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Adsorption methods for arsenic removal in water bodies: a critical evaluation of effectiveness and limitations

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The presence of arsenic in water bodies poses a significant risk to both human health and the environment. Arsenic (As) contamination in water sources is a global environmental concern caused by both natural processes and human activities. Due to its toxic and persistent nature, arsenic has detrimental effects on ecosystems and human wellbeing. This study aimed to elucidate the mechanisms behind arsenic accumulation in water bodies. In aquatic environments, arsenic concentrations in drinking water have been reported as high as 1,320 $\mu\text{g/L}$ (Nicaragua), while groundwater levels exceeded 5,000 $\mu\text{g/L}$ (Thailand), and wastewater contained up to 134,000 $\mu\text{g/L}$ (landfill leachate in Brazil). Furthermore, bioaccumulation of arsenic ($\mu\text{g/g}$) in fish species ranges from 0.4 (catfish in the Paraná River Delta, Brazil) to 362 (*Pteromylaeus bovinus*, Northern Adriatic Sea). Recent research has predominantly focused on removing arsenic from aqueous solutions through adsorption methods. Notably, nanoparticle adsorbents and graphene-based adsorbents demonstrate a high capacity for arsenic removal from water bodies.

KEYWORDS

adsorption, arsenic removal, bentonite, graphene, nanoparticles, zeolite

1 Introduction

As the global demand for clean water rises, the importance of water resources is increasing. Factors such as population growth, pollution, and climate change are reducing the usability of limited water resources. Urbanization, agriculture, and industry have a significant impact on water quality. Pollutants, such as heavy metals, disrupt aquatic ecosystems, thereby undermining their sustainability (Sener et al., 2023). The presence of metal pollution, such as arsenic (As), has a harmful impact on the ecosystem because of its toxic, non-biodegradable, and persistent characteristics. Arsenic rapidly accumulates in the soil, water, and sediment, posing a threat to the environment (Prasad Ahirvar et al., 2023). The presence of As in the water is a result of weathering processes that occur in rocks and sulfidic minerals. One of these processes involves the oxidation of sulfidic minerals like arsenopyrites, which releases arsenic into water sources. Additionally, anthropogenic activities such as agriculture, mining, and industrial waste also contribute to the release of As into the environment (Irunde et al., 2023).

Arsenic, a harmful contaminant, is a toxic substance that is both colorless and odorless (Abedi and Mojiri, 2020). It can be found in elevated concentrations in water bodies across different countries worldwide. Exposure to high levels of arsenic can result in severe immediate and long-term health issues, including diarrhea, vomiting, diabetes, heart disease, cancer, miscarriage, childhood cancer, and even fatalities (Nguyen and Mulligan, 2023). To ensure safe drinking water and protect public health, the World Health Organization (WHO) recommends a maximum allowable concentration (MAC) of As in drinking water at 10 $\mu\text{g/L}$. This limit is legally defined in the European Union Drinking Water Directive. However, certain countries like Denmark and Ireland have set even lower limits (5 $\mu\text{g/L}$ and 7.5 $\mu\text{g/L}$, respectively). Arsenic concentrations in groundwater and surface water have been found to range from 100 to 5,000 $\mu\text{g/L}$. In certain cases, particularly near hydrothermal systems close to shorelines, extremely high concentrations of 1,386 and 5,850 $\mu\text{g/L}$ have been reported. This widespread contamination of As in aquatic systems is recognized as a significant global environmental issue, with impacts observed worldwide (Zhang et al., 2022).

Increasingly, more states are adopting stricter As standards based on the severity of As issues, population exposure, and the availability of improved removal technologies (Nikić et al., 2023). Adsorption is a widely used and promising technique for removing As from water (Fang et al., 2023). Adsorption has gained significant recognition as a cost-effective method for As removal. It offers several advantages, including high removal efficiency, lower cost, and simpler operation compared to alternative methods (Moradi et al., 2023). In this review manuscript, our study aims to study the fate of As in water bodies around the world as well as comprehensively evaluate the effectiveness of adsorption as a method for As removal from water.

2 Arsenic contamination in water bodies

Once released into the environment, arsenic becomes part of the biogeochemical cycle and cannot be degraded. In aquatic environments, arsenic undergoes complex chemical speciation, resulting in various inorganic and organic arsenic species. Inorganic forms include arsenite As(III) and arsenate As(V), while organic forms encompass methylarsonate (MMA), dimethylarsinate (DMA), tetramethylarsine (TMA), trimethylarsine oxide (TMAO), arsenocholine (AsC), arsenobetaine (AsB), thiolated arsenic, arsenosugars (As-Sug), and arsenolipids (Zhang et al., 2022).

Commonly, in water, As exists in various redox states, predominantly as inorganic species such as As(III) and As(V). However, the presence of organic species is more common when there is anthropogenic contamination (Fuoco et al., 2022). As(III) is widely recognized for its higher toxicity compared to As(V) (Wu et al., 2023). The fate and mobility of arsenic in water bodies are influenced by various processes: (1) redox reactions, (2) adsorption and desorption, (3) competitive adsorption (ion exchange), (4) solid-phase precipitation and dissolution, and (5) biological activity. These interconnected processes, along with factors such as redox potential (Eh), pH,

chemical composition, and reaction kinetics, collectively determine the behavior of arsenic under specific conditions (Mohan and Pittman, 2007; Cheng et al., 2009). This redox sensitivity of arsenic is depicted in Figure 1, which illustrates the Eh-pH diagram for arsenic species. In oxidizing conditions, arsenate (H_2AsO_4) is the predominant form at low pH levels (below approximately 6.9), while at higher pH levels, it exists in the form of HAsO_4^{2-} . Conversely, under reducing conditions at pH levels below approximately 9.2, the dominant form is the uncharged arsenite species (H_3AsO_3) (Smedley and Kinniburgh, 2002).

The problem of arsenic contamination in aquatic environments is a worldwide concern (Singh et al., 2020). In the area near Lomé (in Togo), surface water samples exhibited elevated concentrations of arsenic, reaching as high as 6,460 $\mu\text{g/L}$. Similarly, in Prestea, Ghana, a wide range of arsenic concentrations ranging from 150 to 8,250 $\mu\text{g/L}$ were reported in surface waters (Ahoulé et al., 2015). Table 1 presents the concentration of As reported in aquatic environments worldwide. According to the data presented in Table 1, the highest recorded arsenic concentration ($\mu\text{g/L}$) in drinking water was 1320 in Nicaragua. In Taiwan and Thailand, the maximum arsenic concentrations ($\mu\text{g/L}$) in groundwater were 5,000 and 1,820, respectively. Furthermore, the highest arsenic concentration ($\mu\text{g/L}$) in surface water was 2,650 in Nicaragua. In Brazil, the maximum arsenic concentration ($\mu\text{g/L}$) in wastewater was 134,000, observed in landfill leachate. Additionally, a notable concentration of arsenic (14.8 $\mu\text{g/L}$) was reported in seawater near Tarut Island in the Persian Gulf.

When aquatic organisms come into contact with arsenic through their diet and various sources like water, they have the ability to accumulate, retain, and convert different forms of arsenic within their bodies (Azizur Rahman et al., 2012). The primary bioaccumulation processes encompass the uptake, assimilation, biotransformation, and elimination of arsenic (Figure 2). Once absorbed by an organism, arsenic has the potential to undergo biotransformation, converting into a less toxic form. Notably, AsB is recognized for its low toxicity among aquatic organisms across different trophic levels (Zhang et al., 2022).

The toxicity of arsenic varies among living organisms depending on their resistance capabilities and detoxification mechanisms. In terms of toxicity, inorganic arsenic (iAs) is more harmful than organoarsenic and has been classified as a proven carcinogen for humans. Arsenite (AsIII) is typically more toxic than arsenate (AsV), whereas dimethylarsinous acid (DMAAIII) and monomethylarsonous acid (MMAAIII) exhibit higher toxicity compared to their respective parent compounds (Azizur Rahman and Hasegawa, 2012). Azizur Rahman et al. (2012) stated that while it is commonly known that iAs is typically more toxic than organoarsenic species, the toxicity of iAs species toward aquatic organisms is still a topic of debate. Interestingly, there are exceptions to this generalization. Marine phytoplankton, for instance, often exhibit higher sensitivity to arsenite (AsIII), whereas freshwater phytoplankton are highly sensitive to arsenate (AsV). As an illustration, the marine phytoplankton species *Dunaliella* sp. and *Polyphysa peniculus* demonstrate greater sensitivity to AsV compared to AsIII. Table 2 shows the bioaccumulation As in aquatic organisms, which indicates that up to 362 $\mu\text{g/g}$ of As reported in fish tissues.

