

Bioremediation of heavy metal-contaminated environment: developed strategies and potential use of biosurfactants as chelators

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ABSTRACT

Heavy metal pollution damages the ecosystems and presents a major problem for public health. Thus, an urgent need was developed to decrease the high levels of heavy metals in the soil and aquatic environments. With this aim, numerous physico-chemical strategies were developed. However, they are money-consuming, require the use of energy and chemical additives and can release secondary compounds that can pollute and cause great damage to the environment. Then, biological methods based on the investigation of bacteria, fungi and plants along with their derived secondary active metabolites became the best alternatives. Using plant capacities, different phytoremediation strategies were developed such as phytoextraction, phytovolatilization, rhizofiltration and phytostabilization. Regarding bioremediation, bacterial biosorption of heavy metals, biolixiviation and lagooning offer great potential for their environmental cleaning. Additionally, the use of secondary active metabolites, such as biosurfactants, is well-studied. Generally, they are a class of structurally very varied molecules commonly synthesized by many microorganisms with amphiphilic character. Owing to their anionic charge, they have the capacity to sequester heavy metals permitting their elimination. Glycolipids and lipopeptides are among the most recognized biosurfactants with interesting heavy metal chelating properties.

Key words: bioremediation, chelation, glycolipids, heavy metals, lipopeptides, phytoremediation, toxicology

HIGHLIGHTS

- Studied heavy metal pollution and toxicology.
- Bioremediation and phytoremediation technologies for heavy metal depollution.
- Use of biosurfactants as heavy metal chelators.

INTRODUCTION

Generally, heavy metals (HMs), defined as natural elements with a density above 5 g/cm³, are well characterized by their high toxicities (Briffa *et al.* 2020). Copper (Cu), arsenic (As), nickel (Ni), zinc (Zn), cobalt (Co), manganese (Mn), lead (Pb), cadmium (Cd) and mercury (Hg) are among the most recognized elements. They are usually present in the aquatic and terrestrial environment as traces (Sarker *et al.* 2023). However, nowadays, due to mine and industrial exploitation, HMs have become highly omnipresent in the environment to display a principal source of contamination (Sumiahadi & Acar 2018; Sarker *et al.* 2023). Being nonbiodegradable and having great toxicity at very low levels, HM pollution became a particular worry (Rajeshkumar & Li 2018; Maurya *et al.* 2019). On the basis of these considerations, their decontamination became a pressing requirement. In this aim, various physicochemical techniques were investigated for the cleaning of HM-polluted soil and aquatic media (Akhtar *et al.* 2020; Ahmed *et al.* 2022; Sarker *et al.* 2023). Nevertheless, they are expensive and highly energy consuming (Selvi *et al.* 2019). Hence, biological methods mainly bioremediation and phytoremediation were exploited for the cleaning of contaminated areas (Sarker *et al.* 2023). Being clean, safe and low-energy engrossing, they became the best alternative for physicochemical methods (Sarker *et al.* 2023). Different processes like phytoextraction, phytovolatilization, rhizofiltration, phytostabilization, bioleaching, bacterial biosorption and biological chelation appeared as great replacement strategies to cleanup HM contamination

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(Shawai *et al.* 2017; Selvi *et al.* 2019; Akhtar *et al.* 2020; Mishra *et al.* 2021; Ahmed *et al.* 2022; Sarker *et al.* 2023).

Biosurfactants (BioS) are secondary active metabolites synthesized by various microbial strains and can be investigated in HM bioremediation (Mishra *et al.* 2021). Having the ability to decrease the surface tension, possessing emulsifying and oil-dispersing activities along with their efficiency under extreme conditions of salinity, pH and temperature, microbial surfactants have become very popular (Mnif & Ghribi 2015a, 2015b; Markande *et al.* 2021; Debnath *et al.* 2022; Gayathiri *et al.* 2022; Sharma *et al.* 2022). They have become the best replacement for synthetic surfactants in various areas including medical and therapeutic fields (Mnif & Ghribi 2015a, 2015c; Markande *et al.* 2021), cosmetic (Bouassida *et al.* 2017; Gayathiri *et al.* 2022), chemical industry like detergent formulation (Bouassida *et al.* 2018a; Mnif *et al.* 2023), food technology (Mnif *et al.* 2013a; Ribeiro *et al.* 2020; Debnath *et al.* 2022; Gayathiri *et al.* 2022), agriculture (Gayathiri *et al.* 2022) and for the biocontrol of plant pests (Mnif *et al.* 2013b; 2015a; Markande *et al.* 2021). Moreover, they are widely used in environment and bioremediation as surface tension reducers, microbial growth inducers and flocculating agents, to improve hydrocarbon solubilization, mobilization and biodegradation, to increase dye biodecolorization and biodegradation and to remove HMs (Mnif *et al.* 2013c, 2013d, 2014, 2015b, 2015c, 2015d, 2016, 2017a, 2017b; Mnif & Ghribi 2015b; Busi & Rajkumari 2016; Bouassida *et al.* 2018b; Markande *et al.* 2021; Sharma *et al.* 2022). In particular, being biodegradable and non-toxic, research on BioS is stimulated by major ecological concerns as they appear as the best choice to better fight against pollution. Being derived from microbial cells, BioS have great possibilities to be produced from renewable substrates with easy purification procedures (Mnif *et al.* 2013b, 2013d, 2021a, 2021b, 2021c; Zouari *et al.* 2014; Mnif *et al.* 2021a, 2021b; Gayathiri *et al.* 2022). Thus, having an anionic character, they have great opportunities to be exploited for metal sequestration enabling consequently metal desorption from soil particles and elimination from contaminated water (Mnif & Ghribi 2015a; Mnif *et al.* 2021a). With this aim in view, we will discuss the exploitation of bacterial strains and plants for HM elimination along with the particular use of BioS as chelating agents.

HM CONTAMINATION AND TOXICOLOGY

Soil and aquatic contamination

As largely recognized, HM pollution is one of the most interesting environmental inquiries. Diverse natural and anthropogenic facts can be the cause of HM release and dispersion in soil and aquatic environments. Commonly, rocks are the main reservoir of HMs. When they are altered, they discharge huge quantities of HMs (Sumiahadi & Acar 2018; Akhtar *et al.* 2020; Ahmed *et al.* 2022). Also, numerous anthropogenic activities release huge quantities of metals into the ecosystems and disturb the ecological balance. They are mainly caused by industries actuating in mining fields, metallurgical plants, chlorine manufacturing, tannery and paper mills (Sumiahadi & Acar 2018; Akhtar *et al.* 2020; Ahmed *et al.* 2022). Also, there exist numerous sources of soil contamination including the application of trace elements or remaining slurry from wastewater treatment plants to agricultural soils, the use of certain fungicides and pesticides that contain mercury, the land accumulations of compost, the application of animal excrement, the irrigation by wastewater, the use of charcoal combustion residues and the leakage of petrochemicals (Sumiahadi & Acar 2018; Akhtar *et al.* 2020; Ahmed *et al.* 2022). In addition, Zn, Cu and Pb are discharged from the incineration of waste and the combustion of motor gasoline. They are deposited from the atmospheric dust emitted by the runoff of rainwater on roofs and roads causing soil contamination (Sumiahadi & Acar 2018; Akhtar *et al.* 2020; Ahmed *et al.* 2022). Table 1 summarizes some data reporting actual HM concentrations in soil and water in various locations.

