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REVIEW ARTICLE

Influence of Different Conditions on the Sorption of Potentially Toxic Elements by Selected Sorbents: A Review

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Abstract
Information about how different conditions affect the course of sorption is variously scattered and needs to be consolidated. The paper primarily focuses on identifying the optimal sorption conditions for acid and neutral mine drainages. In this review, key parameters were assessed, including temperature, pH level, sorbent dose, initial metal concentrations, and sorption duration. This overview also includes a comparison of the advantages and disadvantages of selected types of sorbents. The sorption of many metals tends to be optimal at circumneutral pH values. The adsorptive capacity decreases with an increase in temperature for exothermic processes, whereas it increases in the case of an endothermic one. Increasing the initial concentration has a positive effect on adsorption until the sorbent is fully saturated, leading to a plateau in adsorption capacity. The knowledge gained from this research extends the spectrum of the potential sorption applications, especially in the processes of recovering the metals and sorbents by desorption.

Keywords Mining wastewater · Optimal conditions · Efficiency · Water purification · Mine drainage

Introduction

Mine effluents, also referred to as mine drainage, can contain substances such as cyanide and various metals that have serious potential health and environmental consequences (Azapagic 2004). Acid mine drainage (AMD) production typically, but not exclusively, occurs in rocks containing iron sulphide. Although this process occurs naturally, mining can promote the generation of AMD simply by increasing the amount of sulphide exposed to oxygen and water (Akcil and Koldas 2006). In many cases, due to the neutralizing ability of waste minerals or human intervention, such as the spreading of limestone to precipitate metals, drainage may have circumneutral pH values (4.5–8.5) and is referred to as neutral mine drainage (NMD). NMD can also cause environmental problems in mining environments because many metals and metalloids can remain soluble at alkaline pH under appropriate redox conditions (Banks et al. 1997; Lindsay et al. 2009).

Currently, the main methods for the treatment of wastewater containing metal ions include chemical precipitation, electrochemical methods, reverse osmosis, ion exchange, and adsorption (Johnson and Hallberg 2005; Wang et al. 2022). Adsorption is often used to remove metals from mine water, especially when natural and inexpensive adsorbent materials are available in large quantities or certain waste products from industrial or agricultural activities can serve as inexpensive sorbents (Bailey et al. 1999). Examples include dead biomass, blast furnace slag, fly ash, clay, tree bark, tea leaves, and natural zeolite (Bailey et al. 1999; Bhattacharyya and Gupta 2006; Günay et al. 2007). These can be combined to obtain the desired adsorption properties (Crini and Morcellet 2002). Some of the more cost effective natural and renewable sorbents are chitosan, tea leaves, brown coal,

However, there is currently no survey developed to monitor the effect of sorption conditions on metal removal from the water system. Due to the lack of information in this field, we have focused our review of the sorption conditions in terms of temperature, pH, inlet concentration as well as sorbent availability in this review article.

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sewage sludge, agricultural waste, and biomass (Ahmaruzzaman 2008; Kalderis et al. 2008; Munagapati et al. 2010). Some studies have focussed on the sorption of Hg from mine effluents (e.g. Bae et al. 2003; Fiaizullina et al. 2017; Gómez-Giménez et al. 2017).

The sorption properties of adsorbents, methods to determine the sorption properties of metal ions, and sorption mechanisms (batch sorption techniques and spectroscopic techniques, such as XPS, FTIR, EXAFS, and ATR-IR) have been addressed (Zhao et al. 2010). Also, the physical basis of the sorption isotherm, with a detailed description of the experimental methods and various empirical or mechanistic models to obtain the sorption isotherm have been summarized (Limousin et al. 2007). Theoretical models describing the kinetics of pollutant adsorption was reviewed (Plazinski et al. 2009), as have sorption models (Basu et al. 2006). In addition, the influence of temperature on sorption equilibrium and sorption kinetics for organic micropollutants was previously compiled (Hulscher and Cornelissen 1996). However, there is no published survey on the effect of other sorption conditions on metal removal, so this review is an effort to unify the available scientific information from the field on the influence of conditions on the sorption process.

Description and Division of Sorbents

Sorption of metals under laboratory conditions is currently applied using natural sorbents, natural biosorbents (Zhao et al. 2010), waste materials (Apak et al. 1998; Koshy and Singh 2016) or modified sorbents. In many cases, these are innovative ways of using sorbents (Hagarová and Nemček 2021; Pereira et al. 2019).

Natural Non-biological Sorbents

Natural sorbents include bentonites and zeolites (Malamis and Katsou 2013). In addition to these commonly used sorbents, vermiculite can also be used. Vermiculite is a type of clay that is usually found in nature in the form of magnesium. It consists of two layers of silicon tetrahedra, in which silicon is partially replaced by aluminium, and a layer of hydroxy (-OH) groups and magnesium ions forming a strongly bonded mica layer (Fleischer 1975; Malamis and Katsou 2013).

Currently, the adsorption of pollutants by natural materials is widely reported, such as diatomite (Shabani et al. 2014; Yurak et al. 2021), glauconite (Flieger et al. 2020), perlite (Ghassabzadeh et al. 2010), red mud (Lopes et al. 2013), zeolite (Egashira et al. 2012), and clay (Sheikhhosseini et al. 2013). These adsorbents are environmentally friendly and most of them can be regenerated or used in various products and are commonly used for synthetic wastewater treatment (Shabani et al. 2014). In addition to the above sorbents, a dolomite sorbent was prepared for the metals sorption (Ivanets et al. 2016). Several natural sorbents including kaolinite were described for wastewater treatment (Adebowale et al. 2006), sludge (Genç-Fuhrman et al. 2004), diatomite (Al-Degs et al. 2001; Li et al. 2009) and laterite (Yu et al. 2008). The sorption mechanism of natural non-biosorbents and biosorbents is shown in Figs. 1 and 2.

Conditions for Sorption Processes by Natural Non-biological Sorbents

In some cases, the pH of the solutions was adjusted with acids and hydroxides during the experiments. Tables 1, 2 and 3 focus on the sorption of potentially toxic elements from wastewater and artificially produced aqueous solutions. The aqueous solutions were prepared by the addition of a salt containing the metal of interest. The sorption conditions—pH, temperature, inlet concentration, sorbent addition—are reported depending on the embedding experiment.

Natural and Waste Biosorbents

In recent years, many economically available biosorption materials have been used and have proven to be promising methods for metal removal. The major benefits of biosorbents for metal sorption are low operating costs, high efficiency, and minimal generation of toxic sludge (Bailey et al. 1999; Kratochvil and Volesky 1998).

Based on the results of previous studies, it is possible to use biosorbents such as tea waste (Orhan and Büyükgüngör 1993), coffee husks (Quyen et al. 2021), orange peel (Altunkaynak et al. 2022), banana peel, potato peel (Nathan et al. 2021), pomegranate peel, pineapple peel (Turkmen Koc et al. 2021), coconut, peanut, almond, walnut shells (Das et al. 2020; Kali et al. 2024), mango and guava bark (Krishnani et al. 2021), and many other agricultural by-products. Thus, the waste streams of the agricultural sector is one of the major sources of natural biosorbents. Composted manure has also been used as a biosorbent (Zhang 2011). In addition, waste digested activated sludge from wastewater treatment plants (Appels et al. 2008) and industrial waste such as ladle furnace slag, fly ash (Mohammed et al. 2017), and steel slag (Pfeifer et al. 2021) can be used as low-cost adsorption materials for the sorption of metals from mine effluents. The sorption mechanism of waste biosorbents is shown in Fig. 3.

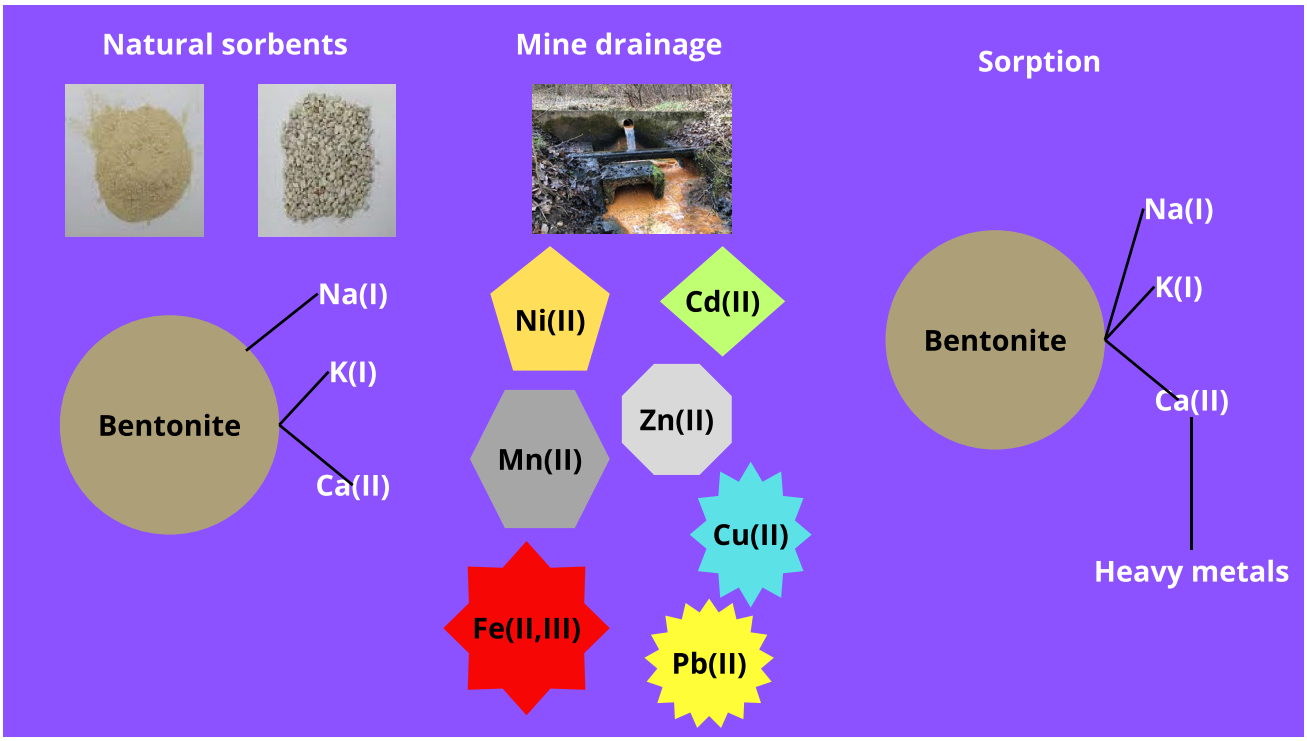

Fig. 1 Graphical scheme of metal sorption with the use of natural non-biological sorbents