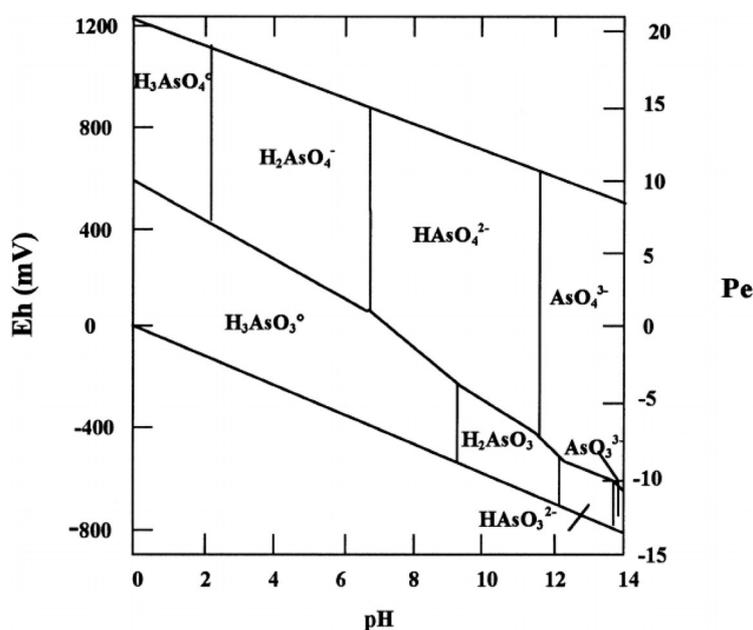


FIGURE 1

Redox potential (Eh)–pH diagram for aqueous arsenic species in the AsO₂–H₂O system at 25°C and 1 bar total pressure (Akter et al., 2005; copyright permission received from Elsevier).

Garai et al. (2021) stated that prolonged exposure of freshwater fish to low concentrations of arsenic leads to the accumulation of this toxic element primarily in the liver and kidney tissues. The exposure to arsenic causes histopathological changes in the gills and liver tissue of tilapia (*Oreochromis mossambicus*), a type of freshwater fish. The observed alterations in the gills include epithelial hyperplasia, lamellar fusion, epithelial lifting, edema, desquamation, and necrosis.

3 Removal of As with Adsorption method

In recent decades, various treatment techniques have been used to remove heavy metals like As, including chemical precipitation, membrane filtration, extraction, and electrochemistry. However, these methods have drawbacks such as poor selectivity, high energy consumption, and high costs, limiting their application in engineering. As a result, adsorption has become a favored choice for heavy metal removal due to its broad applicability, low cost, and environmental friendliness (Di et al., 2023). On the other hand, the main advantages of the adsorption technique include its adaptability, user-friendly nature, cost-effectiveness, and its ability to accommodate a diverse range of adsorbents sourced from minerals, as well as biological and organic origins (Rajendran et al., 2022). Adsorption begins by transferring heavy metal ions from the aqueous solution to the surface of adsorbents. Subsequently, these ions bind to the surface through physical or chemical interactions. To facilitate this process, adsorbents must possess a large accessible surface area, which ensures the presence of numerous exposed active

sites for effective and selective binding with heavy metal ions (Fei and Hu, 2023). A wide range of adsorbents have been employed for the adsorption of heavy metals (such as As) from wastewater and natural water (Chakraborty et al., 2022). These include engineered adsorbents like nano-adsorbents and metals coated adsorbents, as well as low-cost options. Among low-cost adsorbents, natural materials such as zeolites, and clay are commonly used. Additionally, agricultural waste materials and biochar serve as prominent sources of bio-adsorbents (Abdollahi et al., 2022; Oladoye, 2022). According to Ariffin et al. (2017), the most advantageous aspects of adsorption methods include their simplicity, flexibility in design, ease of operation, and resistance to toxic contaminants. However, a notable disadvantage of this method is the need for regeneration processes. The adsorption mechanism can involve physical entrapment (absorption) or chemical binding through weak Van der Waals forces, dipole-dipole and ion-dipole interactions, cation exchange, or strong covalent bonding (Mojiri et al., 2020). Physisorption occurs when the adsorbent and adsorbate are united by van der Waals forces. Chemisorption takes place when the adsorbate forms chemical bonds with the surface of the adsorbent. Electrostatic attraction occurs when the adsorbent surface carries negative or positive charges. Ion exchange happens between divalent metal cations and oxygen-containing functional groups, taking advantage of the cation exchange capacity. Surface complexation occurs when heavy metals exceed the available sites on the adsorbent surface, and this involves the formation of multiatom assemblies during the reaction activities (Mahesh et al., 2022). The removal of As through adsorption methods is presented in Table 3. According to Table 3, the maximum adsorption capacity of engineered

TABLE 1 Occurrence of As in water bodies.

Water sources	Concentrations (ranges or mean; $\mu\text{g/L}$)	Locations	References
Drinking water	0–220	Great Hungarian Plain, Hungary	Rowland et al., 2011
	2 to 250	Vojvodina, Serbia	Jovanovic et al., 2011
	<1 to 610	Osijek-Baranja and Vukovar- Syrmia counties, Croatia	Rowland et al., 2011
	<0.5–175	Romania	Neamtii et al., 2015
	30–105	Kutahya, Western Turkey	Sener and Karakuş, 2017
	<1–118	Spain	Medrano et al., 2010
	0.02–27.2	Italy	Zuzolo et al., 2020
	0.03–25.3	Aarhus, Denmark	Alarcón-Herrera et al., 2020
	0.88–17.9	San Rafael Las Flores, Guatemala	Marcillo et al., 2020
	10–1,320	Nicaragua	Delgado Quezada et al., 2020
Groundwater	40.0 (Hand pump water) 49.0 (Tube Well water:)	District Jamshoro, Sindh, Pakistan	Baig et al., 2009
	481 (Boreholes)	Vietnam	Sø et al., 2018
	209 (well)	Kandal, Cambodia	Lawson et al., 2016
	13.8 (well)	R. Y. Khan, Pakistan	Farooqi et al., 2017
	285.6 (Tube-well)	Nawalparasi, Nepal	Mueller and Hug, 2018
	36.8 (well)	Kurdistan, Iran	Sharifi and Sinegani, 2012
	104.3 (well)	Kuitun, China	Chen et al., 2018
	160 (tube-well)	Ganges, Bangladesh	Mukherjee et al., 2018
	1146	Anhui, China	Sanjrani et al., 2019
	up to 300	South China	Luo et al., 2021
	As(III): <0.5–208	Pannonian Basin, Hungary	Rowland et al., 2011
	0.10–168	West Romania	Senila et al., 2017
	<0.05–2,230	Tampere, Finland	Pedretti et al., 2019
	<0.4–431	Italy	Carraro et al., 2013
	0.1–1320	Nicaragua	Delgado Quezada et al., 2020
	0.1	Plateria Puno, Peru	George et al., 2014
	3.2–116.6	Okavango delta, Botswana	Mladenov et al., 2014
	21.4–278	Ethiopia	Rango et al., 2010
	<5.2–69.4	Ghana	Bhattacharya et al., 2012
	0.5–123	Central Tanzania	Nwankwo et al., 2020
	53.8±93.4	Sindh, Pakistan	Ahmed Baig et al., 2010
	10–1,820 1–>5,000	Groundwater of 200,000 population, Taiwan Groundwater of 15 000 population, Thailand	Sarkar and Paul, 2016
	<2–1,300	Verde Valley, Arizona, USA	Camacho et al., 2011
	450 (maximum)	Baja California Sur, Mexico	Wurl et al., 2014
	25.7–137.8	Banat, Serbia	Devic et al., 2014
	41.4	Bolivia	Ramos Ramos et al., 2012
	As(III): 12.24 As(V): 6.54	Rayong province, eastern Thailand	Boonkaewwan et al., 2021
253	Namaqualand, South Africa	Abiye and Bhattacharya, 2019	

(Continued)

TABLE 1 (Continued)

