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Can bioleaching of critical metals from secondary resources be a sustainable extraction solution?

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ABSTRACT

The demand for critical metals (CMs) has significantly increased due to their indispensable roles in advanced manufacturing. Bioleaching, when coupled with secondary resources, offers a promising alternative that addresses both environmental and economic concerns. This review aims to highlight the potential of environmentally friendly bioleaching technologies and the use of secondary resources for the sustainable extraction of CMs. This review covers: (i) characterization of low-grade waste resources (LGWR) for potential CMs recovery, (ii) current progress in bioleaching of CMs extraction, (iii) a strength-weakness-opportunity-threat (SWOT) analysis of bioleaching for LGWR; (iv) relevant market trends, and (v) a case study on bioleaching of CMs from LGWR. Comprehensive data analysis from 102 peer-reviewed papers indicates that rare earth elements are notably present in red mud (0.13%) and tailings (0.11%). Mutant and mixed cultures were found to be crucial for scaling up due to higher recovery efficiencies with faster processing, addressing industrial demand for time- and cost-efficient solutions. Moreover, SWOT analysis highlighted infrastructure incompatibility and capital cost inertia as key threats, whereas opportunities were achieved around process integration. Ultimately, red mud was found to be a strategic waste in future supply chains due to its high CM content, indicating the direction of future research to meet the demands.

Introduction

Critical metals (CMs), as defined by the International Energy Agency (IEA) and the U.S. Department of Energy, are essential elements for energy transition and technological innovation [1,2]. The typical CMs include Cu, Co, Li, Ni, and rare earth elements (REEs), which are widely used in the development of low-carbon emission and advanced technologies, including automotive catalytic converters, hybrid electric vehicle batteries, computer hard disks, smartphones, petroleum-refining catalysts, permanent magnets, and superconductors [3–5]. Therefore, the demand for CMs, specifically REEs, is estimated to grow three to seven times by 2040 [6]. For instance, in the case of Cu, its market value is projected to increase by USD 88 billion [7]. However, current supply capacities cannot keep up with the growing reliance on CMs. This is mainly due to limited resources and geopolitical risks [8–10]. Li and Cu are especially susceptible to supply constraints, in contrast to Co, Ni, and REEs, whose availability is more strongly shaped by geopolitical instability (Fig. 1). The impact of these risks is evident in the case of Li, which has the highest risk score among CMs.

In this sector, production has traditionally depended on extracting CMs from primary geological deposits, mainly through ore mining. Such dependence, however, comes at a considerable cost: it generates high greenhouse gas emissions, produces vast amounts of tailings, limits access to high-grade ores, and discharges toxic compounds into wastewater. Thus, shifting toward more sustainable alternatives is imperative.

In response, attention is shifting from primary ores to secondary resources; LGWR has emerged as a promising option. These include end-of-life devices (like e-waste), mining by-products (e.g., residues, tailings, and slags), and even wastewater sludge, are now recognized as viable secondary resources of CMs [11,12]. These resources are considered untapped. They can contribute to metal recovery, sometimes without the need for new infrastructure. However, extracting CMs from LGWR remains technically challenging. It is also economically unfeasible when using conventional methods [13–15]. Bioleaching has attracted increasing attention as a biological technique because it combines cost advantages with environmental compatibility and operational flexibility [16]. The historical implementation of bioleaching began in the 1960 s with projects in Utah and New Mexico, followed by projects in China and

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Uganda [17]. These efforts demonstrated their technical feasibility and applicability for LGWR. Nevertheless, it has so far seen only limited adoption, as persistent technical constraints and economic uncertainties impede broader uptake [17].

The overarching aim of this review is to provide a comprehensive technical and strategic evaluation of bioleaching for recovering CMs from LGWR. The specific objectives are to: (i) characterize the diversity of LGWR, (ii) review current advances in bioleaching techniques for LGWR, (iii) develop a corresponding strength–weakness–opportunity–threat (SWOT) analysis, (iv) analyse market trends, and (v) discuss a case study of bioleaching in practice.

Low-grade waste resources of critical metals

LGWR of CMs originates from diverse industrial activities and can be categorized into: (i) mining waste and (ii) other solid wastes. Mining waste includes materials generated at different stages of mining, such as exploration, extraction, and closure. Common types are tailings, slags, waste rock, and acid mine drainage (AMD). Other solid wastes include coal combustion products, phosphogypsum (PG), and wastewater sludge from industrial and domestic activities.

Mining waste

Mine tailings

Mine tailings are the finely ground residue from the enrichment process and consist of a slurry with a high water content composed of fine-grained silt. In most cases, tailings are transported by pipeline and discharged into purpose-built storage facilities. Worldwide, more than 282 billion tons of tailings remain abandoned, and roughly 16 billion tons continue to be added annually [18]. The volume of tailings generated is closely tied to the ore grade. For example, Fe extraction from Fe-ore produces relatively small volumes of tailings, as the ore typically contains up to 50% Fe, a significantly higher grade compared to copper sulfide or Au-ores. In contrast, gold and silver mines usually operate at much lower grades (measured in grams per tonne), resulting in the generation of large volumes of tailings after extraction [19,20].

Tailings contain significant residual concentrations of metals such as Zn (up to 8000 mg/kg), Cu (up to 4700 mg/kg), Ni (up to 1500 mg/kg), Co (up to 750 mg/kg), and REEs (200–900 ppm) [20–27]. However, tailings are subject to a high variation in the mineral structures and characteristics. Mine tailings host metals in distinct mineral phases. Fe is commonly in pyrite, pyrrhotite, hematite, goethite, and jarosite [23,24]. Cu is mainly associated with chalcopyrite, bornite, chalcocite, covellite, and weathered carbonates such as malachite and azurite [23]. Zn typically occurs in sphalerite. Under oxidative conditions, it may also transform into smithsonite, hemimorphite, or soluble sulfates [24]. Ni and Co are present in pentlandite, cobaltite, carrollite, and gersdorffite [23], while REEs are retained in monazite, xenotime, bastnäsite, allanite, or adsorbed onto Fe/Mn oxides [25]. The role of processing history (e.g., flotation, roasting, cyanidation) and subsequent weathering is strongly reshaping these mineral associations by controlling their speciation and recoverability [24,28]. For example, roasting transforms chalcopyrite into Fe-oxides and sulfates that passivate metal release. Flotation leaves pyrite- and sphalerite-rich residues that weather into Fe/Mn oxides [23,24]. These transformations show that mineralogical context is critical. It is just as important as bulk metal grades when evaluating tailings as secondary resources.

Beyond processing history, environmental factors can also influence the extent and nature of geochemical variability in mine tailings. For instance, composite covers alter CM profiles (e.g., Al, Mg, Mn) by limiting sulfide oxidation and the associated release of metals [27]. They act mainly by suppressing oxygen ingress and rainfall infiltration, thereby reducing acidity and metal fluxes [29]. Field data show oxygen can fall below 0.01%, porewater levels drop, and oxidation weakens, consistent with reduced mobilization [27]. Well-sealed systems stabilize metals in Fe(III) oxides and hydroxysulfates, whereas defects allow seasonal re-oxidation [27,29]. Multilayer systems with tuned thickness, tight sealing, and reactive additives offer a pathway to controlled pH–redox regimes that release target metals for recovery.

The potential for metal extraction from tailings is closely tied to the mineralogical hosts and chemical speciation of CMs, which are primarily hosted in sulfide and oxide minerals (such as chalcopyrite, scheelite, sphalerite, and pyrrhotite) [21,24,27]. For instance, Zn and Cu occur in soluble sulfates and are weakly bound in outer-sphere complexes. Metals such as Co and Ni associate with Fe/Mn oxides or sulfides. These associations can be released under acidic or oxidative conditions, which depend on the specific characteristics of the host material. The presence

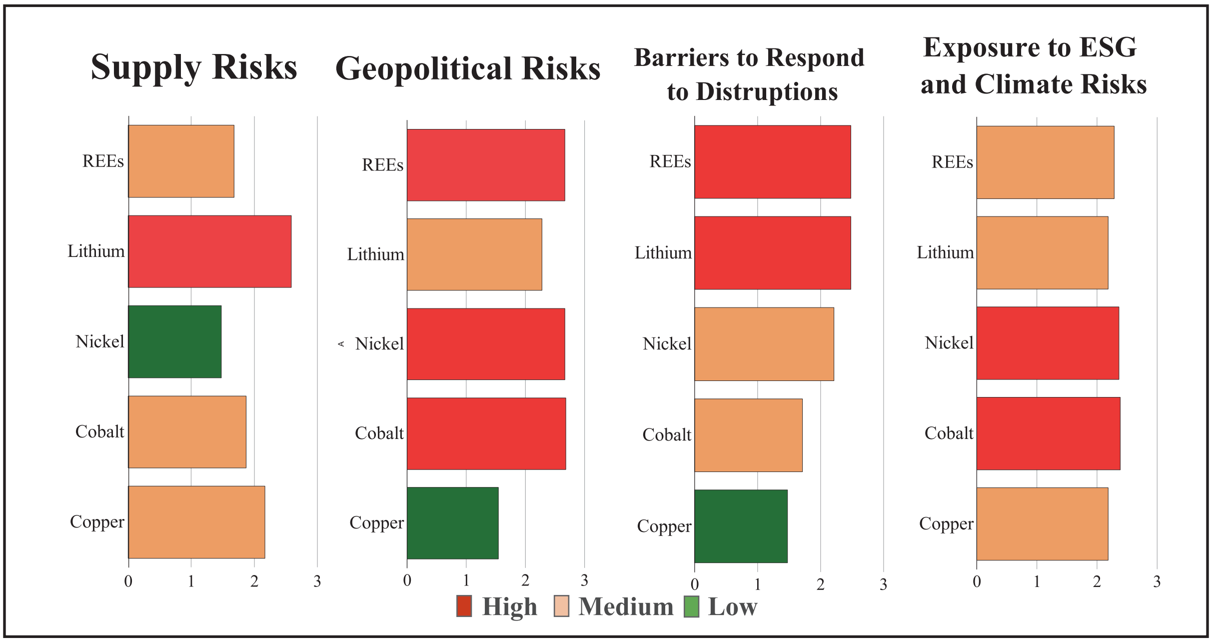


Fig. 1. Risk score of important CMs by category (scale from 0 to 3). Data are extracted and redrawn from [7]. ESG: Environmental, Social, and Governance.

of these metals in such phases, combined with partial liberation and oxidation, suggests they retain high residual value and are viable targets for reprocessing initiatives [20,23,25]. For instance, Zn and Cu occur not only as soluble sulfate species but also as surface-bound Me–SO_4 complexes on Fe phases [21,30]; ATR-IR bands at around 1223, 1142, and 1070 per cm indicate predominantly bidentate inner-sphere sulfate complexes in oxidation zones, whereas adjacent acidic waters show monodentate (weaker) binding consistent with more labile, outer-sphere association [21,23,26]. Organic ligands form stable carboxylate complexes after bioleaching that shift speciation toward soluble organic-metal forms, most notably copper–citrate and cobalt–oxalate [19]. Consistent with sequential extraction, Zn and Cu partition largely into exchangeable and reducible pools associated with Fe and Mn oxides (e.g., ferrihydrite, schwertmannite), while Ni and Co are preferentially retained in silicate and sulfide hosts (e.g., substitution in pyrite/pyrrhotite; Ni in gersdorffite) that confer lower inherent mobility [20,23]. Acidic or oxidative disturbances trigger the dissolution and recrystallization of Fe-oxyhydroxides and drive sulfide oxidation, processes that liberate associated metals. Passivation layers such as jarosite or elemental sulfur can also be destabilized, particularly when biochar amendments or Ag^+ act as catalysts. It is thereby accelerating fluxes from both sorbed pools and mineral-bound reservoirs [21,31]. Microtextural and phase mapping further refine host–metal pairings, with REEs occurring mainly in monazite and residual silicate matrices [23]. In addition, Principal Component Analysis patterns together with observations from oxide-rich deposits underscore the post-processing and oxidative concentration of Au, As, Cu, Ni, and Ag in Fe-oxides [20]. These mineralogical features translate directly into bioleaching performance and pathway selection.

