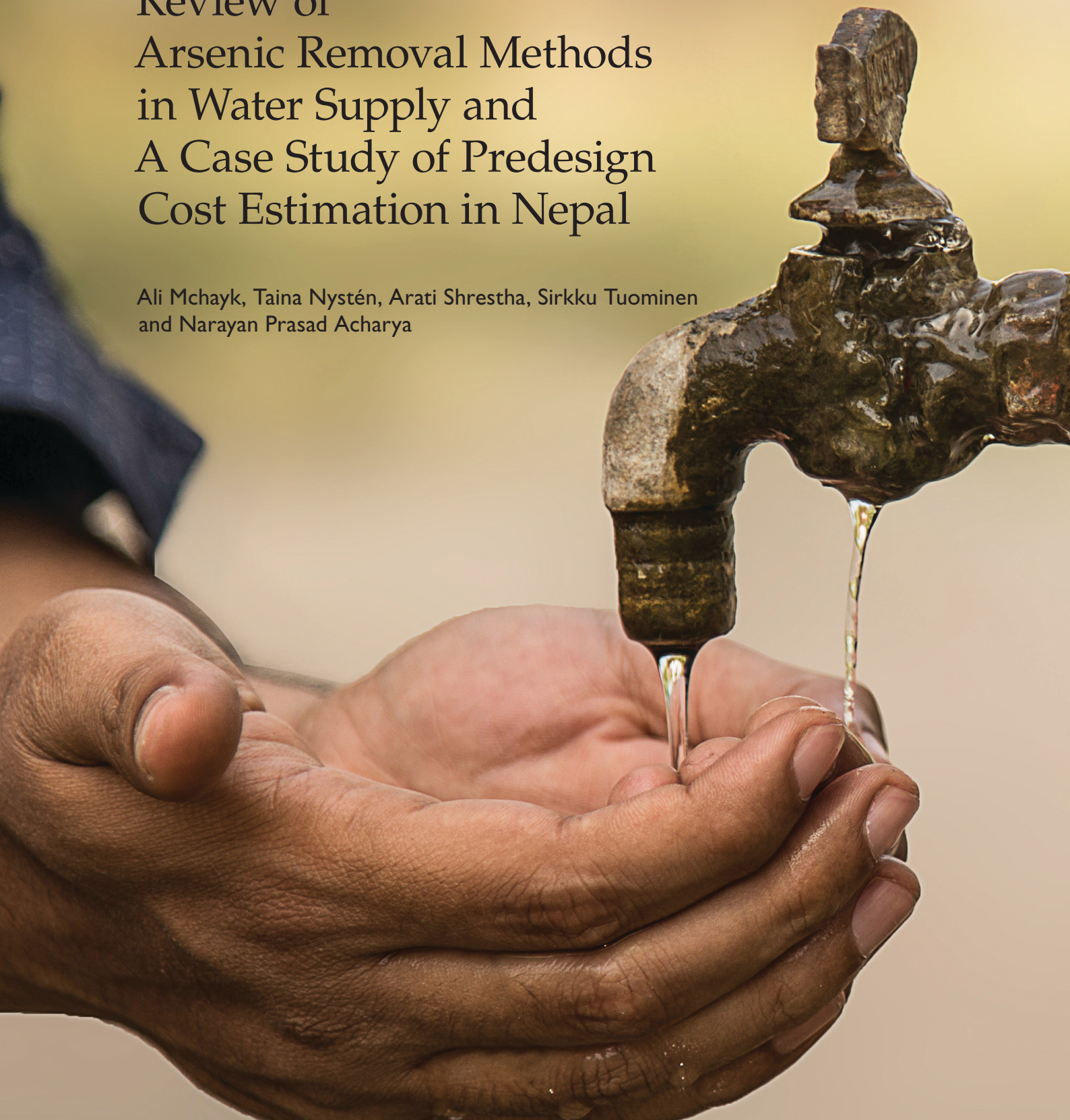# Review of Arsenic Removal Methods in Water Supply and A Case Study of Predesign Cost Estimation in Nepal

Ali Mchayk, Taina Nystén, Arati Shrestha, Sirkku Tuominen and Narayan Prasad Acharya

Finnish Environment Institute

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Abstract

Review of Arsenic Removal Methods in Water Supply and A Case Study of Predesign Cost Estimation in Nepal

The consumption of arsenic contaminated drinking water leads to adverse impacts on human health; arsenic is classified by the World Health Organization as a human carcinogenic substance. In the early 2000s, Nepal was added to the list of countries suffering from arsenic contaminated water.

This report reviews the efficiencies and limitations of arsenic oxidation techniques, conventional technologies used for arsenic treatment along with the recent application of nano-particle adsorbents. The arsenic treatment methods reviewed in this report include coagulation-flocculation, membrane filtration, adsorption-ion exchange, and nano-particle adsorbents. These treatment technologies generate arsenic bearing residuals, which can be managed by using several techniques presented in this review. The influence of the water composition on arsenic removal efficiencies is also addressed, along with the most important parameters to consider when choosing the best available technology for arsenic removal.

As well as, this report provides tools for the cost effectiveness and environmental evaluation of different treatment systems, according to the capital costs, operational costs, and the environmental impact of greenhouse gas emissions and arsenic disposal. Additionally, a predesign cost estimation for a waterwork system with the capacity of 10 mgd (37,854 m^3/day) in Nepal was applied, based on historical data. The intention of the predesign cost estimation is to focus research and resources onto cost effective unit processes and technologies.

Keywords: drinking water, arsenic, contaminants, removal, water supply, water treatment, costs, geogenic, geochemistry, Nepal, Asia

Tiivistelmä

Katsaus vesihuollon arseeninpoistomenetelmiin ja niiden ennakkosuunnittelun kustannusarvioon Nepalissa

Arseenin saastuttaman juomaveden käyttö aiheuttaa haitallisia vaikutuksia ihmisten terveyteen ja Maailman terveysjärjestö on luokitellut sen ihmisille syöpää aiheuttavaksi aineeksi. Nepal lisättiin 2000-luvun alussa niiden maiden luetteloon, jotka kärsivät arseenin saastuttamasta vedestä.

Tässä kirjallisuuskatsauksessa tarkastellaan arseenin hapetustekniikoiden tehokkuutta ja rajoituksia, tavanomaisia arseenin poiston käsittelyyn käytettyjä tekniikoita sekä uutena tekniikkana nanohiukkasten adsorbentteja. Tarkasteltuihin arseenin käsittelymenetelmiin kuuluvat koagulaatio-flokkulaatio, kalvosuodatus, adsorptio-ioninvaihto ja nanopartikkeliadsorbentit. Nämä käsittelytekniikat tuottavat arseenipitoisia jäämiä, joita voidaan hallita useilla tässä yhteydessä esitellyillä tekniikoilla. Kirjallisuuskatsauksessa käsitellään lisäksi veden koostumuksen vaikutusta arseeninpoistotehokkuuteen sekä tärkeimpiä parametreja, jotka on otettava huomioon valittaessa parasta käytettävissä olevaa arseeninpoiston teknologiaa.

Tässä julkaisussa myös johdatellaan eri arseenipoiston käsittelymenetelmien kustannustehokkuuden ja ympäristövaikutusten arviointiin, mihin vaikuttavat pääomakustannukset, käyttökustannukset sekä kasvihuonekaasupäästöjen ja arseenipitoisen jätteen ympäristövaikutukset. Lisäksi esimerkinomaisesti Nepalissa 10 mgd:n (37 854 m^3/vrk) kapasiteetilla toimivalle vesilaitokselle tehtiin arseeninpoiston ennakkosuunnittelun kustannusarvio, joka perustuu aiempiin kustannustietoihin. Esisuunnittelun kustannusarvion tarkoituksena on keskittää tutkimusta ja resursseja kustannustehokkaisiin yksikköprosesseihin ja teknologioihin.

Asiasanat: juomavesi, arseeni, haitalliset aineet, poistaminen, vedenhankinta, vedenkäsittely, kustannukset, geogeeninen, geokemia, Nepal, Aasia

Sammandrag

Översikt över metoder för avlägsnande av arsenik i vattenförsörjning och deras kostnadsberäkning för förhandsplanering i Nepal

Konsumtionen av arsenikförorenat dricksvatten leder till negativa effekter på människors hälsa och arsenik klassificeras av Världshälsoorganisationen som ett cancerframkallande ämne för människor. I början av 2000-talet lades Nepal till listan över länder som lider av arsenikförorenat vatten.

Denna rapport granskar effektiviteten och begränsningarna hos arsenikoxidationstekniker, konventionell teknik som används för avlägsnande av arsenik samt, som ny teknik, tillämpningen av nanopartikeladsorbenter. De arsenikbehandlingsmetoder som granskas i denna rapport inkluderar koagulation-flockning, membranfiltrering, adsorptionsjonbyte och nanopartikeladsorbenter. Dessa behandlingstekniker genererar arsenikhaltiga rester, som kan hanteras med hjälp av flera tekniker som presenteras i denna översyn. Även vattensammansättningens inverkan på effektiviteten av avlägsnandet av arsenik behandlas i litteraturstudien, samt de viktigaste parametrarna som måste beaktas vid val av bästa tillgängliga teknik för avlägsnande av arsenik.

Denna rapport ger också verktyg för en bedömning av kostnadseffektiviteten och miljökonsekvenserna av olika metoder för avlägsnande av arsenik. Bedömningen baserar sig på uppgifter om kapitalkostnader, driftskostnader samt miljöpåverkan av utsläpp av växthusgaser och behandling av arsenikhaltigt avfall. Dessutom gjordes en kostnadsuppskattning för en förhandsplanering av ett vattenverkssystem i Nepal med en kapacitet på 10 mgd (37 854 m^3/dag) baserat på historiska data. Avsikten med kostnadsuppskattningen är att fokusera forskning och resurser på kostnadseffektiva enhetsprocesser och tekniker.

Nyckelord: dricksvatten, arsenik, skadliga ämnen, borttagning, vattenförsörjning, vattenbehandling, kostnader, geogen, geokemi, Nepal, Asien

Preface

Arsenic is a highly toxic substance, with the World Health Organization (WHO) recommending a 10 µg/l concentration value in drinking water. More than seventy countries, including Nepal, are exposed to elevated arsenic concentrations in drinking water.

This report reviews the efficiencies and limitations of arsenic removal methods in water supply and provides tools for the cost effectiveness and environmental evaluation of different treatment systems. The aim of this report is to improve the capacity of the water sector to collect, share and use the information and know-how on the removal of arsenic in drinking water. The immediate objective of this literature review is the ability of the Nepal Water Supply and Sewerage Management (DWSSM) to guide water supply operators in cost effectively treating arsenic-contaminated drinking water in Nepal.

In the Finnish Environment Institute (Syke), trainee Ali Mchayk, Development Manager Taina Nystén and Senior Research Scientist Sirkku Tuominen, and in DWSSM Engineer Arati Shrestha and Senior Division Engineer Narayan Prasad Acharya participated in planning and writing this report.

The report is one of the outputs of the NeAs project, i.e., 'Towards arsenic safe drinking water in Nepal' syke.fi/projects/neas. This cooperation project by Syke, DWSSM and Geological Survey of Finland (GTK) started in 2020 and is funded by the Ministry for Foreign Affairs of Finland (MFA). The aim of this project is the improved capacity of DWSSM to guide and support the water supply actors in arsenic mitigation.

Thank you to everyone who contributed to the promotion of this publication.

Helsinki, October 2022

Taina Nystén, Development Manager (PhD), Finnish Environment Institute (Syke)

Contents

1 Introduction

1.1 General Background

Arsenic (As), a naturally occurring metalloid abundant in the earth's crust, is gaining international concern for being a hazardous element to human health (Hughes et al. 2011; Fatoki et al. 2022). Moreover, arsenic is used across multiple sectors including agriculture, electronics, and medicine (Nicomel et al. 2016). Arsenic can be released into the environment as a result of natural events or anthropogenic sources. This is done through natural processes (e.g., weathering of arsenic rocks, and floods) due to its presence in hundreds of minerals, mainly metal ores such as copper, nickel, and cobalt ores (Akin et al. 2011). It can also be released from a wide range of human activities like agriculture and processing metals or through the use of arsenic in industry and other applications. Furthermore, Arsenic mobility is governed by biogeochemical conditions such as the parent mineral form, oxidation state, and others which could favor high mobility (Nicomel et al. 2016). As such, arsenic could contaminate water, soil, and air thus allowing it to find its way to humans, animals, and plants (Amen et al. 2020). In the developing world, human exposure to arsenic is mainly attributed to the consumption of arsenic contaminated water, especially when relying on groundwater (Amen et al. 2020). This groundwater contamination is often associated with geothermal processes, mineral dissolution, desorption in oxidizing conditions and, reductive desorption and dissolution (Nicomel et al. 2016).

1.2 Arsenic Toxicity and Speciation

Arsenic is classified either as organic or inorganic and is globally recognized as highly toxic. It is classified by the World Health Organization (WHO) as human carcinogenic substance and associated with skin, lung, kidney, and bladder cancer when inorganic arsenic concentrations exceed 50 µg/l in drinking water, which is five times greater than those limits recommended (10 µg/l) by WHO (WHO 2018; Nicomel et al. 2016). In addition, arsenic has a multitude of adverse impacts on human health, collectively named arsenicosis, such as, skin lesions, cirrhosis, and melanosis (Amen et al. 2020). Toxicity levels are related to arsenic speciation and oxidation states (Amen et al. 2020). The most common arsenic species in aquatic environments are inorganic arsenite (As(III)) and arsenate (A (V)), as well as organic dimethyl arsenic acid (DMA) and monomethyl arsenic acid (MMA) (Amen et al. 2020). Both redox potential and pH levels are determining factors, of which arsenic chemical species are prevalent. As(III) is the favored state of arsenic under anoxic reducing conditions (e.g., subsurface water) and in alkaline water, whereas As(V) prevails under aerobic oxidizing conditions (e.g., surface water) and acidic pH levels. However, due to slow redox transformations, As(III) and As(V) could exist in both reduced and oxidized conditions (Nicomel et al. 2016; Amen et al. 2020). Given that it is more genotoxic, cytotoxic, soluble, and mobile (Nicomel et al. 2016), the inorganic trivalent As(III) ranks as the most toxic among arsenic species. As(V) follow in the toxicity rankings, where MMA, and DMA are the least toxic of the four mentioned species (van Halem et al. 2009).

1.3 Arsenic contamination in Nepal

It is estimated that hundreds of millions of people in more than 70 countries are exposed to elevated arsenic concentrations in drinking water (Nicomel et al. 2016; Singh et al. 2015). Although, arsenic contamination is a global threat to human wellbeing, it is reported to be the most severe in Asian countries (Nicomel et al. 2016; Singh et al. 2015). In the early 2000s, Nepal joined the list of Asian countries suffering from arsenic-contaminated water, which has been reported to be a significant health concern affecting districts with high population densities (Thakur et al. 2011).

The permissible arsenic level for drinking water is below 50 µg/l, as stated by National Drinking Water Quality Standard (NDWQS) of Nepal. However, the arsenic blanket test (2003–2008) carried out by Government of Nepal in partnership with various international development agencies and non-governmental organizations, covering 1,101,536 wells in 20 districts in Nepal, showed that 5.37 and 1.73% were contaminated with arsenic by 10–50 µg/l and > 50 µg/l, respectively (GoN/NASC 2011). Other similar studies carried out at various part of Nepal also depicted the exceedance of permissible value: in Bara district, 50.00% and 12.50% of the total samples (n=24) exceeded WHO guideline value (10 µg/l) and NDWQS (50 µg/l), respectively; in Kathmandu valley, 76.20% and 47.60% of the total samples (n=20) showed the arsenic level to be above 10 µg/l and 50 µg/l, respectively (Kayastha 2015; Gwaccha et al. 2020). Nepal was categorized in the "Sear D" region, indicating a severely arsenic affected country in the WHO comparative risk assessment, with approximately 12.50 Disability Adjusted Life Years (DALYs) per 1,000 population (Fewtrell et al. 2005). Moreover, around 33.40% of the population rely on groundwater from shallow and deep tube wells for daily domestic water use in Nepal, which signifies arsenic poisoning from contaminated wells as a major public health threat (Shrestha 2012; GoN/NMIP 2014).

In view of the above, adequate water treatment strategies should be put in place to alleviate the threat of arsenic contamination. The following sections aim to provide a comprehensive view of arsenic treatment methods while evaluating their efficiencies and costs.

2 Arsenic Pre-treatment

2.1 Oxidation Techniques

The basis of arsenic oxidation techniques is the employment of oxidation as a pretreatment step to oxidize As(III) into As(V), which is less mobile and easier to adsorb to surfaces (Giles et al. 2011; Aremu 2020). Subsequently, oxidation is followed with a suitable As(V) treatment method (Giles et al. 2011). Oxidation pretreatment is especially important for anaerobic water, since As(III) is dominant at neutral pH levels and remains mostly uncharged below 9.2 pH values, making processes such as precipitation, adsorption, and ion exchange ineffective in arsenic removal (Nicomel et al. 2016). However, several arsenic oxidation techniques exist including chemical oxidation, photochemical oxidation, photocatalytic oxidation, biological oxidation, and *in situ* oxidation (Singh et al. 2015).

2.1.1 Oxidation by oxygen, ozone, and chemicals

Water composition, concentrations of As(III) and oxidants are deciding factors for arsenic removal efficiencies (Singh et al. 2015; Aremu 2020). As such, oxidation could be performed using oxygen, chemicals, bacteria, or ozone (Johnston et al. 2001). Although atmospheric oxygen is readily available to be used as an oxidant, arsenic oxidation by air is believed to be very slow (Johnston et al. 2001). Alternatively, using pure oxygen increases the concentration of dissolved oxygen, resulting in faster oxidation rates (Johnston et al. 2001). On the other hand, As(III) oxidation rate is further accelerated by using ozone (O_3), chlorine (Cl_2), hypochlorite (ClO^-), chlorine dioxide (ClO_2) (Bissen et al. 2003). Although complete oxidation of As(III) is achievable by using O_3, it is an expensive process, requiring high energy input, handling of post-treatment toxic residuals and removal of residual O_3 in treated water. (Mou et al. 2020; Aremu 2020). Iron pentacarbonyl (Fe(0)), ferrous oxide (Fe(II)), and ferrate (Fe(VI)) are other commonly used chemicals in arsenic oxidation. Upon the oxidation of the lower valent Fe(0) and Fe(II) by oxygen or hydrogen peroxide, the formation of Fe(III) hydroxide takes place to act as adsorbent to arsenic (Giles et al. 2011), while ferrate offers a unique advantage of both oxidizing As(III) and coagulating As(V) (Sharma et al. 2007). According to multiple studies, ferrate is one of the fastest As(III) oxidizers at pH 7 or below (Giles et al. 2011). Furthermore, ferrate still outperforms hydrogen peroxide and chloramine at alkaline pH levels (Giles et al. 2011). Table 1 illustrates the efficiencies and limitations of various oxidants.

2.1.2 Photochemical Oxidation

Photochemical oxidation is a procedure that utilizes UV irradiation to enhance As(III) oxidation, given the presence of oxygen in the solution (Mou et al. 2020; Johnston et al. 2001). This procedure is widely studied for water that naturally contains iron, where UV or solar light is utilized to supplement the photolysis of Fe(III) species to create hydroxyl radicals ($FeOH^{2+}$) which are a strong oxidant (Singh et al. 2015). It is reported that the presence of both hydroxyl radicals and oxygen amplifies the As(III) oxidation rate (Singh et al. 2015). Additionally, studies estimate that the As(III) oxidation rate can be increased by four orders of magnitude (10^4) by using UV irradiation in the presence of Fe(III) (Bissen et al. 2003).

