0.05 A, energy consumption = 0.070 kWh/m^3, operating cost = $ 0.017/m^3, electrode consumption = 0.052 kg/m^3	Omwene et al. (2019)
(Al scrap electrode)			pH = 8.01, removal efficiency = 93.5%, t = 30 min, applied current = 0.1 A, energy consumption = 0.876 kWh/m^3, operating cost = $ 0.181/m^3, electrode consumption = 0.066 kg/m^3	
Adsorption (Graphene oxide-hydrated zirconium oxide nanocomposites)	As(III) and As(V) both of 80 mg/l individually	De-ionized water	Solution pH range = 3–9 for As(III) and 5–11 for As(V), adsorption capacity = 95.15 mg/g for As(III) and 84.89 mg/g for As(V), t = 15 min for both As species, for 1:100 mass ratio of composite Brunauer–Emmett–Teller (BET) surface area = 420.9 m^2/g, average pore size = 2.27 nm, pore volume = 0.299 cm^3/g	Luo et al. (2013)

Table 1 (continued)

Technology used	Initial as concentration	Type of water used	Observations	References
Adsorption (Chitosan ZVI encapsulated nanospheres)	As(III) and As(V) both of 500 mg/l individually	De-ionized water	pH = 7 for both As species, adsorption capacity = 94 mg/g for As(III) and 119 mg/g for As(V), $t = 180$ min, BET surface area = 69 m^2/g, average pore size = 5.943 nm, pore volume = 0.103 cm^3/g, successfully recycled 5 times without significant change in adsorption capacity	Gupta et al. (2012)
Adsorption (MOF called MIL-53(Al))	0.054–2.428 mg/l as As(V)	De-ionized water	Maximum adsorption capacity = 105.6 mg/g at pH = 8, adsorption capacity at equilibrium concentration of 10 µg/l (safe limit by WHO in drinking water) = 15.4 mg/g, $T = 25$ °C, adsorbent dose = 20 mg/l, surface area = 920 m^2/g, $t = 11$ h to reach 80% of the maximum adsorption capacity	Li et al. (2014)
Catalytic adsorption (Fe and Mn based bimetallic oxides)	10 mg/l as As(III) and As(V) individually	De-ionized water	optimum Fe/Mn molar ratio = 1:1, adsorption capacity = 45.3 mg/g for As(III) and 49.7 mg/g for As(V) at pH = 3, maximum adsorption capacity = 74.05 mg/g for As(III) and 38.09 mg/g for As(V) at 40 mg/l initial concentration, pore radius = 18.778 Å, BET surface area = 388.6 m^2/g, calcination temperature = 300 °C, pore volume = 0.40 cm^3/g, $T = 25$ °C, adsorbent (Fe_1Mn_1-300) dose = 200 mg/l, $t = 24$ h, stirring speed = 200 rpm	Xie et al. (2022)

Table 1 (continued)

Technology used	Initial as concentration	Type of water used	Observations	References
Ion exchange (Anion exchange synthetic resins)	100 µg/l as As(V)	Ultrapure water	pH = 4.69 for Selion ASRFG3300 and 4.41 for Lewatit FO36, sorption capacity = 0.095 µg As/mg of ASRFG3300 and 0.108 µg As/mg of FO36, $t = 90$ min, optimum resin dose = 40 mg for both, sorption = 90% for FO36 and 80% for ASRFG3300 at optimum resin dose	Karakurt et al. (2019)
Ion exchange (Amine doped acrylic ion-exchange fiber)	50 mg/l as As(V)	Ultrapure water	pH = 3.04, sorption capacity = 205.32 mg/g at $t = 120$ min, diameter of fibers = 37–54 µm, ion-exchange capacity = 7.5 meq/g, removal efficiency > 83% during 9 regeneration cycles	Lee et al. (2017)
Membrane process (Negative charged polyamide membrane-based nanofiltration)	42.6 µg/l as total As	Real groundwater	pH = 6.9, $T = 43.6$ °C, low operating pressure = 1.2 bar, total As rejection = 0.87–0.90, [As] in permeate 5 µg/l, low recovery rate chosen = 10%, flux = 16 l/m^2hr, feed flowrate = 0.4 m^3/h, permeate flowrate = 0.7 l/min, membrane active area = 2.6 m^2	Gonzalez et al. (2019)
Membrane process (Granular media filter followed by RO)	230–780 µg/l as As(III)	Real groundwater	pH = 7.2, operating pressure = 9–12 bar, low recovery rate chosen = 10%, RO feed flowrate = 600 l/h, permeate flowrate = 60–70 l/h, energy consumption per liter of permeate = 3.4 Wh/l for RO, As(III) removal efficiency ≈ 99%	Schmidt et al. (2016)