Water sources	Concentrations (ranges or mean; $\mu\text{g/L}$)	Locations	References
	6,150	West of Johannesburg, South Africa	
	500	Karoo, South Africa	
	As(V): 20.29 Total As: 251	Mexico	Ortiz Letechipia et al., 2022
Surface water (River Lake)	0.23–105	El Salvador	Bundschuh et al., 2021
	14.6–42.5	Plateria Puno, Peru	George et al., 2014
	200–2,200	Ankobra, Volta and Densu river, Ghana	Gbogbo et al., 2017
	4.0±1.60	Sindh, Pakistan	Ahmed Baig et al., 2010
	7.04	Changjiang River, china	Wang et al., 2011
	23.88	Diyarbakir, Turkey	Varol and Sen, 2012
	2.354	Tigris River, Turkey	
	1.08 (Upriver:) 1.93 (Downriver:) 2.14 (Wet season:)	Pardo River, Brazil	Alves et al., 2014
	0.938	Oghji, river, Armenia	Gabrielyan et al., 2018
	2.72 ± 2.51	27 subtropical rivers, China	Deng et al., 2018
	0.66–4.01	Bandama river, West Africa	Ouattara et al., 2018
	0.56–5.06	Comoé river, West Africa	
	1.45–6.19	Bia river, West Africa	
	13.4 ± 1.2–13.8 ± 1.8	Sinos River (Brazil)	Weber et al., 2013
	9.7	the Antequera River, Bolivia	Ramos Ramos et al., 2012
	4.6	Lasha River (L.R.), downstream, China	Li et al., 2013
	52.75	The middle parts of the Zarshuran stream, IRAN	Sharifi et al., 2016
	23.45	the Sarouq River in the downstream station, IRAN	
	130	Yamuna River, Delhi, India	Rahman and Singh, 2018
	39 ± 3	Snake River below Lewis River	Wang et al., 2010
	45 ± 4	Snake River above Lewis River	
	<6	Yangzonghai Lake, Yunnan, China	McCleskey et al., 2022
	97.5 ± 28.5	Manchar lake, Sindh, Pakistan	
	As(V): 2.16 As(III): 0.34	Zhushan Bay, Lake Taihu, china	Yan et al., 2016
	4.18	Waihai, Lake Dianchi, china	Zhang N. et al., 2013
	170	Handle Lake, Canada	Palmer et al., 2019
	60	Lower Martin, Canada	
	35	Long Lake, Canada	
	0.99–2650	Nicaragua	Delgado Quezada et al., 2020
	<1–188	Okavango delta, Botswana	Mladenov et al., 2014
	0.21–3	Ethiopia	Dsikowitzky et al., 2013
	<1–82	Tanzania	Nyanza et al., 2014
	3.26–22.7	Rift lake (Lake Elementaita and Naivasha), Kenya	Yang et al., 2017
	1.47–1.89	Lake Victoria, Kenya	Outa et al., 2020
	12.0±2.20	Sindh, Pakistan	Ahmed Baig et al., 2010
	305.6 20.6	Mapam Yumco, China Yangzuoyongcuo, China	Li et al., 2013

(Continued)

TABLE 1 (Continued)

Water sources	Concentrations (ranges or mean; $\mu\text{g/L}$)	Locations	References
Wastewater and landfill leachate	972	Ontario, Canada	Sprague and Vermaire, 2018
	200.1–1,588.3	River Niger Upstream, Onitsha, Nigeria River Niger Central Drainage, Onitsha, Nigeria	Ezeabasili et al., 2014
	Total As: <6.0	Taihu, Chaohu, and Dianchi lakes, China	Yang et al., 2016
	As(III): 0.8	Utinoie lake (hot-springs), Mutnovsky volcano on the Kamchatka peninsula, Russia	Ilgen et al., 2011
	Gohagoda leachates: ND-2 Acetogenic leachates: <100–300 Methanogenic leachates: <1–2	Sri Lanka	Wijesekara et al., 2014
	45.2	Simpang Renggam Landfill Site, Johor, Malaysia	Mohd-Salleh et al., 2020
	1.6520	Backwash Sludge, West Bengal, India	Koley, 2021
	1,360	China	Liu et al., 2018
	As(V): 42,000	South Korea	Lee et al., 2015
	134,000	the Maringá landfill (Paraná, Brazil)	Scandelai et al., 2021
	Minimum:0.2 Maximum: 928	Romania	Schiopu and Gavrilesco, 2010
	Landfill A: 1,180 Landfill B: 2,870	South Africa	Mosai et al., 2022
	As(III): 108	Pennsylvania (PA), USA (effluent)	Zhao et al., 2013
	Seawater	T-As: 1.71 ± 0.35	Kagoshima Bay, southwestern Japan
0.46–1.55		Jordanian Gulf of Aqaba	Al-Taani et al., 2014
1.2 to 4.3		southern Taiwan	Lin et al., 2013
8.55–14.88		Tarut Island, Arabian Gulf	El-Sorogy et al., 2016
As(III): 0.09–0.51 Total As: 0.20 to 0.66		Todos os Santos Bay, Salvador, Bahia, Brazil	dos Santos et al., 2018
0.6		China Sea	Wang et al., 2022
0.6–1.6		Atlantic Ocean	
0.8–1.1		Indian Ocean	
0.94–1.56		Atlantic Ocean	
1.0		Pacific Ocean	
1.4		Galway Bay, Ireland	
1.8		Krka estuary, Yugoslavia	
2.81–4.48 1.78–3.42		surface seawater, subtropical bay of China bottom seawater, subtropical bay of China	
1.26		Caspian Sea	Abadi et al., 2018
2.41		Al-Khobar	Alharbi et al., 2017
0.300		Mediterranean Sea	El-Sorogy and Attiah, 2015
1.18		Dingzi Bay, Yellow Sea	Pan et al., 2014
2.19		Jinzhou Bay	Wang et al., 2012
2.60		World Ocean (mean)	Famil Yusif et al., 2015
7.420 ± 0.742		the Jordanian coastline of the Gulf of Aqaba, Red Sea	Al-Absi et al., 2019

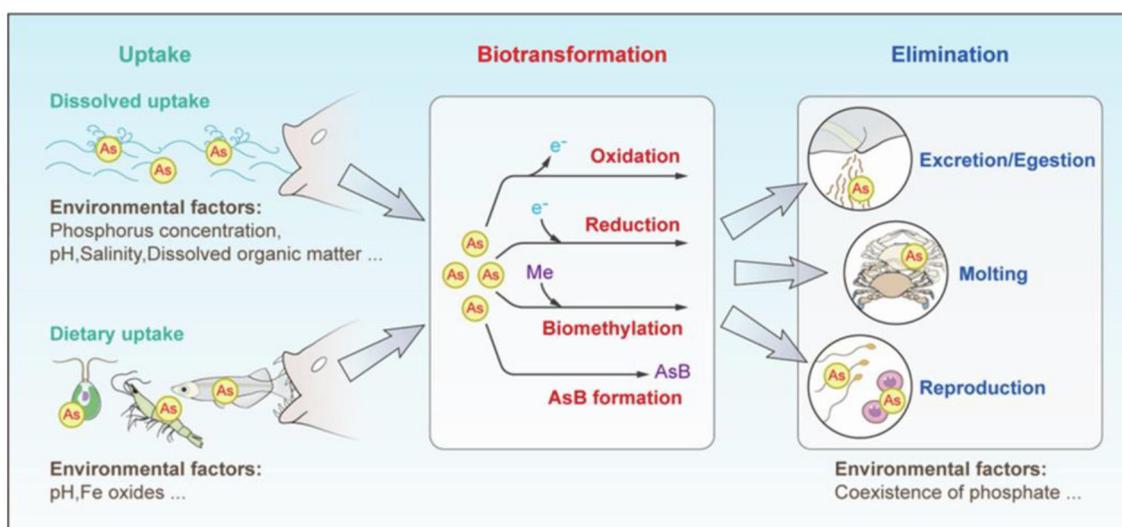


FIGURE 2 Processes of arsenic bioaccumulation in aquatic organisms (Zhang et al., 2022; open access paper under the terms of the Creative Commons CC-BY license).

adsorbents is significantly higher than that of single adsorbents. For example, engineered nanoparticles can exhibit adsorption capacities exceeding 100 mg/g.

3.1 Removal of As with carbon-based adsorbents

Carbon-based adsorbents possess significant potential as adsorbent materials due to their large specific surface area, high porosity, substantial pore volume, and adjustable morphological and functional group properties. These characteristics make them suitable for various applications, including water purification. The adsorption efficiency of carbonaceous materials is influenced by factors such as the raw material, production technique, and environmental variables (Mahesh et al., 2022). The main mechanisms of As removal with carbon-based adsorbents are shown in Figure 3. Sabzehmeidani et al. (2021) reported that activated carbon, biochar, and graphene-based adsorbents are commonly utilized for the removal of arsenic (As) from water sources.