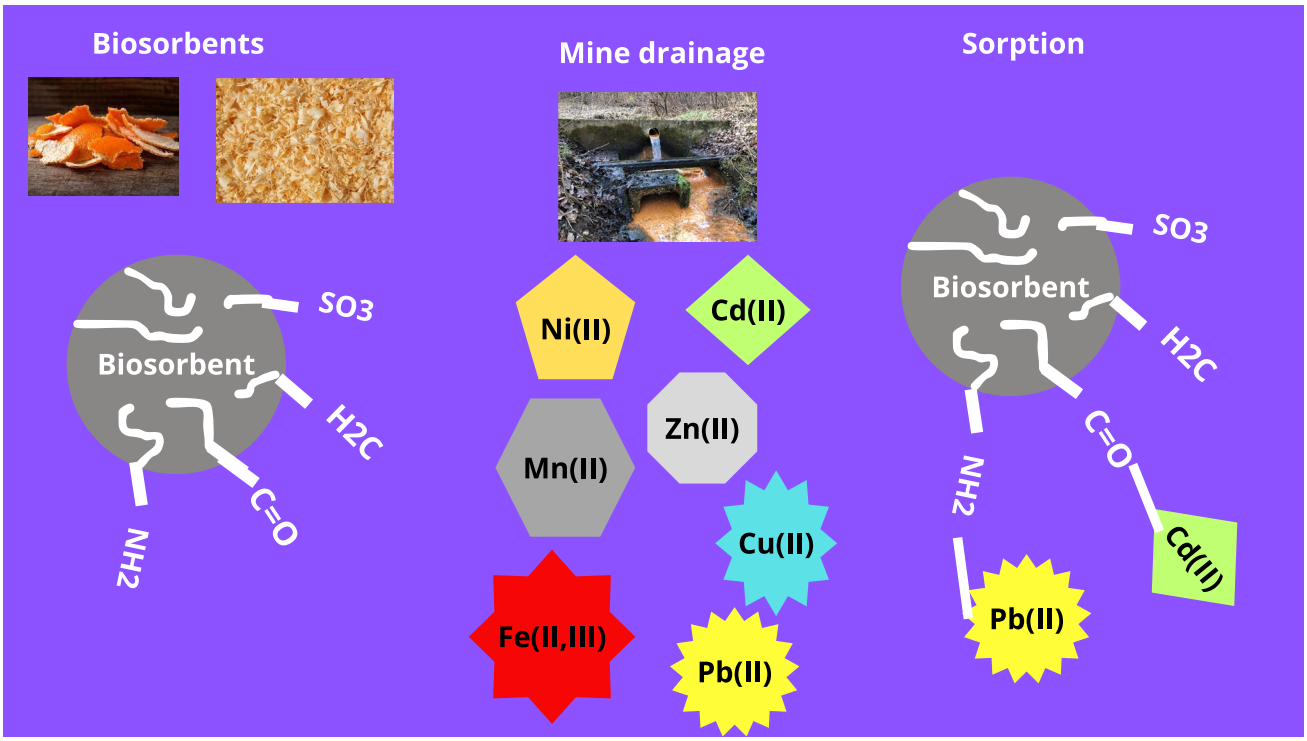

Fig. 2 Graphical scheme of metal sorption with the use of biosorbents

Table 1 Sorption of PTEs by natural non-biological sorbents

Metal	Sorbent	Type of water	pH	Temperature (°C)	Initial concentration (mg/l)	Contact time (h)	Removal efficiency [%]	Uptake sorbent capacity mg/g	Dosage (g)	Isotherm model	References
Cu(II)	Zeolite	Aqueous metal ion solution	7	25	111.1	24	98.47	13.69	0.4	Langmuir	Pfeifer et al. (2021)
Cu(II)	Zeolite	Acidic solutions	4.2	25	50	24	89	3.8	1	–	Balintova et al. (2012)
Cu(II)	Zeolite—300	Acidic solutions	2	24	90	12	17.1 ± 1.5	–	5.0 ± 0.2 g/L	Langmuir and Freundlich	Ryu et al. (2019)
Cu(II)	Zeolite—400	Acid mine drainage	2	24	90	12	22.8 ± 1.2	-	5.0 ± 0.2 g/L	Langmuir and Freundlich	Ryu et al. (2019)
Cu(II)	Zeolite—500	Acid mine drainage	2	24	90	12	31.3 ± 1.9	-	5.0 ± 0.2 g/L	Langmuir and Freundlich	Ryu et al. (2019)
Cu(II)	Zeolite—600	Acid mine drainage	2	24	90	12	29.1 ± 2.1	3.15	5.0 ± 0.2 g/L	Langmuir and Freundlich	Ryu et al. (2019)
Cu(II)	Zeolite	Acid mine drainage	2.5–2.86	18–22	136	6	35.88	–	3.2	–	Olegario-Sanchez and Pelicano (2017)
Cu(II)	Slovakite	Synthetic solution of Cu(II)	4	Room temperature	50	24	99.7	4.99	1	Brunauer–Emmett–Teller	Balintova et al. (2014)
Cu(II)	Bentonite	AMD	2.56–3.72	25 ± 2	181.42	3	93.30	–	10 g/L	–	Gitari (2014)
Cu(II)	Bentonite	Aqueous metal ion solution	7	25	111.1	24	–	13.87	0.4	Langmuir	Pfeifer et al. (2021)
Cu(II)	Magnetic natural composite Fe_3O_4⁻chitosan@ zeolite	AMD	5	25	0.16	2.5	> 89	–	0.05	Langmuir and Freundlich	Feng et al. (2019)
Cu(II)	Zeolite	AMD	3	22	100	2	98.16	30.03	1.5 g/L	Langmuir and Freundlich	Wulandari et al. (2020)
Cu(II)	Synthesised zeolite	AMD	3	22	100	2	93.98	23.8	21 g/l	Langmuir and Freundlich	Wulandari et al. (2020)
Fe(II)	Zeolite	AMD	2	24	340	12	12.8 ± 0.9	–	5.0 ± 0.2 g/L	Langmuir and Freundlich	Ryu et al. (2019)
Fe(II)	Zeolite—300	AMD	2	24	340	12	17.9 ± 1.6	–	5.0 ± 0.2 g/L	Langmuir and Freundlich	Ryu et al. (2019)
Fe(II)	Zeolite—400	AMD	2	24	340	12	25.3 ± 2.2	–	5.0 ± 0.2 g/L	Langmuir and Freundlich	Ryu et al. (2019)
Fe(II)	Zeolite—500	AMD	2	24	340	12	34.9 ± 2.7	11.95	5.0 ± 0.2 g/L	Langmuir and Freundlich	Ryu et al. (2019)
Fe(II)	Zeolite—600	AMD	2	24	340	12	31.0 ± 2.5	–	5.0 ± 0.2 g/L	Langmuir and Freundlich	Ryu et al. (2019)

Table 1 (continued)

Metal	Sorbent	Type of water	pH	Temperature (°C)	Initial concentration (mg/l)	Contact time (h)	Removal efficiency [%]	Uptake sorbent capacity mg/g	Dosage (g)	Isotherm model	References
Fe(II)	Zeolite	Aqueous metal ion solution	5	25	97.7	24	96.11	12.13	0.4	Langmuir	Pfeifer et al. (2021)
Fe(III)	Zeolite	AMD	2.8	22 ± 0.5	112	24	99.11	–	5	BET—Brunauer–Emmett–Teller isotherm	Varvara et al. (2013)
Fe(III)	Zeolite	AMD	2.5	22 ± 2	400	6	59.9	6	3.7	Langmuir and Freundlich isotherms	Motsi (2010)
Fe(II)	Modified bentonite clays	AMD	2–6	25	100–800	24	30–44	–	0.1	Langmuir and Freundlich	Gumede and Musonge (2022)
Fe(III)	Magnesite-bentonite	AMD	< 3	26	100–2000	6	97–100	199.9	0.1–8	Langmuir and Freundlich	Masindi et al. (2017)
Fe(III)	Bentonite	AMD	2.56–3.72	25 ± 2	179.94	3	65.63	–	10 g l^{-1}	–	Gitari (2014)
Fe(III)	Bentonite	Aqueous metal ion solution	1.65–2.37	23 ± 0.2	100	1	–	3.9	25 g/L	Langmuir	Flieger et al. (2020)
Fe(III)	Glauconite	Aqueous metal ion solution	1.65–2.37	23 ± 0.2	100	1	–	2.5	25 g/L	Langmuir	Flieger et al. (2020)
Fe(III)	Perlite	Aqueous metal ion solution	1.65–2.37	23 ± 0.2	100	1	–	0.9	25 g/L	Langmuir	Flieger et al. (2020)
Fe(II)	Magnetic natural composite Fe O_{34}^{-} chitosan@ bentonite	AMD	2	25	102.9	2.5	84		0.05	Langmuir and Freundlich	Feng et al. (2019)
Fe(II)	Bentonite	Aqueous metal ion solution	7	25	97.7	24	–	11.99	0.4	Langmuir	Pfeifer et al. (2021)
Fe(II)	Bentonite clay	AMD	2.7	30	40 $g.l^{-1}$	6	83	~80	4	Langmuir, Freundlich, Dubinin–Radushkevich (D–R), and Temkin	Orakwue et al. (2016)
Fe(III)	Bentonite	AMD	< 3	26	100–2000	0.5	> 99	–	8 g/L	Langmuir and Freundlich	Masindi et al. (2015)

Table 1 (continued)

Metal	Sorbent	Type of water	pH	Temperature (°C)	Initial concentration (mg/l)	Contact time (h)	Removal efficiency [%]	Uptake sorbent capacity mg/g	Dosage (g)	Isotherm model	References
Fe(III)	Bentonite—13 µm	Aqueous metal ion solution	–	25	4000	2	> 99	16.65	1	Langmuir, Freundlich, Redlich Peterson	Bakalár et al. (2020)
Fe(III)	Zeolite—20 µm	Aqueous metal ion solution	–	25	4000	2	> 95	10.19	1	Langmuir, Freundlich, Redlich Peterson	Bakalár et al. (2020)
Pb(II)	Zeolite	AMD	2.5–2.93	18–22	210	6	99.03	–	3.2	–	Olegario-Sanchez and Pelicano (2017)
Pb(II)	Natural clinoptilolite	Aqueous metal ion solution	5	25 ± 1	1000	6	63.5	–	1	Langmuir and Freundlich	Oter and Akcay ((2007)
Zn(II)	Zeolite—300	AMD	2	24	120	12	15.6 ± 2.4	–	5.0 ± 0.2 g/L	Langmuir and Freundlich	Ryu et al. (2019)
Zn(II)	Zeolite—400	AMD	2	24	120	12	21.1 ± 2.0	–	5.0 ± 0.2 g/L	Langmuir and Freundlich	Ryu et al. (2019)
Zn(II)	Zeolite—500	AMD	2	24	120	12	29.3 ± 2.3	–	5.0 ± 0.2 g/L	Langmuir and Freundlich	Ryu et al. (2019)
Zn(II)	Zeolite	AMD	2.8	22.5 ± 0.5	9.6	24	71.94	–	5	BET	Varvara et al. (2013)
Zn(II)	Zeolite	AMD	3.5	22 ± 2	120	6	68.1	2.1	5	BET	Motsi (2010)
Zn(II)	Zeolite	AMD	2	24	120	30	9.4 ± 1.4	–	5.0 ± 0.2 g/L	Langmuir and Freundlich	Ryu et al. (2019)
Zn(II)	Bentonite	AMD	2.56–3.72	25 ± 2	148.43	3	51.03	–	1 g/L	Brunauer–Emmett–Teller (BET)	Gitari (2014)
Zn(II)	Zeolite—600	AMD	2	24	120	30	27.0 ± 2.3	–	5.0 ± 0.2 g/L	Langmuir and Freundlich	Ryu et al. (2019)
Mn(II)	Zeolite	AMD	4.5	22 ± 2	20	6	95	0.52	37 g/L	Langmuir and Freundlich	Motsi et al. (2009)
Mn(II)	Zeolite	AMD	2.8	22 ± 0.5	20	24	45	–	5	BET	Varvara et al. (2013)
Mn(II)	Zeolite	AMD	3.5	22 ± 2	20	6	18.9	0.5	15	BET	Motsi (2010)
Mn(II)	Bentonite	AMD	< 3	26	5–100	6	84–100	3	20 g/L	Langmuir and Freundlich	Masindi et al. (2015)