It is worth noting that variable structures and characteristics in tailings also affect further metal extraction processes, especially bioleaching. This means that sulfide-rich residues favor iron- and sulfur-oxidizers with efficient ferric/acid attack [24], where carbonate-rich tailings show slower release and higher acid consumption [23]. Secondary phases (jarosite, Fe-oxyhydroxides) can immobilize or re-adsorb metals, reducing yields [23,24], toxic metals (such as As and Pb) may inhibit microbial growth and fungal systems [24]. In fact, *Acidithiobacillus* strains show markedly reduced activity at 600 mg/L As(III) and 200 mg/L Pb, with fungi such as *Aspergillus niger* tolerating higher than 1000 mg/L [32,33]. Acid type and substrate choice further dictate which metals are solubilized most efficiently [19]. For instance, citric acid dissolved up to 95% of Cu, while oxalic acid favored Co (80%). Fungal cultures using molasses or winery waste as substrates achieved up to 60% Co and 65% Zn solubilization, respectively [19]. Among all, red mud and coal refuse represent promising opportunities among other tailings for CM recovery, since they exhibit a more consistent profile with fewer CMs but substantially enriched concentrations (elements like Al, Ti, and REEs) [34,35].

Red mud

Red mud or bauxite residue is the by-product of alumina production, when treating bauxite ore with caustic soda [36]. It is highly alkaline, due to its high sodium hydroxide content, leading to significant concentrations of various metal oxides [37].

An assessment of red mud across multiple global studies reveals that maximum concentrations reported for elements include Al (up to 40 g/kg) [38], V (up to 730 mg/kg) [39], Y (up to 150 mg/kg) [39,40], Sc (up to 120 mg/kg) [35,41], and Ga (up to 80 mg/kg) [35]. Ti is also present at much higher levels, reaching up to 25 wt% as TiO_2. This reflects its major oxide status and its potential for industrial-scale recovery [39]. Furthermore, REEs are considerably enriched, with total ΣREEs reaching up to 1050 mg/kg and Ce alone reported at 408 mg/kg [40,42,43]. These data clearly highlight red mud as a rich secondary source of Al, Ti, and REEs. The metals are generally hosted within mineral matrices such as cancrinite, goethite, and Ti- or Fe-bearing phases [41,43], which makes them suitable for removal through leaching methods. Notably, Sc and REEs often occur through isomorphic substitution within these host minerals. This is a feature that can still be exploited through targeted extraction strategies, yet poses certain processing challenges [35].

However, despite its metal richness, red mud also presents significant challenges that complicate its direct utilization. For instance, the concentration of NaOH in red mud is typically reported in the range of 100–400 g/L [44,45]. This high alkalinity poses significant risks, such as handling challenges, if the material is not properly managed. The very high sodium hydroxide content poses a major barrier to bioleaching, since most efficient leaching microorganisms are acidophilic and fail to survive in strongly alkaline media [46]. High pH also promotes the precipitation of metal hydroxides, which limits solubilization and lowers recovery efficiency. To address these issues, pretreatment steps such as neutralization or the use of alkaliphilic microbial species (such as *Alkaliphilus metalliredigens* and *Bacillus vedderi*) have been considered to enable effective bioleaching of red mud [47,48].

Coal refuse

Coal refuse, including coal, clay, and other organic and inorganic material, is a by-product of coal mining and downstream processing activities. Several CMs in coal refuse have been reported at notably high concentrations, including Mn (up to 820 ppm), Sr (up to 415 ppm), Zn (up to 230 ppm), Ni and Li (up to 130 ppm), and Cu (up to 110 ppm) [49–51].

A direct correlation was observed between the metal content and ash content. This indicates the potential of high-ash-content coal refuse as feedstocks for CMs extraction [51]. Recent studies show that ash composition strongly influences bioleaching efficiency. High contents of silicates and aluminosilicates (e.g., quartz, mullite) hinder recovery because REEs remain locked in resistant Si-O-Al frameworks [52,53]. This typically corresponds to SiO_2 levels of 50–60% and Al_2O_3 of 30–40 wt%. In contrast, pyrite and other sulfides are highly favorable since microbial oxidation generates Fe^{3+} and H_2SO_4, enhancing REE dissolution [54,55]. Ashes rich in carbonates and sulfate phases (especially from circulating fluidized bed combustion, CFB) typically yield recoveries in the range of 41–63%. This is much higher than pulverized coal fly ashes (PCF), which are more quartz- and aluminosilicate-rich and show only 9–17% recovery [53].

Moreover, metal distribution is structured rather than random, being controlled by mineralogy, redox state, and secondary phase formation in coal refuse [49,50]. In high-ash coal refuse, most CMs are hosted in inorganic minerals rather than organic matter, consistent with the strong metal–ash correlation [56]. Clay minerals such as kaolinite, illite, and chlorite commonly accommodate Ga, Li, and some REEs either structurally or through adsorption [52]. REEs are mainly concentrated in phosphates and accessory minerals: monazite and apatite (LREEs), xenotime and zircon (HREEs), and secondary crandallite-group phases like gorceixite or florencite [52,57]. Both the weathering and redox fluctuations drive the transformation of sulfides into Fe oxyhydroxides and stimulate the precipitation of secondary REE-phosphates. These phases scavenge released metals and frequently become encapsulated in clay aggregates, which in turn affects their extractability [57]. A large proportion of CM content is sequestered in refractory phosphates and oxides, while the extractable portion is retained on clays and organic matter in adsorbed or exchangeable form. Effective leaching strategies are shaped by this distribution [57].

Given the high concentrations and mineral associations, the CMs are considered technically recoverable. With their high abundance and occurrence in pyritic, carbonate, or oxide forms, the metals are technically recoverable under proper processing conditions [51,58].

Slag

Slag is a by-product of pyrometallurgy, a high-temperature technique used to extract metals from ores. It typically forms as impurities in the ore react with flux during smelting and is broadly categorized into ferrous and non-ferrous types. This waste has attracted growing interest for recovery, particularly from Cu, Ni, and Sn smelting slags [59].

Ferrous slags, which are derived from Fe and steel production, have been studied as LGWR for the extraction of elements such as Nb, Ta, and REEs. Their inherently high pH contributes to elemental stabilization and potential selectivity in recovery processes [60]. Among non-ferrous slags, copper slag has shown promising results for metal extraction. For instance, it may contain Cu (1.2 wt%), Co (0.4 wt%), Ag (1.6 wt%), and Zn (1.8 wt%). This metal profile underscores its potential as a valuable resource for reprocessing [61].

Waste rock

Open-pit and underground mining operations produce considerable quantities of barren rock. These materials are commonly deposited in waste dumps, and when exposed to weathering, they are vulnerable to the onset of acid rock drainage (ARD). REEs have been identified in such drainage, along with other minerals associated with pegmatites, carbonatites, and peralkaline igneous rocks [62,63].

Although studies in this area are limited to a few sources, various waste rock deposits have shown different quantities of CMs. For example, analysis from Mount Bonnie (Canada) confirmed the presence of extractable CMs such as Zn (538 ppm) [64]. Additionally, relatively unaltered pyrite and elevated pyrrhotite contents, together with Si and Al, were identified across bulk samples and microstructures [65].

Acid rock drainage and acid mine drainage

Acid mine drainage (AMD) is widely recognized as one of mining's most destructive legacies. It originates in the oxidation of sulfide minerals. The process liberates protons and dissolved metals; for example, oxidation of Zn-bearing sulfides generates acidic discharges heavily loaded with metals [66]. This process also occurs naturally in ARD, where sulfide-bearing rocks undergo weathering, releasing acidity and metals [67]. However, in host rocks containing carbonates, acidity is counteracted through neutralization, whereby both the levels and mobility of metals in the drainage are altered [68]. Although these drainage sources are still considered LGWR for CMs such as Al, Mg, and REEs, the concentrations in mine drainage systems vary widely depending on geochemical conditions and mineral phases [69]. For instance, total REE concentrations range from 100 to 4300 µg/L in solution and up to 1900 mg/kg in solids, with the highest aqueous levels observed in Cu-AMD [70] and the highest solid-phase REE in Fe-rich residues [71], respectively. In contrast, coal-derived AMD provided moderate average REE concentrations (370 µg/L), with heavy REEs such as Y, Ce, Nd, and Sc forming the dominant fraction [70]. Peak concentrations of CMs in mine drainage systems have been reported as Co (3100 µg/L), Ni (3200 µg/L), and Cu (190 µg/L), with these values primarily observed in Mn- and Fe-rich AMD [71]. Overall, Mn- and Al/Fe-bearing treatment residues are key secondary reservoirs of CMs for AMD, with their distribution strongly influenced by mineral composition and the acid-neutralizing capacity of the host rock.

Other solid wastes

In addition to mining waste, other solid waste streams such as coal ash, PG, and wastewater sludge have been identified as potential LGWR for CMs. These materials originate from power generation, fertilizer production, and wastewater treatment processes.

Coal combustion byproducts

Coal combustion byproducts are residues generated in power plants during the burning of coal. In this process, components are transformed into newly formed phases, including fly ash (fine particles), furnace bottom ash (coarser particles), and slag [72]. The chemical composition of ash depends on both the type of original coal and the combustion conditions, and metals can sometimes become more concentrated in the ash than in the raw coal itself [73]. The main chemical constituents of both bottom ash and fly ash originate from non-combustible fractions and contain metal oxides such as Si and Al, along with minor amounts of metals like Co, Cu, Ni, and REEs [74]. The fly ash, which has finer particles, contains higher amounts of many CMs than bottom ash (often by a factor of 1.1 to 3) due to the elemental concentration in the fine particles and the involved processes [75,76]. For example, a global mean near 445 ppm was reported for REEs in fly ash [77]. Lithium is typically concentrated from 45 to 81 ppm in feed coal to 77–359 ppm in ash, about 1.5–5 times enrichment [73]. Vanadium can vary between 12 and 570 ppm, but exceptional cases reach more than 1000 ppm [75,78]. Average Ga concentrations have been reported at about 33 ppm globally, although values of 90–110 ppm have been measured in bituminous ash [73]. Concentrations of Co and Ni have also been recorded up to 79 ppm and 258 ppm, respectively [75].

The distribution and concentrations of CMs in coal ash vary substantially with the coal type and the operating conditions of combustion. More Ga, V, and REEs are present in the ash of bituminous coals than in that of lignite, whereas lignite ashes contain higher levels of Ca and Sr [78]. Combustion temperature also influences partitioning: at higher flame temperatures (PC boilers, 1200–1500 °C), volatile elements like Se preferentially enrich in fly ash, leaving bottom ash depleted. By contrast, fluidized bed combustion at 850–900 °C retains more metals in bottom ash due to reduced volatilization [73]. For this reason, coal combustion byproducts need to be carefully examined for their origin and systematically assessed for their resource potential.

Phosphogypsum

PG, which is a by-product generated during the production of phosphoric acid, is where phosphate rock is reacted with sulfuric acid. Despite lower concentrations compared to primary ores, PG has shown levels of Zn, Cu, Al, and REEs ranging from 4 to 2000 ppm [79].

Recent studies demonstrated significant geographical variability in elemental concentrations within PG, with REEs such as Y reaching up to 200 ppm, La up to 90 ppm, Nd up to 120 ppm, and Ce up to 2400 ppm. This shows notable enrichment depending on the phosphate rock source [80–82]. A challenge, however, is that REEs are often encapsulated within gypsum crystals [83]. In this context, three primary mechanisms governing the retention of REEs in PG: (1) isomorphous substitution for Ca^{2+} in the gypsum crystal lattice, (2) adsorption onto particle surfaces, and (3) precipitation as secondary oxide, phosphate, or sulfate phases [80,82]. Additionally, Fe and Mn oxides also act as host phases for other CMs such as Co and Zn. This supports a geochemical model in which PG retains these metals both structurally and physically [81]. The presence of fine-grained and highly porous gypsum crystals further enhances surface retention, particularly in aged PG, where natural leaching has reduced soluble impurities [84].