HM bioaccumulation

In general, soil contamination with HMs may pose a danger and threat to human being and the ecosystem equilibrium. Diverse factors can promote the toxic impact of HMs such as the direct contact and/or ingestion of contaminated soil and the drinking of contaminated ground water. Also, HMs can be dispersed and transmitted via the food chain from the soil to plant to human or from soil to plant to animal to human. Moreover, we remark on the reduction of land relevance for agricultural production (Selvi *et al.* 2019; Akhtar *et al.* 2020; Ahmed *et al.* 2022). Also, soil contamination may decrease the food quality reducing its safety and marketability via phyto-toxicity. Furthermore, the biodegradation of organic pollutants may be altered in the presence of toxic metals that inhibit microbial growth and activities (Lu *et al.* 2020; Batkhuyag *et al.* 2022). Additionally, HM presence causes great damage to agricultural soil decreasing crop productivity (Rai *et al.* 2019; Alengebawy *et al.* 2021).

Table 1 | HM concentrations in soil and water in various locations

Location	Wastewater	Soil	References
Open drainage channels in Nairobi, Kenya	Mean levels ranged from 0.0001 to 0.015 mg/kg in an ascending order of Tl < cd < Hg < Ni < Cr < Pb.	Mean levels ranged from 0.085 to 199.99 mg/kg, with those of Hg, Pb, Cr, Cd & Ni	Kinuthia <i>et al.</i> (2020)
Surakarta		Dumpsites soil (in mg/kg): Mean levels ranged from 283.56 to 610 for Mn; 33.96 to 48.03 for Cr; 0.74 to 1.09 for Cd; 9.98 to 31.05 for Pb; 8,261.70 to 9,112.33 for Fe	Nyiramigisha & Sajidan (2021)
Surakarta		Control soil (in mg/kg): Levels are around 210.91 for Mn; 23.25 for Cr; 0.14 for Cd; 0.6 for Pb; 5,251.92 for Fe	Nyiramigisha & Sajidan (2021)
Sediment Sample of Bartin River		Mean levels (in mg/kg) are around 7 for Co; 16 for Cr; 22 for Cu; 20 for Fe; 352 for Mn; 21 for Ni; 9 for Pb; 53 for Zn; 7 for As; 73 for Ba; 26 for Li	Gunes (2022)
Water Sample of Bartin River		Mean levels (in mg/kg) are round 2.53 for Cr; 2.535 for Cu; 290 for Fe; 28 for Mn; 7.7 for Ni; 7.45 for Zn; 211 for Al; 105 for Ba; 70.1 for B; 12.16 for Li	Gunes (2022)
Rudňany (Slovakia)	Water (in µg/l) : Mean values are around 8 for Cu; 5 for As; 4.5 for Pb; 6.9 for Zn and 0.8 for Hg	Soil (in mg/kg): Mean values are around 294.75 for Cu; 38.125 for Pb; 194.875 for Zn and 27.66 for Hg	Angelovicova & Fazekasova (2014)
King George Island, Antarctica		Soil (in mg/kg): Mean values ranged from 61.36 to 562.2 for Cu; 0.52 to 1.95 for Pb; 54.61 to 577.9 for Zn; 0.04 to 3.76 for Cd and 6.83 to 25.9 for Cr Sediments (in mg/kg): Mean values ranged from 58.55 to 498.3 for Cu; 0.60 to 2.51 for Pb; 56.22 to 345.9 for Zn; 0.07 to 5.77 for Cd and 7.76 to 39.5 for Cr	Lin <i>et al.</i> (2021)
Coastal area in Northern Vietnam	Surface water and well water (in mg/L) respectively: Mean values are about 0.02 and 0.03 for Cd; 0.17 and 0.12 for Pb; 0.19 and 0.06 for As and 2.56 and 0.25 for Cr		Minh Ngoc <i>et al.</i> (2020)
Industrial area in West Algeria		Soil collected from waste dumpsite of al zinc ursine (in mg/kg): Mean values are about 21.64 for Pb; 394.79 for Zn; 13.25 for Ni; 8.54 for Co; 5.41 for Cd; 138.65 for Mn; 37.74 for Cr; 6,276.66 for Fe and 15.06 for As	Tahar & Keltoum (2011)
Tanda Dam kohat	Water (in mg/L): Mean values are about 0.081 for Cd; 0.35 for Pb; 0.0043 for Ni; 0.23 for Zn; 2 for Fe; 2.122 for Cr and 0.432 for Cu	Soil (in mg/kg): Mean values are about 0.173 for Cd; 0.215 for Pb; 0.46 for Ni; 0.212 for Zn; 183.07 for Fe; 4.942 for Cr and 0.803 for Cu	Nazir <i>et al.</i> (2015)
Xiangjiang River in South China		Sediment (in mg/kg): Mean values are about 59.71 for Cr; 16.97 for Co; 36.29 for Ni; 71.29 for Cu; 257.17 for Zn; 98.38 for As; 23.31 for Cd and 102.52 for Pb	Huang <i>et al.</i> (2020)

(Continued.)

Table 1 | Continued

Location	Wastewater	Soil	References
River Kabul (Pakistan)	Water (in mg/L): Mean values are about 0.29 for Cr; 1.204 for Ni; 0.028 for Cd and 0.476 for Pb	Sediments (in mg/kg): Mean values are about 53.8 for Cr; 137.2 for Ni; 3.2 for Cd and 16.9 for Pb	Afzaal et al. (2022)
Dhaleswar River	Water (in µg/L): Mean values are about 7.21 for Ni; 50.05 for Pb; 6.49 for Cd; 441.34 for Cr and 154.69 for Cu	Sediment (in mg/kg): Mean values are about 181.06 for Ni; 64.22 for Pb; 3.23 for Cd; 117.56 for Cr and 44.05 for Cu	Ahmed et al. (2009)
Northern Ningxia, China	Irrigation water (µg/L): Mean values are about 0.11 for Cd; 1.12 for Pb; 41.03 for Cr; 0.91 for Zn; 1.6 for Cu and 3.6 for As	Soil (in mg/kg): Mean values are about 0.14 for Cd; 18.16 for Pb; 37.25 for Cr; 138.2 for Zn; 19.61 for Cu and 14.18 for As	Liu et al. (2020)
District Kasur, Pakistan		Soil (in mg/kg): Mean values are about 22.78 for As; 4.35 for Cd; 7.44 for Co; 24.53 for Cr; 22.84 for Cu; 53.56 for Fe; 0.23 for Hg; 1,042.56 for Mn; 21.56 for Ni; 32.99 for Pb and 92.36 for Zn	Ashraf et al. (2021)