Table 1 (continued)

Metal	Sorbent	Type of water	pH	Temperature (°C)	Initial concentration (mg/l)	Contact time (h)	Removal efficiency [%]	Uptake sorbent capacity mg/g	Dosage (g)	Isotherm model	References
Mn(II)	Modified bentonite clays	AMD	2–6	25	10–50	24	18–40	–	0.1	Langmuir and Freundlich	Gumede and Musonge (2022)
Mn(II)	Magnesite–bentonite	AMD	< 3	26	100	6	100	9.99	10 g/L	Langmuir and Freundlich	Masindi et al. (2017)
Mn(II)	Bentonite	AMD	2.56–3.72	25 ± 2	2863.36	3	18.14	–	10 g/L	BET	Gitari (2014)
Mn(II)	Bentonite	Aqueous metal ion solution	5.66–6.89	23 ± 0.2	100	1	–	3.6	25 g/L	Langmuir	Flieger et al. (2020)
Mn(II)	Glauconite	Aqueous metal ion solution	5.66–6.89	23 ± 0.2	100	1	–	2.55	25 g/L	Langmuir	Flieger et al. (2020)
Mn(II)	Perlite	Aqueous metal ion solution	5.66–6.89	23 ± 0.2	100	1	–	3.15	25 g/L	Langmuir	Flieger et al. (2020)
As(V)	Iron hydro (oxide) modified zeolite	AMD	2.23	23–53	0.38	1.5	99	1.69	3 g/30 ml	Langmuir and Freundlich	Nekhunguni et al. (2017)
As(V)	Natural ZMA–Fe) Clinoptilolite + erionite)	Aqueous solutions	5	22–27	As(III) + As(V)–100 and 20 µg l^{-1}	24	75	–	–	–	Elizalde-González et al. (2001)
Cd(II)	Modified zeolite oxalic acid	AMD	6.81	26.9	1–8	24	46.68	–	8 mg/L	–	Mokgehle et al. (2019)
Cd(II)	Modified zeolite succinic acid	AMD	6.81	26.9	1–8	24	46.57	–	8 mg/L	–	Mokgehle et al. (2019)
Cd(II)	Magnetic natural composite Fe_3O_4–chitosan@bentonite	AMD	2	25	0.972	–	> 89	–	0.05	Langmuir and Freundlich	Feng et al. (2019)
Ni(II)	Zeolite	AMD	2	24	3.5	12	10.8 ± 1.1	–	5.0 ± 0.2 g/L	Langmuir and Freundlich	Ryu et al. (2019)
Ni(II)	Zeolite	AMD	2.5–2.96	18–22	110	6	35.36	–	3.2	–	Olegario-Sanchez and Pelicano (2017)
Ni(II)	Zeolite—300	AMD	2	Ambient 24	3.5	12	15.9 ± 2.0	–	5.0 ± 0.2 g/L	Langmuir and Freundlich	Ryu et al. (2019)
Ni(II)	Zeolite—400	AMD	2	24	3.5	12	19.7 ± 1.9	–	5.0 ± 0.2 g/L	Langmuir and Freundlich	Ryu et al. (2019)
Ni(II)	Zeolite—500	AMD	2	24	3.5	12	21.1 ± 1.3		5.0 ± 0.2 g/L	Langmuir and Freundlich	Ryu et al. (2019)
Ni(II)	Zeolite—600	AMD	2	24	3.5	12	20.0 ± 2.9	–	5.0 ± 0.2 g/L	Langmuir and Freundlich	Ryu et al. (2019)

Table 1 (continued)

Metal	Sorbent	Type of water	pH	Temperature (°C)	Initial concentration (mg/l)	Contact time (h)	Removal efficiency [%]	Uptake sorbent capacity mg/g	Dosage (g)	Isotherm model	References
Ni(II)	Magnetic natural composite Fe_3O_4-chitosan@bentonite	AMD	2	25	–	–	> 89	–	0.05	Langmuir and Freundlich	Feng et al. (2019)
Ni(II)	Bentonite	AMD	2.56–3.72	25 ± 2	2952.34	3	35.97	–	10 g./L	BET	Gitari (2014)
Ni(II)	Bentonite (deposit Slovakia–Stará Kremnička)	Aqueous metal ion solution	5.85	20	50	2	99	9.0	5 g/L	Langmuir	Šuránek et al. (2021)
Ni(II)	Bentonite (deposit Slovakia–Stará Kremnička)	Aqueous metal ion solution	5.85	20	100	2	90	15.1	5 g/L	Langmuir	Šuránek et al. (2021)
Ni(II)	Bentonite (deposit Slovakia–Hliník nad Hronom)	Aqueous metal ion solution	5.85	20	50	2	60	5.0	5 g/L	Langmuir	Šuránek et al. (2021)
Ni(II)	Bentonite (deposit Slovakia–Hliník nad Hronom)	Aqueous metal ion solution	5.85	20	100	2	40	5.2	5 g/L	Langmuir	Šuránek et al. (2021)
Ni(II)	Bentonite	Aqueous metal ion solution	5.66–6.89	23 ± 0.2	100	1	–	3.7	25 g/L	Langmuir	Flieger et al. (2020)
Ni(II)	Glauconite	Aqueous metal ion solution	5.66–6.89	23 ± 0.2	100	1	–	1.1	25 g/L	Langmuir	Flieger et al. (2020)
Ni(II)	Bentonite clay	AMD	4.85	–	14	–	99.5	–	10 g/L	–	Enslin et al. (2010)
Ni(II)	Zeolite	Aqueous metal ion solution	4.0	30	50	4	60.04	75.04	0.4 g/L	Langmuir	Al-Abbad and Al Dwairi (2021)
Cr(III)	Weathered basalt andesite products (WBAP)	Wastewater of the electroplating industry	2.36	22	220	1	93.86	12	30 g/L	Langmuir and Freundlich	Shah et al. (2009)
Cr(III)	Bentonite	Aqueous metal ion solution	1.65–2.37	23 ± 0.2	100	1	–	3.75	25	Langmuir	Flieger et al. (2020)
Cr(III)	Glauconite	Aqueous metal ion solution	1.65–2.37	23 ± 0.2	100	1	–	1.05	25	Langmuir	Flieger et al. (2020)
Cr(III)	Perlite	Aqueous metal ion solution	1.65–2.37	23 ± 0.2	100	1	–	0.2	25	Langmuir	Flieger et al. (2020)

Table 1 (continued)

Metal	Sorbent	Type of water	pH	Temperature (°C)	Initial concentration (mg/l)	Contact time (h)	Removal efficiency [%]	Uptake sorbent capacity mg/g	Dosage (g)	Isotherm model	References
Pb(II)	Magnetic natural composite Fe3O4-chitosan@bentonite	AMD	2	25	0,883	–	> 89	–	0.05	Langmuir and Freundlich	Feng et al. (2019)
Pb(II)	Zeolite	Aqueous metal ion solution	5	25	362.2	24	–	45.14	0.4	Langmuir	Pfeifer et al. (2021)
Pb(II)	Bentonite	Aqueous metal ion solution	8.5	25	362.2	24	–	44.46	0.4	Langmuir	Pfeifer et al. (2021)
Hg(II)	Zeolite	Sewage industrial effluents from copper smelter and refinery	6.28	21	0.0080 mg/kg	48	–	0.0083	0.35 g/L	Langmuir and Freundlich	Chojnacki et al. (2004)
Hg(II)	Zeolite	Sewage industrial effluents from copper smelter and refinery	6.28	21	0.0062 mg/kg	48	–	0.0067	0.70 g/L	Langmuir and Freundlich	Chojnacki et al. (2004)
Hg(II)	Zeolite	Sewage industrial effluents from copper smelter and refinery	6.28	21	0.0028 mg/kg	48	–	0.0039	2.10 g/L	Langmuir and Freundlich	Chojnacki et al. (2004)
Hg(II)	Zeolite	Mixed effluents: industrial and domestic	8.48	21	0.0174 mg/kg	48	–	0.021	0.35 g/L	Langmuir and Freundlich	Chojnacki et al. (2004)
Hg(II)	Zeolite	Mixed effluents: industrial and domestic	8.48	21	0.0143 mg/kg	48	–	0.015	0.70 g/L	Langmuir and Freundlich	Chojnacki et al. (2004)
Hg(II)	Zeolite	Mixed effluents: industrial and domestic	8.48	21	0.0117 mg/kg	48	–	0.0061	2.10 g/L	Langmuir and Freundlich	Chojnacki et al. (2004)
Hg(II)	Granular Bentonite	Aqueous metal ion solution	6	Ambient	100	36	15	1.7	10 g/L	Langmuir and Freundlich	Fernández-Nava et al. (2011)
Hg(II)	Bentonite	Aqueous metal ion solution	4	25	40	0.67	–	8.14	1	Langmuir and Freundlich	Guerra et al. (2011)

Table 1 (continued)

Metal	Sorbent	Type of water	pH	Temperature (°C)	Initial concentration (mg/l)	Contact time (h)	Removal efficiency [%]	Uptake sorbent capacity mg/g	Dosage (g)	Isotherm model	References
Hg(II)	Ca–bentonite	Simulated wastewater	–	Room temperature	0.5	6	94	–	1	Freundlich	Đozić et al. (2022)
Hg(II)	Ca–bentonite	Simulated wastewater	–	Room temperature	1	6	94	–	1	Freundlich	
Hg(II)	Ca–bentonite	Simulated wastewater	–	Room temperature	1.5	6	94	–	1	Freundlich	Đozić et al. (2022)
Hg(II)	Ca–bentonite	Simulated wastewater	–	Room temperature	2	6	94	–	1	Freundlich	Đozić et al. (2022)

Conditions for Sorption Processes by Waste Sorbents

Advantages and Disadvantages of Using Sorbents

Effective removal of PTEs has been reported using many different sorbents. The porosity of the adsorbent, its surface exchange capacity, and the presence of several functional groups play a major role in sorption. The adsorbent dose, pH, temperature, the presence of other cations in the water or the initial concentration and the type of metal to be sorbed also affect the adsorption efficiency (Bilal et al. 2021). The advantages and disadvantages of using different types of sorbents are summarized in Table 4.