Activated carbon (AC) is a versatile and widely available carbon material used for adsorption. It comes in different forms, such as granular, powder, pellet, and spheres. These materials exhibit efficient and selective adsorption of toxins and heavy metals. Compared to other adsorbents, activated carbon-based materials are cost-effective, highly effective, and easy to use (Lan et al., 2023). AC has a structure that includes well-defined micro, meso, and macropores, as well as diverse surface functional groups. The surface area of activated carbon ranges from 500 to 3,000 m^2/g (Sultana et al., 2022). In a study, Gao et al. (2023) utilized modified AC to effectively remove both As(III) and As(V). The maximum adsorption capacities were found to be 10.9 mg/g for As(III) and

16.0 mg/g for As(V). The maximum adsorption capacity (mg/g) observed for As(V) removal using As was 1.2 (Yürüm et al., 2014).

Graphene is a 2D lattice of carbon atoms with high surface area, thermal conductivity, Young's modulus, and charge carrier mobility. It exhibits the quantum Hall effect due to electron confinement in 2D materials (Sabzehmeidani et al., 2021). Graphene oxide (GO), a derivative of graphene, is a single sheet of graphite with a 2D honeycomb crystal plane structure. It exhibits a high specific surface area of up to 2,620 m^2/g (Bian et al., 2015). GO is characterized by diverse functional groups on its surface, including epoxy, lactol, carboxyl, phenol, and hydroxyl groups, along with large π -stacking. These features enable GO to possess a high sorption capacity through strong interactions such as hydrogen bonding, electrostatic forces, and π - π interactions (Gabris et al., 2022). In the study (Bian et al., 2015), GO was synthesized using a modified Hummers method. A solution containing 1.0 g of $NaNO_3$ dissolved in 50 ml of concentrated H_2SO_4 was prepared, and then 1.0 g of natural graphite was slowly added to the solution while stirring in an ice-water bath. After complete dissolution, 18.0 g of $KMnO_4$ was gradually added over a period of 30 min. The mixture was stirred for 2 h at 309 ± 1 K and then heated in a water bath at 368 K for 30 min. Following this, 180 ml of deionized water was added, and a suspension of 20 ml of H_2O_2 (30 wt%) was slowly introduced. The desired products were washed with 10% v/v HCl and ultrapure water, undergoing centrifugation and ultrasonication until a constant pH value was reached. The resulting GO was dried at 333 K for 24 h. Chandra et al. (2010) removed As(III) and As(V) using modified GO (magnetic GO). The maximum adsorption capacity (mg/g) based on the Langmuir isotherm was found to be 10.2 for As(III) and 5.2 for As(V), respectively. Their study revealed that the adsorption process of As using magnetic-GO primarily followed surface complexation. In a study conducted by Wu et al. (2018b) GO-based adsorbent ($GO/CuFe_2O_4$) was used to remove

TABLE 2 Bioaccumulation of As in fish/shrimp.

Fish/shrimp	Concentrations ($\mu\text{g/g}$)	Remarks	Location	References
<i>Micropterus salmoides</i>	1.39 \pm 0.22	Liver	Emory River mile 1.0; U.S.	Otter et al., 2012
<i>Pomoxis annularis</i>	4.66 \pm 0.82			
<i>Lepomis macrochirus</i>	1.76 \pm 0.21			
<i>Lepomis microlophus</i>	6.83 \pm 0.726			
Rabbitfish <i>S. fuscescens</i>	1.04 \pm 0.09	Muscle	China, under control conditions	Zhang et al., 2016
Seabass <i>L. japonicus</i>	2.59 \pm 0.67			
<i>Pteromylaeus bovinus</i>	36.3–362	Muscle	Northern Adriatic Sea	Šlejkevec et al., 2014
<i>Myliobatis aquila</i>	32.4–69.8			
Freshwater fish	0.748 \pm 0.651	Muscle	Xiang River,	Jia et al., 2018
Green tiger shrimp (<i>Penaeus semisulcatus</i>)	29.254 \pm 0.473	Muscle	Iskenderun Bay, Turkey	Kaya and Turkoglu, 2017
Common sole (<i>Solea solea</i>)	4.397 \pm 0.311			
Whiting fish (<i>Merlangius merlangus</i>)	10.320 \pm 0.518			
<i>E. suratensis</i>	3.0–3.4	Kidney	Sri Lanka	Perera et al., 2016
<i>A. testudineus</i>	4.1–4.7			
<i>C. striata</i>	4.0–4.3			
<i>Meretrix lusoria</i>	7.6–10.7	NR*	Putai county	Liu et al., 2007
Catfish	0.45–4.93	Gill	Paraná River Delta (PRD, Argentina) and in Paranaguá Estuarine Complex (PEC, Brazil)	Avigliano et al., 2020
<i>Anadara</i> sp.	13–23		American Samoa	Peshut et al., 2008
<i>Gafarium</i> sp.	20–68			
<i>Labeo calbasu</i> (Calbasu)	1.86 \pm 0.02	Muscle (wet weight)	Manchar Lake	Shah et al., 2009
<i>Cirrhinus mrigala</i> (Mrigala)	0.54 \pm 0.04			
<i>Mystus gullio</i> (Gullio)	1.72 \pm 0.24			
<i>Oreochromis niloticus</i>	5.70 \pm 2.10 As(III); 8.79 \pm 4.46 As(IV)	Liver; after 1 week analysis	NR	Ferreira et al., 2019

*NR, Not reported.

both As(III) and As(V). The maximum adsorption capacities (mg/g) achieved were 51.64 for As(III) and 124.69 for As(V) removal. Their study demonstrated that both As(III) and As(V) adsorptions on the GO-CuFe₂O₄ adsorbent followed an inner-sphere complex mechanism. In another study conducted by Su et al. (2017), a modified GO called iron oxide-graphene oxide was used to successfully remove 99.9% of both As(III) and As(V). The maximum adsorption capacities were 147 mg/g for As(III) and 113 mg/g for As(V). The results of their study showed that the adsorption of As onto modified-GO takes place through a mechanism known as surface complexation.

Biochar is produced through the thermal treatment of natural organic feedstocks in an oxygen-limited environment. Huang et al. (2019) emphasized that biochar offers numerous advantages, such as its ion exchange capacity, expansive surface area, high porosity, and presence of surface functional groups, which make it promising for wastewater treatment. Biochar is composed of various elements, including carbon, sulfur, hydrogen, oxygen,

nitrogen, and minerals found in the ash fraction. Although biochar shares similar properties and structure with AC, AC has a considerably larger surface area. The primary surface functional groups found in biochar include carboxyl, hydroxyl, phenolic hydroxyl, and carbonyl groups (Mojiri and Zhou, 2023). In a research, Wang et al. (2015) used a modified version of biochar called manganese oxide-modified biochars to successfully remove As(V). They discovered that the biochar had a maximum adsorption capacity of 0.59 mg/g, with the main adsorption mechanism being the interaction between the biochar and As(V). Navarathna et al. (2019) employed a modified biochar known as magnetic Fe₃O₄/Douglas fir biochar composites to effectively remove over 68% of As(III). The maximum adsorption capacity of the modified biochar was determined to be 6.9 mg/g. Their study suggested that the adsorption process was primarily governed by a chemisorption mechanism. In a research study, wood biochar exhibited remarkable efficiency in removing 92–100% of As. The maximum adsorption capacity for As(III) removal was measured

TABLE 3 Removal of arsenic with adsorption method.