Moreover, in the case if they are not adsorbed to soil particles, HMs can be dissolved in groundwater and rivers, and thus they can contaminate many plants ([Rai et al. 2019](#); [Alengebawy et al. 2021](#)). When the levels of vegetable contamination increase greatly, the growth and productivity of plants are affected in addition to the food qualities ([Rai et al. 2019](#); [Alengebawy et al. 2021](#)). Then, these contaminations eventually lead to poisoning of plants, animals and humans. Food, vegetables, cereals, fruits and fish can all be stained by accumulation of the toxins from the soil or water. In addition, throughout the food chain, some are concentrated in living organisms, a phenomena called bio-accumulation ([Rai et al. 2019](#); [Alengebawy et al. 2021](#)). As a result, by the process of bio-amplification, very high levels of HMs are found in diverse species of fish that are consumed by humans ([Ahmed et al. 2019](#); [Ali & Khan 2019](#)). These 'bio-accumulation' and 'bio-amplification' processes explain the propagation of HM contamination to numerous living organisms including human beings, animals and plants ([Ahmed et al. 2019](#); [Ali & Khan 2019](#)).

Toxicology of HMs

As a rule, previously, in the aquatic environment, metals are dispersed among the different compartments namely water, suspended solids, sediments and biota. Diverse physicochemical and biological phenomena govern the distribution of HMs. They include the sedimentation, the adsorption/desorption, the dilution, the advection, the dispersion and the bio-accumulation ([Liu et al. 2022](#); [Sarker et al. 2023](#)). Being fairly widespread in marine environments and bio-accumulated in aquatic organisms, especially in fish, HMs presented a potential hazard for marine consumers ([Ahmed et al. 2019](#); [Maurya et al. 2019](#); [Liu et al. 2022](#)). Seafood is an excellent sensor for metal pollutants which focuses on assimilation organs (liver and kidney equivalents), that are precisely what humans eat ([Ahmed et al. 2019](#); [Maurya et al. 2019](#); [Liu et al. 2022](#)). In fact, as proved by [Rajeshkumar & Li \(2018\)](#), great amounts of HMs were found in the liver, gills and the lowest in the muscle by bio-accumulation. Mostly, fish and aquatic animals accumulate high levels of Cd and to a lesser extent Pb, but little Hg. In addition, food plants and vegetables suffer from HM contamination. For vegetables, the bio-accumulation pattern was the highest in the leaf followed by the root and the stem and the tuber are at the end of the list ([Khan et al. 2015](#); [Rai et al. 2019](#); [Alengebawy et al. 2021](#)). Moreover, HM accumulation greatly affect the nutritional values of foods as they damage greatly plant proteins, amino acids, carbohydrates, fats and vitamins ([Khan et al. 2015](#); [Rai et al. 2019](#); [Alengebawy et al. 2021](#)). As a matter of fact, plants and vegetables grown on metal-contaminated soil suffer from nutrient deficiency. Therefore, their consumption may cause nutritional deficit in the population especially those of the developing countries which are already suffering from the mal-nutrition troubles ([Khan et al. 2015](#); [Rai et al. 2019](#); [Alengebawy et al. 2021](#)).

As largely recognized, the majority of HMs have no known beneficial effect on the cell and are considered poisonous to humans, mainly As, Pb, Hg, Cr and Cd ([Briffa et al. 2020](#); [Balali-Mood et al. 2021](#); [Witkowska et al.](#)

2021). They can cause very serious consequences for our health such as behavioral and developmental disorders, stunting and system disruptions responsible for functional or psychosomatic disturbance (chronic fatigue syndrome), allergies, neurodegenerative problems (Parkinsons and Alzheimers), organic lesions and cancer in addition to autoimmune diseases (Briffa *et al.* 2020; Balali-Mood *et al.* 2021; Witkowska *et al.* 2021; Ahmed *et al.* 2022).

BIOLOGICAL PROCESS DEVELOPED FOR HM DEPOLLUTION

Regarding the issues developed earlier involving the widespread of HMs, their toxicity to humans, animals and plants and their bioaccumulation and bioamplification among the food chain, an immediate necessity was developed for their bio-de-pollution. In this respect, diverse physicochemical methods for HM elimination from water- and soil-polluted areas were investigated (Akhtar *et al.* 2020; Ahmed *et al.* 2022; Sarker *et al.* 2023). Nevertheless, they have some drawbacks such as the employment of toxic products and high energy consumption (Akhtar *et al.* 2020). Accordingly, biological treatment methods implicating the utilization of inert agricultural waste, plants and microbial cells along with their secondary active metabolites became the best replacement (Akhtar *et al.* 2020; Sarker *et al.* 2023). They are well known for their ability to neutralize completely or partially HMs. The principal phenomena applied are bio-sorption, mobilization, immobilization, volatilization and/or extraction (Kapahi & Sachdeva 2019; Akhtar *et al.* 2020; Ahmed *et al.* 2022; Sarker *et al.* 2023).

APPLICATION OF PHYTO-REMEDICATION FOR HM DECONTAMINATION

Phyto-remediation is defined as a biotechnology that uses plants to reduce, remove, degrade or immobilize contaminants in soil, sludge, sediment, surface water, groundwater and polluted water (Sumiahadi & Acar 2018; Ahmed *et al.* 2022; Sarker *et al.* 2023). Typically, the majority of plants have the property to tolerate and accumulate extremely high levels of HMs in their roots, stems and aerial parts (Sumiahadi & Acar 2018; Ahmed *et al.* 2022; Sarker *et al.* 2023). Regarding literature reviews and studies, numerous phyto-remediation strategies are developed including the phyto-extraction, the phyto-volatilization, the rhizo-filtration and the phyto-stabilization (Yan *et al.* 2020; Ahmed *et al.* 2022; Sarker *et al.* 2023). As defined, phyto-extraction corresponds to the absorption of soil contaminants by the roots of the plant. Subsequently, these contaminants are transported and accumulated in the stems and leaves, which are then harvested (Sumiahadi & Acar 2018; Yan *et al.* 2020; Ahmed *et al.* 2022; Sarker *et al.* 2023). As known, hyper-accumulating plants have the ability to accumulate more than 1% of metals in their internal tissues. As largely recognized, the phyto-volatilization requires the employment of plants to extract soil pollutants and to convert them toward less toxic volatile compounds (Sumiahadi & Acar 2018; Yan *et al.* 2020; Ahmed *et al.* 2022; Sarker *et al.* 2023). Therefore, some phyto-extracted contaminants can be transformed into their volatile form(s) and evaporated into the atmosphere via the stoma of the leaves under the effect of photosynthesis. This strategy requires the investigation of plants qualified as hyper-accumulators having the potentiality to cumulate very high amounts of HMs in their underground parts without phyto-toxicity symptoms. As mainly accepted, rhizo-filtration, requiring the use of plant roots to absorb, precipitate and accumulate metals from wastewater is well applied (Sumiahadi & Acar 2018; Yan *et al.* 2020; Ahmed *et al.* 2022; Sarker *et al.* 2023). When the roots of these plants reach their maximum retention capacity, the plants are harvested. Phyto-stabilization corresponds to the immobilization of HMs on the plant roots (Sarker *et al.* 2023). With this ability, roots limit the erosion in surface layers and stabilize HMs preventing, therefore, migration of contaminants to the surface and groundwater and reducing their bioavailability in the soil (Sumiahadi & Acar 2018; Yan *et al.* 2020; Ahmed *et al.* 2022; Sarker *et al.* 2023). Additionally, it is well reported that the bioavailability and uptake of HMs increase considerably by the microorganisms of rhizosphere that have the capacity to secrete enzymes and secondary active metabolites with anionic character (Yan *et al.* 2020; Sarker *et al.* 2023). Consequently, they complex HMs in the rhizosphere leading to the formation of HM aggregates that limit metal elements translocation and uptake (Yan *et al.* 2020).