Nanomaterials (NMs) represent another group of sorbents that could be used in the adsorption of potentially toxic elements from mine drainage or waste water. These materials have at least one dimension of 100 nm or less. Their very high adsorption efficiency is primarily linked to large specific surface area and surface multifunctionality to react chemically and bind specific target ions easily (Kegl et al. 2020). Their other advantageous characteristics are extreme reactiveness, good mechanical stability, powerful solution mobility, dispersibility, hydrophilicity, and hydrophobicity (Saleem and Zaidi 2020). These materials can be divided into several groups: (a) carbon-based nanomaterials like graphene (Ahmad et al. 2020); (b) zeolite nanoparticles (Tahoon et al. 2020); (c) polymer-based (e.g. cellulose- or chitosan-based) nanomaterials (Sayyed et al. 2021); (d) magnetic nanomaterials like iron oxide (Leonel et al. 2021); (e) metal oxides and metal-based nanomaterials (Bashir et al. 2023); (f) silica-based nanomaterials (Ethaib et al. 2022). There is a need for further research, not only in batch experiments, but also in industrial-scaled experiments.

From the point of view of costs, it is difficult to compare individual types of sorbents, as this review article focuses primarily on the sorption of metals using natural and waste sorbents, which are characterized by low costs and their affordability compared to commercial sorbents, which are not addressed in this review. Of course, various chemical modifications and preliminary treatments of sorbents are considered. Instead, we have focused on the efficiency of the sorption process. The repeated use of a sorbent is important. Natural non-biological sorbents can be reused after desorption, while natural and waste biosorbents cannot be used for a long time due to their own decomposition. In addition, the risk of using waste non-biological sorbents is risky because of the potential release of other metals and other potentially toxic compounds that would have to be subsequently removed from the aqueous solution. Natural and waste biosorbents can be highly efficient

Table 2 Sorption of PTEs by natural and waste biosorbents

Metal	Sorbent	Type of water	pH	Temperature(°C)	Initial concentration(mg/l)	Contact time(h)	Removal efficiency [%]	Uptake capacity(mg g)$^{-1}$	Dosage(g)	Isotherm model	References
Cu(II)	Grape stem	Aqueous solutions	5.5–6.0	–	–	1	80	10.1	–	Langmuir isotherms	Villaescusa et al. (2004)
Cu(II)	Riasa–Undaria pinnatifida	Aqueous solution	4	25	–	2	15	38.82	1	Langmuir, Freundlich and Temkine equilibrium isotherms	Chen et al. (2008)
Cu(II)	Barley straw	Aqueous solutions	6	25 ± 1	–	4	69	4.64	1	Langmuir isotherm	Pehlivan et al. (2009b)
Cu(II)	Orange peel—OP	Aqueous solutions	6	30	50	3	68.6	–	2 g/L	Langmuir isotherm	Feng et al. (2009)
Cu(II)	Orange peel—OPAA	Aqueous solutions	6	30	50	3	94.6	–	2 g/L	Langmuir isotherm	Feng et al. (2009)
Cu(II)	Rice bran	Aqueous solutions	5	25 ± 1	–	–	60	–	0.2	Freundlich isotherm	Montanher et al. (2005)
Cu(II)	Biomass Ash(BA)	AMD	2.12–7.09**	–	10.704	1200	99.69	–	–	–	Mohammed et al. (2017)
Cu(II)	Chicken eggshells	AMD	4.31–5.59	25 ± 2	–	–	5.27–55.69	387.51	–	Thomas model	Zhang et al. (2017)
Cu(II)	Sawdust	Aqueous solution	6	25	–	3	60	–	0.5	Freundlich and Langmuir isotherm	Ajmal et al. (1998)
Cu(II)	Banana peel	Aqueous solutions	7.0	25	0.1	72	87.3	1.15	–	–	Nathan et al. (2021)
Cu(II)	Orange peel	Aqueous solutions	7.0	25	0.1	72	79.3	2.09	–	–	Nathan et al. (2021)
Cu(II)	Potato peel	Aqueous solutions	7.0	25	0.1	72	81.1	1.07	–	–	Nathan et al. (2021)
Cu(II)	Mango bark	Aqueous solutions	6	30	200	24	22.24	–	2	–	Krishnani et al. (2021)
Cu(II)	Guava bark	Aqueous solutions	6	30	200	24	30.36	–	2	–	Krishnani et al. (2021)
Fe	Banana Peels	Groundwater	–	–	0.920	0.5	82–90	–	8	–	Baharudin et al. (2018)
Fe(III)	Biomass Ash(BA)	AMD	2.12–7.09**	–	822.029	1200	99.79	–	–	–	Mohammed et al. (2017)

Table 2 (continued)

Metal	Sorbent	Type of water	pH	Temperature(°C)	Initial concentration(mg/l)	Contact time(h)	Removal efficiency [%]	Uptake capacity(mg g)$^{-1}$	Dosage(g)	Isotherm model	References
Fe(II)	Shrimp shell(SS) and mussel byssus	AMD	3.04	25 ± 2	83.24	24	70–90	–	11.46 g L–1 SS and 71.6 g L–1 MB	–	Núñez-Gómez et al. (2020)
Fe(II)	Waste orange and lemon peel activated carbon(WOLAC) Orange:Lemon 1:1	AMD	7*	25	–	3	80	–	0.2	Langmuir, Freundlich, Temkin, and Dubinin–Radushkevich isotherm models	Ramutshatsha-Makhwedzha et al. (2023)
Fe(II)	Waste orange and lemon peel activated carbon(WOLAC) Orange:Lemon 1:4	AMD	7*	25	–	3	70	–	0.2	Langmuir, Freundlich, Temkin, and Dubinin–Radushkevich isotherm models	Ramutshatsha-Makhwedzha et al. (2023)
Fe(II)	Waste orange and lemon peel activated carbon(WOLAC) Orange:Lemon 4:1	AMD	7*	25	–	3	53	–	0.2	Langmuir, Freundlich, Temkin, and Dubinin–Radushkevich isotherm models	Ramutshatsha-Makhwedzha et al. (2023)
Pb(II)	Mango peels	Aqueous solutions	5	25 ± 2	50	1	89.96	17.47	2.5	Langmuir adsorption isotherm	Iqbal et al. (2009)
Pb(II)	Rice husk	Aqueous solutions	5	30	10	1	98.82	–	5 g/L	Dubinin–Radushkevich isotherm	Naiya et al. (2009)

Table 2 (continued)

Metal	Sorbent	Type of water	pH	Temperature(°C)	Initial concentration(mg/l)	Contact time(h)	Removal efficiency [%]	Uptake capacity(mg g)$^{-1}$	Dosage(g)	Isotherm model	References
Pb(II)	Rice husk	Aqueous solutions	5	40	10	1	99.45	–	5 g/L	Dubinin–Radushkevich isotherm	Naiya et al. (2009)
Pb(II)	Rice husk	Aqueous solutions	5	50	10	1	99.76	–	5 g/L	Dubinin–Radushkevich isotherm	Naiya et al. (2009)
Pb(II)	Rice husk	Aqueous solutions	5	30	10	1	97.5	–	25 g/L	Dubinin–Radushkevich isotherm	Naiya et al. (2009)
Pb(II)	Rice husk	Aqueous solutions	5	40	10	1	98.4	–	25 g/L	Dubinin–Radushkevich isotherm	Naiya et al. (2009)
Pb(II)	Rice husk	Aqueous solutions	5	50	10	1	98.86	–	25 g/L	Dubinin–Radushkevich isotherm	Naiya et al. (2009)
Pb(II)	Rice husk	Aqueous solutions	5	30	10	1	97.5	–	50 g/L	Dubinin–Radushkevich isotherm	Naiya et al. (2009)
Pb(II)	Rice husk	Aqueous solutions	5	40	10	1	98.4	–	50 g/L	Dubinin–Radushkevich isotherm	Naiya et al. (2009)
Pb(II)	Rice husk	Aqueous solutions	5	50	10	1	98.86	–	50 g/L	Dubinin–Radushkevich isotherm	Naiya et al. (2009)
Pb(II)	Barley straw	Aqueous solutions	6	25 ± 1	4	88	23.2	–	1	Langmuir isotherm	Pehlivan et al. (2009b)
Pb(II)	Rice bran	Aqueous solutions	–	25 ± 1	–	–	~100	–	0.2	Freundlich isotherm	Montanher et al. (2005)
Pb(II)	Almond shell	Aqueous solutions	6	25 ± 1	207.2	4	68	8.08	0.5	Langmuir, and Freundlich isotherm	Pehlivan et al. (2009a)

Table 2 (continued)

Metal	Sorbent	Type of water	pH	Temperature(°C)	Initial concentration(mg/l)	Contact time(h)	Removal efficiency [%]	Uptake capacity(mg g)$^{-1}$	Dosage(g)	Isotherm model	References
Pb(II)	Hazelnut shell	Aqueous solutions	6	25 ± 1	207.2	4	90	28.18	0.5	Langmuir, and Freundlich isotherm	Pehlivan et al. (2009a)
Pb(II)	Biomass Ash(BA)	AMD	2.12–7.09**	–	1.877	1200	99.42	–	–	–	Mohammed et al. (2017)
Pb(II)	Eggshells	AMD	4.31–5.59	25 ± 2	–	–	30.42–77.09	146.44	–	Thomas model	Zhang et al. (2017)
Pb(II)	Waste orange and lemon peel activated carbon(WOLAC) Orange:Lemon 1:1	AMD	7*	25	–	3	90	–	0.2	Langmuir, Freundlich, Temkin, and Dubinin–Radushkevich isotherm models	Ramutshatsha-Makhwedzha et al. (2023)
Pb(II)	Waste orange and lemon peel activated carbon(WOLAC) Orange:Lemon 4:1	AMD	7*	25	–	3	52	–	0.2	Langmuir, Freundlich, Temkin, and Dubinin–Radushkevich isotherm models	Ramutshatsha-Makhwedzha et al. (2023)
Pb(II)	Waste orange and lemon peel activated carbon(WOLAC) Orange:Lemon 1:4	AMD	7*	25	–	3	42	–	0.2	Langmuir, Freundlich, Temkin, and Dubinin–Radushkevich isotherm models	Ramutshatsha-Makhwedzha et al. (2023)
Pb(II)	Walnut shell	Aqueous solutions	6	–	20	4	100	3.59	7	Langmuir, Freundlich, Dubinin–Radushkevich	Das et al. (2020)

Table 2 (continued)