Arsenic	Adsorbent	Maximum adsorption capacity (mg/g)	Adsorption isotherm	Remarks	References
As(III)	Activated carbon (AC)	184	Langmuir	As concentration was 20 mg/L; Anion exchange and physiochemical adsorption were the primary mechanisms of adsorption; The removal of As(III) reached 92% with powdered activated carbon (PAC).	Sabzehmeidani et al., 2021
As(III); As(V)	Granular activated carbon (GAC)	0.03; 0.05	NR*	As(V) undergoes adsorption on an oxide surface through a ligand exchange mechanism, forming an inner sphere surface complex. The removal of As(III) and As(V) reached 44% and 71% respectively with GAC.	Mondal et al., 2008
As(III)	AC	1.0	Langmuir	Ion exchange, complexation and physiochemical adsorption were the primary mechanisms of adsorption	Budinova et al., 2009
As(III)	AC	6.5	Langmuir	The adsorption of As(III) by AC is a complex process that is not solely controlled by electrostatic attraction.	Mondal et al., 2007
As(III); As(V)	Fe-modified-AC	38.8; 51.3	Langmuir	NR	Chen et al., 2007
As(III)	Fe-Zr@AC	556	Langmuir	Surface functional groups -OH, NH ₂ , COO-, Fe-O, ZrO were involved for As (III) removal.	Sahu et al., 2021
As(V)	Fe-AC	17	Langmuir	Surface functional groups were involved for As(V) removal	Lodeiro et al., 2013
As(III); As(V)	Biochar (derived from peanut shell)	7.6; 7.9	Langmuir	Both hydroxyl (-OH) and aromatic surface functional groups (such as C=O, C=C-C, and -C-H) played a significant role in the sorption of both species of arsenic. The removal of As(III) and As(V) reached 95% and 99% respectively with biochar.	Sattar et al., 2019
As(III)	Magnetic Fe ₃ O ₄ /Douglas fir biochar	5.4	Sips isotherm	Both surface complexation and pH-dependent chemisorption mechanisms played a significant role in the sorption of As(III).	Navarathna et al., 2019
As(V)	Fabrication of engineered biochar (derived from paper mill sludge)	22.8	Langmuir	Specific surface interactions played a significant role in the sorption of As(V)	Yoon et al., 2017
As(V)	Activated biochar	24.4	Langmuir	The primary sorption mechanism is the chemical interaction between the metal ions and the surface functional groups of biochars.	Jin et al., 2014
As	Activated biochar	8.6	Langmuir	The main adsorption mechanism was the functional groups.	Saikia et al., 2017
As(V)	Biochar (derived from sewage sludge)	0.03	Langmuir	53% of As(V) was removed. The primary mechanisms responsible for heavy metal immobilization were the electrostatic interactions between the negative surface charge of biochar and metal cations, as well as metal precipitation in the case of anionic metals.	Agrafioti et al., 2014
As(V)	Zerovalent iron composited biochars	26.5	Langmuir	Electrostatic interactions, surface complexation and diffusion were sorption mechanisms.	Ahmad et al., 2020
As(V)	Magnetic graphene oxide	59.6	Langmuir	The adsorption of As(V) on MGO is mainly governed by the electrostatic attraction or repulsion interaction of As(V) species with the MGO surface.	Sheng et al., 2012
As(III)	Fabricated crumpled graphite oxides	47.3	Langmuir	>98% of As(III) was removed. The both electrostatic interaction and surface complexation were the main sorption mechanism.	Son et al., 2018

(Continued)

TABLE 3 (Continued)

Arsenic	Adsorbent	Maximum adsorption capacity (mg/g)	Adsorption isotherm	Remarks	References
As(III); As(V)	Graphite oxides	19.0; 28.0	Langmuir	Both surface complexation mechanism, rather than electrostatic interactions were the main sorption mechanisms.	Su et al., 2017
As(III)	Combination of Graphene Oxide (GO) and Granular Ferric Hydroxide (GFH)	0.2	Langmuir	Chemisorption was the main sorption mechanism.	Tolkou et al., 2023
As	Modified bentonite	0.6–2.8	Langmuir	The electrostatic adsorption, the ligand exchange and precipitation were the main sorption mechanism.	Hua, 2015
As(III)	Modified bentonite (hydroxyapatite-bentonite clay-nanocrystalline cellulose)	51.0	Langmuir	>95% of As(III) was removed. Ion-exchange mechanism was the main sorption mechanism.	Hokkanen et al., 2019
As	Co-modified bentonite with manganese oxides and poly(dimethyldiallylammonium chloride)	9.1–99.9	Langmuir	Both electrostatic attraction and inner-sphere complexation were main sorption mechanisms.	Hua, 2018
As(V)	Bentonite-Anthracite@Zetag (BT-An@Zetag) composite	38.6	Freundlich	The electrostatic, ion exchange and functional surface groups were main sorption mechanism.	El-Aassar et al., 2022
As(III)	Fe-exchanged zeolite	100	Langmuir	The ion exchange was the main sorption mechanism.	Li et al., 2011
As(III); As(V)	Copper exchange zeolite	1.3; 1.4	Langmuir	The ion exchange was the main sorption mechanism.	Pillewan et al., 2014
As(III)	Zeolitic imidazolate frameworks	108.1–122.0	Langmuir	Hydroxyl substitution was the main mechanism for arsenic removal.	Liu et al., 2015
As(III); As(V)	Cellulose@iron oxide nanoparticles	23.1; 32.1	Langmuir	Both electrostatic and functional groups governed the sorption mechanism.	Yu et al., 2013
As	Copper (II) oxide nanoparticles	10.8	Langmuir	Chemisorption was the main sorption mechanism.	Goswami et al., 2012
As(III); As(V)	Iron oxide nanoparticles	42.0; 83.0	Langmuir	The removal of arsenic on nanosized iron oxide occurs through chemisorption, specifically via inner-sphere surface complexation.	Cheng et al., 2016

*NR, Not reported.

at 3.1 mg/g, while for As(V) removal, it was 3.8 mg/g. The main identified adsorption mechanisms were chemisorption and physisorption (which involve the filling of pores in the biochar) (Niazi et al., 2018).

One of the methods to analyze the structural properties and identify functional groups on biochar and other sorbents is FTIR spectroscopy. It enables the differentiation of functional groups between the original feedstock material and the resulting biochar, as well as the assessment of any changes in functional groups before and after As sorption (Amen et al., 2020). The FTIR spectrum of Oak wood biochar (OW-BC) before and after As removal is shown in Figure 4. In OW-BC, spectral bands indicated the presence of -OH and non-ionic carboxyl groups, as well as aliphatic methylene/methyl groups and C-O/C-O-C surface

functional groups. After As sorption, slight shifts and intensity changes were observed. Notably, in As(III)-loaded OW-BC, peak shifts and transformations were observed at specific wavelengths. These changes were relatively smaller in As(V)-loaded OW-BC (Niazi et al., 2018).

3.2 Removal of As with mineral-based adsorbents

Researchers have been increasingly drawn to natural mineral-based adsorbents as an alternative source. These adsorbents have garnered significant attention due to their low cost, abundance, easy retrievability, and exceptional adsorption capabilities. Among

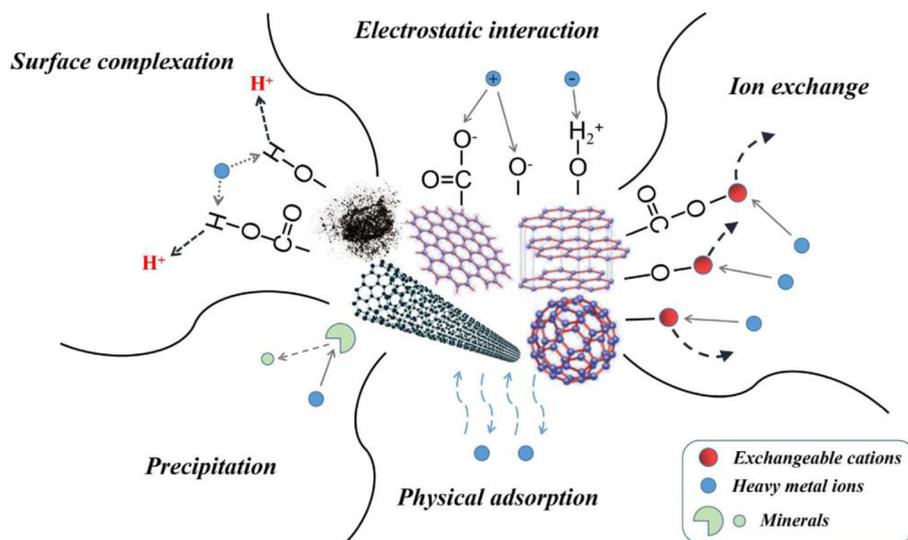


FIGURE 3 Schematic representation of heavy metal adsorption mechanisms on carbon adsorbents (Yang et al., 2019; copyright permission received from Elsevier).