APPLICATION OF BIOREMEDIATION FOR HM DECONTAMINATION

Regarding HM de-pollution, bioremediation appears to be the most promising technology as it presents now drawbacks. As defined, it is qualified as a biotechnological method involving the exploitation of living organisms, like microbes and bacteria, for the elimination of soil and water pollutants, namely hydrocarbons and HMs (Kapahi & Sachdeva 2019). Regarding HM elimination, many strategies are developed including bacterial

bio-sorption, bioleaching and lagooning (Ayangbenro & Babalola 2017; Kapahi & Sachdeva 2019; Akhtar *et al.* 2020; Ahmed *et al.* 2022; Sarker *et al.* 2023). Generally, the application of bacterial cells as a support for the bio-sorption of HM acknowledges a special interest. Passive and active mechanisms managed the interrelation between bacterial cells and metals (Fathollahi *et al.* 2021; Pham *et al.* 2022; Sarker *et al.* 2023). Passive ones are independent of the metabolism and, therefore, of the physiological state of the cells (live or dead). They are also fast and reversible. Occurring at the cell/solution interface, they imply diverse mechanisms including precipitation, surface complexation and ion exchange (Fathollahi *et al.* 2021; Pham *et al.* 2022). The active ones are specific to each bacterial strain and depend on the cell's metabolism. They are generally inducible and slower (Fathollahi *et al.* 2021; Pham *et al.* 2022). During this time, multiple reports studied the bio-sorption of HMs (Yang *et al.* 2016; Oyewole *et al.* 2019; Mwandira *et al.* 2020; Priyadarshane & Das 2021; Sarker *et al.* 2023). Additionally, agriculture waste, lignocellulosic biomass, marine and brown algae and plant-derived biopolymers were reported as efficient and green biosorbents of HMs for waste-water treatment and decontamination (Lindholm-Lehto 2019; Ali *et al.* 2021; Elgarahy *et al.* 2021; Nguyen *et al.* 2022). The second strategy developed is bioleaching. It involves specific bacterial strains that resist metals, oxidizing iron and sulfides. This can reduce the pH and change the oxidation–reduction environment permitting the release of HMs. In this aim, numerous sulfur-oxidizing bacteria efficient for the bioleaching of HMs from mine tailing, rocks and sediments (Gan *et al.* 2016; Wu *et al.* 2020; Gao *et al.* 2021) and from electronic waste (Kaur *et al.* 2022; Venugopal *et al.* 2022) are well reported. The third strategy developed corresponds to lagooning. It requires the passage of effluent through swamps whose vegetation (plants, algae, and bacteria) immobilizes metals, especially by precipitation in sulfide form by sulfate-reducing bacteria (Verma & Sharma 2017).

POTENTIAL USE OF BIOS FOR THE TREATMENT OF HM-CONTAMINATED WATER AND SOIL

BioS: different classes and properties

BioS or biological surfactants correspond to secondary active metabolites produced by microbial strains during their growth on water immiscible substrates (Mnif & Ghribi 2015b; Markande *et al.* 2021; Sharma *et al.* 2022). Constituted by a hydrophobic tail and hydrophilic head, they are characterized by an amphiphilic character. The hydrophobic tail correlates to saturated or un-saturated fatty acids. For the hydrophilic moiety, it consists of polysaccharides (mono or di), amino acids or peptides (Mnif & Ghribi 2015b; Markande *et al.* 2021; Sharma *et al.* 2022).

BioS classification is based on different principles. The first criterion corresponds to their molecular weight for that we discern the high- and low-molecular-weight BioS (Mnif & Ghribi 2015b; Markande *et al.* 2021). The second criterion consists of their ionic charges for which we distinguish the anionic, cationic, non-ionic and neutral ones (Mnif & Ghribi 2015b; Markande *et al.* 2021). The third criterion matches the secretion type. Accordingly, BioS are subdivided into intracellular, extracellular and those attached to microbial cells. The fourth and last requisite corresponds to the biochemical nature of the BioS for that we enumerate six groups, namely glycolipids (GLs), lipopeptides (LPs), phospholipids (PLs), lipopolysaccharides (Lps), neutral lipids (NLs) and polymeric surfactants (Mnif & Ghribi 2015b; Markande *et al.* 2021). On the basis of their higher structural versatility and various functional activities, LPs and GLs are among the most recognized.

Structurally, the combination of a fatty acid chain and a carbohydrate moiety constitutes GL BioS. They are described as a diverse group of compounds differing by the nature of the carbohydrate and lipid parts (Mnif & Ghribi 2015c; Mnif *et al.* 2017a; Eras-Muñoz *et al.* 2022). On the basis of these properties, GLs are partitioned into rhamnolipids (RLs), trehalolipids (TLs), sophorolipids (SLs), cellobiolipids (CLs), mannosylerythritol lipids (MLs), lipomannosyl-mannitols (LMs), lipomannans (Lps) and lipoarabinomannanes (Lams), diglycosyl diglycerides (Dds), monoacylglycerol (MGs) and galactosyl-diglycerides (GDs) (Mnif & Ghribi 2015d; Mnif *et al.* 2017a; Eras-Muñoz *et al.* 2022).

The combination of a fatty acid to a peptide moiety constitutes LPs. They correspond to the second most known class of BioS composed of a group of isoforms differing by the length of the fatty acid chain, the composition of the peptide moiety and the connection between them (Mnif & Ghribi 2015a, 2015d; Carolin *et al.* 2021). To name, surfactin, iturin and fengycin are among the most admitted ones (Mnif & Ghribi 2015a, 2015d).