Metal	Sorbent	Type of water	pH	Temperature(°C)	Initial concentration(mg/l)	Contact time(h)	Removal efficiency [%]	Uptake capacity(mg g)$^{-1}$	Dosage(g)	Isotherm model	References
Pb(II)	Almond shell	Aqueous solutions	6	–	20	4	100	3.58	7	Langmuir, Freundlich, Dubinin–Radushkevich	Das et al. (2020)
Pb(II)	Rice husk ash	Aqueous solutions	6	30	50	1	92.28	–	2.5	Temkin, Dubinin–Radushkevich	Priya et al. (2022)
Pb(II)	Mango bark	Aqueous solutions	6	30	200	24	60.85	–	2	–	Krishnani et al. (2021)
Pb(II)	Guava bark	Aqueous solutions	6	30	200	24	70.25	–	2	–	Krishnani et al. (2021)
Zn(II)	Hazlenut shell	Aqueous solutions	2.5–3.5	20 ± 1	1.308	5	80	–	4 g.l^{-1}	Langmuir and Freundlich isotherms	Cimino et al. (2000)
Zn(II)	Rice bran	Aqueous solutions	5	25 ± 1	–	–	70	–	0.2	Freundlich isotherm	Montanher et al. (2005)
Zn(II)	Biomass Ash(BA)	AMD	2.12–7.09**	–	8.328	1200	94.80	–	–	–	Mohammed et al. (2017)
Zn(II)	Pomegranate peel	Aqueous solutions	–	20	40	2	–	37.17	0.25	–	Turkmen Koc et al. (2021)
Zn(II)	Orange peel	Aqueous solutions	–	20	40	2	–	37.64	0.25	–	Turkmen Koc et al. (2021)
Zn(II)	Pineapple peel	Aqueous solutions	–	20	40	4	–	36.99	0.25	–	Turkmen Koc et al. (2021)
Zn(II)	Rice husk ash	Aqueous solutions	6	30	50	1	95.62	–	2.5	Temkin, Dubinin–Radushkevich	Priya et al. (2022)
As(V)	Agri-food by-products	AMD	5.8	25	–	2	90	–	–	Freundlich isotherm	Carrillo-González et al. (2022)
Cd(II)	Orange peel	Aqueous solutions	5.5	45	250	4	–	138.0	0.04	–	Akinhanmi et al. (2020)
Cd(II)	Banana peel	Aqueous solutions	7.0	25	0.1	72	89.3	1.02	–	–	Nathan et al. (2021)

Table 2 (continued)

Metal	Sorbent	Type of water	pH	Temperature(°C)	Initial concentration(mg/l)	Contact time(h)	Removal efficiency [%]	Uptake capacity(mg g)$^{-1}$	Dosage(g)	Isotherm model	References
Cd(II)	Orange peel	Aqueous solutions	7.0	25	0.1	72	92.4	2.11	–	–	Nathan et al. (2021)
Cd(II)	Potato peel	Aqueous solutions	7.0	25	0.1	72	89.6	1.02	–	–	Nathan et al. (2021)
Cd(II)	Mango bark	Aqueous solutions	6	30	200	24	25.86	–	2	–	Krishnani et al. (2021)
Cd(II)	Guava bark	Aqueous solutions	6	30	200	24	32.54	–	2	–	Krishnani et al. (2021)
Cd(II)	Waste tea	Wastewater	–	22	–	2	98	–	0.3	Freundlich isotherm	Orhan and Büyükgüngör (1993)
Cd(II)	Turkish coffee	Wastewater	–	22	–	2	70	–	0.3	Freundlich isotherm	Orhan and Büyükgüngör (1993)
Cd(II)	Exhausted coffee	Wastewater	–	22	–	2	89	–	0.3	Freundlich isotherm	Orhan and Büyükgüngör (1993)
Cd(II)	Nut shell	Wastewater	–	22	–	2	78	–	0.3	Freundlich isotherm	Orhan and Büyükgüngör (1993)
Cd(II)	Walnut shell	Wastewater	–	22	–	2	90	–	0.3	Freundlich isotherm	Orhan and Büyükgüngör (1993)
Cd(VI)	Hazlenut shell	aqueous solutions	2.5–3.5	20 ± 1	–	5	92.4	–	4 g.l^{-1}	Langmuir and Freundlich isotherms	Cimino et al. (2000)
Cd(II)	Mango peels	Aqueous solutions	5	25 ± 2	50	1	93.5	16.55	2.5	Langmuir adsorption isotherm	Iqbal et al. (2009)
Cd(II)	Rice bran	Aqueous solutions	5	25 ± 1	–	–	66	–	0.2	Freundlich isotherm	Montanher et al. (2005)
Cd(II)	Eggshells	AMD	4.31–5.59	25 ± 2	–	–	5.11 –18.85	1.57	–	Thomas model	Zhang et al. (2017)

Table 2 (continued)

Metal	Sorbent	Type of water	pH	Temperature(°C)	Initial concentration(mg/l)	Contact time(h)	Removal efficiency [%]	Uptake capacity(mg g)$^{-1}$	Dosage(g)	Isotherm model	References
Cd(II)	Waste orange and lemon peel activated carbon(WOLAC) Orange:Lemon 1:1	AMD	7*	25	30 mg/L	3	80	–	0.2	Langmuir, Freundlich, Temkin, and Dubinin–Radushkevich isotherm models	Ramutshatsha-Makhwedzha et al. (2023)
Cd(II)	Waste orange and lemon peel activated carbon(WOLAC) Orange:Lemon 1:4	AMD	7*	25	30 mg/L	3	78	–	0.2	Langmuir, Freundlich, Temkin, and Dubinin–Radushkevich isotherm models	Ramutshatsha-Makhwedzha et al. (2023)
Cd(II)	Waste orange and lemon peel activated carbon(WOLAC) Orange:Lemon 4:1	AMD	7*	25	30 mg/L	3	60	–	0.2	Langmuir, Freundlich, Temkin, and Dubinin–Radushkevich isotherm models	Ramutshatsha-Makhwedzha et al. (2023)
Cd(II)	Biomass ash(BA)	AMD	2.12–7.09**	–	–	1200	99.87	–	–	–	Mohammed et al. (2017)
Cd(II)	Borago officinalis	Wastewater from organized industrial zone	2–7*	25		2	89.54	40.26	0.35	Langmuir isotherm model	Tunali Akar et al. (2016)
Ni(II)	Activated sludge	Synthetic wastewater	7 ± 0.1*	25 ± 0.5	10.6	5	39.7	177.76	–	Langmuir and Freundlich isotherms	Arican et al. (2002)
Ni(II)	Grape stem	Aqueous solutions	5.5–6.0	–	–	1	80	7.677	–	Langmuir isotherms	Villaescusa et al. (2004)

Table 2 (continued)

Metal	Sorbent	Type of water	pH	Temperature(°C)	Initial concentration(mg/l)	Contact time(h)	Removal efficiency [%]	Uptake capacity(mg g)$^{-1}$	Dosage(g)	Isotherm model	References
Ni(II)	Riasa–*Undaria pinnatifida*	Aqueous solution	4	25	–	2	82	24.71	–	Langmuir isotherms	Chen et al. (2008)
Ni(II)	Biomass Ash(BA)	AMD	2.12–7.09**	–	0.861	1200	98.51	–	–	–	Mohammed et al. (2017)
Ni(II)	Banana peel	Aqueous solutions	7.0	25	0.1	72	78.9	0.9	–	–	Nathan et al. (2021)
Ni(II)	Orange peel	Aqueous solutions	7.0	25	0.1	72	80.3	1.84	–	–	Nathan et al. (2021)
Ni(II)	Potato peel	Aqueous solutions	7.0	25	0.1	72	70.9	0.81	–	–	Nathan et al. (2021)
Ni(II)	Walnut shells	Aqueous solutions	6.5	20	200	12	61.8	–	–	–	Cruz-Lopes et al. (2022)
Ni(II)	Chesnut shells	Aqueous solutions	6.5	20	200	12	59.75	–	–	–	Cruz-Lopes et al. (2022)
Ni(II)	Pinewood	Aqueous solutions	6.5	20	200	12	27.2	–	–	–	Cruz-Lopes et al. (2022)
Cr(VI)	WPOOF—waste pomace of olive oil factory	Aqueous solution	2	25 ± 1	2.828g$K_2Cr_2O_7$/l	3	98.6	6.57	15 g/L	Langmuir and Freundlich isotherms	Malkoc et al. (2006)
Cr(VI)	WPOOF—waste pomace of olive oil factory	Aqueous solution	–	25	50	3	61	6.1	–	Langmuir and Freundlich isotherms	Malkoc et al. (2006)
Cr(VI)	WPOOF—waste pomace of olive oil factory	Aqueous solution	–	45	50	3	81.74	8.17	–	Langmuir and Freundlich isotherms	Malkoc et al. (2006)
Cr(VI)	WPOOF—waste pomace of olive oil factory	Aqueous solution	–	60	50	3	100	10	–	Langmuir and Freundlich isotherms	Malkoc et al. (2006)
Cr(VI)	Walnut shells	Aqueous solutions	6.5	20	200	12	52.25	–	–	–	Cruz-Lopes et al. (2022)
Cr(VI)	Chesnut shells	Aqueous solutions	6.5	20	200	12	59.85	–	–	–	Cruz-Lopes et al. (2022)
Cr(VI)	Pinewood	Aqueous solutions	6.5	20	200	12	41.8	–	–	–	Cruz-Lopes et al. (2022)

Table 2 (continued)

Metal	Sorbent	Type of water	pH	Temperature(°C)	Initial concentration(mg/l)	Contact time(h)	Removal efficiency [%]	Uptake capacity(mg g)$^{-1}$	Dosage(g)	Isotherm model	References
Cr(VI)	Rice husk ash	Aqueous solutions	6	30	50	1	87.12	–	2.5	Temkin, Dubinin–Radushkevich	Priya et al. (2022)
Cr(VI)	Waste tea	Wastewater	–	22	–	2	93	–	0.3	Freundlich isotherm	Orhan and Büyükgüngör (1993)
Cr(VI)	Turkish coffee	Wastewater	–	22	–	2	98	–	0.3	Freundlich isotherm	Orhan and Büyükgüngör (1993)
Cr(VI)	Exhausted coffee	Wastewater	–	22	–	2	85	–	0.3	Freundlich isotherm	Orhan and Büyükgüngör (1993)
Cr(VI)	Nut shell	Wastewater	–	22	–	2	88	–	0.3	Freundlich isotherm	Orhan and Büyükgüngör (1993)
Cr(VI)	Walnut shell	Wastewater	–	22	–	2	80	–	0.3	Freundlich isotherm	Orhan and Büyükgüngör (1993)
Cr(VI)	Hazlenut shell	Aqueous solutions	2.5–3.5	20 ± 1	–	5	94.6	–	4 g.l^{-1}	Langmuir and Freundlich isotherms	Cimino et al. (2000)
Mn(II)	Banana peels activated carbon	Groundwater	–	–	0.376	0.5	98.79–99.43	–	8	–	Baharudin et al. (2018)
Mn(II)	Borago officinalis	Wastewater from organized industrial zone	2–7*	25	0.15	2	72.99 ± 0.18	20.42	0.3	Langmuir isotherm model	Tunali Akar et al. (2016)
Mn(II)	Saccharomyces cerevisiae	groundwater	–	28	–	1.5	84.23	2.19	–	–	Fadel et al. (2017)
Mn(II)	Biomass ash(BA)	AMD	2.12–7.09**	–	199.673	1200	99.34	–	–	–	Mohammed et al. (2017)

