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Review

Biobased Strategies for E-Waste Metal Recovery: A Critical Overview of Recent Advances

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Abstract: The increasing e-waste volumes represent a great challenge in the current waste management landscape, primarily due to the massive production and turnover of electronic devices and the complexity of their components and constituents. Traditional strategies for e-waste treatment focus on metal recovery through costly, energetically intensive, and environmentally hazardous processes, such as pyrometallurgical and hydrometallurgical approaches, often neglecting other e-waste constituents. As efforts are directed towards creating a more sustainable and circular economic model, biobased alternative approaches to these traditional techniques have been increasingly investigated. This critical review focuses on recent advances towards sustainable e-waste treatment, exclusively considering studies using e-waste sources. It addresses, from a critical perspective, approaches using inactive biomass, live biomass, and biogenic compounds, showcasing the diversity of strategies and discussing reaction parameters, advantages and disadvantages, challenges, and potential for valorization of generated by-products. While ongoing research focuses on optimizing operational times and metal recovery efficiencies, bioprocessing approaches still offer significant potential for metal recovery from e-waste. These approaches include lower environmental impact by reducing energy consumption and effluent treatments and the ability to recover metals from complex e-waste streams, paving the way for a more circular economy in the electronics industry.

Keywords: bioremediation; e-waste; metals; non-living biomass; bioleaching; microorganisms; biogenic lixiviants; waste utilization

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1. Introduction

Waste electrical and electronic equipment (also known as WEEE, or e-waste, as referred to henceforth) encompasses a wide range of waste materials derived from electrical and electronic equipment (EEE) without the intent of reuse [1,2]. As technology progresses and EEE becomes obsolete at an ever-faster pace, the generated waste volumes increase. This is further amplified by ever broader EEE markets and lax regulations on the safe disposal and treatment of this hazardous type of waste. E-waste volumes in 2022 were reported to be 62 billion kg and are predicted to reach 110 Mt in 2050 if current trends hold, making this the fastest-growing waste stream worldwide [1,3]. This is of particular concern, given the inadequacy of the current e-waste management infrastructure and regulations—for

2022, it was reported that only around 22% (or 13.8 billion kg) of e-waste was formally recycled, compared to over 48 billion kg that remained undocumented, potentially ending up informally treated, incinerated, or unsafely dumped [1]. This information can be observed in Figure 1.

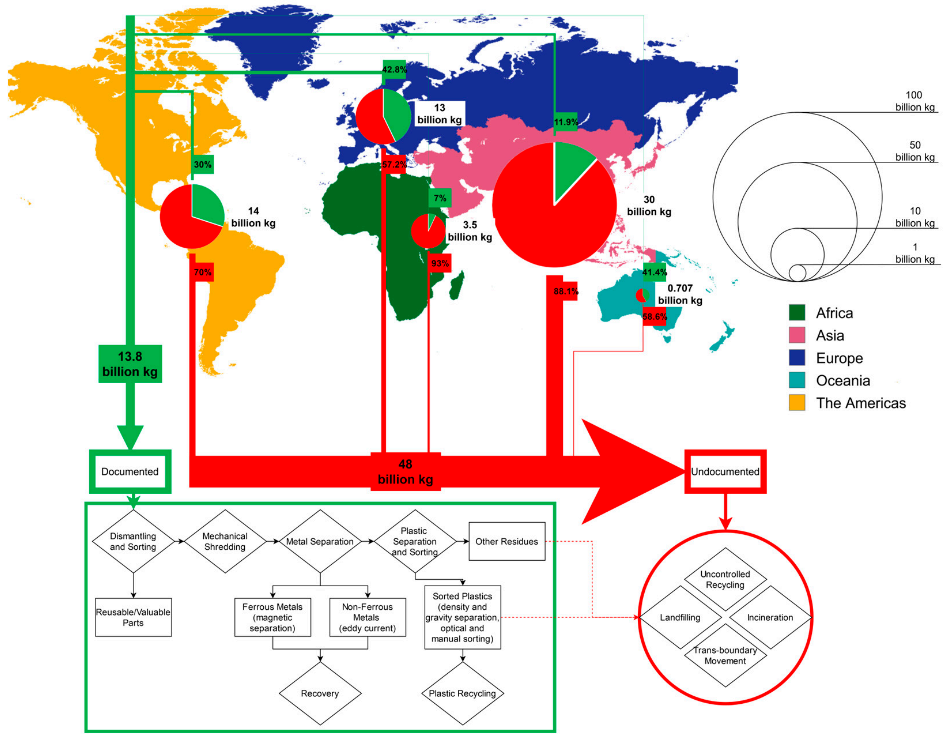

Figure 1. Overview of e-waste generation by region, including the respective proportions of documented and undocumented treatment. Examples of end-of-life treatment methods are also provided. Data and information sourced from references [1,4].