Additionally, sulfite, ferric ions, and citrate could be used as catalyzers to further enhance the oxidation of arsenic (Mou et al. 2020). The addition of citrate (lemon juice) after applying UV/solar light would create a reaction between the citrate and the newly formed radicals due to UV irradiation to produce further arsenic oxidizing radicals (Mou et al. 2020). However, excessive citrate concentration could hinder oxidation due to the formation of acid complexants (Singh et al. 2015). It is reported that 90% of the initial 40 µg/l arsenic concentration was oxidized within 30 to 90 seconds when using high

Table 1. Various oxidants, along with limitations and reported efficiencies

Oxidant	pH	Initial [As] concentratio	Efficiency	Remarks	Limitations	References
Air, Pure O_2	7.6–8.5	46–62 µg/l	54% & 57%	54% & 57% of arsenic (III) is oxidized in 5 days using air and pure O_2 respectively (groundwater)	Slow, incomplete	Nicomel et al. 2016; Bissen et al. 2003
Ozone (O_3)	7.6–8.5	46–62 µg/l	100%	100% oxidation in 20 mins. Slow reaction with ammonia if present, increasing efficiency	Expensive, carcinogenic bromate ion by-product when reacting with bromide & reduced efficiency due to TOC and sulfide	Nicomel et al. 2016; Singh et al. 2015; Mou et al. 2020
Cl2 & ClO-			-	Rapid, effective, inexpensive & available	Trihalomethanes byproduct when reacting with organic matter	Giles et al. 2011; Mou et al. 2020
Chlorine Dioxide (ClO_2)			-	inexpensive & available	Prohibited for use in surface water	Mou et al. 2020
Hydrogen Peroxide (H_2O_2) & Fenton's Reagent		40 mg/l	50% & 100%	at 1:1 (As: H_2O_2) molar ratio, 50% As(III) oxidation in 10 mins. 100% oxidation was achieved at 1:4 molar ratio. Also, Fe(II) enhanced oxidation efficiency	Highly PH dependent	Bissen et al. 2003
Potassium Permanganate ($KMnO_4$) coated sand		200 µg/l	-	The initial concentration of 200 µg/l was reduced to 25 µg/l. Inexpensive, fast, and available		Aremu 2020; Bissen et al. 2003
Ferrate (FeO_4^{2-})			-	Does not react with bromide. Residual Fe(III) is a coagulant to potentially remove As(V). One of the best oxidizers below PH 7. Acts as an oxidizer, coagulant, and disinfectant.		Singh et al. 2015; Giles et al. 2011
Activated Carbon	-	40 µg/l	90%	Dosage of 10 g/l achieved 90% oxidation within 20 to 30 mins.	Natural arsenic leaching from activated carbon	Bissen et al. 2003

Samples are dosed with arsenic in experimental studies in laboratories, elevating arsenic concentrations to the order of mg/l.

pressure mercury UV lamps emitting light at a wavelength between 190 nm and 254 nm (3 to 4 KWh/m³) (Bissen et al. 2003). Furthermore, the addition of Fe(III) to the perchlorate/ perchloric solution followed by UV irradiation proved to enhance arsenic oxidation at pH levels between 0.5–2.5 due to the formation of hydroxyl and dichloro radicals (Singh et al. 2015). Further experimental studies are displayed in Table 2.

2.1.3 Photocatalytic Oxidation

Similarly to the photochemical oxidation procedure, photocatalytic oxidation (PCO) utilizes UV irradiation to enhance As(III) oxidation. However, PCO requires the presence of oxygen and a photocatalyst which is often titanium dioxide (TiO_2). The photocatalyst (TiO_2) serves as an oxidizer to As(III) and adsorbent to As(V). Singh et al. (2015) reported several studies displaying an efficient arsenic removal using PCO and TiO_2 as an adsorbent. The same report also highlighted a decrease in efficiency upon the elimination of UV use in the treatment procedure. Furthermore, arsenic concentrations along with pH values, TiO_2 type, anions, silicate, fluoride, phosphate, and natural organic matter (NOM) are critical factors that affect the adsorption of As(V) on TiO_2 surfaces (Singh et al. 2015), whereas the PCO of As(III) is affected by the presence of humic acid and bicarbonate in water (Singh et al. 2015). Although, the presence of NOM could enhance the PCO of As(III), excessive NOM concentration could create a competition between As(V) and NOM for binding sites on TiO_2 surfaces (Singh et al. 2015).

2.1.4 Biological Oxidation

Biological oxidation methods aim at replicating naturally occurring biological processes performed by plants and micro-organisms to aid the treatment of metals in groundwater (Singh et al. 2015). Given that arsenic contaminated groundwater generally reduces and contains iron (Fe) and manganese (Mn) (Singh et al. 2015), biological oxidation emerged as an arsenic treatment method. This treatment method utilizes Fe or Mn oxidizing bacteria or both to enhance As(III) oxidation. Arsenic is then removed by adsorption or co-precipitation onto biogenic Fe or Mn oxides (Katsoyiannis et al. 2006). The series of reactions that takes place is as follows (Singh et al. 2015; Mou et al. 2020):

    i.   Biotic oxidation of Fe(II) to Fe(III) and Mn (II) to Mn (IV).
    ii.   Biotic oxidation of As(III) to As(V)).
    iii. Abiotic precipitation of manganese oxides (MnO).
    iv. Oxidation of As(III) by MnO.
    v.   Abiotic As(V) sorption by MnO_2.

Table 2. Photochemical and photocatalytic oxidation efficiencies

Oxidation Process	Initial [As] Concentration	Efficiency	Remarks	References
Photochemical, 90 W/m² UV-A Light	500 µg/l	90% in 2–3 hours	initial Fe(II) and Fe(III) concentrations of 0.06–5 mg/l	Singh et al. 2015
Photochemical, 3–4 KWh/m³, High pressure mercury UV light	40 µg/l	90% in 30–90 seconds	wavelength between 190 nm and 254 nm	Bissen et al. 2003
Photochemical, UV/H_2O_2	40 mg/l	Arsenic was oxidized in 10 mins	Molar ratio As: H_2O_2 of 1:1	Bissen et al. 2003
Photochemical, $UV/sulfite-O_2$	0.47 mg/l	100%	8 mg/l of sulfite is used	Bissen et al. 2003
Photocatalytic UV/TiO_2	66.7 µM		10 µg/l arsenic concentration was achieved	Singh et al. 2015
Photocatalytic UV/H_2O_2	100 µg/l	85%	UV dose of 2 000 mJ/cm²	Nicomel et al. 2016

Samples are dosed with arsenic in experimental studies in laboratories.

Bissen et al. (2003) reported that As(III) oxidation can be performed by microbial organisms given the presence of 1 mg/l of O_2 in the solution. Moreover, several studies have suggested that microbial organisms enhance arsenic removal from water to different extents depending on the selection of As(III) oxidizing microbial organisms (Mou et al. 2020).

Gallionella ferruginea and *Leptothrix ochracea* are said to oxidize As(III) into As(V) and aid the biotic oxidation of Fe, resulting in a better As(V) adsorption onto Fe oxides. Indeed, it is reported that 95% arsenic removal efficiency was achieved (initial [As]=200 mg/l) by using a filter medium with iron oxides and the above-mentioned microorganisms (Singh et al. 2015). Furthermore, Katsoyiannis et al. (2006) reported a significant increase in arsenic removal efficiencies when using iron oxidizing bacteria, opposed to when the same experiments were performed under sterilized conditions. However, biological iron oxidation is primarily affected by the solution's redox potential, noting that the residence time is usually in the order of minutes which indicates that the oxidizing effect of oxygen is minimal (Katsoyiannis et al. 2006). It is also worth mentioning that aeration and the presence of bacteria enhances As(V) removal (Katsoyiannis et al. 2006). In addition to avoiding chemical by-products in treated water, another advantage of the biological oxidation of Fe is the continuous production of adsorbents, preventing adsorbate exhaustion, which is a common feature when using physiochemical methods (Katsoyiannis et al. 2006). Arsenic removal efficiencies of multiple biological and physiochemical iron oxidation experiments under different conditions are illustrated in Table 3.

Biological oxidation could also be applied to water only containing Mn (Katsoyiannis et al. 2006). Under such conditions, Mn oxidizing bacteria is utilized to facilitate the generation of reactive manganese oxide surfaces capable of oxidizing dissolved As(III) and removing As(V) (Singh et al. 2015). Bacterial As(III) oxidation is reported to have a higher rate than abiotic As(III) oxidation by manganese oxides (Singh et al. 2015). However, arsenic sorption on Fe oxides is more efficient when compared to Mn oxides (Katsoyiannis et al. 2006). The combined removal of iron, manganese and arsenic is achievable using biological oxidation, which involves a two-stage filtration process, preceded by initial aeration (Katsoyiannis et al. 2006). The first filtration stage removes dissolved Fe, As(III) and As(V) by the same mechanisms described above (Katsoyiannis et al. 2006), while the second filtration stage removes dissolved Mn, and the remainder of the arsenic by adsorption onto Mn oxides (Katsoyiannis et al. 2006). Arsenic sorption onto Fe or Mn oxides is reduced by the presence of anions in the water, namely phosphate anions (Katsoyiannis et al. 2006). Phosphate competes with arsenic for sorption sites presented by Mn oxides. As such, arsenic sorption onto Mn oxides is substantially reduced by phosphate, where an 80% arsenic removal efficiency was reduced to 30% in previous research (Katsoyiannis et al. 2006). On the other hand, the extent to which phosphate affect arsenic sorption onto Fe oxides is displayed when using coagulation as a treatment method. In the presence of phosphate, a 40:1 ratio of Fe:As is required to achieve efficient arsenic removal, as opposed to 12:1 in the absence of phosphate (Katsoyiannis et al. 2006). However, the phosphate effect on As(III) oxidation was dismissed by Katsoyiannis et al. 2006.

Table 3. Biological oxidation efficiency in different operating conditions (Katsoyiannis et al. 2006)

Oxidation Process	pH	Initial [As] (µg/l)	Efficiency	Remarks
Biological iron oxidation	7.2	≤150	80% As(III), 95% Fe	optimal Fe removal conditions (Redox Potential 270–280 mV, dissolved oxygen 2.7 mg/l)
Biological iron oxidation	7.2	>150	65% As(III)	optimal Fe removal conditions (Redox Potential 270–280 mV, dissolved oxygen 2.7 mg/l)
Biological iron oxidation	7.2	35–40	95% As(III)	Redox Potential 320 mV, dissolved oxygen 3.7 mg/l, residual arsenic below 5 µg/l
Physiochemical iron oxidation	-	20–50	80% As(V)	-
Physiochemical iron oxidation	-	188	60% As(V)	-

2.1.5 *In Situ* Oxidation

In situ oxidation processes reduce arsenic and iron concentrations at the source water aquifer before extraction (Singh et al. 2015; Mou et al. 2020). The process is performed by introducing oxygen to water extracted from the arsenic-contaminated well, which is later circulated back into the same well. This allows the O_2 to oxidize As(III) into As(V) and ferrous iron into ferric iron (Mou et al. 2020). Arsenic is subsequently adsorbed onto ferric iron, thus reducing the arsenic content in the source water aquifer (Mou et al. 2020). Biological oxidation could also be promoted by pumping oxygenated water into the aquifer. Thus, this process enhances the growth of Fe oxidizing bacteria which facilitates arsenic removal (Katsoyiannis et al. 2006). Although, *in situ* oxidation might lead to clogging the aquifers pore spaces if the dissolved iron content is high, it still can reduce arsenic content in both tube wells and deep wells (Mou et al. 2020).

3 Conventional Arsenic Removal Methods

3.1 Coagulation and Flocculation

Coagulation-flocculation, one of the most studied water treatment methods, has the potential to treat arsenic-contaminated water when combined with filtration and/or sedimentation (Nicomel et al. 2016; Aremu 2020). In addition to arsenic removal, turbidity, Fe, Mn, phosphate and fluoride, as well as odor and color reduction are possible when using coagulation & flocculation followed by filtration (Johnston et al. 2001). As such, both suspended and dissolved constituents in water could be treated simultaneously. Coagulation involves the rapid mixing of an added coagulant (Nicomel et al. 2016). The cationic coagulant destabilizes the colloids suspension by neutralizing the particles' surface charge, thus reducing their repulsion forces which ultimately allow for particle agglomeration (Aremu 2020). As the agglomerated particle size is still insufficient to undergo solid/liquid separation processes (e.g., filtration and sedimentation), the need for flocculation is necessary, which is performed through gentle mixing and the addition of anionic flocculants, allowing the formation of larger aggregates known as flocs (Baloch et al. 2020). Consequently, a portion of the dissolved arsenic is converted into settable insoluble solid, while the remaining of dissolved arsenic is co-precipitated by metal hydroxides to then be removed by sedimentation and/or filtration (Nicomel et al. 2016).

Due to their low cost and ease of handling (Singh et al. 2015), the most used coagulants in this treatment method, are aluminum salts (i.e., aluminum sulfate ($Al_2(SO_4)_3$)) and ferric salts (i.e., ferric chloride ($FeCl_3$), ferric sulfate ($Fe_2(SO_4)_3$)). However, arsenic removal efficiencies vary according to pH values, coagulant type and dosing, arsenic concentrations and valency (Baloch et al. 2020). The general consensus is that both aluminum and iron coagulants have equal As(V) removal efficiencies at pH values equal to or lower than 7.6, but iron coagulants have a wider pH range and perform better at pH values greater than 7.6 and when As(III) is present in water (Garelick 2005). Hering et al. (1997) reported that As(V) removal efficiencies could be increased with higher $FeCl_3$ or $Al_2(SO_4)_3$ coagulant doses. Nevertheless, this treatment method is considered to be more efficient in removing As(V) than As(III) since As(III) has a neutral charge at pH 4–10 (Nicomel et al. 2016). Additionally, the raw water quality has a share in determining arsenic removal efficiencies (Mou et al. 2020). For instance, the presence of sulfate (pH 4 & 5) and organic matter (pH 4 to 9) substantially reduce As(III) removal by ferric chloride, while As(V) removal is slightly affected (Garelick et al. 2005; Hering et al. 1997). Yet, while the presence of calcium at pH values higher than 7 enhances As(V) removal, the presence of orthophosphates and carbonate in West of Bengal waters was found to have a negative effect on As(V) removal (Garelick et al. 2005). Responses to various operating conditions are better presented in Table 4.

By using kaolinite and $FeCl_3$ as a coagulant/flocculants along with 9.2 mg/l of Fe^{3+}, 90% and 77% removal efficiencies of As(V) and As(III), respectively, were achieved (Singh et al. 2015). Hering et al. (1997) added that As(III) removal using Aluminum coagulants is ineffective, thus requiring the oxidation of As(III) into As(V) for effective treatment, while the same is also advisable when using ferric chloride. In addition, Nicomel et al. (2016) reported that aluminum-based coagulants were able to reduce an initial arsenic concentration of 280 µg/l to 10 µg/l. Table 4 displays reported removal efficiencies of different coagulants. However, it is important to consider the large amounts of arsenic concentrated sludge produced by the coagulation-flocculation method (Singh et al. 2015; Nicomel et al. 2016). Though arsenic sludge management is costly, it is important in order to prevent secondary environmental pollution (Nicomel et al. 2016). Furthermore, it is important to mention that the overall arsenic removal efficiency also relies on the solid/liquid separation process that follows; for instance, inadequate filtration might allow arsenic adsorbed onto fine colloids to pass through (Hering et al. 1997).

Table 4. Arsenic removal efficiency of coagulation-flocculation and ferric chloride performance

Coagulant	pH	Initial [As] Concentration	Efficiency As(V)	Efficiency As(III)	Remarks	References
Ferric Chloride	5–7	20 µg/l As(V)	80%	-	River water dosed with 4.9 mg/l of $FeCl_3$. Final arsenic concentration of 2 µg/l. Efficiency decreased at higher pH levels (8 & 9) in the presence of organic matter,	Hering et al. 1997
Ferric Chloride	7	20 µg/l As(V)	90%	-	River water dosed with 8.1 mg/l of $FeCl_3$. Final [As(V)] concentration of 5 µg/l, was achieved even when the initial [As(V)] concentration was up to 50 µg/l. As(V) removal efficiency is not strongly linked to initial [As(V)].	Hering et al. 1997
Ferric Chloride	7	20 µg/l	-	30%	River water dosed with 8.1 mg/l of $FeCl_3$. As(III) removal efficiency is strongly linked to initial [As(III)]. As(III) removal efficiency was increased using higher coagulant dose.	Hering et al. 1997
Ferric Chloride	7	-	75–90%	-	When considering both arsenic and iron residuals the optimum dosages were found to be 15, 25, 30 & 30 mg/l for the initial As(V) concentrations of 0.2, 0.5, 1 & 2 mg/l respectively. As(V) removal efficiencies were around 80%, 90%, 85% and 75% when using optimum dosage for its respective initial arsenic concentration.	Hesami et al. 2013
Ferric Chloride	7	-	-	45–60%	When considering both arsenic and iron residuals the optimum dosages were found to be 20, 30, 30 & 30 mg/l for the initial As(III) concentrations of 0.2, 0.5, 1 & 2 mg/l respectively. As(III) removal efficiencies were around 60%, 60%, 50% and 45% when using optimum dosage for its respective initial arsenic concentration.	Hesami et al. 2013
Ferric Chloride & Chitosan	7	-	80–90%	-	When using a 0.5 mg/l chitosan as a coagulant aid, the optimum dosages were found to be 10, 20, 25 & 30 mg/l for the initial As(V) concentrations of 0.2, 0.5, 1 & 2 mg/l respectively. As(V) removal efficiencies were around 80%, 90%, 90% and 90% when using optimum dosage for its respective initial arsenic concentration.	Hesami et al. 2013
Ferric Chloride & Chitosan	7	-	-	70–90%	When using a 0,5 mg/l chitosan as a coagulant aid, the optimum dosages were found to be 15, 25, 30 & 30 mg/l for the initial As(III) concentrations of 0.2, 0.5, 1 & 2 mg/l respectively. As(III) removal efficiencies were around 70%, 90%, 80% and 70% when using optimum dosage for its respective initial arsenic concentration.	Hesami et al. 2013
Aluminum Sulphate	6 & 7	20 µg/l	80%	-	River water dosed with 40 mg/l of $Al_2(SO_4)_3.18H_2O$. Final arsenic concentration of 2 µg/l. At pH 8 and above As(V) removal efficiency decreased. As(III) removal was negligible even when increasing the coagulant dose.	Hering et al. 1997
Ferric Sulphate	7	1 mg/l	-	80%	Ferric sulphate dosage of 25 mg/l.	Nicomel et al. 2016; Sun et al. 2013
Titanium Sulfate	7	1 mg/l	-	90%	Titanium sulphate dosage of 25 mg/l. Also, titanium sulphate As(III) removal efficiencies were better than ferric sulphate at pH levels 4–9.	Nicomel et al. 2016; Sun et al. 2013

3.2 Membrane Technologies

3.2.1 Background

Membrane filtration is a water treatment method capable of removing various contaminants (e.g., bacteria, salts) from water, including arsenic (Nicomel et al. 2016; Johnston et al. 2001). Membranes are usually synthetic materials comprised of billions of pores which reject some constituents in water while others are allowed to permeate, hence the term selective barriers (Aremu 2020). Pressure driven membranes require a pressure difference between the feed and they permeate to act as a driving force to convey water through the membrane (Nicomel et al. 2016; Singh et al. 2015). Generally, membrane technologies are expensive when compared to other methods, while also producing high residual volumes (Singh et al. 2015). The two classes of pressure driven membranes are (Johnston et al. 2001):

i.   low-pressure membranes (i.e., Microfiltration (MF) and Ultrafiltration (UF))
ii. high-pressure membranes (i.e., Nanofiltration (NF) and Reverse Osmosis (RO)).