Table 2 (continued)

Metal	Sorbent	Type of water	pH	Temperature(°C)	Initial concentration(mg/l)	Contact time(h)	Removal efficiency [%]	Uptake capacity(mg g)$^{-1}$	Dosage(g)	Isotherm model	References
Hg(I)	Sugarcane Bagasse	Aqueous solution	1.8	30	76	3	92.89	–	5 g/L	Langmuir and Freundlich isotherms	Khoramzadeh et al. (2013)
Hg(II)	Sugarcane Bagasse	Aqueous solution	4	30	76	3	97.58	–	5 g/L	Langmuir and Freundlich isotherms	Khoramzadeh et al. (2013)
Hg(II)	Banana peels	Aqueous solution	6	22 °C ± 1	50 µg/L	72	73	–	0.15	Langmuir and Freundlich isotherms	Fabre et al. (2020)
Hg(II)	Banana peels	Aqueous solution	6	22 ± 1	50 µg/L	72	91	–	0.50	Langmuir and Freundlich isotherms	Fabre et al. (2020)
Hg(II)	Nutshells	Contaminated solutions	6.5–7.0	22	50 µg/L	48	> 90%	–	0.5	–	Dias et al. (2021)
Hg(II)	*A. flocculosa*	Aqueous solution	6	Room temperature	1000 ppm	24	–	456	–	Langmuir and Freundlich isotherms	Cain et al. (2008)
Hg(II)	*S. platensis*	Aqueous solution	6	Room temperature	1000 ppm	24	–	428	–	Langmuir and Freundlich isotherms	Cain et al. (2008)

Table 3 Sorption of PTEs by waste non-biological sorbents

Metal	Sorbent	Type of water	pH	Temperature(°C)	Initial concentration(mg/l)	Contact time(h)	Removal efficiency(%)	Uptake capacity(mg g)$^{-1}$	Dosage(g)	Isotherm model	References
Cu(II)	Fly ash	AMD	2.12–7.09*	–	10.704	50 days	99.69	–	–	–	Mohammed et al. (2017)
Cu(II)	Ladle Furnace Slag(LFS)	AMD	2.12–7.09 *	–	10.704	50 days	99.69	–	–	–	Mohammed et al. (2017)
Cu(II)	Modified fly ash	Aqueous solutions	4	room	500	24	96	42.89	10	BET	Buema et al. (2021)
Cu(II)	Modified fly ash	Aqueous solutions	5	room	500	24	98	43.85	10	BET	Buema et al. (2021)
Cu(II)	Coal fly ash	Aqueous solutions	5.0–6.5	–	500	2	97.1	11.65	3	Modified Langmuir	Lekgoba et al. (2021)
Cu(II)	Activated steel slag	Aqueous solutions	5	20	100	0.5	63.93	–	0.4	Langmuir	Nikolić et al. (2020)
Cu(II)	Steel slag	Aqueous solutions	7	–	–	24	99.96	4.98	8	Langmuir	Pfeifer et al. (2021)
Fe(II)	Fly ash	AMD	2.12–7.09*	–	822.029	50 days	99.97	–	–	–	Mohammed et al. (2017)
Fe(II)	Ladle Furnace Slag(LFS)	AMD	2.12–7.09*	–	822.029	50 days	99.78	–	–	–	Mohammed et al. (2017)
Fe(II)	Lignite	AMD	2	22	3.5	48	99.99	18.7	6 g/L	–	Mohan and Chander (2006)
Fe(II)	Steel slag	Aqueous solutions	7	–	–	24	99.98	48.08	8	Langmuir	Pfeifer et al. (2021)
Pb(II)	Fly ash	AMD	2.12–7.09**	–	1.877	50 days	99.65	–	–	–	Mohammed et al. (2017)
Pb(II)	Ladle Furnace Slag(LFS)	AMD	2.12–7.09*	–	1.877	50 days	99.95	–	–	–	Mohammed et al. (2017)
Pb(II)	Modified fly ash	Aqueous solutions	6	25	100	2	–	49.9	2	Langmuir, Freundlich	Huang et al. (2020)
Pb(II)	Modified coal fly ash	Aqueous solutions	–	27 ± 2	100	4	–	31.38	1	Modified Langmuir	Astuti et al. (2021)
Pb(II)	Steel slag	Aqueous solutions	7	–	–	24	85.92	59.81	8	Langmuir	Pfeifer et al. (2021)
Zn(II)	Fly ash	AMD	2.12–7.09*	–	8.328	50 days	98.05	–	–	–	Mohammed et al. (2017)
Zn(II)	Ladle Furnace Slag(LFS)	AMD	2.12–7.09*	–	8.328	50 days	97.18	–	–	–	Mohammed et al. (2017)
Ni(II)	Modified slag of electric arc furnace	Aqueous solutions	8	50	–	3	99.8	–	30	Langmuir, Freundlich, hybrid	Changalvaei et al. (2021)

Table 3 (continued)

Metal	Sorbent	Type of water	pH	Temperature(°C)	Initial concentration(mg/l)	Contact time(h)	Removal efficiency(%)	Uptake capacity(mg g)$^{-1}$	Dosage(g)	Isotherm model	References
Zn(II)	Modified coal fly ash	Aqueous solutions	–	27 ± 2	100	4	–	26.98	1	Modified Langmuir	Astuti et al. (2021)
As(V)	Fly ash	AMD	2.12–7.09*	–	0.024	50 days	78.20	–	–	–	Mohammed et al. (2017)
As(V)	Ladle Furnace Slag(LFS)	AMD	2.12–7.09*	–	0.024	50 days	88.80	–	–	–	Mohammed et al. (2017)
As(V)	Biomass Ash(BA)	AMD	2.12–7.09*	–	0.024	50 days	86.40	–	–	–	Mohammed et al. (2017)
As(V)	Steel slag	Aqueous solutions	2	–	10	24	95	–	0.1	Langmuir	Liem–Nguyen et al. (2020)
Cd(II)	Fly ash	AMD	2.12–7.09*	–	–	50 days	98.78	–	–	–	Mohammed et al. (2017)
Cd(II)	Ladle Furnace Slag(LFS)	AMD	2.12–7.09*	–	–	50 days	98.81	–	–	–	Mohammed et al. (2017)
Cd(II)	Modified coal fly ash	Aqueous solutions	4	–	–	–	90.02	90.02	–	Langmuir, Freundlich, Temkin	Zhao et al. (2021)
Cd(II)	Modified fly ash	Aqueous solutions	6	25	100	2	–	43.44	2	Langmuir, Freundlich	Huang et al. (2020)
Cd(II)	Fly ash wood pellet-based thermal power plant	Aqueous solutions	–	–	100	24	41	62.3	0.4	Langmuir, Freundlich	Park et al. (2020)
Cd(II)	Fly ash wood pellet-based thermal power plant	Aqueous solutions	–	–	100	24	100	32.4	2	Langmuir, Freundlich	Park et al. (2020)
Cd(II)	Steel slag	Aqueous solutions	5.5	20–80	20–100	15 s–5 min	–	123.45	0.5	Langmuir, Freundlich, Temkin	Dizaj Khalili and Ghaemi (2021)
Ni(II)	Fly ash	AMD	2.12–7.09*	–	0.861	50 days	99.84	–	–	–	Mohammed et al. (2017)
Ni(II)	Ladle Furnace Slag(LFS)	AMD	2.12–7.09*	–	0.861	50 days	99.71	–	–	–	Mohammed et al. (2017)
Ni(II)	Coal fly ash	Aqueous solutions	7.0–8.0	–	500	2	65.1	5.0	5	Modified Langmuir	Lekgoba et al. (2021)
Ni(II)	Modified slag of electric arc furnace	Aqueous solutions	8	50	–	3	96.3	–	30	Langmuir, Freundlich, hybrid	Changalvaei et al. (2021)

Table 3 (continued)

Metal	Sorbent	Type of water	pH	Temperature(°C)	Initial concentration(mg/l)	Contact time(h)	Removal efficiency(%)	Uptake capacity(mg g)$^{-1}$	Dosage(g)	Isotherm model	References
Mn(II)	Fly ash	AMD	2.12–7.09*	–	199.673	50 days	99.99	–	–	–	Mohammed et al. (2017)
Mn(II)	Ladle Furnace Slag(LFS)	AMD	2.12–7.09*	–	199.673	50 days	99.89	–	–	–	Mohammed et al. (2017)
Mn(II)	Lignite	AMD	2	22	3.5	48	99.99	18.7	6 g/L	–	Mohan and Chander (2006)
Hg(II)	Coal fly ash	Industrial wastewater	2.5	Room temperature	10	24	≈ 100	–	100 g/L	Langmuir, Freundlich, Temkin	Attari et al. (2017)
Hg(II)	Coal fly ash	Industrial wastewater	2.5	Room temperature	10	24	> 90	–	50 g/L	Langmuir, Freundlich, Temkin	Attari et al. (2017)
Hg(II)	Coal fly ash	Industrial wastewater	2.5	Room temperature	10	24	> 75	–	10 g/L	Langmuir, Freundlich, Temkin	Attari et al. (2017)
Hg(II)	Silico–Aluminous Fly Ash	Demineralized water	5	30	602	24	53	3.2	5	–	Rio and Delebarre (2003)
Hg(II)	Coal fly ash	Aqueous solutions	2	25	10	–	10	–	0.75	–	Tauanov et al. (2018)
Hg(II)	Synthetic zeolites from coal fly ash	Aqueous solutions	2	25	10	–	89	–	0.75	–	Tauanov et al. (2018)
Hg(II)	Ag-Synthetic zeolites from coal fly ash	Aqueous solutions	2	25	10	–	99	–	0.75	–	Tauanov et al. (2018)
Hg(II)	Carbide Slag + Fly Ash	Wastewater	11	30	–	40 min	97.1	–	2	Freundlich	Li and Sun (2014)