In 2022, most e-waste was generated in Asia (30 billion kg), while the continent that generated the most per capita was Europe (17.6 kg per capita), followed by Oceania (16.1 kg per capita) and the Americas (14.1 kg per capita), whereas Asia and Africa generated, respectively, 6.6 and 2.5 kg per capita [1]. Notwithstanding, Europe has the highest documented formal e-waste collection and recycling rate (42.8%), whereas in the remaining continents, the volume of e-waste formally collected and recycled remains substantially lower [1]. Due to higher labor costs and stricter environmental regulations, some financially developed regions (e.g., North America, Europe) export uncontrolled e-waste to the global south (e.g., Africa, Southeast Asia, and South and Central America, the former importing up to 14% of its total e-waste share, with less than 5% of such trade being classified as controlled) [1]. In these importer countries, e-waste may be processed in less regulated conditions [5]: for example, only around 1% of the e-waste in Africa was documented to be soundly managed, although an even smaller share of 0.1% is documented in West Asia [6], a significantly richer region than Africa [7].

The improper disposal and treatment of e-waste poses significant environmental and health risks due to the presence of hazardous components. For example, soils collected in farmlands near e-waste industrial sites in southeast Zhejiang Province, China, revealed exceeded risk screening values of cadmium, copper, lead, and zinc [8], linked to compromised crop quality and potential health risks for humans [9] and local fauna [10]. This raises environmental and health risk concerns in the importing countries (as reviewed by Chakraborty et al. [11,12]), leading to international agreements, such as the Basel Convention, to control and oversee shipments [13]. However, since its inception, the Basel Convention has been plagued with problems related to data inaccuracy and non-standardization [14,15].

Irrespective of a country being developed or developing, a considerable share of e-waste may end up in landfills after recovering its more valuable constituents, like metals (e.g., gold, silver, copper). E-waste is often found as a complex mixture of different constituents, including hard-to-recover hazardous metals (e.g., lead, lithium, mercury), organic contaminants (e.g., polybrominated diphenyl ethers used as flame retardants), glass (including leaded glass), and plastics (e.g., acrylonitrile butadiene styrene, polystyrene, and polypropylene) [16]. The non-recovery of e-waste metals represents an economic loss of roughly USD 57 billion in 2019 alone, leading to efforts in urging governing bodies worldwide to regulate the disposal and recycling of e-waste [3]. The People's Republic of China and the European Union, for example, have planned significant expansions in e-waste recycling and infrastructure in the near future to recover critical raw materials (CRMs) [17,18].

Current recycling approaches for CRM/metals recovery mainly focus on conventional methods [19], which usually go through a sorting step (including separation and dismantlement [20], followed by pyrometallurgical (e.g., pyrolysis, smelting) or hydrometallurgical techniques [20]. However, as economies transition into a green, circular model, the high energy demand and release of hazardous compounds from conventional methods, often allied to potential co-treatments for effluents, increase overall costs and environmental footprint. The unsustainability of these approaches is further compounded, given the focus on CRMs to the detriment of less valuable components. Thus, this does not align with green chemistry principles [21] and the circular economy model [22]. This would instead require increased focus on the sustainable management of these resources as well as other constituents of e-waste towards their industrial (re)application, as per Figure 2. To address these challenges and move towards a more sustainable circular economy, bioprocessing of e-waste is emerging as a promising alternative to conventional methods. In this context, the bioprocessing of e-waste is being investigated as a viable alternative to these conventional procedures.