Table 5 displays the characteristics of different membrane processes.

3.2.2 Microfiltration (MF)

In view of the above, MF has the capability to remove colloids and particles in the range of 0.1–10 μm, which is insufficient for the removal dissolved As(V) and As(III) from arsenic-contaminated water (Nicomel et al. 2016; Singh et al. 2015). MF efficiency is highly dependent on the size of arsenic bearing particles, thus coagulation-flocculation is a common process used to increase the size of these particles (Mou et al. 2020). When a 7.5 mg/l $Fe_2(SO_4)_3$ dose was used as a flocculant at pH 6.8 and an initial arsenic of 60 ppb prior to MF, it was reported that the arsenic removal efficiency exceeded that of MF followed by sedimentation (Han et al. 2002). Han et al. (2002) also reported that membranes with a nominal pore size of 0.22 μm had a better arsenic removal efficiency than those with a 1.2 μm nominal pore size. Furthermore, microfiltration has a low energy requirement and a high flux relative to other membrane processes, making it potentially cost effective (Han et al. 2002). However, the arsenic removal efficiency of MF coupled with flocculation is linked to pH levels and the presence of ion in the

Table 5. Membrane processes characteristics (Van der Bruggen et al. 2003 & Ezugbe et al. 2020)

Membrane Process	MWCO* (Kilo Dalton)	Retained Diameter (μm)	Average Permeability (l/m² h bar)	Pressure Required (bar)	Separation Mechanism	Applications	Solutes Retained
MF	100–500	10^{-1}–10	500	1–3	Sieving	Clarification, pre-treatment, removal of bacteria	colloids, particles
UF	20–150	10^{-3}–1	150	2–5	Sieving	Removal of macromolecules, bacteria, viruses	microplastics, macromolecules, particles
NF	2–20	10^{-3}–10^{-2}	10–20	5–15	Sieving charge effects	Removal of multivalent ions, small organics	Sulfates, sodium chloride, multivalent ions, macromolecules, particles
RO	0.2–2	10^{-4}–10^{-3}	5–10	15–75	Solution Diffusion	Ultrapure water, Desalination	multivalent ions, macromolecules, particles, small organics, and monovalent ions

*MWCO Molecular Weight Cut-off is the lowest molecular weight that can be retained by the membrane at 90% efficiency.

water. Thus, it is better suited to treat As(V) that remains negatively charged over a pH range of 4–10, unlike As(III) which has a neutral charge at this pH range preventing it from binding to surfaces (Nicomel et al. 2016). Thus, As(III) oxidation into As(V) is required for a successful arsenic treatment (Singh et al. 2015).

3.2.3 Ultrafiltration (UF)

In a similar manner to MF, UF pore size is big enough to allow dissolved arsenic to pass through the membrane, making UF alone ineffective in treating arsenic-contaminated water (Aremu 2020). Singh et al. (2015) reported 53% and 65% removal efficiencies for As(III) and As(V) respectively when using a charged UF membrane. Brandhuber et al. (2001) performed a study on arsenic removal using a charged UF membrane, showing that As(V) removal slightly increased with the increasing flux, contrary to the slight decrease in As(V) removal as the temperature was increased. The same study showed that at all pH levels, As(V) is better rejected by the membrane than As(III) (Brandhuber et al. 2001). In addition, the presence of calcium and magnesium divalent cations (i.e., Ca^{2+}, Mg^{2+}) significantly reduced As(V) rejection to nearly zero by locally neutralizing the membranes negative charge (Brandhuber et al. 2001).

The use of surfactants in UF processes could enhance arsenic removal efficiencies such as Micellar enhanced ultrafiltration (MEUF) (Nicomel et al. 2016). MEUF involves the addition of a cationic surfactant to a concentration level that exceeds the critical micelle concentration of the water thus allowing micelles to form which in turn binds the negatively charged arsenic (Nicomel et al. 2016). By using hexadecylpyridinium chloride hexadecyltrimethyl ammonium bromide as surfactants in the MEUF process, Singh et al. 2015 reported an arsenic removal efficiency of 96% and 94%, respectively. However, these efficiencies are reduced as pH levels decrease (Nicomel et al. 2016). Additionally, the effluent needs to be treated before it is discharged due to high surfactant concentrations (Nicomel et al. 2016).

3.2.4 Nanofiltration (NF)

Given that suspended solids in feed water are very low, NF is capable of reducing dissolved arsenic concentration in water (Nicomel et al. 2016; Singh et al. 2015). NF membranes generally possess a negative charge in neutral and alkaline pH levels, which is lost in acidic media (Singh et al. 2015). As such, NF has two separation mechanisms, referred to as charge exclusion and size exclusion (Nicomel et al. 2016; Uddin et al. 2007). Size exclusion (i.e., sieving) is achieved when the membrane's pore size is smaller than the targeted ionic species (Uddin et al. 2007), while charge exclusion relies on the electrostatic repulsion between the charged membrane and the anionic arsenic species in water (Uddin et al. 2007; Singh et al. 2015). Charge exclusion is reported to be the main mechanism responsible for arsenic rejection in NF processes and is reported to be more efficient when a membrane is highly charged, especially in treating charged solutes with a similar charge as the membrane (i.e., co-ions) (Uddin et al. 2007; Singh et al. 2015). However, the charge of NF membranes is dependent on the total concentration of ions in the solution which may alter the membrane's charge by adsorbing onto it (Singh et al. 2015). Furthermore, if the membrane has no charge, it may become positively charged if the pH level is low (Singh et al. 2015). Uddin et al. (2007) studied the removal of As(V) and As(III) using commercially available NF membranes. Their study reported effective As(V) reduction to values below 10 µg/l (98% efficiency), when the feed concentration was in the range of 100–1,000 µg/l. As(V) removal efficiency increases along increasing pH and pressure levels, and along decreasing temperatures, and initial arsenic concentrations (Uddin et al. 2007; Singh et al. 2015). In their study, Uddin et al. (2007) also reported a 59% As(III) removal efficiency but concluded that the oxidation of As(III) onto As(V) is required, since As(III) concentrations in the permeate remained above the allowable 10 µg/l even at a 50 µg/l initial feed concentration.

3.2.5 Reverse Osmosis (RO)

RO membranes have very small pores ranging from 10^{-4}–10^{-3} μm and are highly capable of rejecting compounds and ions with low molecular weight (Van der Bruggen et al. 2003; Singh et al. 2015). Thus, given that the suspended solids in the water feed is very low (Nicomel et al. 2016), RO is capable of reducing dissolved arsenic concentration in water. The earliest studies investigating arsenic removal using RO reported a 90% As(V) removal efficiency by using a cellulose-acetate RO membrane operating at 400 psi pressure (Shih 2005). Although As(III) removal efficiency was less than 70%, it is not as simple to oxidize As(III) into As(V) due to oxidants being potentially damaging to RO membrane (Shih 2005).

The removal of As(V) and As(III) using sea water high rejection (SWHR) and brackish water (BW-30) RO flat-sheet membranes was investigated by Akin et al. (2011). The study demonstrated that initial arsenic concentrations in feed water have no major impact on both arsenic species rejection efficiencies, whereas membrane type, operation pressure and pH levels are related to arsenic rejection efficiencies. Results showed that the SWHR membrane type had better As(III) and As(V) removal efficiency than BW-30 membrane. When using the SWHR membrane, As(V) and As(III) rejection increased from 90% to 95% and 70% to 80%, respectively, along with the increasing pH level (i.e., 2 to 10). While the increasing pH had the same effect when using for BW-30 membrane, As(V) and As(III) rejection increased from 65% to 70% and 60% to 68%, respectively.

3.3 Adsorption and Ion Exchange

Adsorption is a phenomenon that occurs in natural physical, chemical, and biological systems. Adsorption processes employ solids that are capable of attracting substances from gaseous or liquid solutions. As such, adsorption is the separation of substances from one phase that are accumulated onto the surface of the adsorbent. When selecting the adsorbent material, adsorbent surface properties (i.e., polarity, surface area) are of importance, as physical adsorption is driven by van der Waals forces and electrostatic forces between the adsorbent and adsorbate (Choong et al. 2007). For Arsenic removal, usually an ion exchange resin placed in a vessel is flushed with HCl so chloride ions occupy the available exchange sites on the resin (Choong et al. 2007; Singh et al. 2015). When arsenic-contaminated water is pumped through the vessel, arsenic adsorbs onto the resin by detaching the chloride and occupying the exchange sites (Choong et al. 2007). Thus, the arsenic content in the treated water is reduced while the chloride content increases (Choong et al. 2007). Ultimately, most of the exchange sites will be occupied by arsenic or other anions present in the contaminated water. Thus, a periodic backwash is required to regenerate the available adsorption sites.

Arsenic treatment using adsorption is a cost-effective process offering several advantages including simple operation and handling, and no sludge production (Nicomel et al. 2016). Strong base anion resins have better efficiencies over a wide pH range than weak-base resins (Singh et al. 2015); Table 6 illustrates efficiencies of various adsorption processes studied for arsenic-contaminated water treatment. Wan et al. (2010) reported how the biological oxidation of As(III) into As(V) improved arsenic removal efficiency when using a reactor of sand mixed with zero-valent iron powder. However, to avoid potential damage to sensitive resins, excess oxidants should be removed prior to the water entering the resin vessel (Singh et al. 2015). Furthermore, a concentration of competing ions (e.g., sulfate and nitrate), resin type and affinity to arsenic, dissolved solids levels, and pH levels are critical parameters controlling the adsorption efficiency of arsenic (Singh et al. 2015). The presence of competing ions and totally dissolved solids in high levels hinders the system's performance, while the presence of Fe(III) could also form arsenic complexes that are not adsorbed (Singh et al. 2015). Lenoble et al. (2002) reported that under acidic conditions (i.e., $3 < pH < 5$), As(III) and As(V) adsorption was close to 100% when using Amorphous iron hydroxide (HFO), goethite, and pillared polycation solutions (i.e.,

Table 6. Sorption capacity, efficiencies, and properties of various adsorbents

Adsorbent	Optimum pH	Dosage (g/l)	Surface Area (m²/g)	Temp. (C°)	Sorption Capacity As(III)	Sorption Capacity As(V)	Initial [As] Concentratio	Efficiency	Remarks	References
Coconut-shell carbon	5	5	1,150–1,250	25	-	2.4	-	-	Adsorbent: 3% ash content, 0.70–0.85 cm³/g pore volume.	Lorenzen et al. 1995
Coal based carbon	5	5	1,050–1,200	25	-	4.09	-	-	Adsorbent: 5–6% ash content, 0.94 cm³/g pore volume.	Lorenzen et al. 1995
Fly Ash	4	1	0.8 (BET)*	20	-	30	[As(V)] = 50 mg/l	80% As(V)	-	Diamadopoulos et al. 1993
Activated Alumina	7.6	1–13	370	25	0.18	-	[As(V)] = 0.5 mg/l	98.4% As(V) & 96.2% As(III)	94.4% As(V) removal at pH 6 to 8, 96.2% As(III) and 98.4% As(V) removal at pH 7.6.	Singh et al. 2004
Titanium Dioxide	7	1	250.7	13.4	32.4	41.4	[As] = 300 µg/l	95% As(V) & 90% As(III)	[Fe] < 0.02 mg/l. As(V) & As(III) removal 95% and 90% respectively. Silica (20 mg/l) and phosphate (5.8 mg/l) had no significant effect on arsenic removal at neutral pH level.	Bang et al. 2005
Granular ferric hydroxide	6.5	0.25	240–300	20	-	1.1	[As(V)] = 100 µg/l or [As(III)] = 100 µg/l.	95% As(V) & 97% As(III)	80–95% As(V) removal within 30 min, 3–15% additional removal in the next 24 hours. 90–97% As(III) adsorption at 24 hours (i.e., equilibrium). Efficiency increased along with temperature increase.	Banerjee et al. 2008
Iron oxide-coated sand	7.5	20	-	27	0.02857	-	[As(III)] = 100 µg/l	93% As(III)	93% maximum As(III) removal efficiency in 180 mins (65% removed in 40 mins). As(III) removal efficiency increased as pH level increased from 4.5 to 7.5 and then remained almost constant up to pH level of 10.5.	Gupta et al. 2005
Iron-modified activated carbon	7.6–8.0	0.1–20	723	20–23	38.8	51.3	As content = 1 mg As/L in 100 ml solution	80% As(V) & 80% As(III)	more than 80% As(V) removal at pH 6 and 8 after 3 hours contact, however after 24 hours at pH 6 As(V) removal was greater than pH 8. After 24 hours, 80% As(III) removal was achieved at pH 6 and 8.	Chen et al. 2007
Red Mud		20	-	25	0.884	0.941	[As] = 133.5 µmol/l	26% As(V) & 48% As(III)	Adsorption were 48% and 26% for As(III) and As(V) respectively. As(III) is better adsorbed at 9.1 pH level, while As(V) is better adsorbed at 1.1–3.2 pH levels.	Altundogan et al. 2000

Samples are dosed with arsenic in experimental studies in laboratories, elevating arsenic concentrations to the order of mg/l.
*BET: Brunauer-Emmett-Teller is a commonly used method to calculate specific surface area.

$Fe_x(OH)_y$, Ti_xH_y). In contrast, at near neutral pH levels (i.e., $6 < pH < 8$) and while using the same adsorbents, As(III) was better adsorbed than As(V). In addition, factors such as spent regenerant, resin disposal requirements, and secondary water quality effects are important to consider when selecting the adsorption process (Singh et al. 2015). Table 6 illustrates the adsorption capacity and removal efficiencies of the most common adsorbents.

3.4 Handling of Arsenic Sludge

3.4.1 Background

Several techniques are available for the safe disposal of arsenic contaminated sludge, out of which the most common technique is stabilization which is then followed by landfill disposal (Ruj et al. 2021). However, if landfill is used, it should be in accordance with the regulations of arsenic waste landfill disposal by the Toxicity Characteristic Leaching Procedure (TCLP). Furthermore, due to the increasing emphasis of sustainability, several techniques are emerging as safe and green solutions to dispose of arsenic contaminated sludge. The following sections review the handling of arsenic sludge by natural, biochemical, chemical and solidification processes.

3.4.2 Natural and Biochemical Processes

3.4.2.1 Landfill disposal

Landfill disposal is often used, especially in the absence of clear disposal guidelines to dispose of arsenic contaminated sludge, as it is considered one of the easiest and cheapest methods. The sludge is air dried to reduce its water content and then dumped into a sand covered $1m^3$ brick-lined pit (Sullivan et al. 2010). When using landfill pits, it is possible for arsenic to leach into the soil, to ultimately re-contaminate groundwater aquifers (Ruj et al. 2021). Even if the arsenic leachate concentration is small, long-term leaching could increase arsenic concentrations in the soil and groundwater, making it hazardous (Ruj et al. 2021). Furthermore, faulty seals are often reported to make the pits prone to floods which leads to environmental contamination (Sullivan et al. 2010). As such, stabilization/solidification discussed in the following sections is to be considered prior to landfill.

3.4.2.2 Mixing with Livestock Waste

Mixing arsenic concentrated sludge with livestock waste is a bioremediation technique that involves a bio-reductive reaction carried out by micro-organisms to transform As(III) into As(V) which are then transformed into arsine gas (Sullivan et al. 2010). In India and Bangladesh, cow manure is often used in open spaces to dispose of sludge, thus making the technique vulnerable to re-contamination (Ruj et al. 2021). Furthermore, in open areas, it is not possible to manage conditions to maintain the aerobic and anaerobic reactions to ensure the complete transformation of the sludge into non-leachate forms (Ruj et al. 2021). As such, digesters fitted with gas outlets could be a proper technical way to manage sludge mixed with waste livestock.

3.4.3 Stabilization by Chemical Processes

3.4.3.1 Stabilization by Chemical Encapsulation

A stable arsenic oxide could be formed by using chemical encapsulation. First, the process involves the addition of calcium hydroxide (i.e., $Ca(OH)_2$) to form a Non-gaseous calcium arsenate by reacting with As(V). Followed by the addition of silica, calcium oxide, alumina, feldspar, sodium carbonate and oxide

are then heated up to 1,200 °C, thus forming a stable arsenic oxide (Ruj et al. 2021). Although, this process is reported to satisfy the lower limits of arsenic leaching requirements, the stabilization efficiency only ranges between 1 and 20%. In addition, high energy input is required for heating, along with the formation of environmentally hazardous by products.

3.4.3.2 Stabilization by chemical precipitation/coagulation

Camacho et al. (2009) investigated the use of lime (i.e., calcium hydroxide) as a binder for arsenic solidification/stabilization (S/S). Lime was incrementally dosed at 1g per 10 g of air-dried arsenic residuals, and then the solidified material was cured for seven days at air temperature before analyzing arsenic leaching. Camacho et al. (2009) reported that arsenic leaching increased when lime addition was in the range of 1 to 5 g, which was expected since lime increases the pH of the solution, thus favoring arsenic leaching. When more than 6 g of lime were added, arsenic was successfully stabilized. However, the calcium-arsenic compounds formed by the lime treatment decomposed under atmospheric conditions, making lime treatment a short-term solution unless the residuals are shielded from exposure to atmospheric CO_2. On the other hand, when using a combination of lime and cement, long-term stabilization of arsenic sludge could be achieved without significant differences in arsenic leaching when compared to lime treatment alone. Camacho et al. (2009) reported a leaching of 0.0001 and 0.018 mg/l of As(III) and As(V), respectively, when using 6 g of lime and 3 g of type 1 Portland cement (i.e., ordinary Portland cement) per 10 g of air-dried residuals. Liang et al. (2017) applied intensive mechanical stress to co-ground a commercial iron powder with manganese dioxide to create a modified zero valent iron (ZVI) which is then used in arsenic sludge treatment. The modified ZVI was applied to an acidic arsenic sludge (i.e., pH 4.8) with a moisture content of 85%. The sludge was composed of 33% arsenic, 23.1% Fe and 6.2% Na. The study reported a very effective stabilization of arsenic sludge: when using modified ZVI, arsenic leaching was reduced from 72.5 mg/l to 0.62 mg/l, as opposed to arsenic leaching of 10.48 mg /L arsenic when using unmodified ZVI.

3.4.4 Stabilization by Cementation

Cementation is a popular method for arsenic sludge S/S prior to landfill disposal and has been reported to reduce arsenic leaching. Stabilization by cementation is performed by replacing sand with arsenic waste either in mortar or concrete mixes (i.e., cement, sand, water, and aggregates). The ionic arsenic species are adsorbed and co-precipitated by the calcium-silicate-hydrate matrix of cement, thus reducing arsenic leaching (Ruj et al. 2021). Clancy et al. 2015 replaced 1 or 3% of sand in the mortar mix with iron oxide arsenic waste and performed a long-term arsenic leaching experiment to simulate arsenic leaching under natural flooding events. The study reported that after 406 days of subjecting the mortar cubes to chemically simulated rainwater, less than 0.4% of arsenic in the mortar cubes leached. However, when performing a TCLP test over an 18-hour period, the study reported that cement stabilization reduced arsenic, leaching at high pH levels, whereas at pH levels less than 4.2 arsenic, leaching was greater than that of non-stabilized wastes. However, it is important to mention that if cement containing arsenic is used in construction, then the soaking and drying cycles, carbonation, weathering, and atmospheric vapor content of cement will eventually increase arsenic leaching from cement (Sullivan et al. 2010; Ruj et al. 2021). In addition, the emission of dust containing arsenic during construction would create adverse health impacts on exposed individuals (Sullivan et al. 2010). Arsenic sludge is also stabilized by incorporating the sludge into the manufacturing process of bricks. Like cement, arsenic leaching from bricks increases with time, especially when exposed to acidic conditions (Ruj et al. 2021).