The geochemical heterogeneity and complex retention mechanisms of CMs in PG highlight the need for process-specific characterization to effectively address recovery obstacles. Approaches like mineralogical analyses (XRD, SEM-EDS) and sequential chemical extractions have been applied to overcome these obstacles. For instance, aggressive acid leaching with HCl or HNO_3 showed 30–50% higher efficiency than H_2SO_4 in REE leaching recovery. The underlying reason is largely because $CaSO_4 \cdot 2H_2O$ is more soluble in these acids, preventing the formation of insoluble REE double sulfates [85]. Besides, mechanical pretreatment via ball milling to increase surface area and defects, and ultrasonic-assisted leaching to reduce particle size and accelerate dissolution, were both shown to markedly enhance REE recovery from PG [79]. Likewise, recrystallization of gypsum phases enabled nearly complete REE release with simultaneous resin capture [86]. In contrast, chemical conversion routes, such as transforming PG into CaS and $CaCO_3$, concentrated REEs into more leachable residues (retaining over 60% of the original REE content) [87]. These studies highlight that effective recovery of REEs from PG is inseparable from a mechanistic understanding of their retention forms.

Wastewater sludge

CMs in urban wastewater originate from the daily use of detergents, cosmetics, personal care products, paints, and industrial chemicals, as well as mining and processing activities [88–90]. These contaminants enter the hydrosphere and concentrate in wastewater and sludge [91]. Studies have reported concentrations of CMs in sewage sludge and industrial wastewater, such as Mg (2.3%), Mn (0.1%), Cu (0.02%), Zn (0.08%), and REEs ranging from 0.02 to 41.3 mg/kg dry matter [12,92,93]. Although the concentrations in wastewater are low (μg/L levels), large-scale wastewater production and the enrichment of metals in sludge underscore the importance of recovery initiatives [94]. Countries such as the Netherlands and Belgium are actively testing approaches to recover metals from WWTP outflows [89]. In particular, the use of sludge for resource recovery aligns with the broader shift of treatment plants toward circular economy practices [95].

Comparative assessment of CMs distributions across LGWR

A broader comparison across LGWR can provide a clearer perspective on the concentration patterns of CMs and help identify enriched fractions within each stream. Therefore, Fig. 2 summarizes the distribution of CMs across various types of LGWR. Overall, the concentrations of critical metals show a wide range of variation. Fig. 2A and Fig. 2B compare the content of 12 CMs, indicating that CMs such as Al are the most abundant element in the LGWR, ranging from 8% to 13%. In turn, Ga (<0.01%), Co (<0.6%), and Ni (<0.3%) are among the least abundant. AMD has the lowest metal content compared to other waste types (<0.1%). Red mud shows the highest Al content (about 13%) due to its origin from Al production, which also makes it the most likely choice for high-yield Al recovery. Coal ash and coal refuse also contain notable Al levels (9.07% and 8.7%, respectively). However, in practice, the metal content of each waste type is distinct and may directly influence the

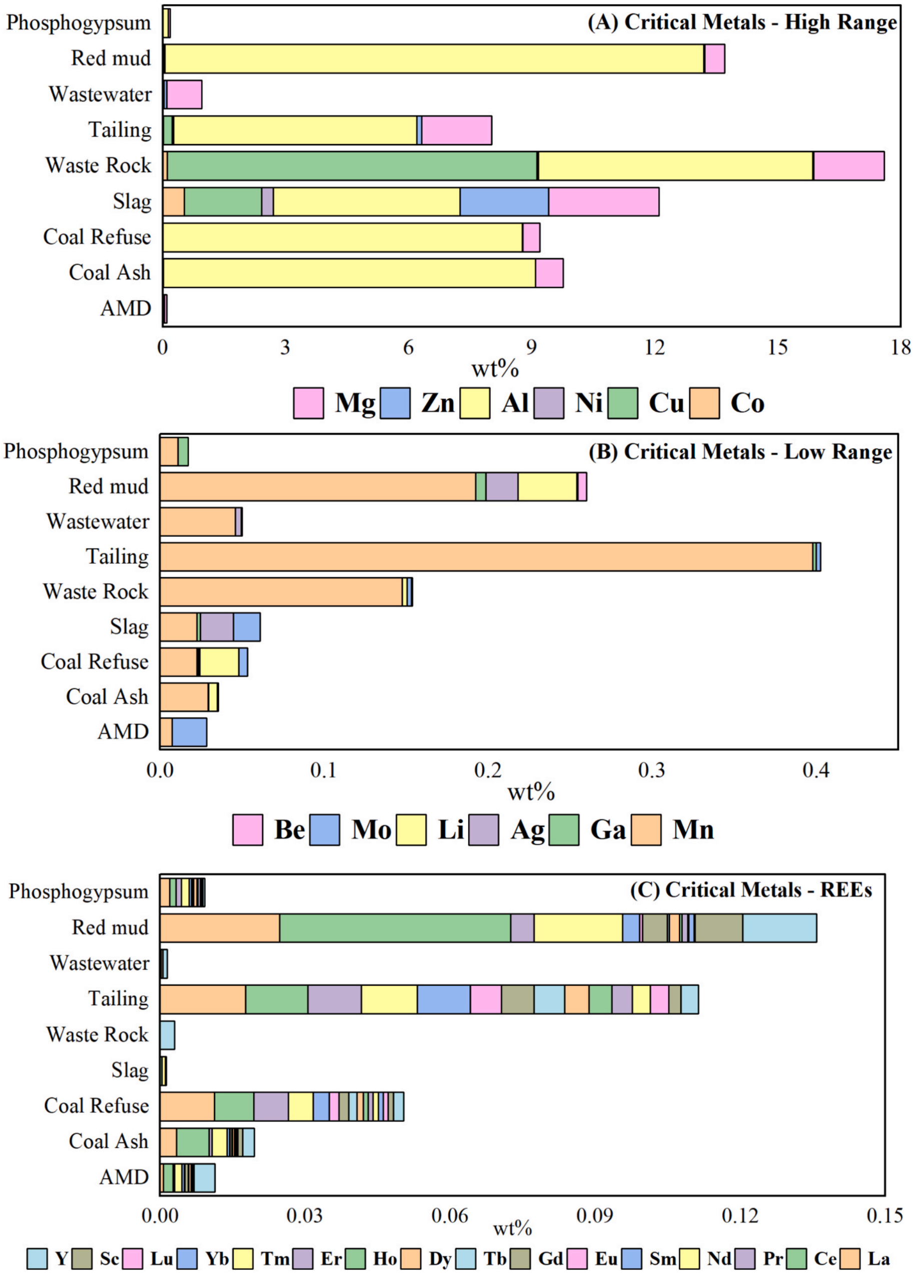


Fig. 2. Metal weight percentage (wt%) in different LGWR: CMs – High Range (>1%) (Panel A), CMs – Low Range (<1%) (Panel B), Rare Earth Elements (Panel C). Data are collected from 102 peer-reviewed scientific publications (Appendix A). The range of variation for each element is provided in the Appendix B.

outcomes of the analyses. More detailed charts for individual waste types are provided in Appendix B.

At lower concentration ranges, Mn exhibits relatively higher abundance, ranging approximately from 0.01% to 0.4%. Following that, Li shows notable relative abundances in red mud and coal refuse (0.03% and 0.02%). Slag contains high concentrations of several metals, notably Zn (2.1%), Mg (2.7%), Cu (1.8%), Co (0.5%), and Ni (0.28%). Thses values make it potentially valuable for CMs recovery. Waste rock is particularly rich in Cu, with values reaching 9%. However, significant data gaps persist for several CMs, especially Li, Pt, and Be, as even recent studies rarely provide detailed or quantitative measurements for these elements [20,24,96–98]. Quantifying these metals with accuracy is hindered by analytical challenges. Lithium suffers from matrix interferences, which demand ion-exchange purification [99]. Pt occurring only at ultra-trace levels is further complicated by nugget effects, so NiS fire-assay pre-concentration is typically required [100–102]. Be is strongly bound in silicates, demanding HF digestion and boric-acid complexation to prevent fluoride losses [103]. Future research can focus on optimizing purification protocols, pre-concentration techniques, and advanced digestion strategies so that more robust datasets can be obtained. For Li, improved cation-exchange chromatography yields 100% recovery while suppressing Na/Ca interferences to negligible levels [104]. Ultra-trace Pt benefits from NiS fire-assay pre-concentration, which enriches Pt by more than 1000 times. It also enables detection limits of 0.1 ng/g with 86–98% recoveries [105]. Be bound in silicates requires microwave-assisted HF–HNO_3 digestion with boric acid, achieving 86–97% recovery compared to less than 5% without fluoride complexation [103]. Such efforts would provide statistically robust concentration ranges and improve comparability across studies. The limitation is also evident in waste types, including red mud, slag, and waste rock (as mentioned earlier) [35,40,106,107].

The distribution of REEs is presented in Fig. 2C. Notable concentrations of Ce, La, and Nd are present in red mud, coal ash, coal refuse, and wastewater. Tailings also show significant levels of many REEs, particularly light REEs. The lowest REE content belonged to AMD, except for Y, which has a relatively high concentration (0.004%). However, the cost of Y in the market is relatively low compared to other REEs such as Sc, Nd, and Pr [108]. Overall, these trends support the role of red mud and tailings as promising REE sources. It also highlights that having a prescreening step to select a specific waste that contains high-value REE is important. At this stage, the absence of data for many REEs in waste types like slag and waste rock might pose challenges for an effective screening step and hence emphasize the need for further characterization of REEs.

Current advances in bioleaching for CM recovery

In the bioleaching process, the acidophile microorganisms (e.g., bacteria, fungi) produce acidic biolixiviants that initiate the breakdown of solid matrices. The leaching mechanisms involve several biochemical and microbial pathways, including acidolysis, complexolysis, and redoxolysis, that collectively facilitate the dissolution and mobilization of CMs. Once in solution, additional processes such as enzymatic reduction, biosorption/bioaccumulation, and bioprecipitation can further recover or stabilize the metals. Understanding how these processes function, both independently and sequentially, is essential for developing effective recovery strategies. The specific mechanisms of bioleaching and the ways to optimize them for CM recovery are summarized in Fig. S1 and Eqs. 1–8 in Appendix C. This section discusses the current advances of bioleaching associated with four primary organism groups: (i) bacteria, (ii) fungi, (iii) mixed cultures, and (iv) mutant strains. The selection among these microbial groups for bioleaching primarily depends on balancing key environmental parameters such as LGWR composition, pH, temperature, and metal tolerance. Different microbial groups employed in the extraction of CMs are reviewed in the following sections, with a particular focus on the role of these environmental parameters.

Bacterial bioleaching

Bacterial bioleaching is primarily carried out by Fe- and S-oxidizing bacteria, notably *A. ferrooxidans* and *A. thiooxidans* [109]. These strains grow in extreme environments, such as low pH and high metal concentrations. The *Acidithiobacillus* species have been applied to recover a range of CMs. For example, *A. thiooxidans* demonstrated effective bioleaching performance of 100% Co, 44% Mo, 70% REEs, and 70% V from copper slag under optimized particle size and pulp density conditions [110]. Likewise, *A. ferrooxidans* increased copper tailings recovery by 13.6%, 23.7%, and 11.0% when ferrous sulfate, sodium thiosulfate, and pyrite were added as energy sources for oxidation reactions [111].

To better understand how these bacteria interact with solid matrices, two major leaching mechanisms are typically described: (i) contact and (ii) non-contact leaching, based on how the bacteria interact with the substrate. In the contact process, bacterial cells are cultured directly intact with the LGWR, facilitating metal extraction through localized biochemical reactions. Regarding non-contact leaching, the redox reactions, such as sulfur oxidation and ferric ion reduction, are the main driving forces that occur in the surrounding solution without direct cell-surface interaction. Both direct and indirect oxidation reactions are employed depending on sulfur content and matrix composition.