Regarding their functional properties, BioS are well acknowledged by their surface activity, namely the ability to decrease the surface and interfacial tension at the surface and the interface, respectively. Moreover, they are endowed with diverse functional activities, namely the emulsification/de-emulsification, dispersing, foaming, viscosity reducers, solubilizing and mobilizing abilities and pore-forming capacity. In addition, BioS are talented by

numerous biological potencies including antimicrobial; hemolytic; antiviral and antioxidant properties enabling their use in many fields (Mnif & Ghribi 2015b; Carolin *et al.* 2021). Compared to chemical surfactants, they are qualified by a low critical micelle concentration (CMC). Thus, given the incontestable concern of these molecules, their higher biodegradability (Darlane *et al.* 2016), biological and ecological compatibility (Gayathiri *et al.* 2022; Mnif *et al.* 2023), lack of toxicity (Camacho-Chab *et al.* 2013; Fei *et al.* 2020) and superior effectiveness against higher temperature, pH and salinity (Mnif *et al.* 2021a, 2021b, 2021c), BioS are very advantageous toward chemical surfactants. Moreover, they have the property to be generated from renewable and cheap substrates offering a low cost of production (Mnif *et al.* 2013b, 2013d, 2021a, 2021c; Samykanu & Achary 2017; Sarubbo *et al.* 2018; da Rocha Junior *et al.* 2019; De Araujo Freire *et al.* 2020; Carolin *et al.* 2021; Gayathiri *et al.* 2022; Kalvandi *et al.* 2022). Furthermore, the optimization of culture media components and conditions increase the yield of production greatly and reduces their cost (Marzban *et al.* 2016; Purwasena *et al.* 2019; Wanjari *et al.* 2019; Mnif *et al.* 2021c). In addition, we report the increase and decrease of the production yield and cost, respectively, by the innovation of novel fermentation systems including the submerged and solid state ones (Mnif *et al.* 2013b, 2013d, 2021a, 2021c; Zouari *et al.* 2014; Carolin *et al.* 2021). As well, we note the improvement of BioS production by random mutagenesis tools (Bouassida *et al.* 2018b). Therefore, biological surfactants afford interesting prospects as ecological and effective replacements for synthetic ones (Mnif & Ghribi 2015b; Carolin *et al.* 2021; Eras-Muñoz *et al.* 2022). They appear as possible supplements in the food, cosmetic and pharmaceutical industries. Also, they are applied in environmental biotechnology for the enhancement of hydrocarbon biodegradation and HM sequestration (Mnif & Ghribi 2015a; Carolin *et al.* 2021; Eras-Muñoz *et al.* 2022).

BioS and HM decontamination: modalities of chelation

During the biological remediation of HMs, different modalities of interaction were defined namely the complexation and/or ion exchange and the solubilization-dissolution (Mishra *et al.* 2021). Moreover, we distinguish interactions between cellular surfaces, enabling the metal's adsorption to bacterial cell and/or their entering into cells (Mishra *et al.* 2021). As recognized, BioS improve bacterial bio-sorption. In addition, they can be used alone as excellent chelating agents of HMs owing to their binding capacity (Carolin *et al.* 2021; Mishra *et al.* 2021; Eras-Muñoz *et al.* 2022). For contaminated soils, microbial cells and their derived polymers find it difficult to move throughout the soil. In those cases, purified or partially purified BioS having generally smaller sizes below 1,500 Da, have great solubility and mobility. Also, owing to their surface activity, they penetrate easily through soil particles (Mnif & Ghribi 2015d; Mishra *et al.* 2021).

BioS binds to metals by different mechanisms including electrostatic interaction, counter-ion association, ion exchange and precipitation-dissolution (Carolin *et al.* 2021; Mishra *et al.* 2021; Eras-Muñoz *et al.* 2022). Therefore, anionic BioS improve the desorption of metals from soils in two ways. Firstly, aggregates were formed between the BioS and the free and non-ionic forms of metals in solution (Ravindran *et al.* 2020; Sun *et al.* 2020; Mishra *et al.* 2021). Moreover, owing this property, they can help in the release of metal ions from the soil. Particularly, having anionic character, certain BioS form ionic bonds with metals. This interaction generates nonionic complexes that are more stable than those created between the metals and the soil (Ravindran *et al.* 2020; Mishra *et al.* 2021). Apparently, the charge interaction between the HMs and the charged polar head groups of the BioS on the micelles surfaces facilitate metal removal (Mishra *et al.* 2021; Zhang *et al.* 2022). Being positively charged, the metals attach to the external hydrophilic surface of the BioS, instead of being embedded into the micelles structure (Mishra *et al.* 2021; Mnif *et al.* 2021a).

Secondly, under reduced interfacial tension conditions, HMs can accumulate at the solid-solution interface by the direct contact between the BioS and the absorbed metals (Mishra *et al.* 2021; Mulligan 2021; Fatima *et al.* 2022). Consequently, having a neutral charge along with potential integration of the metal into micelles, the metal-BioS complexes desorb from the soil matrix and drive into the soil solution (Mishra *et al.* 2021; Mulligan 2021; Fatima *et al.* 2022). In contrast to anionic BioS, a competition between cationic BioS and charged metal ions can occur on the surface of soil particles leading to the replacement of the HMs in some cases (Franzetti *et al.* 2014; Sarubbo *et al.* 2015; Ayangbenro & Babalola 2018).

Prospective use of BioS for HM chelation

LP biosurfactants as HM chelators

As described in numerous literature reviews and studies, LP BioS offer great opportunities for soil and water HM decontamination (Table 2). With this aim in view, Mulligan *et al.* (2001) reported the effective use of *Bacillus*