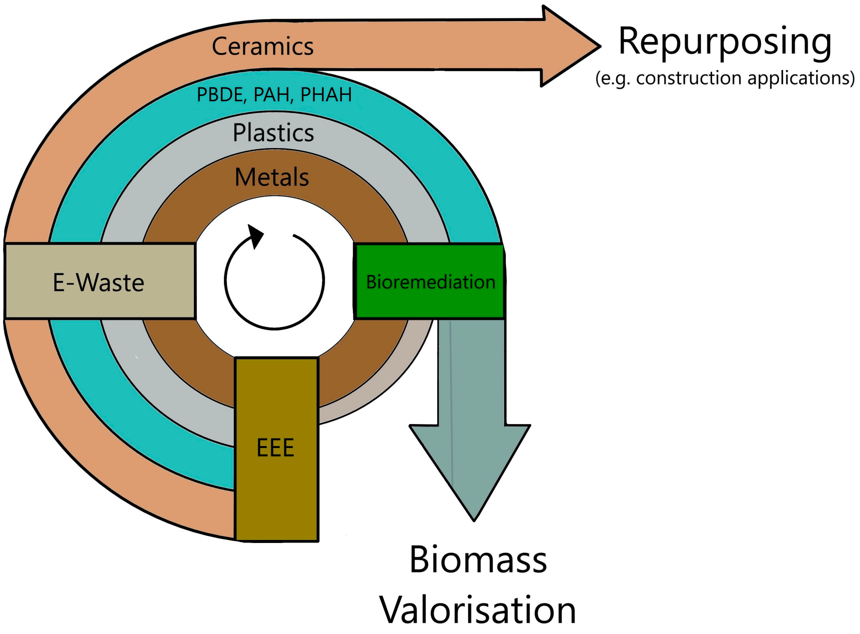

Figure 2. Potential management of electrical and electronic equipment (EEE) and e-waste constituents under a circular economic model. Abbreviation list: polybrominated diphenyl ether (PBDE); polycyclic aromatic hydrocarbon (PAH); polyhalogenated aromatic hydrocarbon (PHAH). Information sourced from references [23,24].

The urgency for sustainable e-waste recycling is further underscored by the geopolitical landscape surrounding critical raw materials [25]. The concentration of production and potential disruptions in supply chains highlight the need to diversify sourcing and recover valuable metals from existing waste streams while opening opportunities for innovative biotechnological approaches like bioprocessing. Given the increased academic interest in and research on e-waste, several reviews and reports have been released focusing on (i) characterizing the global and regional trends in e-waste generation [3,26]), (ii) e-waste composition and its environmental and health impacts [27,28], (iii) policies and regulations related to e-waste management at the national and international levels [29,30], and (iv) recycling technologies and methods for recovering valuable materials (mainly metals) from e-waste to embrace sustainability and circularity (e.g., [31,32]). More recently, efforts have been focused on more sustainable approaches to tackle e-waste (mainly through the recovery of metals), including the treatment of several e-waste form factors, from printed circuit boards to liquid crystal displays, using a variety of microorganisms and processing strategies, as reflected in recent reviews [33,34]. Strategies for the bioremediation of metal-contaminated sites have also been mentioned in such reviews (e.g., through biotransformation or bioaccumulation) [35]. While existing reviews provide valuable insights, they often focus on individual e-waste constituents (previously separated), or even non-e-waste substrates, rather than complete samples, while also overlooking critical aspects such as scalability, hazards, and biomass valorization. This review addresses these gaps by exclusively evaluating recent advances in metal processing directly from e-waste using biobased approaches. It critically addresses the efficiency, scalability, and potential of biobased strategies (using live biomass, dead biomass, and bioactive compounds) while exploring options to add value to by-products and residual biomass. To that end, studies, mainly from 2019 onwards, were collected from several platforms, such as Google Scholar and Scopus, and verified one by one as to the waste material used in their biobased experimental approaches. Only studies using samples of electronic equipment and recovered e-waste were considered for this review, differentiating them from other publications; studies using synthetic materials and laboratory-synthesized samples to simulate e-waste in its entirety or e-waste fractions (e.g., laboratory-produced metal solutions) were thus excluded.

2. Biobased Strategies for E-Waste Metal Recovery—A Critical Overview of Recent Advances

2.1. Use of Inactive Biomass in E-Waste Bioprocessing

The bioprocessing of e-waste constituents in a medium or environmental matrix can be conducted from naturally occurring metabolic processes or by physical–chemical reactions. Using biomass and other biological agents has the potential advantage of decoupling process efficiency from the toxicity of the medium. Given the high toxicity of some e-waste constituents, such as metals and brominated compounds, biobased alternatives using non-living biomass have been the subject of some attention for bioprocessing, with a particular focus on metals. This section explores non-living biomass and other biobased approaches, such as biogenic reagents for bioprocessing contaminants sourced directly from e-waste. An overview of recent literature focusing on these approaches can be found in Table 1, including the main bioprocesses used, waste type, pulp density (i.e., waste solid-to-liquid ratio), recovery of select metals as reported by each study, instances of sorption/desorption and their significance to the recovery of metals, optimal operational conditions, and explored avenues for valorization of generated products. We endeavored to maintain information on the tables in this review as faithful to the original source materials as possible.