Among aerobic sulfur-oxidizers, *A. thiooxidans* and *Gluconobacter oxydans* have shown strong performance. After 14–20 days, G. oxydans achieved REE recovery in the range of 36% (Ce, Nd) to 94% (Sc), which highlights its potential for efficient extraction of REEs [112,113]. However, environmental factors (temperature, pH, and inhibitors) can impact the efficiency [114]. To address these variations, a 2% pulp density has been used for REE bioleaching from PG using *A. thiooxidans* that achieved over 60% recovery [115]. In addition, a typically lower pH, between 1.5 and 3, enhanced REE solubility from PG [116]. Despite these improvements, further optimization is still required for large-scale feasibility. In particular, microbial activity must be optimized and overall recovery efficiency enhanced to make the process economically viable. For example, although sulfur oxidation using *Sulfobacillus thermosulfidooxidans* enabled recovery of Al, Y, and Ce from red mud [117], the strain performed suboptimally below pH 4.5. Nevertheless, some progress has already been made in this regard. For example, a two-step bioleaching strategy (a 5-day pre-culture of *Acidiphilium cryptum* LT followed by inoculation with *Acidithiobacillus ferrooxidans* CL) has shown promising results (over 80% recovery of La, Nd, and Y) for large-scale applications under optimized condition (1.0% (w/v) pulp density, 2% (v/v) inoculum, 0.5% (w/v) glucose, and operational pH of 1.95–2.04) [118]. Similarities (acidic operation at pH = 2; acidophilic consortia relying on pH drop) and differences (pulp density 1% vs 10–20%; temperature 30 °C vs 40–45 °C; residence time 30 days batch vs a few days in multi-stage CSTRs; minimal control vs forced aeration with active pH and DO) between the lab process and industrial bioleaching indicate that, the following need to be implemented to industrialize REE bioleaching for PG: raise solids loading and oxygen transfer; implement active pH and DO control; and shift to continuous multi-stage tanks to cut residence time (even if extraction % drops slightly) [119–121].

In addition to environmental factors, certain bacterial strains also facilitate metal solubilization by conditioning their surrounding environment through the secretion of metabolites. For example, *Bacillus nitratireducens* has demonstrated high extraction efficiencies, yielding over 80% for Lu, 80.6% for Tb, and up to 46% for Sc, through the secretion of intracellular metabolites such as organic acids and biosurfactants, which effectively mobilize REEs from recalcitrant mineral matrices [122,123]. This advantage is particularly evident under controlled conditions of the bacterial pure culture, where growth and metabolism can be more precisely regulated and the targeted production of leaching agents is maximized. The mechanism of action of organic acids is such that protons are donated to acidify the environment and chelate metal ions. The bonds in the mineral lattice are weakened by this process, and the solid phase is dissolved as a result [123]. The acids also form stable complexes with released metal ions, keeping them soluble even under moderately alkaline conditions. Biosurfactants (such as lipopeptides from *Bacillus* spp.), however, use a different mechanism. They enhance metal recovery by increasing the wetting of mineral surfaces and directly complexing with or chelating metal ions [124]. Heterotrophic bacteria, such as *Sphingomonas* spp., are able to deploy key metabolic enzymes which contribute to the secretion of organic acids and surface-active compounds are PQQ-dependent glucose dehydrogenase, oxaloacetase, lactate dehydrogenase, and non-ribosomal peptide synthetases responsible for biosurfactants [31,125–127].

Fungal bioleaching

Fungi produce organic acids (e.g., oxalic and gluconic) that facilitate metal mobilization through ligand-mediated complexation on metal surfaces. To better contextualize these fungal leaching strategies, the underlying mechanisms of metal mobilization are illustrated in Fig. S1 and further detailed in Appendix C. Depending on the leachate production stage, leaching by fungi can occur in one- or two-stage fermentation processes. In a one-stage process, fungi directly contact tailings, with pH declining to 3.5–4. This approach is simple but often less efficient in metal recovery because the fungi are immediately exposed to metal toxicity or high pH conditions, which can inhibit their growth and performance [16]. In two-stage systems, acids are produced first in fermentation broth, and then applied to the tailings. This sequential approach typically yields higher metal recovery efficiency, since the leaching agents are produced in abundance without stressing the microorganisms [16]. Two-stage bioleaching can also offer scalability advantages. For example, the leachate can be used on larger quantities of material (higher pulp densities), and each phase can be optimized independently. However, it involves additional operational steps and equipment (making it more complex and potentially costlier than one-stage processing). Bayraktar (2005) used a two-stage process to incubate *A. niger* for Ni bioleaching, which resulted in an improved version of this method. The positive results were attributed to a combination of the secretion of citric acid and the biosorption capacity of *A. niger* mycelia [128]. Ilyas et al. (2012) studied the potential of *Penicillium chrysogenum* in the production of a variety of organic acids (citric, oxalic, tartaric, and malic acid) to solubilize Ni, Cu, Mg, Co, and Zn from a mine tailing. Cultivation of the fungus on a medium containing glucose, tea leaves, molasses, and waste winery grapes as substrates resulted in maximum solubilization of Ni (55%), Cu (67%), Mg (69%), Co (60%), and Zn (65%) [19].

The acid type and acidity of fungi influence leaching efficiency due to different functional groups. For instance, Mn formed complexes with fungal acids. This enabled 58–74% Mn and 40–67% Ag recovery from tailings in Coahuila, México, using *Cladosporium* and *Penicillium* isolates [129]. Metal complexes can also be formed by metal chelating agents (like oxalate and citrate) that can be provided by heterotrophic metabolism.

The production of organic acids by fungi is highly dependent on the growth conditions. A readily metabolizable carbon source at high concentration (on the order of 100–150 g/L glucose or sucrose) is essential to fuel acid synthesis [130]. The carbon source and environment tune the acid spectrum: glucose commonly supports citric and gluconic acid, while oxalic acid is favored by excess carbon, suitable Ca availability, and acidic pH, reflecting its role in metal complexation/precipitation [131,132]. Citric acid often dominates and can exceed 600 mM, gluconic acid may reach more than 500 mM, while oxalic acid can accumulate up to 400 mM (driving pH range of 2–3) [131]. The initial pH of the medium is typically adjusted to neutral or mildly acidic, which allows robust fungal growth before the medium acidifies; for instance, one study found neutral pH optimal for *Aspergillus* to acidify the medium to pH 3–4 over time [133]. Temperature is maintained around 28–30 °C, near the optimum for filamentous fungi enzyme activity [134]. To ensure oxygen supply for aerobic organic acid production, adequate aeration and agitation over an incubation period of about 1–2 weeks is used.

Bioleaching with mixed culture

Mixed microbial cultures offer distinct advantages in bioleaching applications. Their metabolic diversity strengthens resilience to environmental fluctuations. At the same time, synergistic interactions between species (such as iron-oxidizers and sulfur-oxidizers) increase metal solubilization from tailings. Additionally, the use of mixed cultures removes the need for isolating and enriching single microbial strains, which is often time-consuming and resource-intensive [135]. For example, in Cd leaching, a mixed culture achieved 9% higher efficiency than single strains [136]. In a study on Cu flotation tailings, bioleaching with mixed moderate thermophiles outperformed traditional acid leaching by 13.2%. The process utilized *L. ferriphilum* and *A. caldus*, both Fe- and S-oxidizing bacteria, to enhance Cu sulfide dissolution [137]. A mixed acidophilic culture, mainly containing *Acidithiobacillus*, *Leptospirillum*, *Sulfobacillus,* and some *Alicyclobacillus* species, was able to extract 90% Ni, 60% Co, 86% Zn, and 67% Fe from low-grade flotation tailings [138].

Quorum sensing (QS) also helps facilitate metabolite exchange and enzyme secretion. This further supports bioleaching efficiency [139,140]. In *A. ferrooxidans*, the AfeI/R QS system upregulates genes involved in sulfur oxidation and extracellular compound production. It enhances both leaching metabolite generation and biofilm stability [141]. QS signals such as autoinducer-2 mediate cross-species metabolite exchange, supporting synergistic interactions in the mixed cultures [142]. Studies also show that QS activation increases secretion of extracellular polymeric substances (ranging from about 20% to over 60%) [143], and proteins essential for mineral attack, thereby boosting overall leaching performance, including enzymes and structural proteins involved in EPS biosynthesis and export such as glycosyl transferase (AFE_0233), polysaccharide export protein Wza (AFE_1339), and glucose-6-phosphate dehydrogenase (zwf) [139,144].

The composition of the mixed culture influences the efficiency. It also affects the rate of bioleaching. A comparative study on Zn bioleaching from Pb–Zn tailings showed that a culture with *A. ferrooxidans* and *S. thermosulfidooxidans* achieved 94.0% higher bioleaching efficiency and a 30% shorter processing time than a culture of *L. ferriphilum* and *A. thiooxidans* [31]. These findings emphasize the importance of engineering effective microbial consortia for optimal bioleaching outcomes.

For industrial applications, mixed microbial consortia can be implemented in heap leaching and stirred-tank reactors [145]. Pilot column or tank studies are typically used first to optimize experimental conditions. After this step, the cultures are gradually expanded to industrial heaps or tanks. Continuous solution recirculation is then applied to maintain a high microbial density. Heap operations rely on targeted inoculation and irrigation systems to ensure even microbial colonization, while tank systems require improved aeration and agitation designs to sustain dense communities at scale [146,147]. Past industrial experience with the BIOX process and copper/nickel heap leaching demonstrates that mixed cultures of *Acidithiobacillus* and *Leptospirillum* can thrive without sterile conditions, underscoring their scalability and robustness [147,148].

Maintaining uniform environmental conditions in industrial applications is a major difficulty. In heaps or tanks, gradients in pH, redox potential, temperature, and oxygen can create inactive dead zones, reducing kinetics [146,149]. For refractory ores such as chalcopyrite, controlling redox potential is critical to prevent passivation layers that halt dissolution [150,151]. Community stability is another challenge: mixed consortia must compete with indigenous microbes, and shifts in population structure can reduce performance, requiring monitoring and occasional reinoculation [147,152]. For example, in chalcopyrite heaps, mesophiles are gradually replaced by thermophiles as temperatures rise. This phenomenon often creates a gap in microbial activity around 50–60 °C that necessitates secondary inoculation [147]. Likewise, the growth of key iron oxidizers, such as *Leptospirillum ferriphilum,* is suppressed by high sulfate concentrations in heaps [152].

From an engineering and economic standpoint, issues include mass-transfer and gas-transfer limits, equipment corrosion in acidic slurries, the need for producing large inocula, and the inherently slower kinetics of bioleaching compared to chemical leaching [147,149]. Overcoming these constraints will require optimized reactor and heap design, predictive control systems, and demonstration that mixed cultures can deliver reliable, cost-effective recovery at an industrial scale.

Bioleaching with mutants

Mutant strains are engineered to enhance specific traits that improve microbial performance in industrial bioleaching. Key improvements include improved microbial stability, increased production of acids or metabolites (like polysaccharides, proteins), and faster leaching rates [153,154].

Mutagenesis techniques include chemical, plasma, ultraviolet (UV), and microwave treatments, with effectiveness depending on specific operating parameters. In chemical mutagenesis, the mutagenic agents, such as diethylsulfate (DES) and hydroxylamine, are used under controlled exposure times to induce a wide range of point mutations with straightforward, well-established protocols. DES-induced *A. ferrooxidans* mutants improved phosphorus leaching by 85.8% from rock phosphate containing pyrite [155]. Other examples include *P. polymyxa*, *A. tumefaciens*, and *Bacillus mucilaginosus*, which showed improved acid production and reduced leaching time following chemical treatment [156,157]. Chemically treated *A. ferrooxidans* also achieved a 20.7% increase in Cu recovery from low-grade tailings over wild-type strains, and 85% higher than acid leaching, with a 32-hour reduction in processing time [158]. This method also has limitations. Since changes are random, dose/time must be tightly controlled and large libraries screened [159]. Moreover, reagents are hazardous and require strict safety [160].

Plasma-based mutagenesis, using high-voltage pulses at atmospheric pressure, damages DNA and triggers repair mechanisms that can improve genetic traits [161]. Wu et al. (2019) found that plasma DNA mutation enhanced inhibitor tolerance and organic acid production of *Enterobacter cloacae* [162]. This approach shows potential in developing microbial strains to improve bioleaching efficiency for mineral extraction and recovery. However, it requires specialized equipment and parameter tuning. In addition, initial kill rates are high, meaning that multi-generation stability checks are important [160].