Table 2 | Heavy metal chelation by diverse bacterial-derived BioS

Strains	BioS nature	BioS concentration	Heavy metal efficiency	Study
	Lipopeptides			
<i>B. subtilis</i>	Surfactin	–	15% of Cu and 6% of Zn	Mulligan <i>et al.</i> (2001)
–	Surfactin and lichenysin	–	Zn and Cr	Zouboulis <i>et al.</i> (2003)
<i>B. subtilis</i> A21	Surfactin and fengycin	[BioS] > CMC value	44.2% of Cd, 35.4% of Co, 40.3% of Pb, 32.2% of Ni, 26.2% of Co and 32.07% of Zn	Singh & Cameotra (2013)
<i>B. subtilis</i> SHB 13	Surfactin	–	74 and 98% of Cr	Swapna <i>et al.</i> (2016)
–	Surfactin	–	47.2–95.82% of Fe = when using crude BioS preparation 37.01–91.94% of Fe = when using purified surfactin	De Araujo Freire <i>et al.</i> (2020)
<i>B. circulans</i>	Lipopeptide	–	Pb and Cd	Das <i>et al.</i> (2009)
–	Lipopeptide	–	Co, Zn, Cd, Pb and Hg	Zhu <i>et al.</i> (2013)
<i>Bacillus</i> sp. HIP3	Lipopeptide	–	13.57% of Co, 12.71% of Pb, 2.91% of Zn, 1.68% of Cr and 0.7% of Cd	Md Badrul <i>et al.</i> (2019)
<i>Bacillus</i>	Lipopeptide	[BioS] = 2.0 × CMC	75.5% of Hg, 97.73% of Pb, 89.5% of Mn and 99.93% of Cd	Ravindran <i>et al.</i> (2020)
<i>Bacillus subtilis</i>	Lipopeptide	–	Remove 164.4 mg/g of Pb from manually metal-contaminated water and 130.4 mg/g of Pb in actual wastewater	Zhao <i>et al.</i> (2020)
<i>B. cereus</i> NWUAB01	Lipopeptide	–	Remove 69, 54 and 43% of Pb, Cd and Cr, respectively, from polluted soil	Ayangbenro & Babalola (2020)
<i>B. subtilis</i> ZNI5	Lipopeptide	[BioS] = 0.05%	Chelation of Cu and Co	Mnif <i>et al.</i> (2021a)
<i>Bacillus</i> sp. SHA302	Lipopeptide	[BioS] = CMC	Release 53.8 and 39.3% of Zn and Pb from soil	Kalvandi <i>et al.</i> (2022)
	Glycolipid			
–	Rhamnolipid	0.1%	As, Co, Pb and Zn	Wang & Mulligan (2009)
<i>Pseudomonas</i>	Rhamnolipid	80 ppm	53, 62, 56, 28, 20 and 7% of Cd, Pb, Ni, Ba, Zn and Sr	Elouzi <i>et al.</i> (2012)
–	Rhamnolipid	0.4 mg/mL 1 mg/mL	11% Fe and 25% Zn 19% Fe and 52% Zn	Diaz <i>et al.</i> (2015)
<i>P. aeruginosa</i> AMB AS7	Rhamnolipid	[BioS] = CMC	66.95% of Cr	Samykanu & Achary (2017)
–	Rhamnolipid	–	Chelation of Ni, Cu and Cd	Lee & Kim (2019)
–	Rhamnolipid and Saponin	Mixture of 62.5% of Rhamnolipid and 37.5% of Saponin	87, 77 and 70% for Ni, Cr and Vanadium (V)	Khiyavi <i>et al.</i> (2020)
<i>P. aeruginosa</i> PTCC1340	Rhamnolipid	–	43.05, 34.73 and 52.81% for Ni, Cr and Cd, respectively	Kholghi <i>et al.</i> (2021)
–	Rhamnolipid	[BioS] = CMC	Remove 72.4% of Cd from soil	Zhang <i>et al.</i> (2022)

(Continued.)

Table 2 | Continued

Strains	BioS nature	BioS concentration	Heavy metal efficiency	Study
–	Rhamnolipid	–	Remove 94.07, 81.93, 99.81, 76.00, and 41.91% for Zn, Mg, Cu, Mn and Na, respectively	Munoz-Cupa <i>et al.</i> (2022)
<i>Starmerella bombicola</i>	Sophorolipid	[BioS] = 8%	83.6% of Cd and 44.8% of Pb	Qi <i>et al.</i> (2018)
–	Sophorolipid	–	Cu and Zn	Castelein <i>et al.</i> (2021)
<i>Burkholderia</i> sp. Z-90	Glycolipid	–	Remove 44.0, 32.5, 52.2, 37.7 and 24.1% for Zn, Pb, Mn, Cd, Cu and As, respectively	Yang <i>et al.</i> (2016)
<i>Pseudomonas</i> sp. CQ2	Glycolipid	–	Remove 78.7% of Cd, 65.7% of Cu and 56.9% of Pb from contaminated soil	Sun <i>et al.</i> (2020)
Various biosurfactants				
<i>Candida sphaerica</i> UCP0995	–	[BioS] < CMC	Remove Fe, Zn and Pb	Luna <i>et al.</i> (2016)
<i>Rahnella</i> sp. RM	–	–	70% of Cr at the 100 mg/L	Govarthanan <i>et al.</i> (2017)
<i>Candida lipolytica</i>	–	–	30–40% of Cu and Pb	Santos <i>et al.</i> (2017)
<i>Candida guilliermondii</i> UCP 0992	–	–	Remove 98.9% of Zn, 89.3% of Fe and 89.1% of Pb from soil	Sarubbo <i>et al.</i> (2018)
<i>Citrobacter freundii</i>	–	–	80, 67, 66, 55, 45, 44, and 41% of Al, Pb, Zn, Cd, Fe, Cu, and Mn	Gomaa & El-Meihy (2019)
<i>Candida tropicalis</i>	–	–	Remove Zn, Cu and Pb from contaminated sand and packed columns	da Rocha Junior <i>et al.</i> (2019)
–	–	–	78.82–95.73% for Cu, from 96.78 and 98.98% for Zn and 84.98–97.68% for Pb	Ogbulie (2020)

subtilis-derived surfactin and lichenysin for the removal of 15 and 6% of Co and Zn, respectively. Used as flocculation agents for the separation of the metal-filled absorbents, surfactin and lichenysin were effective when eliminating Zn and Cr from aqueous-polluted solution (Zouboulis *et al.* 2003). Examined under the same conditions, they proved better efficiency than conventional or chemically derived surfactants namely Sodium dodecyl sulfate (SDS) or dodecylamine (Zouboulis *et al.* 2003). Similarly, Das *et al.* (2009) announced the efficiency of a *Bacillus circulans*-derived lipopeptide (LP) BioS for the elimination of HMs (Pb and Cd) from aqueous media. Similarly, *B. subtilis* A21-derived surfactin and fengycin demonstrated an efficient HM soil washing especially Cd, Co, Pb, Ni, Co and Zn with removal percentages of about 44.2, 35.4, 40.3, 32.2, 26.2 and 32.07%, respectively (Singh & Cameotra 2013). Different parameters including the concentration of surfactant, the temperature, the agitation speed and the pH of the washing solution can influence the pollutant removal ability of the BioS mixture (Singh & Cameotra 2013). The obtained results indicate the application of a mixture of LP BioS at concentrations above its CMC for soil washing. This can be a successful and eco-friendly procedure to remove pollutants (Singh & Cameotra 2013). As proposed by Swapna *et al.* (2016), surfactin, derived from HM-tolerant *B. subtilis* SHB 13, was capable of eliminating Cr contamination within 72 h at optimized conditions of pH 7, 37 °C temperature and 4% NaCl. *Bacillus* sp. HIP3-derived LP BioS has the ability to remove 13.57, 12.71, 2.91, 1.68 and 0.7% of Co, Pb, Zn, Cr and Cd, respectively, from synthetically polluted water, underlining its prospective use for biological remediation (Md Badrul *et al.* 2019). These BioS display the capacities to decrease the surface and interfacial tension, stabilize emulsions and form foam with a high selectivity in addition to a specific resistance toward extreme conditions of temperatures, pH and salinity. Moreover, BioS are qualified