4 Arsenic Removal using Nanoparticles Technology

4.1 Background

Advances in nanotechnology and nanoscience have progressed the engineering of nanomaterials intended for the treatment of contaminated water (Otero-Gonzalez et al. 2020). Due to their small particle size, nanomaterials have high surface area to volume ratio which enhance their adsorption capacities when compared to bulk adsorbents (Otero-Gonzalez et al. 2020; Tang et al. 2013). In addition, the high reactivity, specificity, and catalytic potential of nanoparticles makes their use in water treatment promising (Hristovski et al. 2007; Habuda-Stanic et al. 2015). As such, nanoparticles (NPs) have raised the interest of researchers to study their adsorption capabilities of contaminants (i.e., heavy metals, As(III), and As(V)) in aqueous solutions. As nanotechnology is relatively new, it is important to consider the availability of nanomaterials in adequate quantities and reasonable prices to enable their use in water treatment systems (Habuda-Stanic et al. 2015). Another major factor to consider is the safe disposal of arsenic contaminated NPs, along with the separation of NPs from treated water as they are toxic and hazardous to human health and the environment (Otero-Gonzalez et al. 2020).

4.2 Iron-Based Nanoparticles

Iron-based NPs are the most popular magnetic NPs, namely Iron oxide NPs and Zero-valent iron NPs (nZVI), both of which are efficient adsorbents to a broad range of organic compounds, metals and metalloids including arsenic (Hristovski et al. 2007; Tang et al. 2013). Although, nZVI and iron oxide NPs possess different chemical properties (i.e., oxidation state of iron) making their efficiencies differ, various conditions could also interfere in the removal mechanisms taking place (Hristovski et al. 2007). Multiple removal mechanisms are involved when using iron-based NPs, such as adsorption, co-precipitation, and redox reactions (Hristovski et al. 2007). In addition to the high surface area, specificity, and reactivity, iron-based NPs are also magnetic (Tang et al. 2013). Although it is thought that a magnetic field could be used to separate and recover magnetic NPs from treated water, it is also suspected that magnetism of NPs causes aggregation of NPs which reduces the separation and recovery of NPs from treated water (Tang et al. 2013). Thus, a stabilizer or surface modifier (i.e., starch and carboxymethyl cellulose) is required to prevent NPs agglomeration into larger aggregates (Habuda-Stanic et al. 2015).

4.2.1 Zero-valent Iron Nanoparticles (nZVI)

The most common methods for nZVI production are the reductive precipitation of $FeCl_3$ with $NaBH_4$ or reduction of goethite and hematite particles with H_2 at a high temperature (200–600 °C) (Tang et al. 2013). The nZVI has a different iron phase, particle shape, and size depending on the production method used, which ultimately affects the removal efficiencies of NPs (Tang et al. 2013). The strong reductive properties of nZVI, along with surface sorption and co-precipitation capabilities due to the formation iron oxides/hydroxides shell when in contact with water or air makes nZVI one of the most popular nano-adsorbents (Tang et al. 2013). Furthermore, Ramos et al. (2009) reported the formation of As(0), As(III) and As(V) on the surface of the NPs when nZVI reacted with As(III). Thus, As(III) is both reduced and oxidized in the presence of nZVI, which shows the versatile potential of nZVI in arsenic treatment. These redox reactions are attributed to the core-shell structure of nZVI which contains a highly reducing metallic core, in addition to an oxidizing thin layer of amorphous iron hydroxide (Ramos et al. 2009).

In their research, Morgada et al. (2009) reported that a commercially available nZVI displayed outstanding arsenic removal efficiencies. However, operating conditions such as pH, dissolved oxygen, hardness, humic acid (HA), UV light and nZVI dosage are of importance, as they may influence arsenic removal efficiencies. Morgada et al. (2009) concluded that HA has a detrimental effect on nZVI performance in the absence of UV light, due to the obstruction of adsorption. The same study proceeded to examine the effect of UV Light irradiation on As(V) removal with and without the presence of HA while using nZVI. It was reported that in both cases, although UV light enhanced As(V) removal, when UV light irradiation was used in the presence of HA the final As(V) removal was higher than that without HA. In contrast, Mak et al. (2009) studied the effects of hardness (i.e., Ca^{2+}) and alkalinity (i.e., HCO_3^-) on As(V) removal using ZVI with and without HA. The study reported that As(V) removal efficiencies increased in the simultaneous presence of Ca^{2+} and HCO_3^-, though this increase was diminished when HA was introduced to the solution. Gupta et al. (2012) studied the use of chitosan nZVI in arsenic removal and reported that As(III) and As(V) concentrations were reduced from 2 mg/l to less than 5 µg/l. Additionally, Gupta et al. (2012) reported that chitosan nZVI has an adsorption capacity of 4 ± 1.5 mg/g and 119 ± 2.6 mg/g at pH 7 for As(III) and As(V), respectively, while anions such sulfate, phosphate, and silicate had no significant interference in adsorption. Table 7 illustrates arsenic removal efficiencies when using nZVI in different operating conditions.

4.2.2 Iron Oxide Nanoparticles

Iron oxide NPs are said to be five to ten times more effective than micron-sized adsorbents in arsenic removal (Habuda-Stanic et al. 2015). The high surface area to volume ratio, along with the magnetic properties, low toxicity, and low price make iron oxide NPs a suitable option for metal removal from water (Habuda-Stanic et al. 2015). Tuutijärvi et al. (2009) examined the As(V) adsorption capacity of three maghemite NPs (γ-Fe_2O_3) of different particle size. The highest adsorption capacity of 50 mg/g at pH 3 was displayed by a synthesized maghemite, yet this capacity decreased as pH levels increased (i.e., 33.3, 20.0 & 12.5 mg/g at pH 5, 7 and 9 respectively), indicating a higher positive surface charge at acidic pH levels. As(V) adsorption capacities of the other two maghemites also decreased along pH increase. However, Tuutijärvi et al. (2009) reported a decline in magnetic properties as the particle size of maghemite decreased suggesting that the removal efficiency of maghemite NPs is not only dependent on pH levels (surface charge) and surface area, but also on particle size. Tang et al. (2011) reported a 98% As(III) removal using an ultrafine maghemite NPs, synthesized by a solvent thermal process at low temperature. Table 7 illustrates the performance of maghemite under different operating conditions.

The high arsenic adsorption capacity of magnetite (i.e., Fe_3O_4) NPs, along with its magnetic properties which allow for its simple dispersion and removal from aqueous solutions, makes researchers interested in its application for the treatment of arsenic-contaminated water (Habuda-Stanic et al. 2015). Akin et al. (2012) reported a 99.2% As(V) removal efficiency from a natural groundwater sample when using a synthesized magnetite NPs from waste red mud under optimal conditions. In contrast, Shipley et al. (2009) displayed that increasing magnetite concentrations enhance arsenic removal efficiencies. The same study reported that when using 0.5 g/l of magnetite NPs, it was possible to reduce As(V) and As(III) concentrations below 10 µg/l in less than 1 hour and 0.5 hours respectively. Furthermore, pH variations do not have significant effects on As(III) adsorption by magnetite, unlike As(V) adsorption which decreased as pH levels increased (Shipley et al. 2009).

Table 7. Arsenic removal efficiencies of various nanomaterials under different operating conditions.

Nano Material	pH	Temp. (°C)	Initial [As] concentration	Dosage	Surface Area (m²/g)	Particle size (nm)	Efficiency	Remarks	References
nZVI (NanoFe®)	7.8	≤ 30	As(V) = 1 mg/l	0.005–0.1 g/l	59–63 (BET)*	5–15	90% As(V)	Arsenic removal increased as dosage was increased up to 0.1 g/l. 90% removal of As(V) was achieved in 150 mins, using 0.05–0.1 g/l of NanoFe®. Experiment was performed in the absence of UV light and HA. Final arsenic concentration exceeded 10 µg/l.	Morgada et al. 2009
nZVI (NanoFe®)	7.8	≤ 30	As(V) = 1 mg/l	0.005–0.1 g/l	59–63 (BET)*	5–15	-	2 mg/l of HA present. 50% decrease in As(V) removal efficiency when compared to the same conditions but without HA.	Morgada et al. 2009
nZVI (NanoFe®)	7.8	≤ 30	As(V) = 1 mg/l	0.010 & 0.025 g/l	59–63 (BET)*	5–15	100% As(V)	UV light intensity = 5000 µW/cm². Complete As(V) removal at 240 mins of contact, with and without HA presence.	Morgada et al. 2009
Chitosan nZVI	7	25	As(III) and As(V) 1–20 mg/l	1 mg/l	69 (BET)*	-	-	Chitosan nZVI was shown to be effective in removing both As(III) and As(V) over a wide pH range (i.e., 3–10).	Gupta et al. 2012
Chitosan nZVI	7	25	As(V) = 171 µg/l & As(III) = 18 µg/l	0.1 g in a total aqueous volume of 40	69 (BET)*	-	100% As(V) & As(III)	As(III) and As(V) was not detected in the solution after 4 hours.	Gupta et al. 2012
γ-Fe₂O₃	3	21–25	As(V) = 1,000–11,000 µg/l	250 mg/l	51 (BET)*	18.4 ± 8.4	-	Commercially available (i.e., Sigma-Aldrich maghemite nano powder), pH pzc** = 7.5, As(V) adsorption capacity = 16.7 mg/g.	Tuutijärvi et al. 2009
γ-Fe₂O₃	3	21–25	As(V) = 1,000–11,000 µg/l	60 mg/l	90.4 (BET)*	12.1 ± 3.2	-	γ-Fe₂O₃ synthesized using mechanochemical method, pH pzc** = 57, As(V) adsorption capacity = 50 mg/g.	Tuutijärvi et al. 2009
γ-Fe₂O₃	3	21–25	As(V) = 1,000–11,000 µg/l	100 mg/l	203.2 (BET)*	3.8 ± 0.8	-	γ-Fe₂O₃ synthesized using sol-gel method, pH pzc = 5.7, As(V) adsorption capacity = 25.0 mg/g.	Tuutijärvi et al. 2009
Ultrafine γ-Fe₂O₃	7	-	As(III) = 0.115 mg/l	0.01 & 0.04 g/l	162	4–5	73% & 93% As(III)	As(III) removal efficiency 73% and 98.3% for a dosage of 0.01 and 0.04 g/l respectively. Final [As(III)] = 2 µg/l.	Tang et al. 2011
Ultrafine γ-Fe₂O₃	7	-	As(III) = 1 mg/l		162	4–5	64% & 98% As(III)	As(III) removal efficiency 64% and 98% for a dosage of 0.04 and 0.1 g/l respectively. Final [As(III)] = 2 µg/l.	Tang et al. 2011
Magnetite Fe₃O₄	8	-	As(V) = 100 µg/l	0.05–0.2 g/l	60 (BET)*	20	-	As(V) adsorption capacity is 38.2 and 77.6 µg/l for dosages of 0.05 and 0.2 g/l respectively. Contact time is 2 hours.	Shipley et al. 2009
Magnetite Fe₃O₄	8	-	As(III) = 100 µg/l	0.05–0.2 g/l	60 (BET)*	20	-	As(III) adsorption capacity is 37.5 and 87.9 µg/l for dosages of 0.05 and 0.2 g/l respectively. Contact time is 2 hours.	Shipley et al. 2009

Samples are dosed with arsenic in experimental studies in laboratories, elevating arsenic concentrations to the order of mg/l.

*BET: Brunauer-Emmett-Teller is a commonly used method to calculate specific surface area.

**pH pzc point of zero charge of the adsorbent, when pH > pH pzc the surface charge of γ-Fe₂O₃ and Fe₃O₄ becomes negative, which reduce As(V) adsorption.

4.3 Disposal of arsenic contaminated nanoparticles

NPs must be handled and managed when their saturation capacity is reached. Combustion is used to recover NPs when they are used for the treatment of organics and metals; however, in the case of arsenic bearing NPs this is not a viable option since arsenic oxides are volatile and could reach the atmosphere easily during combustion (Leist et al. 2000). As such, Nicomel et al. (2016) considered the stabilization/solidification (S/S) of arsenic bearing NPs followed by landfill disposal as one of the most attractive methods. Under this method, it is important to abide by regulations on leachability prior to landfill in order to prevent arsenic environmental contamination (Leist et al. 2000).

4.4 Regeneration and Reuse

A cost-effective alternative to disposal is the regeneration of NPs by desorption processes to enable their reuse in water treatment applications. Tuutijärvi et al. (2012) examined five different alkaline desorption solutions (i.e., NaOH, Na_2CO_3, Na_2HPO_4, $NaHCO_3$ and $CH_3COONa·3H_2O$) to regenerate maghemite NPs. The study reported that NaOH is the best desorption solution with >95% desorption efficiency. After six cycles of regeneration, a synthesized maghemite maintained 95% of its initial adsorption capacity, whereas the adsorption capacity of a commercial solution was only maintained for two to four cycles. However, the pH, temperature, desorption time, and concentration of the alkaline desorption solution were reported to affect the regeneration efficiency. Furthermore, Saiz et al. (2014) compared the desorption performance of HCl and NaOH, to regenerate arsenate bearing $Fe_3O_4@SiO_2$. Functionalization steps (i.e., protonation of amino groups and coordination of Fe^{3+}) were performed between adsorption and desorption, and it was reported that NaOH had a better desorption performance than HCl, where after five cycles the re-adsorption capacity decreased by 5.7% while the desorption yield also decreased by 26%.

5 Arsenic Mitigation Parameters

5.1 Background

Choosing the best arsenic removal technology requires the consideration of multiple parameters such as arsenic removal efficiencies, initial water composition, post-treatment residuals, capital cost, operation, and maintenance costs among others. It is important to account for these parameters when planning and designing an arsenic water treatment plant. The following sections provide information to aid the comparison of different treatment technologies performance under the aforementioned parameters.

5.2 Arsenic Removal Efficiencies and Water Composition

Table 8 displays the maximum arsenic removal efficiencies reported by the reviewed literature in this work, these efficiencies are mostly performed under optimum conditions which are detailed in previous sections. However, it is challenging to achieve such removal efficiencies in full scale operations due to multiple variables that could hinder arsenic removal. Table 9 provides a list of variables along with their impact on each of the arsenic removal technologies. The impact of some of these variables is evident (i.e., detrimental or enhancement), whereas the impact of others might not be as clear due to their varying impact on removal of As(III) and As(V). Nonetheless, this should not be a reason to demote the importance of a particular variable, but to refer to previous sections for more detailed information. Furthermore, three of the most important variables to consider are pH, initial As(III) and As(V) concentrations, and chemical dosages (i.e., oxidant, coagulant, and adsorbent) along with their impacts which are difficult to demonstrate in Table 9 alone should be supplemented with the previous sections.

Table 9 highlights the importance of water composition when selecting an arsenic treatment technology and outlines important constituents to measure in water before selecting a particular technology. As such, it is possible to utilize Table 9 as a guideline to predict which treatment technology would be promising based on the water composition of the arsenic contaminated water. Additionally, seasonal variations in water composition and temperature are also important to consider in the planning phase.

Table 8. Maximum arsenic removal efficiencies

Treatment Method	Remarks	Maximum efficiency	References
Coagulation-Flocculation	Coagulant: Ferric chloride & kaolinite	90% As(V) & 77% As(III)	Singh et al. 2015
Flocculation-Microfiltration	Flocculant: Ferric chloride	>90% arsenic	Han et al. 2002
Micellar enhanced ultrafiltration	Surfactant: hexadecylpyridinium chloride	96% arsenic	Singh et al. 2015
Nanofiltration	NF-90 commercial nano filter	98% As(V) & 59% As(III)	Uddin et al. 2007
Reverse Osmosis	SWHR flat sheet membrane	95% As(V) & 80% As(III)	Akin et al. 2011
Adsorption/Ion Exchange	Activated Alumina	98.4% As(V) & 96.2% As(III)	Singh et al. 2004
Nanoparticles	Chitosan nZVI	<100% As(V) & As(III)	Gupta et al. 2012

Table 9. Water composition variables effects on arsenic oxidation and treatment methods

Method Variables	Oxidation				Treatment			
	O₂, O₃, chemicals	Photochemical	Photocatalytic	Biological	Coagulation-flocculation	Membrane Filtration	Adsorption/Ion Exchange	NPs
pH	±±	±±	±±	±±	±±	±±	±±	±±
Chemical Dosage	±±	±±	±±	±±	±±		±±	±±
Organic matter	(-)	±±	±±		(-)	(-)		±±
Suspended Solids					(+)	(-)	(-)	
Manganese	(+)			(+)			(+)	
Nitrogen				±±				
Ammonia	(-)							
Hydrogen Peroxide		(+)						
Cations								
Calcium					(+)	(-)		(+)
Magnesium						(-)		
Iron & Ferric ions	(+)	(+)		(+)			(+)	
Anions								
Sulfide	(-)							
Bromide	(-)							
Chlorine							(-)	
Sulfite		(+)						
Sulfate			(-)		(-)		(-)	0
Silicate			(-)		(-)		(-)	0
Fluoride							(-)	
Phosphate			(-)		(-)		(-)	0
Bicarbonate			(-)		(-)			
Carbonate					(-)			

(+) The variable enhances the method in question; (-) The variable is detrimental for the method in question; ±± Considerable impact; 0 No significant impact.

5.3 Treatment systems: Costs and Environmental Evaluation

5.3.1 Background

Considerations other than arsenic removal efficiencies and water composition should be accounted for, such as costs and the environmental footprint of a treatment system. The cost of a treatment system is divided into two main categories: capital cost and operation and maintenance costs. The capital cost includes construction, machinery, equipment, and land costs among others. Whereas operational and maintenance costs include personnel salaries, energy consumption, water distribution, maintenance, residuals management, and chemical product consumption, transportation, and storage (Wang et al. 2018; Chen et al. 2004). The cost of a treatment system varies based on the employed treatment technology, service life, system design and size, flowrate, material for construction, and site conditions (Chen et al. 2004). Additionally, the environmental impact of greenhouse gas (GHG) emissions and post treatment residuals should be included in the decision-making process. The following sections provide leads for the cost effectiveness and environmental evaluation of different treatment systems that could be employed for arsenic treatment.

5.3.2 Coagulation-Filtration

A treatment system based on coagulation/flocculation would require tanks of adequate volumes to allow the treatment mechanisms to take place. These tanks occupy large land spaces relative to other treatment technologies. Hence, both land ownership and tank construction require significant investments, while operational cost is mainly due to the consumption of electricity and chemicals, which is also the main cause of GHG emissions. As such, choosing the best chemical to apply in optimal dosages is essential to reduce both the operating costs and environmental impact. On the other hand, filtration processes include costs such as membranes, tanks, pumps, pipes, electricity, and valves among others. However, the most significant costs are related to electricity consumption used for pumping and membrane costs. As such, MF and UF reduce the cost of the filtration unit compared to NF or RO, since they require smaller membrane areas and lower pressure requirements to achieve the required water flux. Figure 1 illustrates further the advantages and disadvantages of using a coagulation-filtration system.