UV radiation is another tool for inducing mutations by adjusting power, distance, and exposure time. UV-mutated *A. ferrooxidans* showed a 17% increase in Cu extraction from copper-bearing tailings [159], while UV-treated *A. ferrooxidans* and *Acidiphilium cryptum* mutants raised Cu concentrations up to 0.38 g/L [163]. Although this method is a fast, reagent-free method, limited light penetration and high lethality at excessive doses mean careful calibration and follow-up screening are essential [159].

Microwave irradiation can also cause genetic changes. A 60-second exposure improved Cu leaching rates of *A. ferrooxidans* by 20% and reduced processing time by 18 h [164]. Although this method is extremely rapid, the effects are largely thermal and oxidative, with a narrow window between beneficial and lethal. In addition, reproducibility is sensitive to a combination of power and time, and the evidence base is thinner than that of other methods.

Combining multiple mutagenesis methods can yield better results. Simultaneous plasma and hydroxylamine treatment produced a *Bacillus mucilaginosus* mutant with a 5.5% increase in V extraction [165]. Repeated treatments may further enhance results. However, future work must carefully optimize mutagenic conditions. This is necessary to achieve reliable and reproducible outcomes.

To better illustrate the strengths and weaknesses of the major mutagenesis techniques, their advantages and limitations are presented in Table 1. Each technique induces random genetic mutations in microorganisms, but they differ in practicality, mutation rate, and outcomes. Traditional methods (chemical and UV) have been widely used in strain improvement despite challenges like safety concerns and relatively lower mutation rates.

Compared with their wild type, engineered mutant microbes maintain high activity under stressful conditions. They tolerate fluctuations in pH, temperature, and toxic metal concentrations. For instance, plasma-induced mutation produced an *Enterobacter cloacae* strain with double the inhibitor tolerance of its wild-type parent, allowing sustained acid production in the presence of fermentation toxins [162]. An example is an engineered arsenic-resistant *Acidithiobacillus* variant that can grow in 45 mM As(V), a concentration far above what unmodified strains tolerate [166]. In bioleaching, such enhanced resilience means mutant consortia offer more stable metal recovery performance [167].

Comparative evaluation of microbial strategies for CM recovery

Selecting an appropriate microbial group for CM recovery from LGWR requires a balanced perspective of their advantages and limitations. Table 2 summarizes the main strengths and weaknesses of each microbial approach.

Each microbial group contributes distinct advantages and faces specific constraints in the bioleaching of LGWR for CM recovery. Bacteria are well-established for critical metal recovery because they thrive in extremely acidic environments and oxidize sulfides to liberate metals [168]. Advantages include robustness under harsh conditions, low contamination risk (autotrophic growth), and proven scalability in copper and gold operations [169]. However, bacterial leaching is often slow, and certain refractory metals (e.g. Mo) respond poorly without a very high redox potential [170]. Inhibition by high metal concentrations also remains a constraint.

Table 1
Comparison of Mutagenesis Techniques.

Mutagenesis Method	Advantages	Limitations
Chemical Mutagenesis	- Induces a broad range of point mutations - Relatively simple to implement with established protocols - Proven in bioleaching	- Hazardous reagents - Mutations are random and often deleterious - Results can vary between experiments.
UV Irradiation	- Fast and widely used physical mutagenesis method. - Effective for strain improvement in bioleaching: - Simple setup and short treatment times	- Limited penetration - High lethality at excessive doses. - Like chemical mutagenesis, UV generates random mutations.
Plasma Mutagenesis	- Very high mutation rates and diversity of mutations. - Rapid process - Proven efficacy in bioleaching-related traits	- Specialized equipment required - Tends to cause high initial cell death - Newer technique with limited long-term track record.
Microwave Irradiation	- Extremely rapid mutagenesis - Notable bioleaching improvements from microwave-induced mutants. - No chemical reagents are needed	- Primarily induces thermal and oxidative stress rather than specific targeted mutations. - Tight process control - Less established than UV or chemical methods.

Table 2

A summary of positive and negative points of bioleaching for low-grade tailings.

Microbial Group	Positive Points	Negative Points
Bacteria	• Proven at industrial scale • Efficient sulfide oxidizers; thrive in extreme acidity • Robust under harsh conditions • Autotrophic; low contamination risk; no sterilization needed	• Slow kinetics • Require extreme conditions • Poor on refractory metals • Risk of jarosite precipitates. • Activity inhibited by high dissolved metal loads
Fungi	• Produce organic acids for leaching • Operate at milder pH; biodegradable lixiviants • Preserve crystalline structures; ideal for oxide-rich wastes • Tolerant to toxic substrates • Faster solubilization possible	• Require organic nutrients; higher costs & contamination risk • Need sterilization and strict growth conditions • Limited effectiveness on sulfide ores • Two-step processes add complexity. • Scale-up still unproven
Mixed Cultures	• Synergistic action: iron + sulfur oxidizers boost solubilization • Higher rates than pure cultures • Functional diversity (adaptability) • Greater tolerance to toxic metals • Suitable for complex waste streams with multiple mineral types	• Complex control; different species need different conditions • Risk of one species dominating others • By-products may inhibit community • Hard to maintain stability at scale • Reproducibility issues in inoculum ratios
Mutants	• Adapted mutants tolerate higher metal toxicity • Engineered *G. oxydans* improves acid output and REE recovery Faster kinetics and higher yields possible • Tailored selectivity via synthetic biology • Promising for next-gen bioleaching	• Still lab-scale; no commercial field use • Regulatory and biosafety concerns • Risk of genetic instability under stress • Some engineered strains may require costly nutrients or grow slower • Handling and inoculum maintenance more demanding

Compared to bacteria, fungi offer faster leaching rates, though cultivation is more demanding (more complex and stricter growth conditions) and typically requires organic nutrients that raise contamination risk [171]. Indeed, organic acids are secreted by fungi that not only solubilize metals under milder pH, but also help preserve crystalline structures [172]. Species such as *Aspergillus*, *Penicillium*, *Fusarium*, and *Hypocrea* exhibit strong metal tolerance and adaptability, particularly in acidic copper tailings [173]. *Yarrowia lipolytica* has also been shown to utilize crude sunflower oil to generate biolixiviants and recover REEs from electric arc furnace dust in a green bioleaching approach [174]. Nevertheless, fungi are generally less effective on sulfide matrices. In addition, large-scale deployment remains less proven than for bacterial systems [175]. Overall, fungal bioleaching is most efficient when applied to oxide-rich or complex matrices, as long as cultivation barriers are effectively addressed.

Mixed consortia exploit functional complementarity, for instance, pairing iron-oxidizers with sulfur-oxidizers to boost metal dissolution [176]. Such cultures often show enhanced tolerance to toxic metals and greater adaptability [147,177]. They are especially relevant for complex critical-metal wastes where different fractions require different mechanisms. However, since each organism has distinct requirements, mixed systems are harder to control. In addition, population imbalances can reduce efficiency. Scaling up consistently is also difficult, but the potential for higher yields makes this an active research area [178].

Adapted mutants obtained by repeated exposure can tolerate high concentrations of toxic metals. They sustain leaching where wild types fail [179]. Engineered strains like modified *Gluconobacter oxydans* have shown markedly improved acid production and REE recovery [168]. Such approaches promise faster kinetics and higher selectivity for CMs. Still, no commercial operations yet employ engineered strains, largely due to biosafety and regulatory issues [180]. Genetic stability and robustness in real environments remain challenges, but advances suggest strong future potential. Yet, despite these promising footpath, few studies have integrated strain engineering with system-level process optimization. This presents a significant research opportunity to connect successful lab-scale outcomes with the implementation of bioleaching on an industrial scale.

Performance assessment of microorganisms in bioleaching

Numerous studies have explored microbial performance in the bioleaching of low-grade tailings. Table 3 summarizes experimental results across different tailing types. Sulfidic and Pb–Zn tailings were most frequently examined, often using Fe/S-oxidizing consortia. Additionally, most studies have focused on bacterial bioleaching, particularly using *Acidithiobacillus* species. On average, the bioleaching process takes about 22.5 days and results in a 63% metal leaching rate. However, mixed cultures consistently achieved higher recovery rates (Zn, Ni, Cu > 90%) and shorter leaching durations (typically 10–25 days) compared to single strains. The enhanced bioleaching performance observed in mixed microbial cultures is attributed to their metabolic complementarity. It is also linked to their coordinated community behavior. This highlights a distinct requirement for more extensive investigation in this area.

Strength, weakness, opportunity, and threat (SWOT) analysis

To comprehensively evaluate the potential of bioleaching of LGWR, a SWOT analysis is conducted here. This framework allows a structured appraisal of strengths and weaknesses alongside opportunities and threats. Indeed, it helps identify the key internal and external factors that influence the feasibility and scalability of the technology. The analysis examines low-grade bioleaching technologies from a broader strategic viewpoint. This outlines their key strengths, limitations, opportunities, and risks to inform future directions. A summary of the SWOT analysis is outlined in Table 4.

Strengths

Compared to the common pyro- and hydrometallurgical methods such as smelting, heating, leaching, and oxidizing processes, bioleaching has demonstrated its commercial potential for metal recovery from LGWR due to its lower energy requirement and reduced chemical consumption [181,182]. For example, it was reported that the total acid consumption of conventional processings are in the range of 10 kg to 100 kg of H_2SO_4 per tonne of ore, while their energy demand can reach up to 211 MJ/kg for Al and 114 MJ/kg for nickel production [183–185]. This should not be underestimated, where large volumes of toxic residues of hydrometallurgical processes (for example, concentrated sodium hydroxide or sulfuric acid), are difficult to store and pose environmental risks [186]. Moreover, treating these residues is challenging, especially when trying to neutralize waste streams and manage metallurgical sludge to prevent further pollution [187,188].

Bioleaching can also serve as a bioremediation method to eliminate hazardous metals (e.g., As) in tailings, where conventional approaches are costly [189,190]. Researchers have achieved 94–97% solubilization of As from mine tailings within 3 months using naturally occurring acidophiles in column reactors [191].

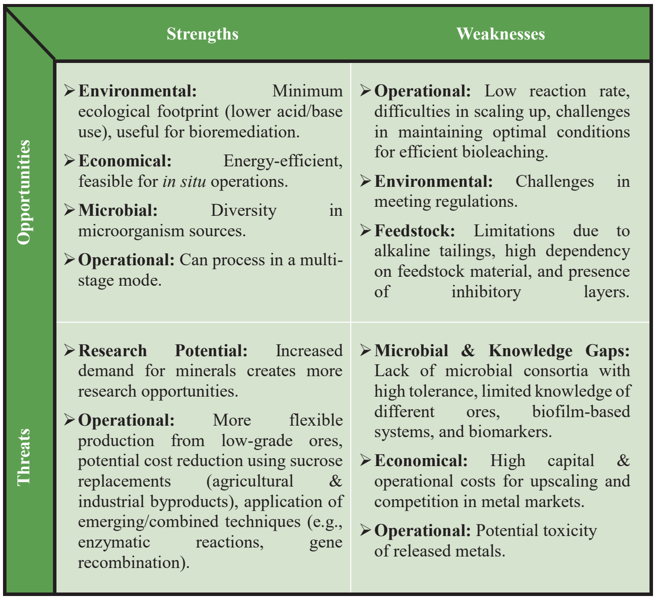
Applications of this method in wastewater sludge management include metal biosorption and the reduction of environmental concerns [192], for land applications [193], and dewaterability improvements [194]. Bioleaching has also been applied to sewage sludge to remove heavy metals and improve sludge quality. In one study, acidophilic

Table 3
A summary of reported bioleaching for low-grade tailings.