by the ability to be produced on renewable resources especially used cooking oil (Md Badrul *et al.* 2019). As suggested by Ayangbenro & Babalola (2020), a *Bacillus cereus* NWUAB01-derived LP BioS was effective in the removal of 69% of Pb, 54% of Cd and 43% of Cr from polluted soil. The metal-chelating property was evidenced by genome analysis. In fact, research reveals diverse genes namely those dependable for the metal transport and resistance along with synthetic gene clusters implicated in the production of different secondary active metabolites (Ayangbenro & Babalola 2020). Zhao *et al.* (2020) proposed the use of *B. subtilis* LP to eliminate Pb from water with maximum removal efficiency of about 130.4 mg/g in natural wastewater and 164.4 mg/g in artificially contaminated water. Additionally, LPs demonstrated the ability to promote the growth of oats under Pb stress in addition to the reduction of its amount in oats grown, but also promote oat growth under Pb stress (Zhao *et al.* 2020). Regarding the mechanism involved, complexes were formed between the Pb and the LPs via –OH, C – O, O = C – O and –NH– functional groups (Zhao *et al.* 2020). Similarly, The N–C–O and C = C/C = N groups were also implicated in metal ions chelating capacity. The respective LPs were able to remove Co, Zn, Cd, Pb and Hg from the aqueous solution (Zhu *et al.* 2013). *Bacillus*-derived LP BioS showed great removal of Hg, Pb, Mn and Cd of about 75.5, 97.73, 89.5 and 99.93%, respectively, when used at $2.0 \times \text{CMC}$. This showed an effective surface treatment of carrot and farm fresh cabbage from HM contamination (Ravindran *et al.* 2020). The Tandem Electron Microscopy (TEM) analysis demonstrated effective capture of the HM. The entrapment was visualized as electron-dense dark spots surrounded by the BioS (Ravindran *et al.* 2020). De Araujo Freire *et al.* (2020) reported the efficiency of Fe capture within a synthetic effluent, utilizing a surfactin produced on papaya peel aqueous extract as substrate. The removal percentages varied between 47.2 and 95.82% when handling crude BioS preparation, and ranged from 37.01 to 91.94% when applying purified surfactin (De Araujo Freire *et al.* 2020). As published by Mnif *et al.* (2021a), having reduced phytotoxicity, *B. subtilis* ZNI5 strain and her derived BioS were checked for Cu and Co sequestration and biosorption. The obtained results indicated an interesting possible investigation of the ZNI5 LP for the bioremediation of HMs. In fact, we observed a great enhancement in the germination index of radish seeds irrigated by the artificially contaminated water after treatment. Higher results were obtained when operating with the BioS alone at 0.05% as a support of sequestration. Moreover, the BioS increase significantly the biosorption of Cu when using ZNI5 living cells as a support.

Kalvandi *et al.* (2022) reported the efficient elimination of HMs from contaminated soil by the employment of an LP BioS derived from a new isolate of *Bacillus* sp. SHA302. The LP produced on sugar beet molasses-based medium can decrease the water surface tension to about 29.2 mN/m and release 53.8% of Zn and 39.3% of Pb, from soil at the CMC and by adjusting HCl treatment (Kalvandi *et al.* 2022).

GL BioS as HM chelators

As observed in Table 1, GL BioS, especially RL and SL were applied extensively as HM chelators. A comparative study done by Mulligan *et al.* (2001) suggested the effectiveness of different treatments by SL and RL BioS for the treatment of HMs. RL and SL derived, respectively, from *Pseudomonas aeruginosa* and *Torulopsis bombicola* were assayed for HM removal from metal-contaminated sediment by 110 mg/kg Cu and 3,300 mg/kg Zn (Mulligan *et al.* 2001). The obtained results indicated that when applying 0.5% RL for soil washing, we can remove 65 and 18% of Cu and Zn, respectively. Although using 4% SL, we are able to eliminate 25 and 60% of Cu and Zn (Mulligan *et al.* 2001). When using each HMs separately, uptake efficiencies were improved greatly. Accordingly, 12% RL were able to remove 19.5% of the Zn. However, 4% SL and 0.7% HCl were able to remove 15.8% of the Zn. For Cu extraction rates, we achieve greater than 25% with 12% RL or with 2% RL and 1% NaOH (Mulligan *et al.* 2001).

In another study, RL and SL BioS were used separately as HM sequesters (Wang & Mulligan 2009). In fact, the use of 0.1% RL solution with an initial pH adjusted to 11 improves significantly As and Co, Pb and Zn elimination simultaneously. Similarly, *Pseudomonas*-derived RL has been effectively exploited for HM cleanup from contaminated water. Results showed that 80 mg/Kg of BioS were required to reduce up to 53% of Cd, 62% of Pb, 56% of Ni, 28% of Ba, 20% of Zn and 7% of Sr (Elouzi *et al.* 2012). As suggested by Diaz *et al.* (2015), RL BioS enhanced mineral waste/polluted soil decontamination. By the employment of a mixed bacterial culture of *Acidithiobacillus thiooxidans* and *Acidithiobacillus ferrooxidans*, we are able to eliminate 50 and 19% of Zn and Fe, respectively. The adjunction of BioS permits to improve the bioleaching efficiency to reach a metal removal between 63 and 70% for Zn and 36% for Fe (Diaz *et al.* 2015). In addition, the increase in RL concentration enhanced the percentage of removal. At a low concentration of about 0.4 mg/mL, we achieve elimination values of about 11% for Fe and 25% for Zn. Even though, using 1 mg/mL BioS, disposal efficacy reached 19% for

Fe and 52% for Zn (Diaz *et al.* 2015). Also, a RL BioS stripped effectively 60.34% of Fe from contaminated soil (Akintunde *et al.* 2015). Similarly, an RL BioS, derived from *P. aeruginosa* AMB AS7 and produced on agro-industry residue was able to extract 66.95% of Cr from an aqueous solution when employed at its CMC (Samykanu & Achary 2017).

With the same aim, RL BioS was demonstrated as a useful tool for extracting Ni, Co and Cd in deep-sea mining tailings (Lee & Kim 2019). With the same goal in mind, Khivyavi *et al.* (2020) studied the concomitant effect of RL and Saponin BioS on the removal of HMs, namely V, Ni and Cr from polluted soil. Maximum removals of 87, 77 and 70% for Ni, Cr and V were obtained when using a mixture of 62.5 and 37.5% of RL and Saponin, respectively (Khivyavi *et al.* 2020). Therefore, these two GL BioS can be better replacements of chemically synthesized surfactants for environmental applications. As reported by Sun *et al.* (2020), a *Pseudomonas* sp. CQ2-derived GL BioS was effective in the treatment of HMs-polluted soil. When compared to the usual chemical surfactants including SDS and Tween-80, BioS exhibited higher efficiency in the bioleaching of three different HMs including Cd, Cu and Pb with a percentage of 78.7, 65.7 and 56.9%, respectively, at optimized conditions. As demonstrated by SEM-EDX, we remark a disappearance of the granular material correlated to the decrease of the content of HMs. Additionally, the surface of the soil became smooth with the formation of perforation (Sun *et al.* 2020). As approved by ATR-FTIR, carboxyl functional groups were involved in the chelation of HMs (Sun *et al.* 2020). Moreover, Zhang *et al.* (2022) demonstrated the same finding that Cd enables the formation of the complex with the carboxyl groups exposed at the exterior of the RL micelles. They removed 72.4% of Cd from poly-aromatic hydrocarbons co-contaminated soil. Similarly, a *P. aeruginosa* PTCC1340-derived RL removed effectively Ni, Cr and Cd from contaminated soil with extraction percentages of 43.05, 34.73 and 52.81%, respectively (Kholghi *et al.* 2021). In addition, RL chelate effectively HMs from desalted effluent with uptake percentages of about 94.07 for Zn, 81.93 for Mg, 99.81 for Cu, 76.00 for Mn and 41.91 for Na (Munoz-Cupa *et al.* 2022). In fact, HM load on RL micelles with better selectivity for Mn followed by Cu, Zn, Mg and Na in the end. As suggested by Yang *et al.* (2016), a GL BioS-producing *Burkholderia* sp. Z-90 strain enhanced the bioleaching efficacy for the remediation of contaminated soil. Elimination values were about 44.0% for Zn, 32.5% for Pb, 52.2% for Mn, 37.7% for Cd and 24.1% for Cu and As. The presence of a high acid-soluble proportion of Mn, Zn and Cd and a high residual fraction of Cu, Pb and As suggested the easy elimination of Mn, Zn and Cd than Cu, Pb and As. Firstly, the adhesion of the Z-90 strain to the contaminated soil minerals contributed to the HM removal in soils. Secondly, we remark on the formation of a metal complex with BioS (Yang *et al.* 2016).