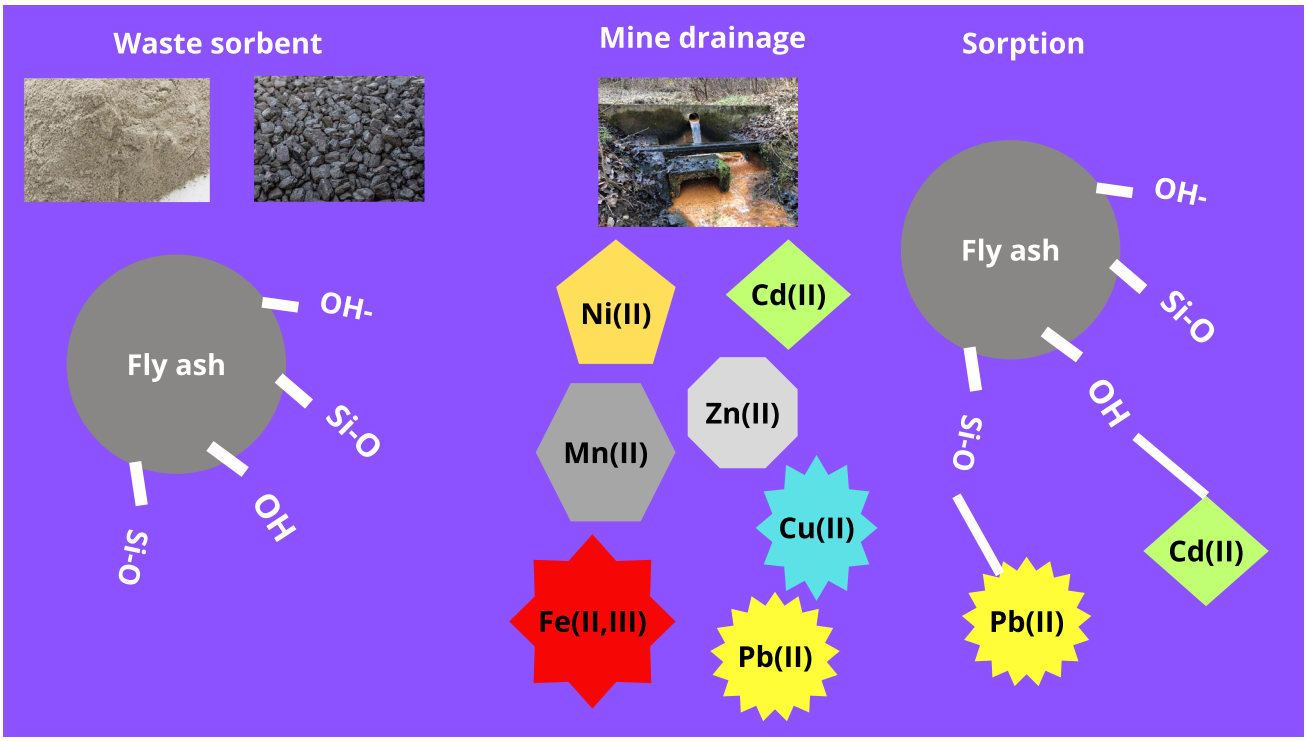

Fig. 3 Graphical scheme of metal sorption with the use of waste biosorbents

but, as can be seen from Table 4, the efficiency of these sorbents is not directly proportional to the dose of sorbent, which can cause certain complications when searching for the optimal dose of sorbent.

Evaluation

The Influence of pH

In previous studies, Cu sorption was evaluated at pH 2–5. The lowest sorption level was in an acidic solution (pH 2) using zeolite (Ryu et al. 2019). By increasing the pH from 2 to 3, the sorption of copper increased to 98.16% (Wulandari et al. 2020). In general, it appears that the sorption of copper is more favourable at moderately low pH values, typically in the range from pH 4 to 6 (Balintova et al. 2012, 2014; Gupta et al. 2018). This applies especially when using waste biosorbents (Feng et al. 2009; Mohammed et al. 2017).

Sorption of Fe depends on the oxidation state of the Fe ions. The sorption of Fe(II) ions tends to be optimal at higher pH values, typically from neutral to slightly alkaline (pH 6 to 8). At these pH levels, the surface charge of most sorbent materials becomes more negatively charged, which facilitates the attraction between the positively charged Fe(II) ions and the sorbent surface. This was confirmed by several studies when Fe(II) was sorbed on waste biosorbents and non-biological sorbents (Mohammed et al. 2017; Núñez-Gómez et al. 2020; Ramutshatsha-Makhwedzha et al. 2023). The sorption of Fe(III) ions is usually optimal at lower pH values. Adsorption effectiveness above 90% was observed for pH values below 5 (Masindi et al. 2015, 2017). However, adsorption of Fe(III) ions depends on several factors, like the sorbent material (materials with negatively charged surfaces, such as certain types of clays or metal oxides, may exhibit higher adsorption capacities for Fe(III) ions at lower pH values) (Kéri et al. 2020), competing ions (the presence of high concentrations of carbonate or phosphate ions may decrease Fe(III) adsorption at certain pH ranges due to the formation of soluble complexes; Senn et al. 2015), redox potential (at higher pH values, the reduction of Fe(III) to Fe(II) may occur more readily, affecting the speciation and adsorption behaviour of the Fe ions).

The oxidation state of Cr plays a key role in its adsorption. The sorption of trivalent chromium ions (Cr(III)) tends to be optimal from pH 4 to 7 (Arim et al. 2018; Batool et al. 2019). The sorption of hexavalent chromium ions (Cr(VI)) is more complex and can depend on the specific sorbent material and solution conditions. It is known that Cr(VI) can exist in various forms in solution, such as chromate (CrO_4^{2-}) or dichromate ($Cr_2O_7^{2-}$), and its sorptive behaviour can be influenced by factors such as pH, redox potential, and the presence of competing ions (Fenti

Table 4 Comparison of advantages and disadvantages of using selected types

Benefits	References	Disadvantages	References
Natural non-biosorbents			
Silica, zeolite, kaolite, bentonite, montmorillonite, clays			
Low cost, availability, and direct use with minimal machining	Erans et al. (2016), Zaimee et al. (2021)	Decrease in reactivity during capture cycles and abrasion	Erans et al. (2016)
They can be modified to achieve a higher adsorption capacity	Tripathi and Rawat Ranjan (2015), Otunola and Ololade (2020), Chakraborty et al. (2022)	Lower efficiency compared to activated carbon	Irannajad and Kamran Haghighi (2021)
Clay minerals have space between the layers, which allows adsorption of PTEs from water, most clays can increase the space between the layers	[CSL STYLE ERROR: reference with no printed form.])	Small pore size	Ramontja (2016)
Natural biosorbents and waste biosorbents			
bacteria, fungi, algae, agricultural and food industry wastes(eggshells, nut shells, fruit peels)			
The cell surfaces of all microorganisms are negatively charged due to the presence of various anionic structures, which allows metal cations to bind	Gavrilescu (2004) and Zaimee et al. (2021)	Biorotation is very fast, which results in faster sorbent consumption and more frequent sorbent replacement/regeneration	Fu and Wang (2011, 2014), Qin et al. (2020)
Economically advantageous	Gupta et al. (2009) and Sabir et al. (2021)	The rapid decomposition of biomass prevents its long-term use in the process	Sahmoune et al. (2011)
Live and dead biomass can be used	Bilal et al. (2013)	One sorbent is not applicable for all metals	Wang and Chen (2009)
Biosorbents are selective for individual metals and regenerable	Fu and Wang (2011)	Sorption is limited by the number and availability of functional groups on the sorbent surface	Beni and Esmaeili (2020)
Sorption properties can be improved by modification of the sorbent or by combination with a chemical material	Qin et al. (2020)	Sorption efficiency is not directly proportional to the sorbent dose, above the optimum dose agglomeration of the bio-agglomerate occurs which reduces the efficiency	Beni and Esmaeili (2020)
Use of waste biosorbents solves the waste management problem	Kanamarlapudi et al. (2018)	Selectivity is low, can be improved by chemical modification	Gupta et al. (2015)
With appropriate selection, sorbed metals can be recovered	Abdolali et al. (2017)	Biosorption is an endothermic reaction, the efficiency of which can be increased by raising the temperature to e.g. 45 °C	Beni and Esmaeili (2020)
High efficiency even at low initial concentration	Gupta et al. (2015)	Microbial-based sorbents are more effective compared to cellulosic or fungal sorbents	Gupta et al. (2015)
Waste non-biosorbents			
Fly ash, red mud, waste slurry			
Suitable for removal of PTEs, dyes	Daneshvar et al. (2017),Morin-Crini et al. (2019)	Sorbents require some modification to increase their adsorption capacity	Ahmaruzzaman (2011)
Low cost, locally available in large quantities	Crini et al. (2019)	Adsorption properties depend on the specific material	Crini et al. (2019)

Table 4 (continued)

Benefits	References	Disadvantages	References
Supports the circular economy	Ahmaruzzaman (2011), Soliman and Moustafa (2020)	Other PTEsand toxic substances trapped in the structure may be released into the water during adsorption	Attari et al. (2017)
The recovery of industrial waste reduces the environmental and economic impact of its disposal	Hossain et al. (2020)	Disposal of spent sorbent not resolved	Anastopoulos et al. (2017) and Hossain et al. (2020)
Nanomaterials			
Effective removal of very low metal concentration(as low as 0.002 mg/L) and high regeneration capability	Saleem and Zaidi (2020), Kolluru et al. (2021)	Nanomaterials may interact with different types of factors in discharged water, which could result into negative effects on the surroundings	Saleem and Zaidi (2020)
Suitable for removal of diverse spectrum of contaminants(metals, dyes, pharmaceuticals etc.)	Tahoon et al. (2020, Saleem and Zaidi (2020)	Certain nanomaterials may form an absolutely new class of non-biodegradable pollutants	Saleem and Zaidi (2020)
Moderate pH is optimal due to deprotonation of sorbent surface and enhanced electrostatic attractions between the surface of adsorbent and metal cations	Wadhawan et al. (2020)	Adsorption effectivity depends on ionic strength and NaCl concentration	Wadhawan et al. (2020)
		Agglomeration tendency and tendency to swell during the adsorption process may decrease adsorption efficiency	Kolluru et al. (2021)

et al. 2020). It was confirmed that Cr(VI) sorption is more likely in acidic conditions (Cimino et al. 2000; Malkoc et al. 2006).

The sorption of other metals (Pb, Zn, Mn, As, Ni) tends to be optimal at slightly acidic to neutral pH values, from pH 5 to 7. However, the sorption depends on the conditions like the metal being sorbed, solution conditions, the presence of competitive ions, and metal speciation. In some cases, Pb sorption can also occur effectively at higher or lower pH values depending on the chemistry of the sorbent and the solution. Adsorption onto zeolite was effective (> 99%) at a pH ≈ 2.5 (Olegario-Sanchez and Pelicano 2017). On the other hand, adsorption onto waste biosorbents was more effective at pH values of 5–6 (Naiya et al. 2009; Pehlivan et al. 2009a; Ramutshatsha-Makhwedzha et al. 2023). The adsorption effectiveness of Zn onto zeolite was < 50% at a pH of 2 (Ryu et al. 2019), but more effective when using hazelnut shell (Cimino et al. 2000). Determining the optimal pH for zinc sorption often involves experimentation with the sorbent material and solution conditions. The same applies to Mn adsorption. However, manganese can exist in various oxidation states (e.g., Mn(II) and Mn(IV)), and its speciation can change with pH. For example, at lower pH values, manganese tends to be present as Mn(II), while at higher pH values, it can form hydrolysis species such as $Mn(OH)_4^{2-}$ (Ilton et al. 2016). Mn(II) can be effectively sorbed at lower pH values on natural non-biological sorbents (Masindi et al. 2017; Varvara et al. 2013). The adsorption of Cd seems to be more effective with increasing pH (Mohammed et al. 2017; Ramutshatsha-Makhwedzha et al. 2023; Tunali Akar et al. 2016). Adsorption of Ni also increased with higher pH values (Enslin et al. 2010; Gitari 2014). Another experimental study also confirmed the effect of increasing pH on better precipitation of metals (Liehr 1995).