Table 2 Advantages and disadvantages of different As treatment process

Sr. no	Type of treatment method	Advantages	Disadvantages
1	Oxidation	Easy to retrofit in current treatment process Method is effective over a broad pH range Good As(III) to As(V) oxidation rate can be achieved using AOPs	High auxiliary chemicals requirements Can only convert As(III) into As(V) and can't remove it which makes pre-oxidation compulsory along with secondary removal Aeration will result in higher operational costs
2	(Electro) Coagulation and flocculation	The technique is straightforward Processes can be operated in a broad pH spectrum EC—An innovative and effective method for As removal Uses resources that are readily accessible nearby Comparatively lower expense with minimal maintenance	Produces extremely As-containing sludge Sludge handling issues—high cost associated The elimination of As(III) is neither appropriate nor effective Greatly influenced by the coagulant's type and the presence of opposing anion sources in the matrix
3	Adsorption	Traditional and competent approach Greater As(V) removal compared to As(III) which fits perfect synergy with pre-oxidation step Low-cost operation and less/no sludge accumulation during the process Significant rate of removal	After few runs, it becomes necessary to regenerate used adsorbent material Regeneration of adsorbents is expensive It just involves moving the target contaminant from one phase to another Appropriate only with minimal As content The effectiveness of the procedure is impacted by the existence of additional salts and organic material Secondary filtration is necessary to get reusable water Coagulating agents must be added continuously
4	Ion exchange	As removal can't be affected by pH of the solution As removal with greater selectivity Great separation potential Capable to reduce As concentration to less than 2 µg/l in treated water Minimized formation of harmful sludge	Great expense of resins which makes operations costlier Significant operating and maintenance costs Brief lifespan of resin Periodically, resin needs to be changed Every resin is tailored to a unique arsenic species Presence of anions hinder the elimination of As
5	Membrane filtration	Widely accepted technique Can be utilized for the household intent Superior effectiveness and minimal power utilization Other impurities can also be removed simultaneously	The installation and upkeep costs are quite significant It is necessary to use numerous membranes Membrane fouling Time intensive technique Generation of large volume of rejection water

of starch-bridged nanoparticles was 26.6 ± 4.8 nm. The sample was prepared in the laboratory using deionized water. Magnetite and starch concentrations were 1.7 g/l and 0.049 wt%, respectively, to remove 280 mg/l of arsenate up to 100% at pH 6.5 in 1 h. It was found that starched nanoparticles remove arsenic 5 times more than bare magnetic particles. The researchers noted maximum arsenate adsorption at pH 4–6 using starched magnetite nanoparticles (in which optimum starch and Fe concentrations were taken). The maximum adsorption capacity of arsenate was 248 mg/g at pH 5 in the presence of 6% NaCl.

A hybrid anion exchange membrane was used in an experiment, executed by Oehmen et al. (2011), to remove arsenic from groundwater. The membranes used in experiments were four different anion exchange membranes, each having size of 11.3 cm^2. The samples were prepared in the laboratory. In this experiment, the sample was added in flow-cell at the flow rate of 46 ml/h. Three hours of residence time was provided in the experiment. The pH of the sample was maintained at 7.8–8.0. The concentration of arsenic as total arsenic in a sample was reduced from 57 to 3 μg/l by using AR204-UZRA anion exchange membrane.