5.3.3 Adsorption / Ion Exchange, NF, RO, Nanoparticles

5.3.3.1 Adsorption / Ion Exchange

To evaluate the economic and environmental feasibility of ion exchange systems, Amini et al. (2015) used a life cycle assessment and cost analysis. The study also compared fixed bed ion exchange systems using conventional resins with fully mixed flow ion exchange reactors. Amini et al. (2015) concluded that fixed bed systems consume smaller amounts of electricity, resin, and require less material transport, thus reducing their operational cost relative to mixed flow reactors. However, the environmental footprint (i.e., eutrophication, ecotoxicity) of fixed beds is higher with elevated waste production. As such, the costs to consider are the regeneration of resins, replacement, and disposal, along with power consumption and the management of backwash water. Figure 2 demonstrates further advantages and disadvantages of using adsorption/ion exchange, NF/RO, and NPs systems.
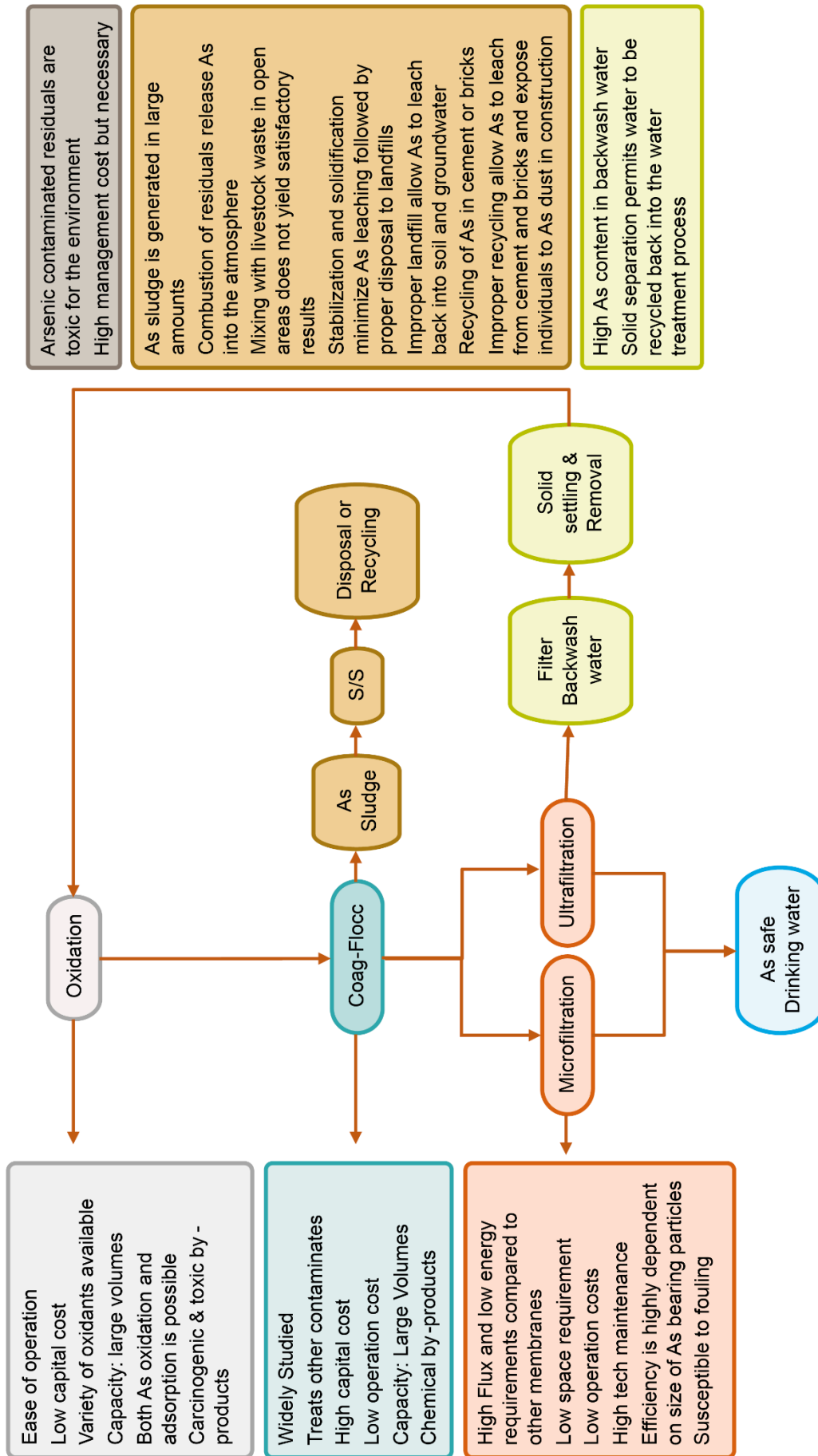
Figure 1. Treatment Systems, Oxidation, Coagulation-Flocculation, Micro and Ultrafiltration.

Arsenic contaminated residuals are toxic for the environment

High management cost but necessary

Residual NPs in treated water should be removed

Managment of arsenic contaminated NPs is costly

After several regeneration and reuse cycels the NPs lose adsorption capacity

Stabilization and solidification minimize As leaching followed by proper disposal to landfills

Improper landfill allow As to leach back into soil and groundwater

High As content in backwash water

Solid separation permits water to be recycled back into the water treatment process

Oxidation → **Removal of excess oxidant**

Nanoparticles → **As contaminated NPs** → **S/S & Disposal**

Reverse Osmosis

Nanofiltration → **Filter Backwash water**

Adsorption / Ion Exchange → **Regeneration Backwash** → **Solid Settling & Removal Treatment** → **Resin disposal**

Excess oxidant can cause damage to resin or membranes

Capacity: Small Volumes
Resin regeneration, replacement & disposal
Resign high cost
High tech maintenance
Secondary water quality effects
Susceptible to clogging

High removal efficiency
Small space requirement
Capacity: large scale (multiple units) RO requires more power than NF
High capital cost
High operation cost
High tech maintenance
Sensitive to oxidants
Susceptible to fouling
Can remove needed ions in water

High Efficiency
NPs can be regenerated and reused
Versatile use in water treatment
High cost of NPs
Secure supply of NPs should be considered
Removal mechanisms not fully understood
Residual NPs in treated water should be removed
Management of arsenic contaminated NPs is costly

Figure 2. Oxidation-Adsorption / Ion Exchange-Ultrafiltration.

5.3.3.2 Nanofiltration and Reverse Osmosis

In their research, Rajendran et al. (2021) assessed the economic feasibility of arsenic removal using nanofiltration. The study reported that nanofiltration is economically feasible for arsenic removal yet also asserted the importance of the membrane cost and service life, energy consumption cost, labor cost, fouling and maintenance of membrane, number of inhabitants and average water consumption per person. On the other hand, RO consumes more energy than NF since the process requires additional pressure, while also being highly sensitive to oxidants and susceptible to fouling.

5.3.3.3 Nanoparticles

Nanoparticles can achieve high arsenic removal efficiencies and also have a versatile role in water treatment applications. However, multiple considerations that have financial and environmental impacts should be taken into account, such as security of NPs supply, NPs residuals in treated water and arsenic bearing NPs residuals. A supply of large quantities of NPs at a reasonable price should be secured prior to their use in large scale water treatment operations. Additionally, proper management schemes should be put in place to ensure an environmentally safe disposal of arsenic bearing NPs.

6 Predesign Cost Estimation

6.1 Definition

The continuous upward trend of energy, labor, and material costs, along with the emphasis from the Safe Drinking Water Act of 1974 on the importance of costs when putting new regulations into effect has raised the interest of stakeholders in the cost effectiveness of unit processes (Clark 1982). However, definitive estimates are costly and require detailed process designs. As such, several preliminary cost estimation methods can be adopted, collectively called "predesign" estimates which rely on historical data and do not require complete designs (Clark 1982). When assessing multiple design alternatives, predesign estimates are very important in eliminating non-cost-effective alternatives and in concentrating research and resources into promising designs.

Predesign cost estimation is a challenging task by itself which is exacerbated further by the lack of data, scope definition, and experience of the analyst. Doloi (2011) reported on the main causes for inadequacies in cost estimation as the lack of practical knowledge by estimators, poor tender documentations, and the broad variability in subcontractor's prices. Doloi (2011) added that a strong consensus exists in the construction industry that cost estimates does not work, not only for technical reasons but also systematic biases due to psychological and political reasons. Furthermore, it is widely acknowledged that every water treatment plant is unique, which makes predesign estimates vulnerable since they rely on historical data. As such, it is very important not to confuse the objective of predesign cost estimates which is to provide an insight of the costs involved in water treatment applications and to prioritize cost effective unit processes that comply with water quality specifications.

6.2 Methodology

It is challenging to estimate the cost of a water treatment facility in the absence of a detailed design for treatment processes. As such, a less accurate but feasible preliminary cost estimate could be derived from published cost curves. Cost curves are developed by the U.S Environmental Protection Agency (USEPA), public utilities, consulting engineers and numerous publications which have accumulated a large amount of cost data that could be utilized to estimate the cost of various types of unit processes. The cost data of the unit processes could be aggregated to develop a Preliminary cost estimation for a water treatment facility. However, the data should be modified and updated according to the appropriate time, location, and size of the project. The data accumulation started in the 1900s, making it crucial to modify the data equivalent to the currency of 2022. Additional modifications are also necessary if the data are to be used in locations outside the United States. Sharma (2010) updated the cost data equivalent to September 2009 by using average yearly indexes provided by the Engineering News Record (ENR) and Bureau of Labor Statistics (BLS), as well as updating energy and labor costs. Furthermore, the cost data is split into two categories, with the first being suitable for facilities of 2,500 gallons per day (gpd)[1] to a capacity of 1 million gallons per day (mgd)[1] (i.e., small systems), while the second category suits capacities of 1 to 200 mgd (i.e., large systems). The ENR and BLS indexes could be further used to update the data for use in 2022, while operation, maintenance, labor, and energy prices from Nepal could be used to update the cost data for the desired location. However, construction cost indexes (CCI) are not available in Nepal, which creates the need for an alternative technique to adjust the construction costs for Nepal. As such, a geographical based surface interpolation method could be used to adjust the construction cost for Nepal (Zhang et al. 2014). However, this spatial interpolation is not in the scope of this work. Consequently, the construction cost according to location

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[1] The references used in developing the methodology relies on US customary units (e.g., mgd and gpd).

will be adjusted based on the difference in labor costs, since it is reasonable to expect cheaper labor costs in Nepal compared to the USA. The labor component of the construction cost will be adjusted according to Nepal, while the remaining construction cost components in Table 10 will be considered to remain the same.

Sharma (2010) used a forward regression analysis, which is a statistical tool to develop a relationship between independent variables and generate a single continuous dependable variable which is cost in cost estimation applications. By using the forward regression analysis, Sharma J. developed mathematical quantitative equations to describe the cost of water treatment facilities (i.e., dependable variable) as a function of feed capacities, surface area and volume of tanks among others (i.e., independent variables). The mathematical equations used to estimate the construction, operation, and maintenance costs are defined in Tables 11 and 12. Additionally, Equations 1, 2 & 3 are used to modify construction costs (Cc) to suitable time and location, while Equations 6, 7, 8, 9 and 10 are used to modify operation and maintenance costs (O&Mc). Whereas Equation 11 is used to calculate the chemical cost. However, some costs which are listed as special costs in Table 10 are excluded from the Cc equations due to large variations in different countries. As such, Equation 4 can be used to account for the special costs, while the total capital cost is calculated by using Equation 5.

It is important to highlight the shortcomings of this methodology that could reduce the accuracy of this predesign cost estimate. Firstly, since no Nepalese CCI exist, the adjustment of the material cost for Nepal 2022 was based on the index from the USA. This does not significantly reduce the accuracy of the estimate since the material cost is not very large when compared to the total of operation and maintenance (O&M) cost. Additionally, electricity costs in Nepal were assumed to be from 2021. Furthermore, no sufficient electricity and labor historical data for membrane filtration unit processes were found, which considerably affects the accuracy for O&M costs for membrane filtration.

All costs mentioned in Section 3, Table 4, and Table 5 are in US dollars. Furthermore, the currency conversion rate from Nepalese Rupee (NRs) to US dollars ($US) is 1 NRs = 0.0078 $US as of September 21, 2022.

Table 10. Construction, operation, and maintenance cost components (Sharma 2010)

Abbreviation	Construction Components	Abbreviation	Operation & Maintenance Components
A	Excavation & Site work	I	Electricity
B	Manufactured Equipment	J	Natural gas
C	Concrete	K	Diesel
D	Steel	L	Labor
E	Labor	M	Maintenance material
F	Pipes & Valves	X	Chemicals
G	Electrical & Instruments		
H	Housing		
Special Costs	Contractors' overhead & Profit Engineering costs Land costs Legal fiscal costs		

Table 11. Construction cost equations for 1 to 200 mgd capacity as of September 2009 in US customary units (metric system in Appendix) (Sharma 2010)

Treatment Unit	Cost Equation	Definitions	Cost Components
Raw water pumping	$Cc = 12{,}627\,x + 68{,}364$	x: plant capacity, (mgd), 1<x<200	B: 45%, E: 15%, F: 27%, G: 13%
Ozone generation system	$Cc = 0.0002x^3 - 1.3451\,x^2 + 4{,}147.8\,x + 212{,}878$	x: ozone generation capacity (lb/day), 10<x<3,500	B: 83%, E: 15%, H: 2%
Ozone contact chamber	$Cc = 6E{-}6\,x^2 + 5.181\,x + 41{,}901$	x: tank volume (ft³), 460<x<92,000	B: 6%, C: 21%, D: 24%, E: 49%
Liquid Alum. Feed System	$Cc = -0.0249\,x^2 + 280.21\,x + 54{,}288$	x: feed capacity (lb/hr), 5.4<x<5,400	B: 64%, E: 14%, F: 2%, G: 14%, H: 16%
Dry Aluminum Feed System	$Cc = 240.78\,x + 71{,}071$	x: feed capacity (lb/hr), 10<x<5,000	B: 41%, E: 4%, F: 5%, G: 3%, H: 47%
Ferric Sulfate Feed System	$Cc = -0.001\,x^2 + 177.92\,x + 63{,}605$	x: feed capacity (lb/hr), 13.3<x<6,600	B: 41%, E: 4%, F: 5%, G: 3%, H: 47%
Rapid Mix Coagulation Tank	$Cc = 0.0002\,x^2 + 55.443\,x + 29{,}756$	x: Tank volume (ft³), 100<x<20,000	A: 2%, B: 68%, C: 5%, D: 5%, E: 16%, G: 4%
Flocculation Tank	$Cc = -0.000004\,x^2 + 9.3239\,x + 160{,}468$	x: Tank volume (ft³), 1,800<x<500,000	A: 4%, B: 35%, C: 11%, D: 12%, E: 30%, G: 8%
Sedimentation Tank	$Cc = -0.0031\,x^2 + 155.61\,x + 78{,}329$	x: Surface area (ft²), 240<x<4,800	A: 4%, B: 26%, C: 11%, D: 22%, E: 24%, F: 12%, G: 1%
Membrane Filtration	$Cc = 188{,}15\,x^{0.7418} \times y$	x: plant flow (mgd), x>0.3; y: membrane flux (gpd/ft²)	B: 100%
Reverse Osmosis	$Cc = -0.0007\,x^2 + 1203.1\,x + 2{,}000{,}000$	x: plant capacity (1,000 gpd), 1<x<200	B: 81%, E: 6%, G: 6%, H: 7%
Backwash pumping Facility	$Cc = 11.94\,x^3 - 624.73\,x^2 + 23{,}021\,x + 67{,}631$	x: plant capacity (gpm), 1.8<x<33	B: 49%, E: 8%, F: 30%, G: 13%
Ion Exchange for softening	$Cc = -170.44\,x^2 + 283{,}732\,x + 37{,}413$	x: plant capacity (mgd), 1.1<x<122.6	B: 52%, C: 1%, E: 12%, F: 17%, G: 10%, H: 8%
Finished water pumping facility	$Cc = 0.1909\,x^3 - 85.9\,x^2 + 28{,}173\,x + 53{,}608$	x: plant capacity (mgd), 1<x<200	B: 17%, C: 2%, D: 2%, E: 39%, F: 28%, G: 6%, H: 6%
Gravity sludge thickener	$Cc = 0.0039\,x^4 - 1.0079\,x^3 + 82.537\,x^2 + 2{,}833.3\,x + 63{,}877$	x: diameter (ft), 20<x<150	A: 4%, B: 39%, C: 13%, D: 12%, E: 31%, G: 1%
Chemical sludge pumping	$Cc = 0.0004\,x^3 - 0.7412\,x^2 + 494.82\,x + 22{,}130$	x: pumping capacity (gpm), 5<x<1,250	B: 68%, E: 16%, F: 4%, G: 3%, H: 9%
In plant pumping	$Cc = 11.758\,x^2 + 12{,}402\,x + 84{,}932$	x: plant flow (mgd), 1<x<200	B: 17%, C: 2%, D: 2%, E: 39%, F: 28%, G: 6%, H: 6%
Management: Administration, lab, building maintenance	$Cc = 69{,}195\,x^{0.5523}$	x: plant capacity (mgd), 1<x<200	H: 100%

Table 12. Operation and maintenance Cost Equations for 1 to 200 mgd capacity as of September 2009 in US customary units (metric system in Appendix) (Sharma 2010)

Treatment Unit	Cost Equation	Definitions	Cost Components
Raw water pumping	O&Mc = 8,709.5 x + 23,723	x: plant capacity (mgd), 1<x<200	B: 45%, E: 15%, F: 27%, G: 13%
Ozone Generation System	O&Mc = -0.0093 x² + 354.32 x + 33,867	x: generation capacity (lb/day), 10<x<3,500	B: 83%, E: 15%, H: 2%
Liquid Alum. Feed System	O&Mc = 2,118 x^{0.293}	x: feed capacity (lb/hr), 5.4<x<5,400	I: 52%, L: 45%, M: 3%
Dry Aluminum Feed System	O&Mc = 0.0004 x²+ 44.575 x + 14,170	x: feed capacity (lb/hr), 10<x<5,000	I: 13%, L: 86%, M: 1%
Ferric Sulfate Feed System	O&Mc = 0.0003 x²+ 33.425 x + 14,152	x: feed capacity (lb/hr), 13.3<x<6,600	I: 13%, L: 86%, M: 1%
Rapid Mix Coag. Tank	O&Mc = 36.096 x + 18,928	x: Tank volume (ft³), 100<x<20,000	I: 84%, L: 16%
Flocculation Tank	O&Mc = -3E-7x² + 0.5692 x + 6,748	x: Tank volume (ft³), 18,000<x<500,000	I: 65%, L: 20%, M: 15%
Sedimentation Tank	O&Mc = -0.00003 x²+ 4.2485 x + 7,748	x: Surface area (ft²), 240<x<4,800	I: 3%, L: 88%, M: 9%
Membrane Filtration	Cost of membrane: 20.8/m² every 5 years (Costa et al. 2006)	-	M: 100%
Reverse Osmosis	O&Mc = 391,189 x + 207,533	x: plant capacity (mgd), 1<x<200	I: 57%, L: 1%, M: 42%
Backwash pumping Facility	O&Mc = 3E-9 x³ – 0.0001 x² + 4.8751 x + 10,915	x: pumping capacity (gpm), 140<x<28,000	I: 51%, L: 32%, M: 17%
Ion Exchange for softening	O&Mc = -12.039 x² + 18,861 x + 102,201	x: plant flow rate (mgd), 1.1<x<122.6	I: 14%, L: 36%, M: 50%
Finished water pumping Facilities	O&Mc = 16,097 x + 22,339	x: plant capacity (mgd), 1.5<x<300	I: 90%, L: 8%, M: 2%
Gravity sludge thickener	O&Mc = 0.4225 x² + 84.568 x + 4,554	x: diameter (ft), 20<x<150	I: 9%, L: 73%, M: 18%
Chemical sludge pumping	O&Mc = -0.0443 x² + 117.88 x + 6,447	x: pumping rate (gpm), 5<x<1,250	I: 43%, L: 31%, M: 26%
In plant pumping	O&Mc = 12,388 x + 23,506	x: plant flow (mgd), 1<x<200	B: 17%, C: 2%, D: 2%, E: 39%, F: 28%, G: 6%, H: 6%
Management: Administration, Lab, Building Maintenance	O&Mc = 88,589 x^{0.4529}	x: plant capacity (mgd), 1<x<200	I: 10%, L: 85%, M: 5%