Microbial Culture	Type of Tailing	Process Time (Day)	Efficiency of Targeted Metals	Reference
Bacteria (Single-Species)				
A. ferrooxidans	Pb-Zn mine tailings	30	Fe: 85.4% Pb: 4.1% Zn: 97.8%	[271]
A. thiooxidans or Pseudomonas putida	Chromite tailings	30	Cr_2O_3: 23%	[272]
A. ferrooxidans	Uranium tailings	20	U: >50%	[273]
A. ferrooxidans	Mine tailings	24	Te: 47·7%	[274]
Pseudomonas	Coal ash	30	Cr: 13.7% Cu: 14.6% Mn: 6.3% Zn: 12.2%	[275]
A. thiooxidans	Zn metallurgical leach residues	45	Zn: 78%	[276]
A. ferrooxidans	Sludge mining sediment	42	Zn: 98.7% Fe: 85.4% Cu: 96.4%	[277]
Alcaligenes faecalis	Mine tailings	15	Au: 15%	[278]
A. ferrooxidans	Pb-Zn mine tailings	50	Fe: 71.3% Pb: 0.8% Zn: 97.3%	[279]
Bacterial Consortia				
Fe- and S-oxidizing bacteria	Sulphidic tailing	30	Cu: 66.7%, Ni: 87% Co: 69%	[280]
	Sulfidic tailingj	25	Fe: 22.5%	[281]
	Sulfidic tailing	30	Cu: 55% Ni: 98.2% Co: 59.5%	[282]
	Limonite ores and laterite tailings	30	Co: 42.4% Fe: 57% Ni: 37.4 Mn: 24.4%	[283]
	Mining residues and drainage	21	Ag: 99.1% Zn: 37.9% Fe: 32.2%	[284]
	Uranium Tailings	35	LREEs: 6–14% HREEs: 54–58%	[285]
	Sulfide mining tailings	10	Co: 87% Zn: 100% Ni: 67% Cu: 43%	[286]
A. ferrooxidans and *A. thiooxidans*	Cu/Co-rich sulfidic mine tailings	13	Co: 91% Cu: 57% Fe: 85% Zn: 100% Mn: 89%	[220]
A. ferrooxidans and *S. thermosulfidooxidans*	Pb-Zn sulfide mine tailings	16	Zn: 98.1% Pb: 60.8%	[287]
A. ferriphilus, *A. ferrooxidans*, and *L. ferrooxidans*	Polymetallic sulfide tailings	28	Cu: 58.3% Al:53.1% Mg: 80% Zn: 76.9%	[288]
A. ferrooxidans, *A. thiooxidans*, and *L. ferrooxidans*	Kooshk Pb − Zn tailing dam	14	Zn: 91% Fe:51%	[289]
A. caldus and Sulfobacillus thermosulfidooxidans	Cu tailings	55	Cu: 84.4	[290]
Leptospirillum, Sulfobacillus, Ferroplasma, and *Acidithiobacillus*	Acid-treated pyritic tailing	12	Cu: 68% Zn: 77% Fe: 75% Au: 92%	[291]
Fungi				
Isolated fungi	Mine tailing	24	Ni: 55% Cu: 67% Co: 60% Zn: 65%	[19]
Other Microbial Consortia				
Bacteria: *Roseospira* and *Sphingomonas*; fungi: *Cladosporium* and *Penicillium chrysogenum*	Ag mine tailings	16.5	Ag*: 67% Mn*: 74%	[129]
Mixed moderate thermophiles	Cu tailings	16	Cu*: 70.7%	[292]
Mixed acidophilic culture	Flotation tailings	11–15	Ni: 90% Co: 60% Zn: 86% Fe: 67%,	[138]

*Up to.

Table 4
Summary of SWOT analysis of low-grade bioleaching technology.



	Strengths	Weaknesses
Opportunities	➢ **Environmental:** Minimum ecological footprint (lower acid/base use), useful for bioremediation. ➢ **Economical:** Energy-efficient, feasible for *in situ* operations. ➢ **Microbial:** Diversity in microorganism sources. ➢ **Operational:** Can process in a multi-stage mode.	➢ **Operational:** Low reaction rate, difficulties in scaling up, challenges in maintaining optimal conditions for efficient bioleaching. ➢ **Environmental:** Challenges in meeting regulations. ➢ **Feedstock:** Limitations due to alkaline tailings, high dependency on feedstock material, and presence of inhibitory layers.
Threats	➢ **Research Potential:** Increased demand for minerals creates more research opportunities. ➢ **Operational:** More flexible production from low-grade ores, potential cost reduction using sucrose replacements (agricultural & industrial byproducts), application of emerging/combined techniques (e.g., enzymatic reactions, gene recombination).	➢ **Microbial & Knowledge Gaps:** Lack of microbial consortia with high tolerance, limited knowledge of different ores, biofilm-based systems, and biomarkers. ➢ **Economical:** High capital & operational costs for upscaling and competition in metal markets. ➢ **Operational:** Potential toxicity of released metals.

bacteria removed 62.7% of Al, 80.7% of Cu, 43% of Pb, and 75.5% of Zn from municipal wastewater sludge over 15 days, which allowed the treated sludge to meet land-application standards. Notably, this microbial leaching also improved the sludge dewaterability by about 67% [195]. The operating parameters, such as a continuous approach [192], pH [196], and temperature changes [197] have shown significant influence on the process efficiency, which helps pave the way for future research. Though research on REE bioleaching from wastewater is still emerging due to growing demand, limited work has addressed REE bioleaching specifically using sludge [91,198].

Another strength of low-grade bioleaching lies in its applicability to challenging scenarios where conventional physical or chemical methods have failed to achieve metal recovery, for example, from coal refuse at elevated temperatures [199–202]. In such cases, the proven success of biological methods like bio-oxidation in extracting REEs from coal cleaning waste highlights the unique advantage of bioleaching [203,204]. For example, bio-oxidation of pyrite-rich coal waste generated enough acidity to solubilize up to 40–56% of certain REEs and over 75% of CMs like Co [205]. Enhancements such as mixed cultures, longer digestion time, and temperature optimization can further improve the recovery [206]. For example, by augmenting a consortium with an additional iron- and sulfur-oxidizing strain, the metal recovery jumped up to 15% Cu and 10% Zn from the tailings [207]. Therefore, we recommend exploring the potential for bioleaching processes as a cost-effective metal recovery for coal refuse.

Moreover, as mentioned in Table 2, a distinct advantage of bioleaching lies in the versatility of microbial agents. This allows metal extraction to be applied across different contexts, including the recovery of REEs, precious metals (such as Au and Ag, which are defined as metals with a high economic value), and base metals (such as Pb, Ni, and Zn), which are defined as less valuable or highly reactive metals compared to precious metals), depending on economic and industrial needs.

Weaknesses

Assessments of industrial-scale bioleaching reveal technical and operational challenges, particularly in maintaining optimal conditions. Large-scale facilities and reactors often face unpredictable fluctuations in pH, microbial distribution, and oxygen availability [208]. This leads to incomplete reactions, unwanted byproducts, and reduced efficiency [115,188]. For instance, tailing alkalinity can negatively affect microbial activity, and variation in the mineralogical composition of raw materials can lead to diverse bioleaching outcomes, making the process highly sensitive to feedstock characteristics [209]. In addition, the dissolution of the targeted layers is usually dependent on surrounding passivation layers; for example, the role of S-oxidizing acidophiles in the biomining of chalcopyrite is to remove passivating layers of reduced S on the surface to increase the accessibility of Cu. In one chalcopyrite bioleaching experiment, introducing *A. thiooxidans* into the microbial community eliminated the sulfur passivation layer on chalcopyrite and increased copper extraction rates by over 50% compared to the original culture [210].

Another weakness lies in the slower rate of bioleaching (as presented in section 3.5). Rates are slower than in chemical extraction methods [211]. This limitation becomes even more evident at larger scales, where several bottlenecks remain to be addressed. The main ones are its slow kinetics (often requiring weeks to months compared to hours in chemical leaching) [13] and the need for strict control of biological conditions such as pH, temperature, redox potential, and oxygen supply [170]. At large scales, scalability issues arise. Maintaining uniform microbial activity and solution flow in heaps or reactors is difficult. This often leads to incomplete extraction [212]. For example, an estimate suggests that if a laboratory-scale bioleaching experiment takes about 300 days, the same process at an industrial scale may require at least 900 days (nearly 2.5 years) to reach an acceptable level of recovery [13]. Clearly, such timeframes are hardly practical for a commercial producer. Additionally, metal toxicity to microbes and precipitation of secondary minerals (e.g., jarosite) can inhibit bioleaching performance [13]. Moreover, biological methods may lag behind due to specific technical or operational limitations. For example, while both sequential chemical extraction and bio-oxidation were used as the primary extraction processes of coal refuse in previous studies, there is still a need for significant improvement in the latter process, such as energy-intensive preparation phases, difficulties with the high ash content, and restrictions in the ability to detect REEs during the X-ray sorting process [51,203]. Nevertheless, efficient metal recovery can diminish it. Ultimately, regulations for implementing green technologies also pose hurdles [187].

Opportunities

Bioleaching has a unique opportunity to be integrated in existing metallurgical processes, either as a pretreatment (e.g., BIOX process in refractory gold ores in Fosterville (Australia), where bio-oxidation enables efficient cyanidation [213]), or as the main leaching step coupled with hydrometallurgy (e.g., heap bioleaching of low-grade copper in Chile, followed by solvent extraction and electrowinning to produce cathode copper [214]). It can also serve as a selective step to remove impurities such as arsenic from copper concentrates before smelting, thereby reducing environmental issues [214]. In some cases, hybrid sequences combine thermal treatment with bioleaching to enhance metal recovery. For example, one study reported that heating laterite ore to 600 °C prior to bioleaching converted goethite into hematite, creating conditions more favorable for acid attack and thereby significantly enhancing the extraction yields of nickel and cobalt [215].

Operationally, bioleaching is well-suited for polymetallic and LGWR targeted for mass production [208]. For example, a natural consortium of bacteria was able to simultaneously leach 55–56% of the copper and 63–73% of the zinc from a complex tailings sample [207]. Low metal concentrations have led to innovations, including hybrid systems combining biological processes with CN leaching, roasting, or microbial fuel cells [188,216]. One study showed that using an MFC-bioleaching system could extract copper about 80% from low-grade tailings while simultaneously generating a maximum power density of 30.54 mW/m^2 [216]. Emerging techniques like enzymatic bioleaching could also selectively improve bioreaction rates while minimizing pollution [188]. Moreover, pretreatment methods can enhance process efficiency. Besides, techniques such as acid leaching, microwave treatment, and recrystallization have been shown to improve REE recovery from PG [79,83,114], but ongoing optimization is required for cost-effectiveness at scale [114,116,217].

Bioleaching of LGWR not only meets increasing industrial demand for REEs and precious metals but also presents various avenues for process optimization. For example, future studies could explore cost reduction strategies, such as replacing sucrose, the primary carbon source in culture media, with low-cost agricultural or industrial byproducts [31]. In a recent experiment on fungal bioleaching, molasses was reported to be 386 times cheaper than pure sucrose, yet it supported the microbes just as well [218].

Advanced technologies, such as genetic engineering and transcriptomic sequencing, can enhance microbial performance. This is done by regulating metabolism under harsh conditions like low nutrients, extreme pH, temperature fluctuations, or high metal concentrations [115]. Nonetheless, *in situ* monitoring of microbial intermediate products in the bioleaching process remains a challenge in metabolic studies.

Advanced techniques have promisingly broadened the scope of bioleaching through applications such as DNA extraction, microbial gene exchange, molecular adaptations, and reconstruction of metabolism on a genomic scale [208]. For example, genetic recombination was successfully used to improve the economic feasibility of tailing bioleaching [188]. However, identifying and characterizing genes for acid tolerance and metal solubilization in acidophiles is still an active research field [219]. It should also be noted that applying genetically modified microorganisms (GMOs) in bioleaching requires strict containment measures to prevent unintended environmental impacts.

The synthesis of the above opportunities can shape a roadmap for future research. However, since leveraging the full operational potential requires a comprehensive action, this roadmap should be outlined in two overarching dimensions: (i) the technical dimension, encompassing process optimization through low-cost nutrient substitution, hybrid flowsheets, microbial engineering; and (ii) the regulatory dimension, emphasizing collaboration among academia, industry, and government to secure funding, provide pilot infrastructure, and establish supportive frameworks.