Summary of different As removal technologies with main attention on optimal conditions along with their removal efficiency is listed in Table 1.

The merits and demerits of the treatment approaches mentioned earlier have been condensed based on the thorough review of published article in Table 2.

While conventional methods are inefficient to remove As from groundwater, chemical coagulation with the popular coagulants like iron and aluminum salts is certainly efficient and cost-effective and is being utilized more than other methods. The chemical coagulation method has been observed as an effective arsenate removal compared to arsenite. EC has been regarded as an innovative and advanced remediation technique for As-polluted groundwater. As reported by Can et al. (2014), Kobya et al. (2015), and Song et al. (2016), removal efficiency of the EC process was above 99% for As within the maximum time of 30 min with the help of Fe electrodes and/or Fe-Al hybrid electrodes. The dissolved oxygen present in the water helps to transform metals into their hydroxides, due to which arsenite is oxidized to arsenate and hence, both are eliminated from the water with very high removal efficiency over a brief period.

Proposed ideas for mitigation

- The primary goal and purpose of a mitigation plan is to identify the As-contaminated sources and vulnerable populations and to provide a sufficient amount of uncontaminated drinking water to them.
- To address the issue of groundwater As toxicity, proper As monitoring of the contaminated region (i.e., state or country) is considered an important first measure that can be included in the action plan, with the development of a well-organized Geographic Information System (GIS) directory being suggested in particular to understand the geographical transportation of As (Nath, 2018).
- Many study results showed that the amount of As concentration in shallow aquifers is higher than in deep aquifers. To avoid As contaminants, it is recommended that groundwater from deep aquifers should be prioritized for drinking purposes (Shankar et al., 2014).
- In order to secure agricultural activities, the rising threat of As pollution of land and harvested yield needs to be investigated as soon as possible to make lifelong changes in cultivation and irrigation activities and to limit the utilization of groundwater (Nath, 2018).
- Local government authorities should develop a color code to distinguish between As-contaminated and non-contaminated groundwater to let the local illiterate people know which sources contain safe drinking water and which does not. Various provisions should be made by local bodies to utilize different sources of As-contaminated groundwater for different purposes according to the degree of As contamination (Das et al., 2009).
- Rainwater harvesting should be adopted by the endangered populations as a replacement of As-contaminated groundwater after proper chemical and disinfection treatment (Shankar et al., 2014).
- Thorough maintenance and frequent evaluation of the treated water quality of the groundwater decontamination plants and/or local water treatment plants should be assessed regularly within the affected region by the regulatory authorities.
- Well-organized and coherent examination of the people diagnosed with arsenicosis should be carried out by the respective medical authorities. It is their responsibility to let the affected people of the nearby region know about the indication of chronic As toxicity (Das et al., 2009).
- It is suggested that the respective government should prompt a rule which entails the proper examination of the province and regulatory approval for a well to be established (Chakraborti et al., 2013).
- No medicines which can treat chronic As poisoning have so far been discovered. Only unpolluted drinking water, healthy food, and physical activities (i.e., exercise) can be utilized to combat this epidemic (Chakraborti et al., 2013).
- Conventional surface water is a prodigious source of As-free water. After treating the surface water, it can be distributed through underground pipes or transport vehicles to the As-affected rural regions. NGOs and other organizations should be

appointed for transportation and other expenditure (Zaved Kaiser Khan, 2012).