Construction Cost (Sharma 2010):

$$Cc_{\text{2022 USA}} = Cc_{\text{2009 USA}} \times \frac{CCI_{\text{2022 USA}}}{CCI_{\text{2009 USA}}} \tag{1}$$

where:

$Cc_{\text{2022 USA}}$ = Construction cost in the United States of America (USA) as of September 2022

$Cc_{\text{2009 USA}}$ = Construction cost in the USA as of September 2009

$CCI_{\text{2022 USA}}$ = Construction cost Index in the USA for September 2022 = 13,173.43 (Engineering News Record Unlimited 2022)

$CCI_{\text{2009 USA}}$ = Construction cost Index in the USA for September 2009 = 8,586 (Engineering News Record Unlimited 2022)

On the other hand,

$$Cc_{\text{2022 USA}} = A \times Cc_{\text{2022 USA}} + B \times Cc_{\text{2022 USA}} + C \times Cc_{\text{2022 USA}} + D \times Cc_{\text{2022 USA}} \\ + E_{USA} \times Cc_{\text{2022 USA}} + F \times Cc_{\text{2022 USA}} + H \times Cc_{\text{2022 USA}} \tag{2}$$

where:

A, B, C, D = Construction components listed in Table 10

E_{USA} = Construction labour component in the USA ($US/year)

Similarly,

$$Cc_{\text{2022 Nepal}} = A \times Cc_{\text{2022 Nepal}} + B \times Cc_{\text{2022 Nepal}} + C \times Cc_{\text{2022 Nepal}} + D \times Cc_{\text{2022 Nepal}} \\ + E_{Nepal} \times Cc_{\text{2022 Nepal}} + F \times Cc_{\text{2022 Nepal}} + H \times Cc_{\text{2022 Nepal}} \tag{3}$$

where:

$Cc_{\text{2022 Nepal}}$ = Construction cost in Nepal as of September 2022

E_{Nepal} = Construction labour component in Nepal ($US/year)

$$E_{Nepal} = E_{USA} \frac{E\,tariff_{\text{Nepal 2022}}}{E\,tariff_{\text{USA 2022}}} \tag{4}$$

where:

$E\,tariff_{\text{Nepal 2022}}$ = Tariff of construction labor in Nepal as of September 2022 = 1.33 $US/hour (Paylab 2022a)

$E\,tariff_{\text{USA 2022}}$ = Tariff of construction labor in USA as of September 2022 = 18.8 $US/hour (Salary.com 2022)

$$Total\ Capital\ Cost = \sum(Cc) \times Factor\ for\ special\ costs \tag{5}$$

where:

Total capital cost = Capital required for construction

$\sum(Cc)$ = Summation of construction costs

Factor for special costs = factor that accounts for the special costs listed in table 10

Operation and Maintenance (Sharma 2010):

$$O\&Mc_{\text{ USA 2009}} = Electricity_{\text{ USA 2009}} + Labor_{\text{ USA 2009}} + Material_{\text{ USA 2009}} \qquad (6)$$

where:

$O\&Mc_{\text{ USA 2009}}$ = Operation and maintenance cost in the USA as of September 2009
$Electricity_{\text{ USA 2009}}$ = Electricity cost in the USA as of September 2009
$Labor_{\text{ USA 2009}}$ = Operation and maintenance labour cost in the USA as of September 2009
$Material_{\text{ USA 2009}}$ = Material cost in the USA as of September 2009

$$O\&Mc_{\text{ Nepal 2022}} = Electricity_{\text{ Nepal 2022}} + Labor_{\text{ Nepal 2022}} + Material_{\text{ Nepal 2022}} \qquad (7)$$

where:

$O\&Mc_{\text{ Nepal 2022}}$ = Operation and maintenance cost in Nepal as of September 2022
$Electricity_{\text{ Nepal 2022}}$ = Electricity cost in Nepal as of September 2022
$Labor_{\text{ Nepal 2022}}$ = Operation and maintenance labor cost in Nepal as of September 2022
$Material_{\text{ Nepal 2022}}$ = Material cost in Nepal as of September 2022

$$Electricity_{\text{ Nepal 2022}} = Electricity_{\text{ USA 2009}} \times \frac{Electricity\ tariff_{\text{ Nepal 2022}}}{Electricity\ tariff_{\text{ USA 2009}}} \qquad (8)$$

where:

$Electricity\ Tariff_{\text{ Nepal 2022}}$ = Electricity tariff in Nepal as of September 2022 = 0.071 \$US/kWh
(GlobalPetrolPrices.com 2022)
$Electricity\ Tariff_{\text{ USA 2009}}$ = Electricity tariff in the USA as of September 2009 = 0.0981 \$US/kWh
(Sharma 2010)

$$Labor_{\text{ Nepal 2022}} = Labor_{\text{ US 2009}} \times \frac{Labor\ tariff_{\text{ Nepal 2022}}}{Labor\ tariff_{\text{ USA 2009}}} \qquad (9)$$

where:

$Labor\ tariff_{\text{ Nepal 2022}}$ = Tariff for operation and maintenance labor in Nepal as of September 2022 =
1.92 \$US/hour (Paylab 2022b)
$Labor\ tariff_{\text{ USA 2009}}$ = Tariff for operation and maintenance labor in USA as of September 2009 =
45.82 \$US/hour (Sharma 2010)

$$Material_{\text{ Nepal 2022}} = Material_{\text{ USA 2009}} \times \frac{CCI_{\text{ 2022 USA}}}{CCI_{\text{ 2009 USA}}} \qquad (10)$$

Chemical Cost:

$$Chc = Required\ amount\ of\ chemical\ \times cost\ of\ chemical \qquad (11)$$

where;

Chc = Chemical cost

6.3 Example of predesign cost estimation

Design capacity: 10 mgd; Suitable for a population of 378,541 at 100 l/day/capita (UN 2010)
Design life: 15 years
Design Flow: 10 mgd = 37,854 m^3/day

Liquid Aluminum Feed System

From Literature review dosage 40 mg/l

Required Aluminum /hour: 63.092 kg/hour = 139.1 lb/hr

Construction cost (Cc)

For a Liquid Aluminum Feed System we have in Table 11

$Cc_{\text{2009 USA}}$ $= -0.0249\ x^2 + 280.21\ x + 54{,}288$

$= 92{,}783.50\ \$US;\ September\ 2009$

where:

x = Feed capacity = 139.1 lb/hr; (Table 13)

According to Equation 1 we get

$Cc_{\text{2022 USA}}$ $= 92{,}783.50 \times \dfrac{CCI_{\text{2022 USA}}}{CCI_{\text{2009 USA}}}$

$= 92{,}783.50 \times \dfrac{131{,}73.43}{8{,}586}$

$= 142{,}356.97\ \$US$

According to Equation 2 and together with

Cc components (Table 10): B: 64%, E: 14%, F: 2%, G: 4%, H: 16%

we get

$E_{\text{USA}} = 14\% \times 142{,}356.97 = 19{,}929.98\ \$US$

According to Equation 4 we get

E_{Nepal} $= 19{,}929.98 \times (1.33/18.8)$

$= 1{,}409.94\ \$US$

According to Equation 3 we get

$Cc_{\text{2022 Nepal}}$ $= 123\ 837\ \$US$

Operation and Maintenance Cost (O&Mc)

For a Liquid Aluminum Feed System we have in Table 12

$O\&Mc$ $\quad = 2{,}118\ x^{0.293}$; x: feed capacity (lb/hr); $5.4 < x < 5{,}400$

$\quad\quad\quad = 8{,}994$ \$US/year; September 2009

where:

x = feed capacity = 139.1 lb/hr (Table 13)

O&Mc components: I: 52%, L: 45%, M: 3% (Table 10)

Components:
- Electricity: 4,676.88 \$US/year at 0.0981/KWH
- Labor: 4,047.30 \$US/year at 45.82 \$US/hour
- Maintenance material: 269.82 \$US/year

According to Equations 8, 9 and 10,

$Electricity_{\,Nepal\ 2022} = 4{,}676.88 \times \dfrac{0.071}{0.0981} = 3{,}384.89$ \$US/year

$Labor_{\,Nepal\ 2022} \quad = 4{,}047.30 \times \dfrac{1.92}{45.82} = 169.60$ \$US/year

$Material_{\,Nepal\ 2022} \quad = 269.82 \times \dfrac{13{,}173.43}{8{,}586} = 413.98$ \$US/year

According to Equation 7,

$O\&Mc_{\,Nepal\ 2022} = 4{,}874.30$ \$US/year

Chemical cost (Chc)

Q = 10 mgd = 37,854 m³/day = 13,816,710 m³/year

where:
Q = Water Flow.

Required liquid Al = 40 mg/l = 552.66 tons/ year

Cost of Aluminum sulfate (Al sulfate) product = 0.63 \$US/kg in Nepal = 630 \$US/ton (Acharya 2022)

Or Cost of Poly Aluminum Chloride (Poly (Al) Cl⁻) = 0.90 \$US/kg in Nepal = 900 \$US/ton (Acharya 2022)

According to Equation 11,

Chc $\quad\quad = 552.66 \times$ cost of Al/ton

Aluminum sulfate Chc = 348,175.80 \$US/year

Poly Aluminum Chloride = 497,394 \$US/year

The design criteria of different unit processes are presented in Table 13, while the construction, operation & maintenance, and chemical cost of all other unit processes are presented in Table 14.

Table 13: Design criteria for unit processes in US customary units (metric system in Appendix)

Unit Process	Design Criteria
Raw water Pumping	Total Dynamic Head (TDH): 100 ft
O_3 generation system	O_3 generation capacity: 200 lb/day
O_3 contact chamber	Contact time 20 mins, needed volume = 18,635.55 ft^3, two tanks of Volume: 10,594.4 ft^3 each, for security compliance
Liquid Aluminum Feed	Chemical Dosage: 40 mg/l, Aluminum sulfate = 0.285 \$US/lb or Poly Aluminum Chloride = 0.4 \$US/lb, feed capacity = 139.1 lb/hr
Dry Aluminum Feed	Chemical Dosage: 40 mg/l, Aluminum sulfate = 0.285 \$US/lb or Poly Aluminum Chloride = 0.4 \$US/lb
Ferric sulfate Feed, 25 mg/l	Chemical Dosage: 25 mg/l, Ferric Sulfate = 2.798 \$US/lb (Acharya 2022)
Rapid Mix Coagulation	Gradient velocity (G) = 900 s^{-1}, Retention time = 25 sec, Required volume = 387.04 ft^3, two tanks of volume: 282.52 ft^3 each, for security compliance
Flocculation	G = 80 s^{-1}, Retention time = 35 minutes, Required volume = 41,745.47 ft^3, two tanks of volume = 21,188.8 ft^3 each, for security compliance.
Sedimentation	Retention time = 2.5 hours, Required volume = 139,254.56 ft^3, Depth of tank: 8.2 ft & Width of tank: 32.8 ft (maximum) Length/width ratio = 2 (optional), Length = 65.62 ft, surface area of 1 tank = 2,152.78 ft^2, Volume of 1 tank = 17,657.4 ft^3, Number of tanks: 8
Membrane Filtration	Membrane Flux: 2.05 ft^3/day/ft^2, Membrane cost: 1.932 \$US/ft^2 replaced every 5 years
Reverse Osmosis	Plant capacity
Backwash Pumping facility	Pumping capacity: 2887.5 ft^3/day
Ion Exchange for softening	Plant capacity
Finished water pumping facility	Total Dynamic Head (TDH): 100 ft
In Plant Pumping	Plant Flow
Management	Administration, Laboratory, Building Maintenance
Gravity Sludge Thickener	Diameter: 150 ft
Chemical Sludge Pumping	Pumping capacity: 38500.05 ft^3/day

Table 14. Construction, Operation, and Maintenance Cost Estimates for Unit Processes

Unit Process	Construction cost ($US)			O&M cost ($US/year)						Chemical cost
	Cc 2009 USA	Cc 2022 USA	Cc 2022 Nepal	O&Mc 2009 US			O&Mc 2022 Nepal			Chc 2022 Nepal
				Elec.	Labor	Material	Elec.	Labor	Material (USA)	
Raw water Pumping	194,634	298,625.37	257,000.7	110,818 / 89,762.58	16,622.7 / 16,697.4	4,432.72 / 8,348.72	89,848.64 / 64,965.78	696.55 / 699.68	6,801.09 / 12,809.37	
O₃ generation System	990,234	1,519,308	1,307,535	104,359 / 79,312.8			86,273.09 / 57,402.74			
O₃ contact chamber	194,928.06	299,076.54	299,076.54	- / -	-	-	-	-	-	-
Liquid Alum. Feed	92,783.5	142,356.97	123,837	8,994 / 4,676.88	4,047.3	269.82	4,874.3 / 3,384.89	169.6	413.99	(Al)sulfate: 348,175.8 $US/y; Poly (Al) Cl-: 497,394 $US/y
Dry Alum. Feed	104,563.5	160,430.93	154,467.7	20,378.2 / 2,649.2	17,525.25	203.8	3,477.55 / 1,917.36	734.36	312.69	(Al) sulfate: 348,175.8$/v; Poly (Al) Cl-: 497,394 $US/y
Ferric Sulfate Feed	79,065.8	121,310.02	116,800.9	17,060.24 / 2,217.84	14,671.8	170.6	2,911.27 / 1,605.16	614.8	261.75	2131,179.7 $US/y
Rapid Mix Coagulation	90,924.8	139,505.18	118,763.6	58,286.4 / 48,960.6	9,325.82	-	45,308.78 / 35,435.29	390.78	-	-
Flocculation	712,470	1,093,136.93	788,397.7	37,347.8 / 24,276	7,469.56	5,602.02	31,179.14 / 17,569.79	313	8,595.14	
Sedimentation	3,191,656	4,896,931.854	3,804,818	134,040 / 4,021.2	117,955.2	12,063.6	27,140.95 / 2,910.34	4,942.69	18,509.09	-
Membrane Filtration	1,557,384	2,389,481.61	2,389,481.61	257,920 / -	-	257,920	395,724.56 / -	-	395,724.56	-

Table 14. Continued

Unit Process	Construction cost ($US)			O&M cost ($US/year)						Chemical cost
				O&Mc 2009 US			O&Mc 2022 Nepal			
	Cc 2009 USA	Cc 2022 USA	Cc 2022 Nepal	Elec.	Labor	Material	Elec.	Labor	Material (USA)	Chc 2022 Nepal
Reverse osmosis	13,961,000	21,420,248.8	20,225,955.99	4,119,423			4,810,486.6			-
				2,348,071.11	41,194.2	1,730,157.6	1,699,419.46	1,726.16	2,654,566.74	
Backwash Pumping facility	312,679.25	479,741.24	444,077.3	10,988.1			8,154.62			
				5,603.93	3,516.2	1,868	4,055.85	147.34	2,866.058	
Ion Exchange for softening	2,857,689	4,384,529	3,895,610	289,607.1			263,737.6			
				40,544.99	104,258.6	144,804	29,344.48	4,368.76	222,171.6	
Finished water pumping Facility	326,938.9	501,619.7	319,829	183,309			157,595.54			-
				164,978.1	14,664.72	3,666.18	119,403.11	614.49	5,625	
In Plant Pumping	210,127.7	322,397.23	205,558.1	147,386			123,749.6			-
				126,752	16,212.5	4,421.6	91,736.92	679.35	6,784.024	
Management	246,816.8	378,689	378,689	251,350.5			51,294.56			-
				25,135.05	213,647.9	12,567.6	18,191.52	8,952.51	19,282.37	
Gravity Sludge Thickener	918,667	1,409,503.31	1 003,471	26,745.45			10,412.86			-
				2,407.09	19,524.17	4,814.2	1,742.13	818.13	7,386.39	
Chemical Sludge Pumping	94,646	145,214.59	123,624.1	28,251			22,781.67			-
				12,147.95	8,757.81	7,345.26	8,792.09	366.98	11,269.78	

6.4 Cost of a sample treatment train

Multiple treatment train configurations could be formed out of the several unit processes presented in Table 11. Figure 3 displays one possible treatment train configuration for the purpose of calculating the total construction cost, operation & maintenance cost, chemical cost, and the total capital cost. However, it is important to mention that further statistical and financial analysis should be utilized to assess the cost effectiveness of a treatment train configuration.

$Cc_{\text{2022 Nepal}}$ = 11,564,158.65 $US

$O\&Mc_{\text{2022 Nepal}}$ = 1,054,338.31 $US/year

$Chemical\ cost_{\text{2022 Nepal}}$ = 348,175.80 $US/year

Contractor's overhead & profit, engineering cost, land cost, and legal fiscal cost are not included in the construction cost since they vary widely based on the region or country. Thus, a special cost factor is required to account for these costs. Sharma (2010) considered the factor for special costs to be 28% of the total construction cost. For example, if the special cost factor is assumed to be 28%, the total capital cost would be as follows:

$Total\ Capital\ Cost$ = $\sum(Cc)$ Factor for special costs

= 11,564,158.65 × 1.28

= 14,802,123.072 $US.

However, the special cost factor value must be assigned based on data and existing regulations specific for the location of the water treatment plant.
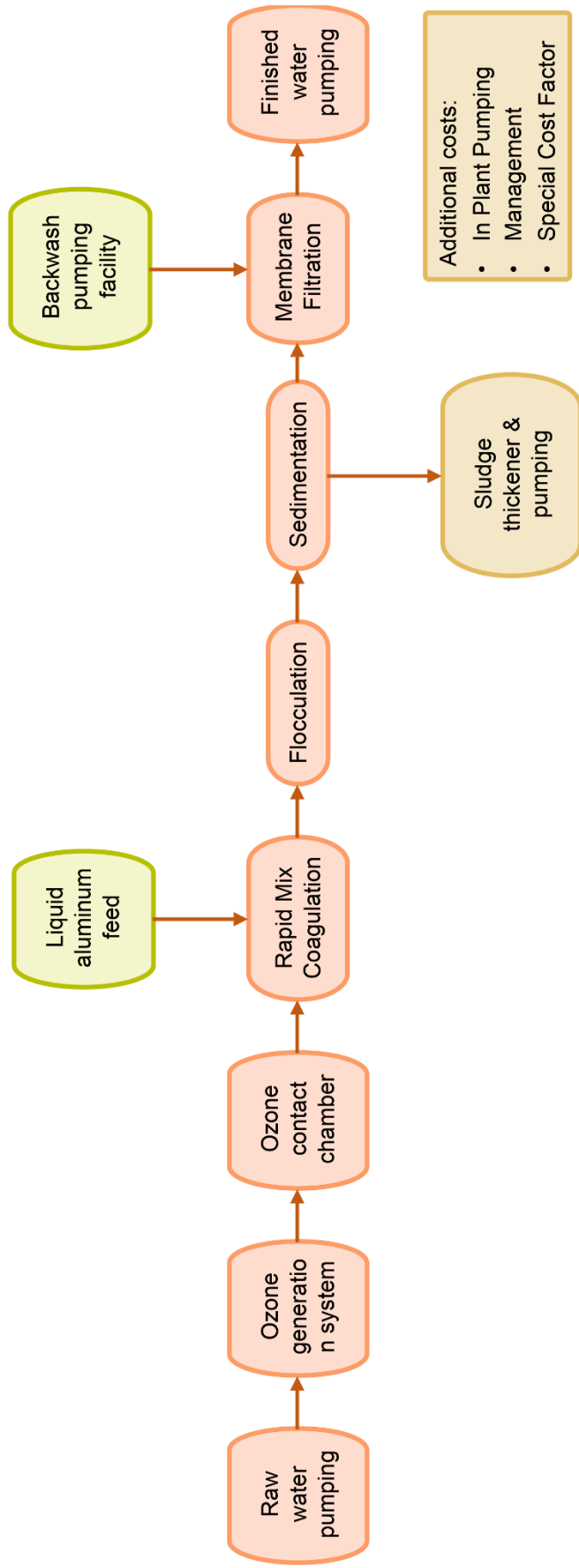
Figure 3. Sample configuration of a water treatment plant.