Threats

Since this technology is still evolving, different challenges can be identified and addressed: Diverse metal grades and complex waste matrices complicate processing [220,221]. The application of pretreatments (such as crushing, milling, and pre-concentration) also incurs extra costs and energy that need to be accounted for in the net balance of the process [222,223]. Therefore, it is vital to provide proper microbial consortia to improve the productivity of bioleaching reactions and ease the operation. This issue becomes more intricate on an industrial scale, where it is difficult to control the operating conditions and site-to-site variability. Although laboratory methods, like microbial strain engineering, can be used to develop more robust and high-performing strains, the process typically requires a lengthy evolution period, and the outcomes are heavily influenced by specific experimental conditions [188]. Another important threat comes from the lack of sufficient facilities to understand the complexities of this process. For example, while there is still room for improvement in microbial metabolism and biodiversity assessment, biomarkers need to be developed to analyse microbial biodiversity within field operations [211].

Despite numerous studies on reaction parameters, such as particle size, temperature, pH, and oxygen availability, threats are still seen from this side [208]. For example, studies need to be conducted on some toxic releases, such as arsenic, which cause a reduction in microbial activity and pose a serious threat to the advancement of the process. In addition, there is a lack of bioleaching research with some low-grade ores (mostly phosphate-based), like xenotime, bastnasite, and monazite [187]. As a result, the knowledge of monitoring the fate of phosphate and REEs within the interfaces is limited [211]. Besides, studies on understanding biofilm formation should be conducted to describe the interactions with mineral surfaces.

There are still some critical obstacles in capital and operational expenditures related to upscaling, as well. This threat must be taken seriously because cost management is the driving force behind the development of bioleaching. For instance, the mining infrastructures of conventional methodologies are not fully adapted to this method. In particular, dump and heap-leaching may often lack the necessary control over some parameters like temperature and microbial distribution [149,188,224,225]. The main threat comes from the fact that any changes to the plant equipment impose some costs on the system [208]. Process optimization is the key to solving this problem, especially when additional parameters, such as limited aeration and poor solution transport through large particle sizes, complicate the leaching process [208,226].

The competitive market of metals prices for extraction from primary sources is itself a disincentive for low-grade bioleaching. For example, the recently commissioned Kamoa-Kakula mine in the Democratic Republic of the Congo owns a high-grade Cu-ore (5%-6%), which accelerates supply growth and reinforces the economic challenges of low-grade bioleaching [7]. Moreover, the grade of resources and the composition of the metals dictate the accessibility, ease of the process, and final metal recovery, and subsequently, the cost of the process [208]. On the one hand, an assessment of the process of economic feasibility (Techno-Economic Analysis (TEA)) must be examined. On the other hand, a reduced expectation needs to be placed on the process revenue and the breakeven of invested capital costs when LGWR is used for metal recovery. Overall, the choice of technology depends on factors such as waste grade and complexity, microbial considerations, environmental regulations, and economic feasibility.

Cross-study comparison of SWOT findings

The SWOT analysis presented in this study is based on previous frameworks [227,228]. A qualitative comparative review was also carried out: first, the SWOT factors reported in the previous studies were extracted, their terminology was standardized (synonyms merged and mapped into the strengths/weaknesses/opportunities/threats), and then this matrix was cross-checked with the findings of the present study to map the deviations. In this way, the analysis reaffirms key aspects such as the lower environmental impacts of bioleaching, microbial diversity and flexibility, and the potential for resource recovery from low-grade materials. However, it deviates in several important aspects that reflect a broader and practice-based perspective on the industrial deployment of bioleaching:

- First, while earlier analyses have mostly focused on incineration residues or REE-bearing ores, the current study extends the application domain to underexplored waste matrices such as wastewater sludge and mine tailings, which pose unique operational and microbial challenges.
- Second, previous SWOT analyses often regard microbial compatibility as a supporting factor. Gomes et al. (2020) lists adaptation of microorganisms and passivation layers as core constraints [227], which the present study elevates to key determinants (strain selectivity, passivation-layer disruption, substrate-specific dynamics). For instance, previous studies have employed *A. ferrooxidans*, *Leptospirillum ferriphilum*, and sulfur/iron-oxidizing archaea in controlled bioreactors (35 °C, pH 1.4), showing that ore mineralogy (e.g., calcite-driven CO_2 release) and operational factors (pulp density, agitation) directly shaped microbial dynamics and passivation outcomes [205,229]. Similarly, Vo et al. (2024) converges with this by calling for high-throughput strain screening and noting that REE biomining prototypes are $\leq$ TRL-4, evidence that strain–matrix fit is a first-order success factor rather than a peripheral one [227,228].
- Third, in addressing economic and infrastructural threats, this analysis moves beyond general references to market volatility or regulatory barriers. Gomes et al. (2020) flags no pilot-scale for anthropogenic MSWI streams and the need for reactor design/scale-up [227]. Vo et al. (2024)'s TEA adds hard numbers: carbon source cost (44%) of total cost and coal ash yielding the highest profit [227,228]. Instead, the present SWOT articulates specific obstacles such as (i) the incompatibility of current heap leaching infrastructure, (ii) capital investment inertia, and (iii) the economic disadvantage posed by high-grade ore competitors (e.g., the Kamoa-Kakula mine).
- The present study's emphasis on these resources maps directly to Vo et al. (2024)'s recommendations, where screening efficient strains, metabolic engineering, pulp-ratio, mineral-processing optimization, and cheaper lixiviants and carbon feeds (e.g., corn-stover wastewater improving net profit) were considered [227,228]. In parallel, Gomes et al. (2020) identifies complementary opportunities (e.g., bio-electrochemical systems, accelerated carbonation) that the present study subsumes under hybrid and integrated routes in the present roadmap [227].

Market trend analysis

As mentioned in Section 4.4, the price competitiveness of metals may limit the economic attractiveness of LGWR bioleaching. For instance, Ga and Ge, were affected by geopolitical risks, which led to the price increase in 2023 [7]. Li has also experienced significant fluctuations due to supply–demand imbalances [230,231]. As a result, understanding market trajectories offers valuable insights into the strategic position of this technology in addressing both short-term and long-term resource challenges. Predictions show that the market trend of CMs is experiencing a growth of 8% – 30%, and transformation is driven by factors such as increasing demand for renewable energy technologies and government policies [7].

Fig. 3 presents an analysis of supply–demand balance in the Sustainable Development Scenario (SDS) and Stated Policies Scenario (STEPS) scenarios. Both predict rapid expansion. This expansion leads to imbalance and fragility in CMs supply chains [232]. According to the IEA, total demand for critical minerals from clean energy technologies is projected to double by 2040 under the STEPS scenario, whereas it nearly quadruples by 2040 in the SDS scenario [7]. These International Energy Agency's scenarios are based on assumptions about the expansion of renewable energy, the growth of electric vehicle sales, and the development of grid infrastructure and energy storage [233]. In each scenario, these drivers translate into different levels of critical metal demand, with STEPS showing moderate increases and SDS showing much stronger growth. Recent market data also show that demand for

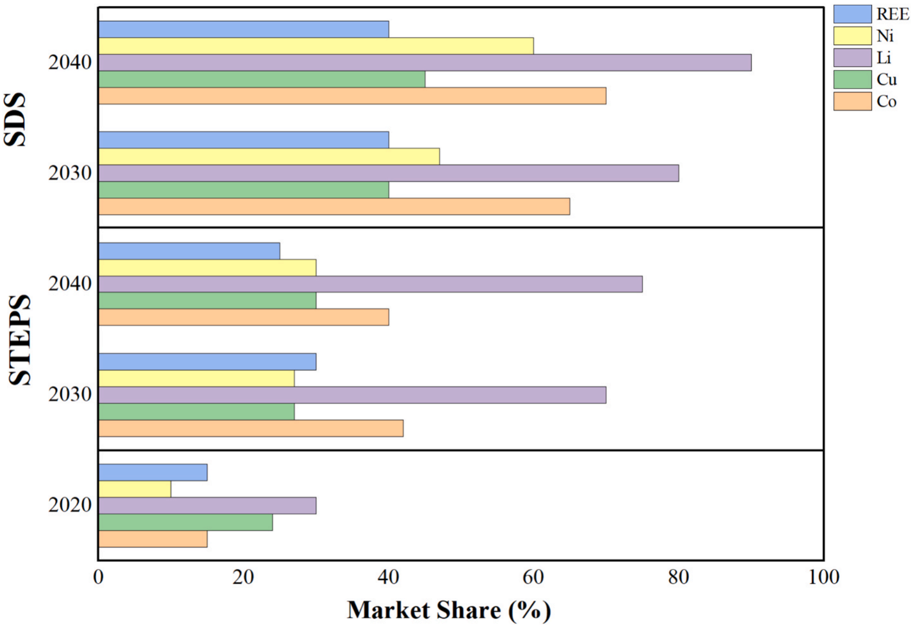


Fig. 3. Projected market share (%) of CMs in the clean energy sector under two different scenarios. Stated Policies Scenario (STEPS): Represents the expected growth of energy systems based on existing government policies and announced commitments. Sustainable Development Scenario (SDS): Envisions an accelerated transition toward clean technologies, emphasizing more decisive policy actions and technological advancements. Extracted and redrawn from [232].

critical minerals has already been rising rapidly (lithium demand alone jumped by 30% in 2023, while nickel, cobalt, and REEs increased by about 8–15% in the same year) [7]. In line with this trend, Li is projected to account for 75% and 90% of the market share by 2040 in the STEPS and SDS scenarios, respectively. Ni and Co will also increase substantially, reflecting their importance in high-performance battery cathodes and energy storage materials [234]. Supporting expanding electricity grids, transmission lines, and renewable infrastructure, Cu demand will rise steadily from 24% in 2020 to 32% market share in STEPS and 45% in SDS by 2040.

At the same time, REEs maintain a critical market share due to their use in clean applications [232]. The SDS scenario shows more pronounced growth in demand for all these minerals, highlighting the urgent need for alternative recovery methods like low-grade bioleaching to meet future resource demands sustainably. The predicted gaps in supply can be seen as an opportunity to mobilize alternative sources within LGWR. For instance, a 20-million-ton slag and tailings dump averaging about 0.66% cobalt in Zambia, and unprocessed PG wastes that are estimated to contain about 200,000 tons of REEs can play an important role in meeting local demand while reducing pressure on primary mining [235].

As shown in Fig. 2, red mud contains over 13 wt% Al and the highest aggregate concentration of REEs (about 0.13 wt%), while slag and tailings exhibit substantial levels of Cu (up to 1.8 wt%), Zn (2.1 wt%), and Co (0.5 wt%). These concentrations not only demonstrate resource equivalency with some primary ores but reveal the unseen potential in these resources, especially when millions of tonnes of them have been dumped as waste. Harnessing this potential could well narrow the gap between supply and demand without having to impose the environmental costs of new mining. This is where the role of LGWR is further reinforced, particularly in jurisdictions where geopolitical exposure and a lack of access to high-grade reserves are present.

To address potential supply constraints, there is a growing interest in enhancing efforts toward CMs bioleaching from LGWR. Countries and companies are actively working to scale up future supplies, diversify sources in response to the growing demand, and reduce dependence on a limited number of dominant producers [236]. For instance, large-scale metal producers such as Aurubis in Germany and Glencore in Canada have incorporated low-grade materials into their smelting operations to recover a broad spectrum of critical and base metals [237]. Forecasts consistently indicate strong justification for expanding bioleaching to low-grade ores. In response, governments are rolling out initiatives that diversify supply bases and reinforce strategic chains. Multilateral initiatives such as the Minerals Security Partnership (MSP) and G7 Critical Minerals Action Plan coordinate investment in mining and processing projects. These provide prioritized partnerships with trusted allies to reduce reliance on dominant producers [238,239]. Regional policies like the EU Critical Raw Materials Act (2023) set specific goals for domestic production, processing, and recycling due to concerns over external dependence [240,241]. In North America, financial and regulatory support are provided by the American Mineral Security Act and related legislation, combined with USMCA coordination to develop secure [242]. Two-party agreements (like the US–Japan Critical Minerals Agreement (2023)) further reinforce these goals. This is done through trade facilitation, aligned standards, and circular-economy measures [243].