- Stakeholders have inadequate coordination with each other as they are the leading organization for the treatment and distribution of As-free water (Milton et al., 2012). Stakeholders include NGOs, institutes, donor agencies, government and social activists (Khan & Yang, 2014).
- An appropriate household study should be undertaken to determine the existing water consumption patterns, as well as the alternatives favored by consumers who require an alternative treatment option. Tube wells were by far the most viable treatment option among the society since they were inexpensive to maintain and simple to operate (Hossain et al., 2015).
- People should be aware of the future aspects and catastrophic effects of consuming As-rich water. The negative health impacts and economic impacts of exposure to high As containment should be exploited publicly (Milton et al., 2012).
- Artificial water recharge is one of the typical mitigation programs for As dilution. As natural replenishment of As is a very slow process; artificial water recharge can be used to increase the pace of replenishment. It can also increase the efficiency of various treatment proposes like filtration, absorption, coagulation as the initial concentration of As will be decreased due to the replenishment.
- Arsenic absorption through the skin is minimal; thus, bathing washing hands or working in the field with arsenic-containment water poses less risk to human health. This poses a high risk to arsenic-contaminated drinking water, and thus mitigation strategies should be more focused on reducing the consumption of arsenic-rich drinking water. Paddy and rice are the staple food in India. Arsenic contamination in these crops is usually typical in India and Bangladesh delta regions. Arsenic accumulation in rice and paddy can be mitigated through agronomics and crop hybrid strategies. Engineered plants with genetic modification can tolerate more As and reduce arsenic uptake for improved food safety (Zhao et al., 2010). In situ treatment process is a well-known mitigation method used in Europe where an aquifer is exposed to atmospheric O_2. This helps metal in the aquifer oxidize and reduces toxic contamination (Kumar, 2015). Rain-Water Harvesting is one of the most common mitigation methods from avoiding drinking arsenic-contaminated water. Biovolatilization of arsenic by efficient bacteria in a culture medium can be used to achieve arsenic-free water (Majumder et al., 2013). Long-term mitigation measures for future generations are to make them aware by giving them the proper education and training. Well Switching is successfully implemented in villages in Bangladesh, where wells are labeled “safe” and “unsafe” based on their arsenic contamination (Jameel et al., 2021).
- Besides different mitigation processes, treatment processes are more reliable for the arsenic removal rate as the cost–benefit analysis report shows more benefits toward the treatment process than the mitigation process. Mitigation cost exceeds treatment cost, and the removal rate also shows significantly more in treatment.

The mitigation measures can further be classified as long-term and short-term which are discussed in Table 3. Any one of them can be applied on the field based on the purpose and required timespan.

Table 3 Suggested short-term and long-term mitigation plans

Sr no.	Long-term mitigation	Short-term mitigation
1	Well switching (Jameel et al., 2021)	Rainwater harvesting and surface water
2	Deep hand tube well	Shallow well
3	Genetic modification in rice (Zhao et al., 2010)	In situ arsenic treatment (Kumar, 2015)
4	Wide-scale education and training about the harmful effects of arsenic and how to avoid them	Arsenic-volatilizing indigenous soil bacteria (Majumder et al., 2013)
5	Treatment technologies	Piped water supply