Raw water pumping → Ozone generation system → Ozone contact chamber → Rapid Mix Coagulation → Flocculation → Sedimentation → Membrane Filtration → Finished water pumping

Liquid aluminum feed → Rapid Mix Coagulation

Backwash pumping facility → Membrane Filtration

Sedimentation → Sludge thickener & pumping

Additional costs:
- In Plant Pumping
- Management
- Special Cost Factor

7 Conclusions

Elevated exposure to arsenic is linked to several adverse health impacts, while arsenic toxicity levels are related to arsenic speciation and oxidation states. The inorganic trivalent As(III) is more genotoxic, cytotoxic, soluble, and mobile than other arsenic species. Hence, the conventional arsenic removal technologies mostly require the oxidation of As(III) into the less mobile As(V), which can be performed by chemical, photochemical, photocatalytic, or biological oxidation. A coagulation-flocculation process can increase the size of arsenic bearing particles which can be separated by precipitation or filtration, though the latter process is considered to yield better arsenic removal efficiencies. The removal efficiency of this process mainly relies on the coagulant type and dosage, while the water composition and pH levels could be detrimental to the filtration membranes. Additionally, this process results in large amounts of arsenic contaminated sludge which is costly to manage. Adsorption onto ion exchange resins is another process that can remove arsenic, while adequate water composition and pH levels are critical for effective arsenic removal. In addition, other considerations should be accounted for such as secondary water impacts, arsenic contaminated backwash water, spent regenerate, and resins regeneration and disposal.

The use of nanoparticle adsorbents is promising in arsenic removal applications, due to their high surface areas, reactivity, and specificity. However, nanoparticles require a surface stabilizer to prevent them from agglomerating in aqueous solutions which significantly reduce their adsorption capacity. Moreover, securing a large-scale supply of nanoparticles could still be challenging due to a lack of availability, while the safe disposal of nanoparticles is of high concern to prevent adverse environmental impacts.

Apart from the initial arsenic concentration and speciation, it is important to identify the constituents and the seasonal variations of the contaminated source water, as it could be detrimental to the treatment process. Additionally, effective and proper management plans to handle arsenic bearing residuals should be put in place to prevent arsenic from leaching back into the soil and water. Furthermore, the cost of the treatment system is also an important consideration in the decision-making process. However, with definitive estimates being costly and requiring detailed process designs, a predesign cost estimate demonstrated in this report could eliminate non cost-effective alternatives. This would aid in focusing research and resources onto cost effective designs that can achieve the desired water quality standards.

Table A1. Construction cost equations for 3,785.4 to 757,080 m3/day capacity as of September 2009 in international system of units (Sharma 2010). (US customary units in Table 11)

Treatment Unit	Cost Equation	Definitions	Cost Components
Raw water pumping	$Cc = 3.3357 \, x + 68,364$	x: plant capacity, (m³/day), $3,785.4 < x < 757,080$	B: 45%, E: 15%, F: 27%, G: 13%
Ozone generation system	$Cc = 2.1430E\text{-}3 \, x^3 - 6.5376 \, x^2 + 9,144.323 \, x + 212,878$	x: ozone generation capacity (kg/day), $4.536 < x < 1,587.575$	B: 83%, E: 15%, H: 2%
Ozone contact chamber	$Cc = 7.4827E\text{-}3 \, x^2 + 182.965 \, x + 41,901$	x: tank volume (m³), $13.025 < x < 2,605.15$	B: 6%, C: 21%, D: 24%, E: 49%
Liquid Alum. Feed System	$Cc = -0.1210 \, x^2 + 617.757 \, x + 54,288$	x: feed capacity (kg/hr), $2.449 < x < 2,449$	B: 64%, E: 14%, F: 2%, G: 14%, H: 16%
Dry Aluminum Feed System	$Cc = 530.83 \, x + 71,071$	x: feed capacity (kg/hr), $4.535 < x < 2,267.96$	B: 41%, E: 4%, F: 5%, G: 3%, H: 47%
Ferric Sulfate Feed System	$Cc = -4.86E\text{-}3 \, x^2 + 392.246 \, x + 63,605$	x: feed capacity (kg/hr), $6.03 < x < 2,993.7$	B: 41%, E: 4%, F: 5%, G: 3%, H: 47%
Rapid Mix Coagulation Tank	$Cc = 0.249 \, x^2 + 1,957.954 \, x + 29,756$	x: Tank volume (m³), $2.83 < x < 566.33$	A: 2%, B: 68%, C: 5%, D: 5%, E: 16%, G: 4%
Flocculation Tank	$Cc = -4.988E\text{-}3 \, x^2 + 329.27 \, x + 160,468$	x: Tank volume (m³), $50.97 < x < 14,158.4$	A: 4%, B: 35%, C: 11%, D: 12%, E: 30%, G: 8%
Sedimentation Tank	$Cc = -0.359 \, x^2 + 1,674.972 \, x + 78,329$	x: Surface area (m²), $22.296 < x < 445.934$	A: 4%, B: 26%, C: 11%, D: 22%, E: 24%, F: 12%, G: 1%
Membrane Filtration	$Cc = 1,023.732 \, x^{0.7418} \times y$	x: plant flow (mgd), $x > 1,135.62$; y: membrane flux ((m³/day)/m²)	B: 100%
Reverse Osmosis	$Cc = -4.885E\text{-}5 \, x^2 + 317.826 \, x + 2,000,000$	x: plant capacity (m³/day), $3,785.4 < x < 757,080$	B: 81%, E: 6%, G: 6%, H: 7%
Backwash pumping Facility	$Cc = 0.07372 \, x^3 - 21.025 \, x^2 + 4,223.26 \, x + 67,631$	x: plant capacity (m³/day), $9.812 < x < 179,883$	B: 49%, E: 8%, F: 30%, G: 13%
Ion Exchange for softening	$Cc = -1.1895E\text{-}5 \, x^2 + 74.9543 \, x + 37,413$	x: plant capacity (m³/day), $4,163.94 < x < 464,090$	B: 52%, C: 1%, E: 12%, F: 17%, G: 10%, H: 8%
Finished water pumping facility	$Cc = 3.52E\text{-}12 \, x^3 - 5.995E\text{-}6 \, x^2 + 7.443 \, x + 53,608$	x: plant capacity (m³/day), $3,785.4 < x < 757,080$	B: 17%, C: 2%, D: 2%, E: 39%, F: 28%, G: 6%, H: 6%
Gravity sludge thickener	$Cc = 0.452 \, x^4 - 35.594 \, x^3 + 888.42 \, x^2 + 9,295.6 \, x + 63,877$	x: diameter (m), $6.096 < x < 45.72$	A: 4%, B: 39%, C: 13%, D: 12%, E: 31%, G: 1%
Chemical sludge pumping	$Cc = 2.47E\text{-}6 \, x^3 - 0.025 \, x^2 + 90.78 \, x + 2,2130$	x: pumping capacity (m³/day), $27.255 < x < 6,813.75$	B: 68%, E: 16%, F: 4%, G: 3%, H: 9%
In plant pumping	$Cc = 8.21E\text{-}7 \, x^2 + 3.28 \, x + 84,932$	x: plant flow (m³/day), $3,785.4 < x < 757,080$	B: 17%, C: 2%, D: 2%, E: 39%, F: 28%, G: 6%, H: 6%
Management: Administration, lab, building maintenance	$Cc = 730.95 \, x^{0.5523}$	x: plant capacity (m³/day), $3,785.4 < x < 757,080$	H: 100%

Table A2. Operation and maintenance Cost Equations for 3,785.4 to 757,080 m³/day capacity as of September 2009 in international system of units (Sharma 2010). (US customary units in Table 12)

Treatment Unit	Cost Equation	Definitions	Cost Components
Raw water pumping	O&Mc = 2.3008 x + 23,723	x: plant capacity, (m³/day), 3,785.4<x<757,080	B: 45%, E: 15%, F: 27%, G: 13%
Ozone Generation System	O&Mc = -0.0452 x^2 + 781.14 x + 33,867	x: ozone generation capacity (kg/day), 4.536<x<1,587.575	B: 83%, E: 15%, H: 2%
Liquid Alum. Feed System	O&Mc = 2 670 $x^{0.293}$	x: feed capacity (kg/hr), 2.449<x<2,449	I: 52%, L: 45%, M: 3%
Dry Aluminum Feed System	O&Mc = 1.945E-3 x^2+ 98.272 x + 14,170	x: feed capacity (kg/hr), 4.535<x<2,267.96	I: 13%, L: 86%, M: 1%
Ferric Sulfate Feed System	O&Mc = 1.458E-3 x^2+ 73.689 x + 14,152	x: feed capacity (kg/hr), 6.03<x<2,993.7	I: 13%, L: 86%, M: 1%
Rapid Mix Coag. Tank	O&Mc = 1274.72 x + 18,928	x: Tank volume (m³), 2.83<x<566.33	I: 84%, L: 16%
Flocculation Tank	O&Mc = -3.741E-4 x^2 + 20.10 x + 6,748	x: Tank volume (m³), 50.97<x<14,158.4	I: 65%, L: 20%, M: 15%
Sedimentation Tank	O&Mc = -3.475E-3 x^2 + 45.73 x + 7,748	x: Surface area (m²), 22.296<x<445.934	I: 3%, L: 88%, M: 9%
Membrane Filtration	Cost of membrane: 20.8/m² every 5 years (Costa et al. 2006)	-	M: 100%
Reverse Osmosis	O&Mc = 103.3415 x + 207,533	x: plant capacity (m³/day), 3,785.4<x<757,080	I: 57%, L: 1%, M: 42%
Backwash pumping Facility	O&Mc = 1.852E-11 x^3 – 3.365E-6 x^2 + 0.8944 x + 10,915	x: pumping capacity (m³/day), 763.14<x<152,628	I: 51%, L: 32%, M: 17%
Ion Exchange for softening	O&Mc = -8.402E-7 x^2 + 4.9826 x + 102,201	x: plant capacity (m³/day), 4,163.94<x<464,090	I: 14%, L: 36%, M: 50%
Finished water pumping Facilities	O&Mc = 4.253 x + 22,339	x: plant capacity (mgd), 5,678.1<x<1,135,620	I: 90%, L: 8%, M: 2%
Gravity sludge thickener	O&Mc = 4.5478 x^2 + 277.45 x + 4,554	x: diameter (m), 6.096<x<45.72	I: 9%, L: 73%, M: 18%
Chemical sludge pumping	O&Mc = -1.49E-3 x^2 + 21.625 x + 6,447	x: pumping capacity (m³/day), 27.255<x<6,813.75	I: 43%, L: 31%, M: 26%
In plant pumping	O&Mc = 3.273 x + 23,506	x: plant capacity (m³/day), 3,785.4<x<757,080	B: 17%, C: 2%, D: 2%, E: 39%, F: 28%, G: 6%, H: 6%
Management: Administration, Lab, Building Maintenance	O&Mc = 2,122.525 $x^{0.4529}$	x: plant capacity (m³/day), 3,785.4<x<757,080	I: 10%, L: 85%, M: 5%

Table A3. Design criteria for unit processes in international system of units (US customary units in Table 13)

Unit Process	Design Criteria
Raw water Pumping	Total Dynamic Head (TDH): 30.48 m
O_3 generation system	O_3 generation capacity: 90.7184 kg/day
O_3 contact chamber	Contact time 20 mins, needed volume = 527.7 m³, Two tanks of Volume: 300 m³ each, for security compliance
Liquid Aluminum Feed	Chemical Dosage: 40 mg/l, Aluminum sulfate = 0.63 $US/kg or Poly Aluminum Chloride = 0.90 $US/kg
Dry Aluminum Feed	Chemical Dosage: 40 mg/l, Aluminum sulfate = 0.63 $US/kg or Poly Aluminum Chloride = 0.90 $US/kg
Ferric sulfate Feed, 25 mg/l	Chemical Dosage: 25 mg/l, Ferric Sulfate = 6.17 $US/kg (Acharya 2022)
Rapid Mix Coagulation	Gradient velocity (G) = 900 s⁻¹, Retention time = 25 sec, Required volume = 10.96 m³, two tanks of volume: 8 m³ each, for security compliance
Flocculation	G = 80 s⁻¹, Retention time = 35 minutes, Required volume = 1,182.1 m³, two tanks of volume = 600 m³ each, for security compliance
Sedimentation	Retention time = 2.5 hours, Required volume = 3,943.25 m³, Depth of tank: 2.5 meters & Width of tank: 10 meters (maximum) Length/width ratio = 2 (optional), Length = 20 meters, surface area of 1 tank = 200 m², Volume of 1 tank = 500 m³, Number of tanks: 8
Membrane Filtration	Membrane Flux: 0.611175 m³/day/m², Membrane cost: 20.8 $US/m² replaced every 5 years
Reverse Osmosis	Plant capacity
Backwash Pumping facility	Pumping capacity: 81.765 m³/day
Ion Exchange for softening	Plant capacity
Finished water pumping facility	Total Dynamic Head (TDH): 30.48 m
In Plant Pumping	Plant Flow
Management	Administration, Laboratory, Building Maintenance
Gravity Sludge Thickener	Diameter: 45.72 m
Chemical Sludge Pumping	Pumping capacity: 1,090.2 m³/day

Abbreviations

As	Arsenic
WHO	World Health Organization
As(III)	Arsenite
As(V)	Arsenate
DMA	Dimethyl arsenic acid
MMA	Monomethyl arsenic acid
NDWQS	National drinking water quality standard
DALYs	Disability adjusted life years
O_2, O_3, Cl_2, ClO^-, ClO_2	Dioxygen, ozone, chlorine, hypochlorite, chlorine dioxide
Fe, Mn	Iron, Manganese
Fe(0), Fe(II), Fe(III), Fe(VI)	Iron pentacarbonyl, ferrous oxide, ferric, ferrate
UV	Ultraviolet
TOC	Total organic carbon
PCO	Photocatalytic oxidation
NOM	Natural organic matter
MF	Microfiltration
UF	Ultrafiltration
NF	Nanofiltration
RO	Reverse Osmosis
MWCO	Molecular weight cut off
MEUF	Micellar enhanced ultrafiltration
SWHR	Sea water high rejection
BW	Brackish water
BET	Brunauer-Emmett-Teller
TCLP	Toxicity Characteristic leaching procedure
S/S	solidification/stabilization
ZVI	zero valent iron
NPs	nanoparticles
nZVI	Zero-valent iron nanoparticles
HA	Humic acid
γ-Fe_2O_3	Maghemite nanoparticles
GHG	Greenhouse gas
USEPA	United States Environmental Protection Agency

ENR	Engineering News Record
BLS	Bureau of Labor Statistics
CCI	Construction cost index
Cc	Construction cost
Cc $_{2022\ USA}$	Construction cost in the USA, as of September 2022
Cc $_{2009\ USA}$	Construction cost in the USA, as of September 2009
CCI $_{2022\ USA}$	Construction cost Index in the USA for September 2022
CCI $_{2009\ USA}$	Construction cost Index in the USA for September 2009
Cc $_{2022\ Nepal}$	Construction cost in Nepal, as of September 2022
E $_{Nepal}$, E $_{USA}$	Construction labour components in Nepal and in the USA
E tariff $_{Nepal\ 2022}$	Tariff of construction labor in Nepal as of September 2022
E tariff $_{USA\ 2022}$	Tariff of construction labor in USA as of September 2022
O&Mc	Operation and Maintenance cost
O&Mc $_{USA\ 2009}$	Operation and maintenance cost in the USA, as of September 2009
Electricity $_{USA\ 2009}$	Electricity cost in the USA, as of September 2009
Labor $_{USA\ 2009}$	Operation and maintenance labor cost in the USA, as of September 2009
Material $_{USA\ 2009}$	Material cost in the USA, as of September 2009
O&Mc $_{Nepal\ 2022}$	Operation and maintenance cost in Nepal, as of September 2022
Electricity $_{Nepal\ 2022}$	Electricity cost in Nepal, as of September 2022
Labor $_{Nepal\ 2022}$	Operation and maintenance labor cost in Nepal, as of September 2022
Material $_{Nepal\ 2022}$	Material cost in Nepal, as of September 2022
Electricity Tariff $_{Nepal\ 2022}$	Electricity tariff in Nepal as of September 2022
Electricity Tariff $_{USA\ 2009}$	Electricity tariff in the USA as of September 2009
Labor tariff $_{Nepal\ 2022}$	Tariff for operation and maintenance labor in Nepal as of September 2022
Labor tariff $_{USA\ 2009}$	Tariff for operation and maintenance labor in USA as of September 2009
Chc	Chemical cost
Q	Water Flow
Chc $_{2022\ Nepal}$	Chemical cost in Nepal, as of September 2022
Units	
gpd	Gallons per day
mgd	Million gallons per day
NRs	Nepalese Rupees
$US	United States Dollars

References

Acharya, B. 2022. Personal communication email, September 2022. Supreme Associates Traders. http://www.sat.com.np/index.php

Akin, I., Arslan, G., Tor, A., Cengeloglu, Y. & Ersoz, M. 2011. Removal of arsenate As(V) and arsenite As(III) from water by SWHR and BW-30 reverse osmosis. Desalination 281: 88–92. https://doi.org/10.1016/j.desal.2011.07.062

Akin, I., Arslan, G., Tor, A., Ersoz, M. & Cengeloglu, Y. 2012. Arsenic(V) removal from underground water by magnetic nanoparticles synthesized from waste red mud. Journal of Hazardous Materials 235: 62–68. https://doi.org/10.1016/j.jhazmat.2012.06.024

Altundogan, H. S., Altundogan, S., Tumen, F. & Bildik, M. 2000. Arsenic removal from aqueous solutions by adsorption on red mud. Waste Management 20(8): 761–767. https://doi.org/10.1016/S0956-053X(00)00031-3

Amen, R., Bashir, H., Bibi, I., Shaheen, S. M., Niazi, N. K., Shahid, M., Hussain, M. M., et al. 2020. A critical review on arsenic removal from water using biochar-based sorbents: The significance of modification and redox reactions. Chemical Engineering Journal 396: 125195. https://doi.org/10.1016/j.cej.2020.125195

Amini, A., Kim, Y., Zhang, J., Boyer, T. & Zhang, Q. 2015. Environmental and economic sustainability of ion exchange drinking water treatment for organics removal. Journal of Cleaner Production 104: 413–421. https://doi.org/10.1016/j.jclepro.2015.05.056

Aremu, J. O. 2020. Target separation of arsenic from contaminated raw water (Thesis, Doctor of Philosophy, PhD). University of Waikato, New Zealand. http://hdl.handle.net/10289/13827

Baloch, M. Y. J., Talpur, S. A., Talpur, H. A., Iqbal, J., Mangi, S. H. & Memon, S. 2020. Effects of Arsenic Toxicity on the Environment and Its Remediation Techniques: A Review. Journal of Water and Environment Technology 18(5): 275–289. https://doi.org/10.2965/jwet.19-130

Banerjee, K., Amy, G. L., Prevost, M., Nour, S., Jekel, M., Gallagher, P. M. & Blumenschein, C. D. 2008. Kinetic and thermodynamic aspects of adsorption of arsenic onto granular ferric hydroxide (GFH). Water Research 42(13): 3371–3378. https://doi.org/10.1016/j.watres.2008.04.019

Bang, S., Patel, M., Lippincott, L. & Meng, X. 2005. Removal of arsenic from groundwater by granular titanium dioxide adsorbent. Chemosphere 60(3): 389–397. https://doi.org/10.1016/j.chemosphere.2004.12.008

Bissen, M. & Frimmel, F. H. 2003. Arsenic – a Review. Part II: oxidation of arsenic and its removal in water treatment. Acta hydrochimica et hydrobiologica 31(2): 97–107. https://doi.org/10.1002/aheh.200300485

Brandhuber, P. & Amy, G. 2001. Arsenic removal by a charged ultrafiltration membrane – influences of membrane operating conditions and water quality on arsenic rejection. Desalination 140(1): 1–14. https://doi.org/10.1016/S0011-9164(01)00350-2

Camacho, J., Wee, H. Y., Kramer, T. A. & Autenrieth, R. 2009. Arsenic stabilization on water treatment residuals by calcium addition. Journal of Hazardous Materials 165(1–3): 599–603. https://doi.org/10.1016/j.jhazmat.2008.10.038

Chen, A. S. C., Wang, L., Oxenham, J. L. & Condit, W. E. 2004. Capital Costs of Arsenic Removal Technologies: US EPA Arsenic Removal Technology, Demonstration Program Round 1. US Environmental Protection Agency. EPA/600/R-04/201.