Public–private collaborations complement these governmental efforts by translating policy into actionable projects [244]. Industry groups such as the Global Battery Alliance (GBA) and Rare Earth Industry Association (REIA) coordinate recycling [245], ESG standards [246], and joint investments, often in low-grade resources or emerging technologies like bioleaching. These initiatives collectively aim to expand global supply bases, enhance resilience, and integrate circular-economy approaches, including material recovery and reuse, into the broader critical-materials strategy.

Yet, while these multilateral and national initiatives highlight the strategic case for mobilizing LGWR through circular-economy approaches, their practical implementation must also contend with significant environmental and social risks at the local scale. What is clear is that LGWR resources provide two benefits of supplying CMs and lowering the long-term environmental hazards of the waste. However, these projects carry serious risks such as tailings dam failures, toxic dust emissions, and water contamination if poorly managed [247]. From a social perspective, community acceptance and environmental justice are crucial. Community acceptance depends less on job promises and more on procedural fairness (being listened to, respected, and involved in participatory assessments of environmental risks (notably water)) [248]. Communities near the mining sites report concerns about toxic dust, water contamination, exposure, and property values, and they often distrust agencies [249]. On Indigenous lands, legacy mining wastes and residues (overburden, waste rock, tailings) are linked to land and biodiversity loss, long-lasting chemical risks (including radiation/metals), and restrictions on access to culturally significant places and livelihoods [250]. These issues cannot be underestimated. Therefore, mobilizing LGWR must pair metal recovery with robust environmental safeguards, transparent monitoring, and inclusive consultation to build trust and prevent conflict over water, land, and health.

Case study: Bioleaching of copper from low-grade tailings

This case study focuses on copper tailings, a by-product of the Cu extraction process, which often contain residuals such as As, Cd, Pb, and CMs. These tailings typically have low Cu content, less than 1%, which means the conventional extraction methods are less economically viable in this case, but the bioleaching is. They also present environmental risks like AMD, which further justifies the need for sustainable management strategies. The small particle size and intergrowth of Cu with gangue minerals in copper tailing may also add additional challenges for CMs recovery [251]. There are currently 8 billion tons of tailings produced annually, with 46% coming from the mining of Cu [252].

Several countries have been investing in the bioleaching of copper tailings. In Australia, the Australian Mine Atlas indicates that about 473 mine waste sites have the potential for Cu recovery [253]. In Chile, leading mining companies such as CODELCO have invested in extracting Cu from LGWR using bioleaching technologies [254]. A notable example is the Sociedad Minera Pudahuel (SMP) Project, which developed and commissioned the Lo Aguirre copper mine near Santiago in 1980. The shift from chemical leaching to bioleaching has enabled the recovery of Cu from exhausted oxide minerals, allowing the operation to maintain efficient production until its closure in 2002 [254,255].

In the Chilean deployments relevant to the present case, Cu bioleaching has been implemented mainly as thin-layer in heap systems integrated with solvent extraction – electrowinning. Heaps are built from agglomerated tailings pre-cured with dilute H_2SO_4 and irrigated by acidic raffinate; aeration is supplied from the base to provide O_2 and CO_2 [255]. The active consortia are the canonical acidophiles (*A. ferrooxidans* and *A. thiooxidans* together with *L. ferrooxidans*) with optional inoculation/biomass make-up when native populations are insufficient. Nutrient demand is low but N and P can become limiting in heaps; supplementation via ammonium/phosphate in the irrigant is common practice. Operating pH is maintained in the the range of 1–2 with oxidation-reduction potential control (for chalcopyrite, a value about 450 mV Ag/AgCl is often targeted) and temperature follows the site regime (mesophilic heaps), while higher-temperature (65–80 °C) stirred-tank bioleaching with thermophiles has been piloted for sulfide concentrates [254]. Typical particle sizes are about 1 cm (or run-of-mine for dumps), with leach cycles of months; process control focuses on aeration and irrigation distribution, solution chemistry, and microbial viability to sustain ferric generation and reduced inorganic sulfur compounds oxidation [254].

The main challenge for bioleaching of copper tailings is the comprehensive understanding of the operation cost to assess economic viability. The energy consumption in this method accounted for about 30% of total process expenses (51% electricity and 49% fuel). It underscores the significant influence of energy demand on overall costs. Reprocessing low-grade copper tailings (grade 0.6%) with a recovery rate of 53% can reduce the hydrometallurgical process cost by about 45% (approximately 2953 USD/t of Cu). This cost reduction translated into a gross margin exceeding 65% of the market price (10000 USD/t for the market price) [256].

The result of this gross margin indicates that economic viability is not strictly dependent only on ore grade (0.6%), but rather on the overall recovery efficiency. This highlights the importance of advancing bioleaching science: (i) microbial strain selection, (ii) leaching kinetics, and (iii) process control, to maximize extraction yields. The economic viability is also partly influenced by the fluctuation in metal prices, the increase in energy and other chemical costs. Additional actions need to be taken to make energy use more efficient. As mentioned before, improving energy efficiency might greatly cut the total operating costs. This efficiency can be achieved via several ways: (i) optimizing aeration and agitation systems, (ii) leveraging advanced microbial consortia, and (iii) implementing passive or renewable energy solutions to reduce heating and fuel demands. Optimizing aeration and agitation is crucial, since forced aeration is one of the largest energy sinks in bioleaching [257]. Improving oxygen delivery efficiency (e.g. using advanced sparger systems) can maintain microbial activity with less power input; for example, replacing a conventional sparger with a nano-aeration unit increased metal leaching efficiency while cutting daily energy use by about 1.6% [258]. Leveraging advanced microbial consortia is another strategy: mixed cultures of complementary microbes (for instance, combining mesophilic *Acidithiobacillus* with thermophilic *Sulfobacillus* bacteria) have shown synergistic leaching performance, which helps sustain high extraction yields without requiring extreme operating conditions [258]. Additionally, passive or renewable energy solutions are being implemented to reduce external fuel demands. A notable example is the use of solar thermal heating in heap bioleaching; warming the heaps with solar energy creates optimal temperatures for thermophiles and boosted copper recovery from about 67% to 85%, thereby lessening reliance on conventional [259]. These approaches underscore how smarter engineering and integration of biological and renewable methods can improve energy efficiency in bioleaching.

Bioleaching of low-grade copper tailings presents a promising alternative, offering two notable advantages: (i) 50–80% lower energy demand compared to conventional methods, and (ii) higher recovery rates up to 85% [16,260]. Nutrients and general costs (e.g., utilities and maintenance) account for 60% of total process expenses, with nutrients representing 40% (0.4 USD/kg) and general costs contributing 20% (approximately 615,000 USD/year), respectively [261]. The successful large-scale use of bioleaching depends on a few important factors, like energy consumption, the availability of nutrients, and the costs of maintenance. The existing mining infrastructure often lacks the precision control required for microbial viability. This would make retrofitting a costly necessity. These findings reveal that the success of bioleaching in the future requires an integration of economic planning, infrastructure redesign, and regulatory alignment. Economically, scale-up is limited by operating costs (such as nutrients, acid, and aeration energy) and long residence times, so success depends on improving leach rates while reducing expenses [16,188]. Infrastructurally, most sites lack precise control of pH, aeration, temperature and solids handling; industrial deployment typically needs purpose-built heap pads or stirred-tank bioreactors with robust aeration or heat management, as illustrated by early Chinese heap operations [16]. With regrad to regulatory challenges, tailings are often governed as “waste” rather than secondary resources; clearer permitting pathways and targeted incentives are required to reduce the risk of tailings bioleaching [262]. To address these challenges, the current research agenda needs to be shifted from purely biological optimization to a cross-disciplinary, system-driven approach.

Life cycle assessments showed that bioleaching also provides environmental benefits: (i) reduced greenhouse gas emissions, and (ii) lower waste generation. It consolidates its role as a sustainable and economically viable solution within the mining industry. Table 5 represents the comparative data of bioleaching and conventional methods (Hydrometallurgy and pyrometallurgy). Bioleaching emits between 3 and 6 kg CO_2 eq/kg, which is significantly lower than traditional hydrometallurgy (4.9–7.37 kg CO_2 eq/kg) and pyrometallurgy (3.06–9.00 kg CO_2 eq/kg) [263,264]. These gains come largely from avoiding high-temperature smelting and its associated fuel and CO_2. Biomining instead operates at ambient or modest temperatures (typically 20–50 °C) and atmospheric pressure. These benefits are particularly important in regions (e.g., European Union) where carbon regulations are strict or emission limits are enforced. In these policy-driven markets, bioleaching’s lower carbon footprint may translate not only to regulatory compliance but also to financial incentives such as carbon credits and reduced operational costs. In addition, hydrometallurgy uses concentrated acids and reagents, whereas bioleaching uses only mild acids generated by microbes which result in about 85% lower in acidification [260,264]. Studies have shown that hydrometallurgical effluents can be up to 250 kg/tonne of feed with highly acidic and metal-rich parameter [265]. For example, hydrometallurgical effluents had initial compositions around pH 1.39, COD 14.48 g/L, Fe 8.69 g/L, and Cu 117 mg/L [266]. Energy requirements across metal-recovery routes span roughly 20–211 MJ per kg of product (for example, iron/steel about 22 MJ/kg, Al about 211 MJ/kg, and copper hydrometallurgy about 64 MJ/kg) [184]. In contrast, Bioleaching has the potential to save around 60% of the total energy [267]. CM bioleaching can further be a much more sustainable extraction route by optimizing energy inputs (e.g. solar power, renewable feedstocks). Bioleaching generally achieves 70–90%, while conventional hydrometallurgy and pyrometallurgy reach 90–99% [268]. Although slightly lower, the bioleaching yield remains within the industrially acceptable range. Eventually, bioleaching requires much lower capital investment ($2000–6500 per tonne) and operating expenses ($3–6 per kg) compared to conventional routes ($7000–10000 per tonne CAPEX and $7–10 per kg OPEX) [261,269,270]. These analyses confirm that bioleaching of CMs from secondary resources can substantially reduce energy use, waste generation, and emissions compared to traditional mining and metallurgy.

Conclusion

This review shows that several low-grade mining wastes (red mud, coal ash and coal refuse, tailings, slag, and waste rock) are credible secondary sources of critical metals and REEs. Red mud is consistently enriched in Al (>13%) and REEs (0.1%), while slags contain notable levels of Zn (2.1%), Mg (2.7%), Cu (1.8%), and Ni (0.28%). Bioleaching

Table 5
Comparison of extraction technologies: bioleaching versus conventional methods.

Parameter	Bioleaching	Conventional Method	Ref.
Operating conditions	20–50 °C, ambient pressure	Hydrometallurgy 80–90 °C (sometimes high pressure) Pyrometallurgy > 1200 °C	[260]
Energy demand	25 MJ/kg	20 – 200 MJ/kg	[184,267]
CO_2 emissions	3–6 kg CO_2 eq/kg	3–9 kg CO_2 eq/kg	[126,127]
Recovery rate	70–90%	90–99%	[268]
Costs	CAPEX: $2000 – 6500 / tonne OPEX: $ 3–6 /kg	CAPEX: $7000 – 10,000 / tonne OPEX: $7–10 /kg	[261,269,270]

emerges as a promising extraction route but is currently limited by slow kinetics, scale-up control, and cost. Key gaps include sparse analytics for metals (such as Li, Pt, and Be), incomplete mineral–phase associations, and uncertain translation from bench to industry. Priorities are rigorous, process-specific waste characterization; development of resilient mixed consortia or engineered microbes; scale-up optimization (pulp density, pH/DO control, oxygen transfer); and decision frameworks that integrate techno-economic and life-cycle considerations to ensure sustainable deployment. Addressing these will improve resource accounting and help align supply with demand through environmentally and socially responsible strategies.

CRediT authorship contribution statement

Soroosh Danaee: Writing – original draft, Visualization, Formal analysis, Conceptualization. **Phong H.N. Vo:** Writing – review & editing, Supervision, Conceptualization. **Sana Malik:** Writing – review & editing. **Peter J. Ralph:** Writing – review & editing, Supervision, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jiec.2025.10.015.

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