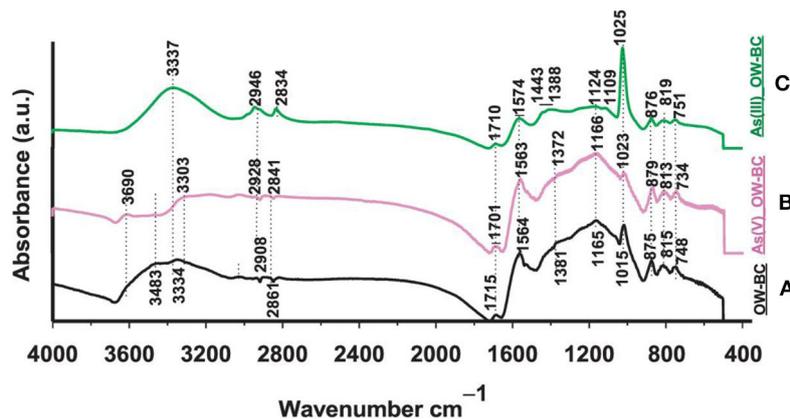
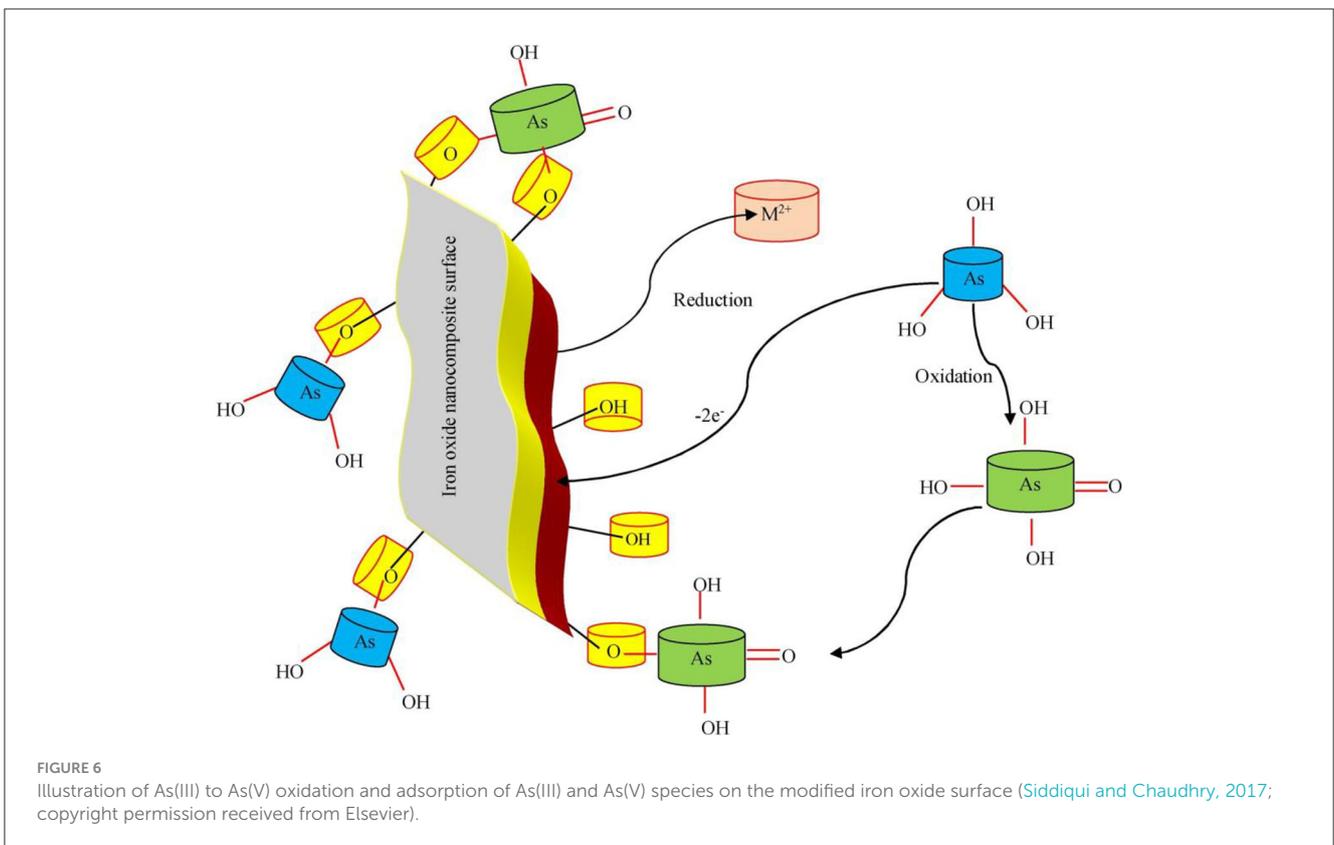
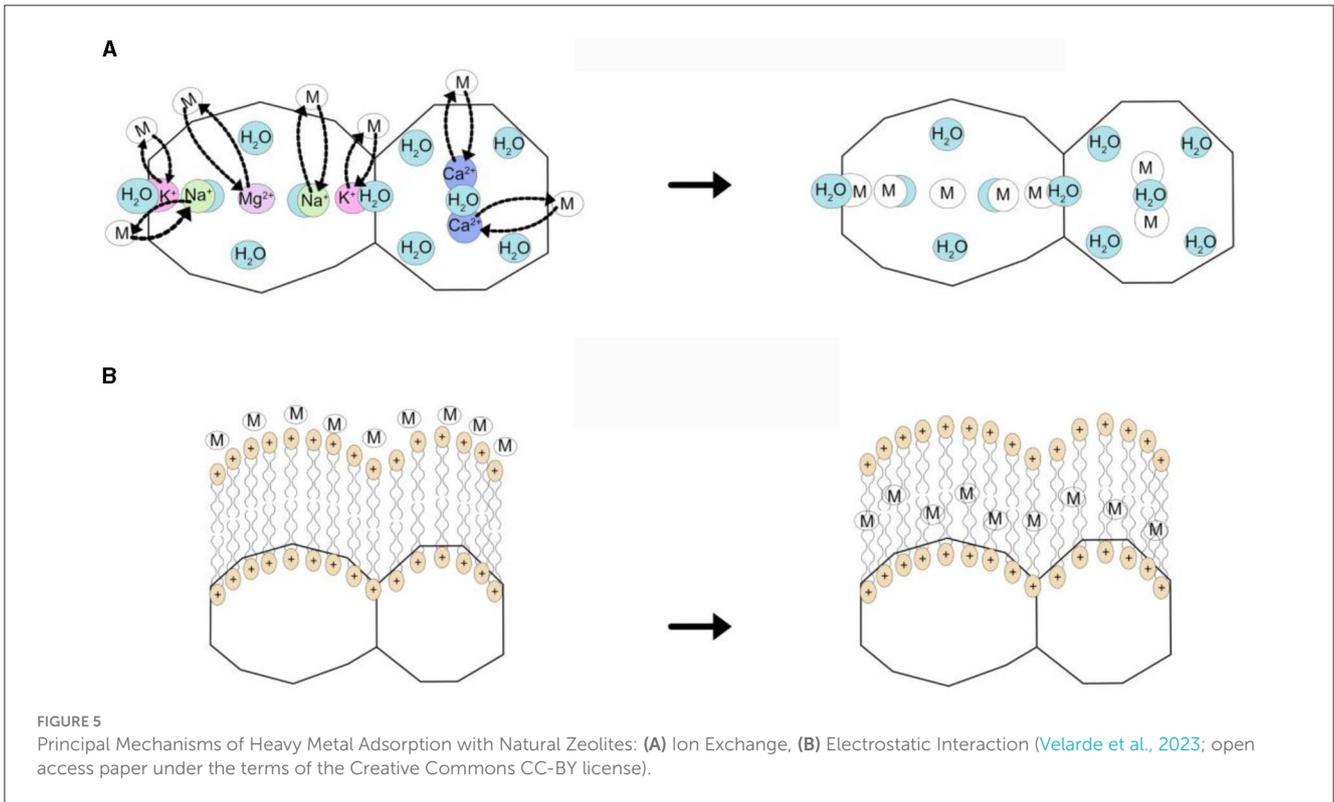


FIGURE 4 FTIR absorbance spectra of Japanese oak wood-derived biochar (OW-BC) prepared at 500°C: (A) OW-BC-As-Unloaded (solid black line), (B) OW-BC-As(V)-Loaded (solid pink line), and (C) OW-BC-As(III)-Loaded (solid green line); (Niazi et al., 2018; copyright permission received from Elsevier).

the numerous natural minerals found on Earth, zeolite and clays, particularly bentonite, have been extensively studied for their effectiveness in removing heavy metals (Zaimae et al., 2021). Clay minerals are phyllosilicates composed of T (tetrahedral) and O (octahedral) sheets in a 1:1 or 2:1 ratio. Due to isomorphous substitution, the layers of certain 2:1 clay minerals (such as smectite and vermiculite) carry a negative charge, which is balanced by cationic counterions in the interlayer space. These counterions can be exchanged, making clay minerals efficient adsorbents for cationic contaminants (Zhu et al., 2016). Bentonite, a clay with a high content of montmorillonite, is widely utilized for ion removal from water due to its bi-dimensional tetrahedral and octahedral sheets and isomorphous substitutions. It is both cheap and abundant, making it an ideal choice for synthesizing functionalized clay (Barakan and Aghazadeh, 2019). Arsenic

was effectively removed using modified bentonite, achieving a maximum adsorption capacity of 9.14–9.99 mg/g. The removal mechanisms involved a combination of electrostatic attraction and inner-sphere complexation, indicating a mixed removal mechanism (Hua, 2018). A modified bentonite [Fe(III)-modified bentonite] successfully removed As(III), exhibiting a maximum adsorption capacity of 0.3 mg/g (Baigorria et al., 2022). A significant removal efficiency of 84.5% was achieved for As(V) using modified bentonite (lanthanum-modified bentonite), with a maximum adsorption capacity of 3.8 mg/g (Cui et al., 2021).