Challenges and future scope

The biggest challenge in the field of As decontamination is to ensure an effective solution that produces less amount of slurry and has the lowest running expenses. In comparison with other treatment methods, the EC is the most efficient As eliminating technique which produces an extremely low amount of slurry. One of the limitations of EC is the passivation/fouling of the electrode surface which dramatically reduces ionic movement between the anode and the cathode that may lead to the reduced removal efficiency after a certain operational period. Passivation negatively affects mass transfer during the process. Surface chemistry, of course, has a critical involvement in the overall operation. van Genuchten et al. (2016) published a paper that included an outstanding clarification of this phenomenon throughout As elimination by EC in which they compared electrodes utilized in lab and field experiments to analyze the surface films developed during As elimination process. To overcome such a situation, there are different approaches through which we can minimize the passivation of the electrodes (Barek, 2021; Hanssen et al., 2016). One of the simplest steps to conquer this problem is to change the electrodes periodically and/or schedule periodic post-treatment of the electrodes to remove the deposition whenever needed. These measures increase the operating and maintenance cost, but, on the other hand, it helps to achieve higher consistent removal efficiency. According to the experiments conducted by Eyvaz et al. (2014), applying the alternating pulse current method not only helps to prevent electrode passivation but also helps in Chemical Oxygen Demand (COD), turbidity, color removal. Moreover, it also helps to reduce operating costs in comparison with the DC mode. At the same time, the amount of sludge formation also increases. Although the passivation film generated on the electrode surface during operation can't be regulated, it can be reduced to certain magnitude by switching polarity after every experiment (Syam Babu et al., 2021). Surface modifications of an electrode and modern electrode materials can be used that are less susceptible to passivation. Rotating disk electrodes have the potential capability to enhance the mass transfer process by washing out the passivation-inducing products from the surface of the electrodes which helps to reduce their accumulation on the electrode surface (Chen et al., 2020; Zanotto et al., 2019). Additionally, EC operation demands the utilization of energy that raises running costs. To optimize electricity costs, the energy produced from clean and sustainable renewable resources (i.e., solar energy) should be investigated. When looking at the EC process, residual sludge handling and disposal is also a significant factor to address. A study was carried out by Syam Babu et al. (2021) to reuse the As-containing As-Fe sludge as a binding agent in concrete with the help of solidification and stabilization processes resulted into small change in concrete's compressive strength. In the same study, leachability studies of As-containing sludge and concrete samples were investigated, and negligible arsenic leaching was found. Low-cost emerging technologies have been used as a promising solution for As removal like laterite-based As removal, use of red mud as an adsorbent, or use of nanomembrane as a filtration technique. Although arsenic is considered a toxic element in many of the studied papers, As and its compounds are used in medicine for the treatment of skin ulcers, psoriasis, diabetes, joint diseases, syphilis (also known as sexually transmitted infection—cured by drug called Arsphenamine or Salvarsan), etc. as well as long-term promyelocytic leukemia, a type of white blood cell cancer (Kulik-Kupka et al., 2016; Santacroce et al., 2021). Dissolved As is a type that is left in the treatment process after the removal of As from water. This dissolved As can be used to obtain arsenous acid that helps to get sodium arsenate, sodium arsenite, copper pyroarsenite, calcium arsenite, and calcium arsenate (i.e., different salts of arsenous acid) which are further used as pesticides and herbicides in agriculture (Bencko & Foong, 2017). Treatment of As-contaminated water is one of the most expensive mitigation strategy. In semi-arid regions, long-term As remediation options like rainwater harvesting and deep aquifers are not reliable. Thus, government established Arsenic Removal Unit (ARU) according to the population and requirements of that regions is preferred (Bhardwaj et al., 2018).

Conclusion

The main source of As poisoning for the individuals is its presence in drinking water and/or food tests substantially beyond the allowed levels. According to the reports, the prolonged exposure to As-contaminated drinking water has high chances of developing chronic diseases including cancer of the lung, skin, kidney, bladder, and liver, arsenicosis, tuberculosis, impaired intellectual functions, diabetes, and cardiovascular diseases including high blood pressure. The arsenic toxicity in the crops highly depends upon the position of the crops grown either above or below the surface. The typical treatment process of As-contaminated water involves two stages in which oxidation stage is considered as a pre-treatment step. Because of the feasible energy consumption, economical operating cost, limited sludge production, and high removal efficiency, EC is the best available technology to remove As from water. To mitigate this global issue, we have put forward some plans of action to be followed. Some of the low-cost efforts are to be put into mitigation of As involving rainwater harvesting, artificial groundwater recharge, finding As-free water from deep aquifers, etc. Due to less rainfall in dry regions, switching to deep aquifers may solve the As-contaminated water issue as rainwater harvesting won't mitigate the problem. Furthermore, following the post-treatment operation, individuals can consume shallow depth water.

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