The speciation of As in solution is crucial because arsenite (As(III)) and arsenate (As(V)) have different affinities for sorbent materials and can exhibit different optimal pH ranges for sorption. Arsenate tends to sorb more strongly under conditions favouring the formation of positively charged species, while arsenite may sorb more readily under the conditions favouring the formation of negatively charged species (Ungureanu et al. 2015). For most sorbents, a neutral or slightly acidic pH (5–7) is the most optimal (Carrillo-González et al. 2022; Mohammed et al. 2017).

According to Verma et al. (2018), for most of the elements with an oxidation state of + 2, such as Co, Ni, Zn, Ba, and Cd (but not Hg), a general trend of enhancement in removal was observed with an increased pH. However, for other elements such as V, As, Mo, Ag, and Bi, which have oxidation states other than + 2, no definite pattern was observed. Precipitation of Ba and As using this method was negligible at all five pH values (2 acidic, 2 basic, and 1 neutral). But, in general, the precipitation and recovery of elements generally depend strongly on the pH of the water sample.

The leaching behaviour of iron and calcium at different pH conditions has a profound effect on the pH-dependent mobilities of most of the trace elements present in the investigated materials. In contrast, the leaching of sodium, potassium, and chloride from slag, particularly in the pH range of 4–12, is pH independent and therefore does not affect the mobilities of potentially toxic elements. Hierro et al. (2014) showed that increasing the pH causes enhance metal removal;. as pH increases, the concentrations of dissolved ore-associated elements are attenuated.

Thus, the pH of the solution is a very important factor since it can affect both the chemistry of the adsorbate and adsorbent. The charges of the adsorbate and adsorbent rey on the pH solution. With the rise of pH from 2 to 5.5, the sorptive capacity of orange peel rose from 24.62 to 44.42%. The highest adsorption was accomplished at a pH solution of 5.5. At lower pH values, the Cd(II) ion elimination is subdued by web of positive charges of the orange peel and the rivalry that exists between Cd(II) ions and the H^+ in solution. At an elevated pH, the negative charge network on orange peel increases due to the deprotonation of the binding sites (Akinhanmi et al. 2020). Metal binding is strongly pH dependent with more metal cations bound at higher pH; the optimum initial pH was found to be 6, where the maximum uptake of metal ions took place. Metal binding by products was reduced at low pH values due to the increasing competition of protons for the same binding sites (Krishnani et al. 2021).

The Influence of Temperature

Solution temperature is also an essential parameter in the adsorption process. In general, an initial increase in temperature reduces the solution viscosity, which enhances the diffusion rate of the adsorbate molecules across the adsorbent surface, resulting in an elevated adsorption efficiency. Subsequent changes in temperature can affect the adsorption process in two different ways, depending on whether the process is exothermic or endothermic. The adsorptive capacity decreases with an increase in the temperature of an exothermic process whereas it increases in the case of an endothermic one (Wadhawan et al. 2020).

The optimal temperature for sorption can vary depending on the specific sorbent material and solution conditions. In general, room temperature (≈ 20–25 °C) is often used in laboratory experiments as it represents a practical and convenient temperature for conducting sorption studies. However, if the sorption process is exothermic (heat-releasing) in nature, lower temperatures may enhance sorption kinetics.

According to analysed studies, the influence of temperature on the efficiency of the process is not clearly confirmed.

For some waste biosorbents, higher temperatures (up to 60 °C) promoted adsorption efficiency (Malkoc et al. 2006). For example, sorption of As was 99% efficient at an elevated temperature of 23–53 °C, compared to an efficiency of 75% at a temperature of 22–27 °C (Nekhunguni et al. 2017). On the other hand, a study of sorption of Cu and Zn onto bentonite and zeolite confirmed that a reduced temperature (10 °C) positively affected the sorption efficiency and increased the adsorption capacity of each sorbent's monolayer (Prepilková et al. 2024). Temperature had an adverse effect on Pb sorption, with an elevated temperature (25 °C) reducing the process efficiency to 63%, while at 18–22 °C, it was up to 99% efficient (Olegario-Sanchez and Pelicano 2017; Oter and Akcay 2007).

It's important to note that higher temperatures can also change the solution chemistry, enhancing the solubility of certain compounds or alterations in the speciation of metal ions. Therefore, the optimal temperature for sorption should be determined experimentally for each specific system to achieve the desired sorption efficiency while considering other factors such as energy costs and practical feasibility.

The Effect of Initial Concentration

Generally, increasing the initial concentration has a positive effect at adsorption until the sorbent is fully saturated, when all the active sites on the sorbent material are occupied by metal ions, leading to a plateau in adsorption capacity. Initial concentrations vary between metals and sorbents.

The initial metal concentration during Cu sorption clearly influenced the sorption outcome. At concentrations of 50 mg/L and above, efficiency values from 89 to 99% were observed for both zeolite and bentonite (Balintova et al. 2012). However, the removal efficiencies at an initial Cu(II) concentration of 9 mg/L only ranged from 17 to 29% (Ryu et al. 2019). Similar results were also recorded for Fe sorption.

The Effect of Sorption Time

The contact time depends on the initial concentration of PTEs in the solution, the availability of active sorbent surface area (sorbents typically accept specifically sized molecules of various metals), and the sorbent dosage. Usually, adsorption occurs rapidly at first, slows down as equilibrium is approached, and eventually reaches a plateau when the adsorption sites are saturated (Soliman and Moustafa 2020).

Sorption of Pb was carried out by a zeolite (heulandite) in 6 h of contact time and 99% efficiency (Olegario-Sanchez and Pelicano 2017) and by another zeolite (clinoptilolite) with 24 h of contact time with 63.5% efficiency (Oter and Akcay 2007). This may indicate decreased efficiency with increased contact time and possibly a reverse desorption process. The contact time of Mn with zeolite was recorded at 6 h with 100% metal removal level (Motsi 2010), while contact time with bentonite was 0.5 h with an efficiency of up to 100% (Masindi et al. 2015).

The contact time differed from one adsorbent to another, and this may be due to one or more of the following: (1) changes in the chemical structure of the adsorbent, (2) the adsorbent surface area, (3) the availability of surface-active sites (4) adsorption binding constants of the adsorbent, and (5) differences in the ionic size of the metal ions (Soliman and Moustafa 2020). Understanding and optimizing the kinetics of adsorption are crucial for designing efficient and effective adsorption processes for metal removal.

The initial sorption phase is followed by a slower rate of ion removal. This is directly related to the availability of binding sites on the surface, which are rapidly occupied by the ions in the initial phase. As the reaction approaches equilibrium, sorption slows and sorption percentage decreases due to a decrease in the ion concentration in the solution. The ion with the shortest equilibrium time and the highest sorption percentage was Cd, followed by Cu, Hg, and Ni (Nathan et al. 2021).

The Advantages and Disadvantages of Using Selected Sorbents

Waste sorbents have proven to be the most economically suitable for use in the sorption process and their use promotes a circular economy. However, waste sorbents typically require some modification to increase their adsorptive capacity before they can be used, and biosorbents and natural non-biological sorbents are also characterised by low operating costs.

Studies show that natural non-biological sorbents and natural sorbents are amenable to modification. For waste sorbents, a significant negative is the possible content of other potentially toxic substances that may be released back into the aquatic ecosystem during sorption.

Biomass, as a sorbent, cannot be used in the long term due to its gradual decomposition. Although regeneration is possible in the sorption of natural non-biological sorbents by the desorption of selected metals, these are non-renewable energy sources. One of the most important factors is the efficiency of sorption. However, high sorption efficiencies were observed even at low initial concentrations.

Waste sorbents are continuously produced, unlike natural ones. Non-biological waste sorbents are also available in large quantities.

The Interdependence of the Assessed Parameters

The pH of the environment can dramatically affect adsorption. At low pH values, a low percentage of absorbed metal

is generally observed, which is attributed to the significant competitive influence of hydrogen ions and the disrupted structure of the adsorbent. A low pH value also increases the mobility and accessibility of PTEs and the content of dissolved substances. Therefore, most sorption studies are performed in the range of pH 4 to 8. The behaviour of metalloids such as arsenic differs significantly from that of PTEs, which are mobile mainly at low pH values, while arsenic is mobile at a wide range of pH values—from extremely acidic to alkaline. The change in pH mainly affects the time of sorption, and it is not directly related to other monitored parameters.

The effect of temperature is related to the thermal course of sorption. If the adsorption reaction is exothermic, the adsorption capacity of the sorbent generally decreases, while the adsorption capacity increases if the adsorption reaction is endothermic. The temperature of the process is clearly related to the type of the sorbent.

Exothermic sorption and reduced adsorptive capacity are negatives in experiments with increasing initial metal concentration, which means that a reduced temperature is suitable for increasing the adsorptive capacity of the sorbent with increased metal concentrations at the beginning of the experiments. On the other hand, it is followed by a decrease in the overall efficiency of metal removal.

The rate of the adsorption process is related to several parameters, including the initial concentration of the metal solution, the type and treatment of the sorbent, the mechanism of sorption and the temperature. Alkali-treated sorbents affect sorption by a chemical mechanism with increased temperature and increased overall metal removal efficiency, and the sorption time is shortened, so that equilibrium is achieved much faster. The adsorption rate is also related to the type of sorbed compound, its size (the metal ion itself in an acidic medium is smaller in size than the oxide hydroxide compound at a higher pH), and also from the charge of the sorbed substance. The type of sorbent determines the availability of active centers on the surface of the sorbent. Their amount is related to the concentration of the sorbed substance and thus also to the time of the ongoing adsorption process. Adsorption usually proceeds quickly at first and then slows down as equilibrium is approached. This is due to the availability of free binding sites and a decrease in the concentration of dissolved ions.

Conclusion

Natural non-biological sorbents can be physically and chemically modified to improve sorption efficiency. Waste sorbents have achieved high efficiencies over a wide pH range and are generally economically favoured, but typically need to be modified to improve their sorption capacity. Waste non-biological sorbents are typically very efficient at low temperatures. In addition, as with other sorbents, an increase in the input metal concentrations increased the rate of adsorption.

Biomass cannot be used for long time periods due to its gradual decomposition. While the sorption of natural non-biological sorbents may allow the recovery of selected metals, it must be noted that natural non-biological sorbents are non-renewable energy sources. Among the waste biosorbents, biomass ash proved to be the most effective for the sorption of PTEs.

Metal sorption is an effective method for the purification of mine effluents. However, the sorbed metals are subsequently contained in the sorbents. Therefore, it is important to direct research not only towards the purification of waste effluents, but also towards the recovery of these metals by desorption processes. This review broadens the spectrum in the field of obtaining metals from sorbents by improving the understanding of the influence of different conditions on metal sorption. Just as it is possible to support the sorption of metals by various sorbents by changing the conditions, it is also possible to support their recovery, by desorption, by reversing the conditions.

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