Chen, W. F., Parette, R., Zou, J. Y., Cannon, F. S. & Dempsey, B. A. 2007. Arsenic removal by iron-modified activated carbon. Water Research 41(9): 1851–1858. https://doi.org/10.1016/j.watres.2007.01.052

Choong, T. S. Y., Chuah, T. G., Robiah, Y., Koay, F. L. G. & Azni, I. 2007. Arsenic toxicity, health hazards and removal techniques from water: an overview. Desalination 217(1–3): 139–166. https://doi.org/10.1016/j.desal.2007.01.015

Clancy, T. M., Snyder, K. V., Reddy, R., Lanzirotti, A., Amrose, S. E., Raskin, L. & Hayes, K. F. 2015. Evaluating the cement stabilization of arsenic-bearing iron wastes from drinking water treatment. Journal of Hazardous Materials 300: 522–529. https://doi.org/10.1016/j.jhazmat.2015.07.051

Clark, R. M. 1982. Cost estimating for conventional water treatment. Journal of the Environmental Engineering Division 108(5): 819–834. https://doi.org/10.1061/JEEGAV.0001345

Costa, A. R. & de Pinho, M. N. 2006. Performance and cost estimation of nanofiltration for surface water treatment in drinking water production. Desalination 196(1–3): 55–65. https://doi.org/10.1016/j.desal.2005.08.030

Diamadopoulos, E., Ioannidis, S. & Sakellaropoulos, G. P. 1993. As(V) removal from aqueous solutions by fly ash. Water Research 27(12): 1773–1777. https://doi.org/10.1016/0043-1354(93)90116-y

Doloi, H. K. 2011. Understanding Stakeholders' Perspective of Cost Estimation in Project Management. International journal of project management 29(5): 622–636. https://doi.org/10.1016/j.ijproman.2010.06.001

Engineering News Record Unlimited. 2022. Construction cost index history. https://www.enr.com/gdpr-policy?url=https%3A%2F%2Fwww.enr.com%2Feconomics%2Fhistorical_indices%2Fconstruction_cost_index_history [Accessed on September 2022.]

Ezugbe, E. O. & Rathilal, S. 2020. Membrane Technologies in Wastewater Treatment: A Review. Membranes 10(5): 89. https://doi.org/10.3390/membranes10050089

Fatoki, J. O. & Badmus, J. A. 2022. Arsenic as an environmental and human health antagonist: A review of its toxicity and disease initiation. Journal of Hazardous Materials Advances 5: 100052. https://doi.org/10.1016/j.hazadv.2022.100052

Fewtrell, L., Fuge, R. & Kay, D. 2005. An Estimation of the Global Burden of Disease Due to Skin Lesions Caused by Arsenic in Drinking Water. Journal of Water & Health 3(2): 101–107. https://doi.org/10.2166/wh.2005.0011

Garelick, H., Dybowska, A., Valsami-Jones, E. & Priest, N. D. 2005. Remediation technologies for arsenic contaminated drinking waters. Journal of Soils and Sediments 5(3): 182–190. https://doi.org/10.1065/jss2005.06.140

Giles, D. E., Mohapatra, M., Issa, T. B., Anand, S. & Singh, P. 2011. Iron and aluminium based adsorption strategies for removing arsenic from water. Journal of Environmental Management 92(12): 3011–3022. https://doi.org/10.1016/j.jenvman.2011.07.018

GlobalPetrolPrices.com. 2022. Nepal Electricity. https://www.globalpetrolprices.com/Nepal/electricity_prices/ [Accessed 27.9.2022.]

GoN/NASC. 2011. The State of Arsenic in Nepal 2011. Government of Nepal, National Arsenic Steering Committee (NASC), Nepal.

GoN/NMIP. 2014. Nationwide Coverage and Functionality Status of Water Supply and Sanitation in Nepal: National Management Information Project (NMIP). Government of Nepal, Department of Water Supply and Sewerage. Kathmandu, Nepal. https://dwssm.gov.np/wp-content/uploads/2019/05/Report-2014.pdf [Accessed 22.12.2022.]

Gupta, A., Yunus, M. & Sankararamakrishnan, N. 2012. Zerovalent iron encapsulated chitosan nanospheres – A novel adsorbent for the removal of total inorganic Arsenic from aqueous systems. Chemosphere 86(2): 150–155. https://doi.org/10.1016/j.chemosphere.2011.10.003

Gupta, V. K., Saini, V. K. & Jain, N. 2005. Adsorption of As(III) from aqueous solutions by iron oxide-coated sand. Journal of Colloid and Interface Science 288(1): 55–60. https://doi.org/10.1016/j.jcis.2005.02.054

Gwaccha, S., Acharya, B. N., Dhakal, A., Shrestha, S. M. & Joshi, T. P. 2020. Assessment of Arsenic Content in Deep Groundwater of Kathmandu Valley, Nepal. Nepal Journal of Science and Technology 19(1): 69–77. https://doi.org/10.3126/njst.v19i1.29785

Habuda-Stanic, M. & Nujic, M. 2015. Arsenic removal by nanoparticles: a review. Environmental Science and Pollution Research 22(11): 8094–8123. https://doi.org/10.1007/s11356-015-4307-z

Han, B., Runnells, T., Zimbron, J. & Wickramasinghe, R. 2002. Arsenic removal from drinking water by flocculation and microfiltration. Desalination 145(1–3): 293–298. https://doi.org/10.1016/s0011-9164(02)00425-3

Hering, J. G., Chen, P. Y., Wilkie, J. A. & Elimelech, M. 1997. Arsenic removal from drinking water during coagulation. Journal of Environmental Engineering 123(8): 800–807. https://doi.org/10.1061/(asce)0733-9372(1997)123:8(800)

Hesami, F., Bina, B., Ebrahimi, A. & Amin, M. 2013. Arsenic removal by coagulation using ferric chloride and chitosan from water. International Journal of Environmental Health Engineering 2: 17. https://doi.org/10.4103/2277-9183.110170

Hristovski, K., Baumgardner, A. & Westerhoff, P. 2007. Selecting metal oxide nanomaterials for arsenic removal in fixed bed columns: From nanopowders to aggregated nanoparticle media. Journal of Hazardous Materials 147(1–2): 265–274. https://doi.org/10.1016/j.jhazmat.2007.01.017

Hughes, M. F., Beck, B. D., Chen, Y., Lewis, A. S. & Thomas, D. J. 2011. Arsenic Exposure and Toxicology: A Historical Perspective. Toxicological Sciences 123(2): 305–332. https://doi.org/10.1093/toxsci/kfr184

Johnston, R. B., Heijnen, H. & Peter Wurzel, P. 2001. Safe water technology. In: United Nations Synthesis Report on Arsenic in Drinking Water. P. 211–316. https://www.ircwash.org/sites/default/files/UNACC_2001-United.pdf [Accessed 22.12.2022.]

Katsoyiannis, I. A. & Zouboulis, A. I. 2006. Use of iron- and manganese-oxidizing bacteria for the combined removal of iron, manganese and arsenic from contaminated groundwater. Water Quality Research Journal of Canada 41(2): 117–129. https://doi.org/10.2166/wqrj.2006.014

Kayastha, S. P. 2015. Arsenic contamination in the deep and shallow groundwater of Bara district, Nepal. Crossing the Border: International Journal of Interdisciplinary Studies 3(1): 65–70. https://doi.org/10.3126/ctbijis.v3i1.14091

Leist, M., Casey, R. J. & Caridi, D. 2000. The management of arsenic wastes: problems and prospects. Journal of Hazardous Materials 76(1): 125–138. https://doi.org/10.1016/s0304-3894(00)00188-6

Lenoble, V., Bouras, O., Deluchat, V., Serpaud, B. & Bollinger, J. C. 2002. Arsenic adsorption onto pillared clays and iron oxides. Journal of Colloid and Interface Science 255(1): 52–58. https://doi.org/10.1006/jcis.2002.8646

Liang, Y., Min, X., Chai, L., Wang, M., Liyang, W., Pan, Q. & Okido, M. 2017. Stabilization of arsenic sludge with mechanochemically modified zero valent iron. Chemosphere 168: 1142–1151. https://doi.org/10.1016/j.chemosphere.2016.10.087

Lorenzen, L., van Deventer, J. S. J. & Landi, W. M. 1995. Factors affecting the mechanism of the adsorption of arsenic species on activated carbon. Minerals Engineering 8(4–5): 557–569. https://doi.org/10.1016/0892-6875(95)00017-k

Mak, M. S. H., Rao, P. & Lo, I. M. C. 2009. Effects of hardness and alkalinity on the removal of arsenic(V) from humic acid-deficient and humic acid-rich groundwater by zero-valent iron. Water Research 43(17): 4296–4304. https://doi.org/10.1016/j.watres.2009.06.022

Morgada, M. E., Levy, I. K., Salomone, V., Farias, S. S., Lopez, G. & Litter, M. I. 2009. Arsenic (V) removal with nanoparticulate zerovalent iron: Effect of UV light and humic acids. Catalysis Today 143(3–4): 261–268. https://doi.org/10.1016/j.cattod.2008.09.038

Mou, S. A., Kabir, M. H., Yasmin, S. & Ahmed, S. 2020. Arsenic Mitigation Technologies from Ground Water: A Brief Review. American Journal of Pure and Applied Biosciences 2(5): 139–158. https://doi.org/10.34104/ajpab.020.01390158

Nicomel, N. R., Leus, K., Folens, K., Van der Voort, P. & Du Laing, G. 2016. Technologies for Arsenic Removal from Water: Current Status and Future Perspectives. International Journal of Environmental Research and Public Health 13(1): 62. https://doi.org/10.3390/ijerph13010062

Otero-Gonzalez, L., Mikhalovsky, S. V., Vaclavikova, M., Trenikhin, M. V., Cundy, A. B. & Savina, I. N. 2020. Novel nanostructured iron oxide cryogels for arsenic (As(III)) removal. Journal of Hazardous Materials 381: 120996. https://doi.org/10.1016/j.jhazmat.2019.120996

Paylab. 2022a. Salaries in the category: Construction & Real Estate. https://www.paylab.com/np/salaryinfo/construction-real-estate#:~:text=The%20gross%20salary%20range%20for,drastically%20among%20different%20job%20positions [Accessed 22.9.2022.]

Paylab. 2022b. Technician Electrical & Power Engineering. https://www.paylab.com/np/salaryinfo/electrical-power-engineering/technician?global_salary_period=monthly [Accessed 22.9.2022.]

Rajendran, R. M., Garg, S. & Bajpai, S. 2021. Economic feasibility of arsenic removal using nanofiltration membrane: A mini review. Chemical Papers 75(9): 4431–4444. https://doi.org/10.1007/s11696-021-01694-9

Ramos, M. A. V., Yan, W., Li, X. Q., Koel, B. E. & Zhang, W. X. 2009. Simultaneous Oxidation and Reduction of Arsenic by Zero-Valent Iron Nanoparticles: Understanding the Significance of the Core-Shell Structure. Journal of Physical Chemistry C 113(33): 14591–14594. https://doi.org/10.1021/jp9051837

Ruj, B., Chakrabortty, S., Nayak, J. & Chatterjee, R. 2021. Treatment of arsenic sludge generated from groundwater treatment plant: A review towards a sustainable solution. South African Journal of Chemical Engineering 37: 214–226. https://doi.org/10.1016/j.sajce.2021.06.003

Saiz, J., Bringas, E. & Ortiz, I. 2014. New Functionalized Magnetic Materials for As5+ Removal: Adsorbent Regeneration and Reuse. Industrial & Engineering Chemistry Research 53(49): 18928–18934. https://doi.org/10.1021/ie500912k

Salary.com. 2022. Construction worker salary in the United States. https://www.salary.com/research/salary/listing/construction-worker-salary [Accessed 22.9.2022.]

Sharma, J. R. 2010. Development of a preliminary cost estimation method for water treatment plants (MS Thesis). The University of Texas at Arlington. http://hdl.handle.net/10106/4924

Sharma, V. K., Dutta, P. K. & Ray, A. K. 2007. Review of kinetics of chemical and photocatalytical oxidation of Arsenic(III) as influenced by pH. Journal of Environmental Science and Health, Part A 42(7): 997–1004. https://doi.org/10.1080/10934520701373034

Shih, M. C. 2005. An overview of arsenic removal by pressure-driven membrane processes. Desalination 172(1): 85–97. https://doi.org/10.1016/j.desal.2004.07.031

Shipley, H. J., Yean, S., Kan, A. T. & Tomson, M. B. 2009. Adsorption of arsenic to magnetite nanoparticles: Effect of particle concentration, pH, ionic strength, and temperature. Environmental Toxicology and Chemistry 28(3): 509–515. https://doi.org/10.1897/08-155.1

Shrestha, R. 2012. Arsenic Contamination of Groundwater in Nepal: Good Public Health Intention Gone Bad. Inquiries Journal 4(09): 1–3. http://www.inquiriesjournal.com/a?id=701

Singh, R., Singh, S., Parihar, P., Singh, V. P. & Prasad, S. M. 2015. Arsenic contamination, consequences and remediation techniques: A review. Ecotoxicology and Environmental Safety 112: 247–270. https://doi.org/10.1016/j.ecoenv.2014.10.009

Singh, T. S. & Pant, K. K. 2004. Equilibrium, kinetics and thermodynamic studies for adsorption of As(III) on activated alumina. Separation and Purification Technology 36(2): 139–147. https://doi.org/10.1016/s1383-5866(03)00209-0

Sullivan, C., Tyrer, M., Cheeseman, C. R. & Graham, N. J. D. 2010. Disposal of water treatment wastes containing arsenic – A review. Science of the Total Environment 408(8): 1770–1778. https://doi.org/10.1016/j.scitotenv.2010.01.010

Sun, Y., Zhou, G., Xiong, X., Guan, X., Li, L. & Bao, H. 2013. Enhanced arsenite removal from water by $Ti(SO_4)_2$ coagulation. Water Research 47(13): 4340–4348. https://doi.org/10.1016/j.watres.2013.05.028

Tang, S. C. N. & Lo, I. M. C. 2013. Magnetic nanoparticles: Essential factors for sustainable environmental applications. Water Research 47(8): 2613–2632. https://doi.org/10.1016/j.watres.2013.02.039

Tang, W., Li, Q., Li, C., Gao, S. & Shang, J. K. 2011. Ultrafine α-Fe_2O_3 nanoparticles grown in confinement of in situ self-formed "cage" and their superior adsorption performance on arsenic(III). Journal of Nanoparticle Research 13(6): 2641–2651. https://doi.org/10.1007/s11051-010-0157-2

Thakur, J. K., Thakur, R. K., Ramanathan, A. L., Kumar, M. & Singh, S. K. 2011. Arsenic Contamination of Groundwater in Nepal –An Overview. Water 3(1): 1–20. https://doi.org/10.3390/w3010001

Tuutijärvi, T., Lu, J., Sillanpää, M. & Chen, G. 2009. As(V) adsorption on maghemite nanoparticles. Journal of Hazardous Materials 166(2–3): 1415–1420. https://doi.org/10.1016/j.jhazmat.2008.12.069

Tuutijärvi, T., Vahala, R., Sillanpää, M. & Chen, G. 2012. Maghemite nanoparticles for As(V) removal: desorption characteristics and adsorbent recovery. Environmental Technology 33(16): 1927–1936. https://doi.org/10.1080/09593330.2011.651162

Uddin, M. T., Mozumder, M. S. I., Islam, M. A., Deowan, S. A. & Hoinkis, J. 2007. Nanofiltration membrane process for the removal of arsenic from drinking water. Chemical Engineering & Technology 30(9): 1248–1254. https://doi.org/10.1002/ceat.200700169

UN (United Nations). 2010. The Human Right to water and Sanitation. https://www.un.org/waterforlifedecade/human_right_to_water.shtml#:~:text=According%20to%20the%20World%20Health,and%20few%20health%20concerns%20arise [Accessed 8.9.2022.]

Van der Bruggen, B., Vandecasteele, C., Van Gestel, T., Doyen, W. & Leysen, R. 2003. A review of pressure-driven membrane processes in wastewater treatment and drinking water production. Environmental Progress 22(1): 46–56. https://doi.org/10.1002/ep.670220116

van Halem, D., Bakker, S. A., Amy, G. L. & van Dijk, C. 2009. Arsenic in drinking water: a worldwide water quality concern for water supply companies. Drinking Water Engineering and Science 2(1): 29–34. https://doi.org/10.5194/dwes-2-29-2009

Wan, J., Klein, J., Simon, S., Joulian, C., Dictor, M. C., Deluchat, V. & Dagot, C. 2010. As-III oxidation by *Thiomonas arsenivorans* in up-flow fixed-bed reactors coupled to As sequestration onto zero-valent iron-coated sand. Water Research 44(17): 5098–5108. https://doi.org/10.1016/j.watres.2010.08.044

Wang, D., Guo, F., Wu, Y., Li, Z. & Wu, G. 2018. Technical, economic and environmental assessment of coagulation/filtration tertiary treatment processes in full-scale wastewater treatment plants. Journal of Cleaner Production 170: 1185–1194. https://doi.org/10.1016/j.jclepro.2017.09.231

WHO (World Health Organization). 2018. Arsenic. https://www.who.int/news-room/fact-sheets/detail/arsenic#:~:text=The%20current%20recommended%20limit%20of,removing%20arsenic%20from%20drinking%2Dwater [Accessed 15.5.2022.]

Zhang, S., Migliaccio, G. C., Zandbergen, P. A. & Guindani, M. 2014. Empirical Assessment of Geographically Based Surface Interpolation Methods for Adjusting Construction Cost Estimates by Project Location. Journal of Construction Engineering and Management 140(6): 04014015. https://doi.org/10.1061/(asce)co.1943-7862.0000850

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