Zeolites, whether naturally occurring or artificially synthesized, are hydrated aluminosilicate minerals. Their porous and cage-like structure enables them to provide significant internal and external surface areas for efficient ion exchange. This characteristic makes zeolites highly effective in adsorbing arsenic from water,



highlighting their potential as a valuable tool for eliminating this toxic substance (Salem Attia et al., 2014; Velarde et al., 2023). The application of a modified zeolite, the Copper Exchange Zeolite, resulted in the removal of over 98% of both As(III) and As(V). The maximum adsorption capacity for As(III) was measured at 1.3 mg/g, while for As(V) it was 1.4 mg/g (Pillewan et al., 2014). Electrostatic Interaction and ion substitution (Figure 5) are

key mechanisms involved in the removal of arsenic (As) using zeolite (Shevade and Ford, 2004; Liu et al., 2015). The maximum adsorption capacity (mg/g) was 0.5, at pH=6, in As(V) removal with a modified zeolite (Macedo-Miranda and Olguín, 2007).

3.3 Removal of As with metal oxide nano-adsorbents

Nanoparticles are tiny particles ranging in size from 1 to 100 nanometers. They exist in the transitional zone between individual molecules and larger materials. Due to their small size, nanoparticles possess unique physicochemical properties, including a high specific surface area, significant energy, and the confinement of quantum effects (Jagwe et al., 2023). Metal oxide nanoparticles (NPs) as adsorbents offer numerous benefits, including high specificity and capacity attributed to the quantum size effect and their large surface area. Moreover, these NPs possess unique structures, diverse pore sizes, low solubility, and can be easily synthesized using cost-effective techniques. Their low solubility, strong mechanical properties, and remarkable stability against organic dyes make metal oxides highly effective adsorbents (Hosny et al., 2023). Zinc oxide (ZnO) nanoparticles, silver oxide (Ag₂O) nanoparticles, copper oxide (CuO) nanoparticles, titanium dioxide (TiO₂) nanoparticles, and iron oxide nanoparticles are widely used metal oxide nanoparticles (Naseem and Durrani, 2021). Numerous iron-based materials and processes have been developed to address the issue of arsenic removal from drinking water. Iron oxides, oxyhydroxides, and hydroxides, such as amorphous hydrous ferric oxide (Fe(O)OH), goethite (FeO-OH), and hematite

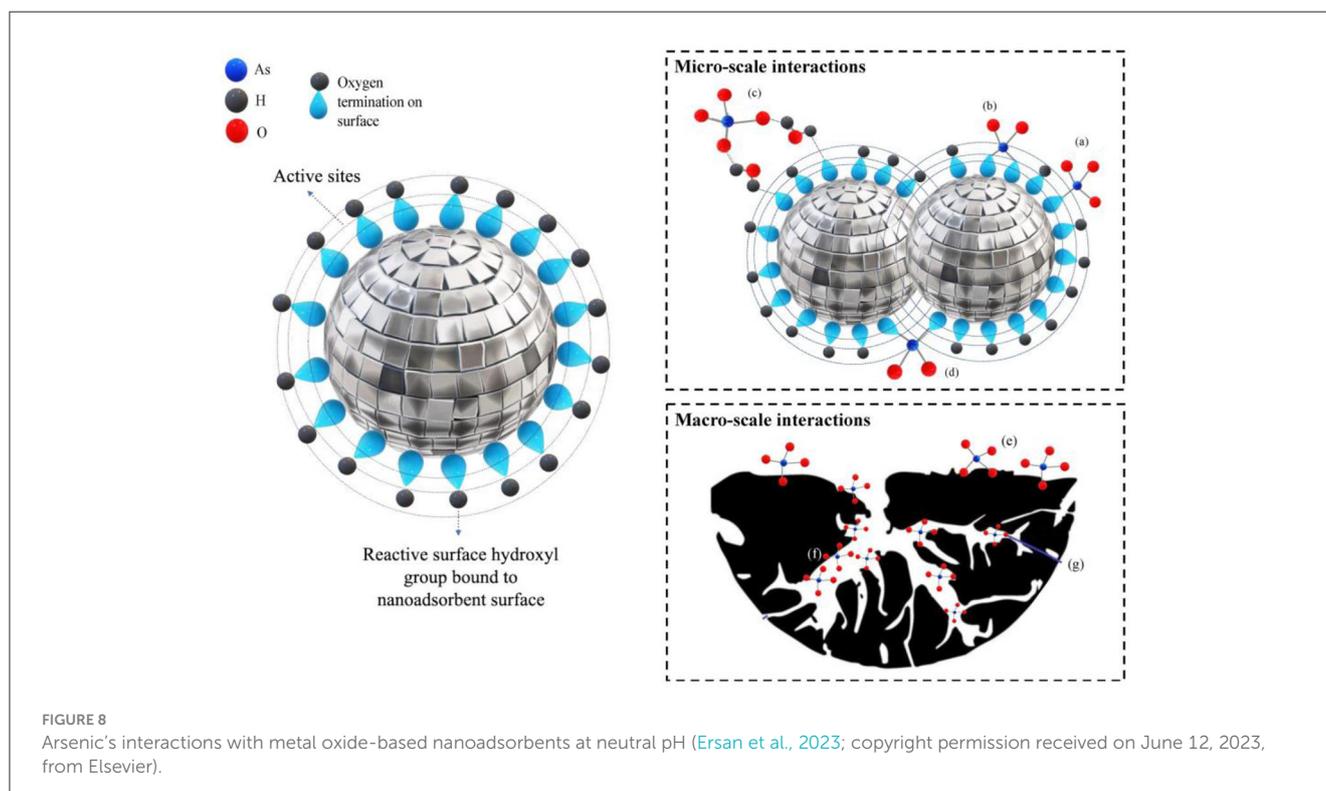
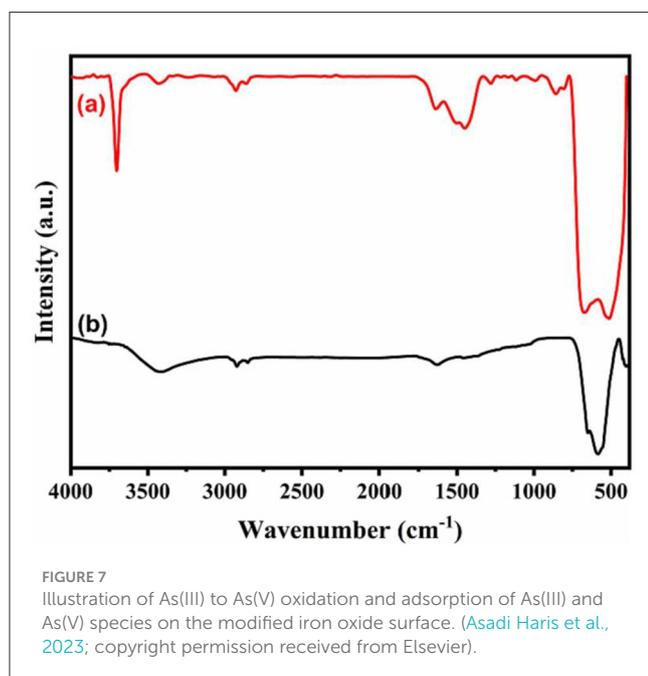


TABLE 4 Nanoparticle adsorbents for As removal.