SL BioS, are well recognized by their capacity to chelate HMs. With this aim in view, *Starmarella bombicola*-derived SLs removed efficiently Cd and Pb from artificially contaminated soil in batch mode. Removal percentages are better than the application of both distilled water and synthetic surfactants. As well as 8% of crude acidic SLs removed 83.6 and 44.8% of Cd and Pb, respectively (Qi *et al.* 2018). Fourier-Transform Infrared Spectroscopy (FTIR) study revealed the sequestration of Cd with the free carboxyl functional group of the acidic SLs (Qi *et al.* 2018). Acidic SLs were more effective than lactonic SLs in improving the remediation of HMs-contaminated soils. Despite this, they present higher water solubility. Accordingly, the sequestration of Co and Zn metals by SLs BioS was investigated by Castelein *et al.* (2021). Optimal Co leaching of about 27 and 53% was obtained from a fayalite slag and a Co sulfide mine tailing, respectively. The mechanism of capture can be explained by the formation of small metallic Co droplets via a corrosion-based process, and/or from Co-Pb sulfides.

Various BioS as heavy metal chelators

As visualized in Table 1, numerous literature studies reported the bioremediation of HM-contaminated water and soil using BioS. Luna *et al.* (2016) demonstrated the effective removal of Fe, Zn and Pb by a *Candida sphaerica* UCP0995-derived BioS with maximal performance at a level inferior to the CMC value. Govarthanan *et al.* (2017) described the uptake of Cr from soil using a *Rahnella* sp. RM-derived BioS with the highest metal extraction rates of about 70% when using 100 mg/L amended flasks for 48 h. A *Candida lipolytica*-derived BioS was demonstrated effective in HM elimination. The crude BioS could remove 30–40% of Cu and Pb from standard sand, in spite of about 30% for the isolated BioS (Santos *et al.* 2017). An anionic BioS produced by the yeast *Candida guilliermondii* have the ability to complex metal elements from contaminated water visualized by the reduction of the electrical conductivity of the solutions. Applied to the elimination of metals contained in soil samples from a battery industry, the crude BioS was able to mobilize 98.9, 89.3 and 89.1% of Zn, Fe and Pb, respectively (Sarubbo *et al.* 2018). Similarly, a *Candida tropicalis*-derived BioS suppressed effectively Zn, Cu and Pb from contaminated sand and packed columns (da Rocha Junior *et al.* 2019). Having an anionic

charge, with potential surface activity, biodegradability and low toxicity, the use of BioS leads to a considerable decrease in the electrical conductivity of HM-contaminated solutions. These encourages their use for the biological remediation of HM-polluted soil and water (da Rocha Junior *et al.* 2019).

As developed by Gomaa & El-Meihy (2019), we propose the effective uptake of different HMs including Al, Pb, Zn, Cd, Fe, Co and Mn by means 80, 67, 66, 55, 45, 44 and 41%, respectively, from wastewater when using a *Citrobacter freundii*-derived BioS. Various analyses including Scanning Electron Microscopy (SEM), Energy Dispersive X-ray Spectra (EDX), and the FTIR demonstrated the implication of hydroxyl and carboxyl groups in the removal of Cd and Pb from aqueous media (Gomaa & El-Meihy 2019).

Ogbulie (2020) reported the investigation of two BioS in the uptake of Cu, Zn and Pb from waste metal dump-site soil. The two BioS demonstrated higher efficiencies with extraction potency ranging from 78.82 to 95.73% for Cu, from 96.78 and 98.98% for Zn and from 84.98 to 97.68% for Pb (Ogbulie 2020). Results are more interesting in comparison to soil washing agents such as KNO_3 , $\text{Ca}(\text{NO}_3)_2$ and NaOH . The increase in BioS concentration improved the extraction percentage (Ogbulie 2020).

CONCLUSION AND FUTURE PERSPECTIVE

HMs, inorganic contaminants with higher toxicity potential, became fairly widespread pollutants. As discussed in our review paper, different bioremediation and phyto-remediation strategies were developed to treat soil and water HM contamination. They became the best alternatives to physicochemical methods implying the use of chemicals and energy to decontaminate the environment. Moreover, here, all the reported studies improved the successful use of rhamnolipid (RL) and LP BioS as an eco-friendly alternative to chemical surfactants for the cleaning of HM-polluted soil and water. These findings implicating the potential use of BioS for the remediation of contaminated areas should be developed. In addition, the use of BioS as chelators of HMs offers a great opportunity for aquatic and soil bioremediation. Being non-toxic, and biodegradable with the property to be produced from renewable substrates, BioS can offer great prospective use in bioremediation processes as a safe and cheap biotechnology. Recently, they have been extensively applied to enhance the bioremediation of xenobiotics and the bio-de-pollution of soils and aquatic environments. However, HM bioremediation is not limited to the use of bacterial strain or BioS to sequester metals separately. The isolation of bacterial strains having dual properties to capture HMs and produce BioS offers great interest. The *in situ* BioS production avoids the long procedure of purification. Also, genetic engineering involving recombinant microorganisms producing BioS can be applied to perform and sequester HMs. Additionally, the elaboration of bioreactors based on immobilized strains offers great advantages. It markedly enhanced the stability of the enzyme and the reduction of HMs. Moreover, immobilization of BioS-producing strain offers a continuous system for HM removal.

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ETHICAL APPROVAL

All procedures performed in studies involving animals were in accordance with the ethical standards of the institution or practice at which the studies were conducted.

CONSENT TO PARTICIPATE

Informed consent was obtained from all individual participants included in the study.

CONSENT TO PUBLISH

All the authors gave the publisher the permission to publish the work.

AUTHOR CONTRIBUTIONS

M.I. elaborated on the plan and wrote this review paper. S.M. helped in the redaction of the paper. G.D. corrected and approved the submission of the review.

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DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

CONFLICT OF INTEREST

The authors declare there is no conflict.

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