Adsorbent	Dia. (nm)	Removal (%)	adsorption capacity mg/g	Initial con. of As mg/L		pH	References
				As (III)	As (V)		
Starch Functionalized Maghemite Nanoparticles (γ -Fe ₂ O ₃ @starch)	9.65	-	8.88	-	1-6	7	Siddiqui et al., 2020
Iron-modified activated carbon fiber (Fe ₂ O ₃ /ACF)	-	-	-	20.33	3	7	Chen et al., 2016
Zinc Oxysulfid Nanomaterials (ZnOxS _{1-x})	5–6	99.9	299.4	299.4	0.01	6-8	Uppal et al., 2019
Activated Carbons Modified with Iron hydro(oxide)F400-M	3–36	-	-	0.3-1.2	0.025-5	6-8	Vitela-Rodriguez and Rangel-Mendez, 2013
SiO ₂ @Fe ₃ O ₄ @MBT	40	95.77 As(V)	-	-	1	6.55	Sheikhmohammadi et al., 2018
Fe ₃ O ₄ loaded activated carbon	-	-	-	204.2	-	8	Liu et al., 2010
Graphene oxide-MnFe ₂ O ₄ nanohybrid	-	-	146	207	-	4-6.5	Kumar et al., 2014
Zirconium-nanoscale carbon	50–70	-	-	110	100	2.5	Mahanta and Chen, 2013
Nano zero valent iron supported on Activated carbon	<100	-	18.2	12.0	2	6.5	Zhu et al., 2009
Nanocrystal line Titanium dioxide	6	95% AS(III) >98% As(V)	-	-	1	4-9.5	Pena et al., 2006
Functionalized Graphene Sheets	-	-	139	142	300	6-7	Mishra and Ramaprabhu, 2011
Core/Shell structural nZVI/Mn Oxide	-	-	29.4	35.7	5	4.8	Bui et al., 2017
Fe (III)-Cu (II) binary Oxide	50	-	123.3	82.7	10	7	Zhang G. et al., 2013
Ultra-small nanoparticles (IONPs) within Electrospun Hydrophilic Poly(vinyl alcohol) (PVA) nanofibers	<5	-	-	3.5±0.3	0.01	5	Torasso et al., 2023
Copper oxide nanoparticles	12–18	-	26.9	22.6	0.1-100	6-10	Martinson and Reddy, 2009
Zr-UiO-66-SH-A	-	-	90.7	98.8	-	-	Shao et al., 2019
Nano-Alumina wrapped Carbon Microspheres (Al-CMs)	-	-	-	68.0	-	2-12	Raj et al., 2023
ZnO Nanorods Coated Porous Ceramic Monolith	100	98 As(V)	-	-	0.2	7	Muensri and Danwittayakul, 2017
Alginate coated Superparamagnetic Iron Oxide Nanoparticles (SPIONs)	25	99	240.08	-	6.5	7	Asadi Haris et al., 2023
Zr-metal-organic framework (MOF) UiO-66	-	-	352.1	147.5	-	3-11	Qu et al., 2022
Zeolite-supported nanoscale zero-valent iron (Z-NZVI)	40–60	-	11.52	-	10	6	Li et al., 2018

TABLE 4 (Continued)

Adsorbent	Dia. (nm)	Removal (%)	adsorption capacity mg/g	Initial con. of As mg/L		pH	References
				As (III)	As (V)		
Copper Ferrite (CuFe ₂ O ₄) Foam (CFF)	-	-	44	85.4	10	8.1	Wu et al., 2018a
Amine functionalization of iron-based metal-organic frameworks MIL-101 [NH ₂ -MIL-101(Fe)]	-	-	153.4	147.7	-	7	Fang et al., 2023
Copper-doped ZIF-8 nanomaterials (Cu-ZIF-8)	-	-	-	238.11	10-350	3-11	Wang et al., 2023
Cerium oxide (CeO ₂) nanoparticles (NPs)	40–50	-	451 As(III) 119 As(V)	-	-	3.6	Bi et al., 2020

(Fe₂O₃), have demonstrated their efficacy in removing arsenic from aqueous solutions (Dhanasekaran and Sahu, 2021). Besides, in a study (Siddiqui and Chaudhry, 2017), a Fe₂O₃ nano-composite adsorbent has been functionalized to enhance arsenic (As) removal. As illustrated in Figure 6, Fe₂O₃ nano-composites have proven to be efficient adsorbents for arsenic cleanup from water. This efficacy is attributed to the presence of a large number of hydroxyl and other functional groups on the surface, enabling a higher capacity for the oxidation of As(III) to As(V). The functional groups attached to the composite material's surface provide strong trapping sites for arsenic across various pH levels.

In a study (Pham et al., 2020), As(V) was removed by a ferric hydroxide-based adsorbent, the maximum removal capacity was 2.9 (mg/g). In a separate study (Asadi Haris et al., 2023), superparamagnetic iron oxide nanoparticles (SPIONs) and alginate-encapsulated SPIONs (SPIONs-Alg) were synthesized for arsenic (As) removal from water. Figure 7 illustrates key bands: 3,400 cm⁻¹ (O–H symmetric vibration), 2,936 cm⁻¹ (aliphatic C–H stretching), 600 cm⁻¹ ([Fe–O] intrinsic stretching), and 400 cm⁻¹ (octahedral [Fe–O] vibration), confirming ferrite formation in spinel form. Notably, SPIONs-Alg exhibited a distinct 585 cm⁻¹ band, indicating the presence of nanoferrite.

Due to the formation of stable bidentate binuclear surface complexes on its surface, TiO₂ shows great promise as a material for the removal of As from industrial wastewater (Qiu et al., 2019). In a study (Deng et al., 2021), the maximum adsorption capacities (mg/g) for the removal of As(III) and As(V) using a TiO₂-based adsorbent were 7.7 and 18.2, respectively. A study utilized a modified metal oxide nanoparticle (Ce_{0.8}Ti_{0.2}O_{2–y}) to remove arsenic. The adsorbent demonstrated a maximum adsorption capacity of 2 × 10⁵ mg/g (Mishra and Rai, 2019). Their study revealed that the removal procedure was primarily governed by two mechanisms. Firstly, the redox reaction involving ceria and titania metals facilitates the partial oxidation of the more toxic As(III) to the less toxic As(V). The resulting As(V) can readily form monodentate and bidentate complexes. Secondly, the adsorption of As(III) occurs through the interaction between the surface hydroxyl groups of the synthesized adsorbent and the As(III) species. In the study (Mishra and Rai, 2019), Titania-doped cerium oxide nanoparticles (Ce_{1–x}Ti_xO_{2–y}) were synthesized using a single-pot aerogel process. Cerium nitrate hexahydrate (40 mM) and stoichiometric oxalic acid were dissolved in a mixture of

ethanol and toluene. Titanium isopropoxide (10% and 20% w/w of cerium nitrate hexahydrate) in 50 mL of ethanol, along with distilled water, were gradually added to the precursor mixture. Vigorous stirring at 300 rpm resulted in a light yellowish gel. The gel was transferred to a Parr reactor and heated at a rate of 1°C per minute. The solvents were supercritically dried under elevated temperature and high pressure, yielding a light yellow, low-density powder of Ce_{1–x}Ti_xO_{2–y} oxide nanoparticles. These nanoparticles were then used as adsorbents for arsenic removal. The use of various metal oxide nanoparticles resulted in the removal of over 92% of As(V), with an impressive adsorption capacity of up to 305 mg/g (Hocaoglu et al., 2019). They mentioned that the primary adsorption mechanism of metal oxides relies on the formation of strong bonds between the surface –OH groups of the metal oxides and the –OH group of the arsenic(V) species. In another study (Sunil et al., 2018), more than 96% of As was removed with Al-Ti₂O₆ nanoparticles. In another study (Powell et al., 2020), carbon-coated iron carbide (Fe₃C@C) was used to remove arsenic (As) from groundwater. The maximum adsorption capacity was 168 micrograms per gram (μg/g). Figure 8 illustrates the interactions between arsenic and metal oxide-based nanoadsorbents under neutral pH conditions. Table 4 shows the removal of As using various nanoparticle adsorbents.

4 Conclusions

The presence of arsenic in water bodies poses a significant risk to both human health and the environment. Arsenic contamination in water sources is a global environmental concern caused by both natural processes and human activities. The high concentrations of arsenic found in drinking water, groundwater, wastewater, and aquatic organisms highlight the urgent need for effective removal methods. This review manuscript critically evaluated the effectiveness and limitations of adsorption methods for arsenic removal from water bodies. Adsorption emerged as a promising technique due to its cost-effectiveness, high removal efficiency, and simplicity of operation. Various adsorbents, including nanoparticle adsorbents and graphene-based adsorbents, demonstrated a high capacity for arsenic removal. Additionally, low-cost adsorbents such as zeolites, clay, and chitosan, as

well as agricultural waste materials and biochar, have shown potential for arsenic adsorption. Future research should focus on optimizing adsorbent materials, understanding the mechanisms of arsenic adsorption, and developing sustainable and efficient regeneration techniques.

Author contributions

AM: Conceptualization, Investigation, Resources, Supervision, Writing—original draft, Writing—review & editing. ER: Conceptualization, Validation, Writing—review & editing. BK: Conceptualization, Validation, Writing—original draft, Writing—review & editing. SR: Writing—review & editing. NK: Funding acquisition, Writing—review & editing. MV: Writing—review & editing. HF: Writing—review & editing.

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Conflict of interest

ER and HF were employed by Envirowise Consultant Limited.

The remaining authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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