Table 1. Recent advances in metal bioprocessing from e-waste sources using inactive biomass ("-"—not explored in study; wPCB—waste printed circuit board; Au—gold; Ag—silver; Cu—copper).

Process(es)	Waste Type	Bioprocessing Agent	Pulp Density	Maximum Recovery of Select Metals			Metal Sorption in Biomass	Operational Volumes and Experimental Conditions	(Re)valorization of Generated Products	Reference
				Au	Ag	Cu				
Bioleaching	wPCB	Oak Tree Biochar	1.6 g/L	-	-	98%	Cu^{2+} adsorbed to biomass at pH higher than 2, adsorbed metal considered not recovered	100 mL (flasks); Optimal Conditions: Oak wood biochar (500 °C, 1 h); biogenic sulfuric acid (15-day 30 °C 160 rpm cell culture, 1 h 600 °C, 10,000 rpm isolation); leaching (3-day 30 °C 160 rpm with 1.6 g/L of biochar)	Undiscussed	[36]
Biosorption	wPCB	Corn Straw Biochar	Not specified	98%	-	-	Up to 98% in a single cycle	10 mL (flasks); Optimal Conditions: Corn straw biochar (700 °C, 3 h); biosorption (5 h, 50 °C, 200 rpm, pH 3 with 0.15 g of biochar/ /3:200 solid-liquid ratio)	Undiscussed	[37]
Bioleaching, Biosorption	wPCB	Cellulose (phosphorylated)	10 g per 100 mL	-	-	35.9 mg/g	Up to 3 sorption/ /desorption cycles	100 mL (flasks); Preparation of crosslinked phosphorylated cellulose (2–3 days, up to 60 °C); simultaneous leaching and sorption (48 h, 60 °C, pH 7)	Reapplication of recovered metals; Reproducible sorption	[38]

Different types of dead biomass may be used to assist e-waste bioprocessing. For example, biochar, a carbon-based product of biomass pyrolysis from various sources, from lignocellulosic material to food waste [39], has been increasingly studied for its potential for metal remediation in different media. Thanks to several attractive characteristics, such as high surface areas, the potential presence of functional groups on said surfaces, and the ability to facilitate electron transfer, biochar can potentially facilitate metal bioleaching and biosorption [36]. Zhou et al. [37] explored the use of biochar properties (particle size, source material, pyrolysis temperature) and experimental conditions (pH, temperature) on gold recovery from pulverized e-waste (in this case, waste printed circuit boards—wPCBs). Furthermore, the use of biochar in this case followed an iodine-iodide leaching process, with increased environmental performance compared to the commonly applied aqua regia or cyanide methods, which make use of more hazardous reagents. Specifying, for this process, pulp density was 10% (*w/v*), the iodine concentration used was 1.0%, in a 1:10 ratio to iodide, with 1.0% (v/v) hydrogen peroxide; the reaction occurred for 4 h, at room temperature. It was found that the inclusion of smaller biochar particles in the iodine–iodide lixiviant after the leaching process generally increased adsorption efficiency (up to 40–60 mesh), but excessively small particles (<200 mesh) reduced efficiency due to decreased material integrity and hindered gold diffusion. Corn straw biochar pyrolyzed at 700 °C showed the highest gold adsorption capacity, likely due to its increased porosity, which in this study was found to correlate positively with charring temperature. Combined with a dose of 0.15 g of biochar per 10 mL, acidic conditions (pH 3, obtained through the addition of potassium hydroxide and hydrogen iodide) were optimal for gold biosorption using this biochar, attributed to the increased surface area and negative surface charge. Higher pH values saw the efficacy of biosorption decrease to as low as 60% in comparison, with a steep decline above pH 7. Biosorption efficiency peaked at 40–50 °C, suggesting a balance between solution viscosity (higher as temperature decreases) and potential temperature-related effects on adsorption mechanisms, resulting in lower efficiencies as tested temperatures deviated from this range (although they remained high, in the 90–95% range). While the maximum adsorption efficiency of corn straw biochar (98%) was comparable to activated carbon (98.6%), the activated carbon approach required less material (0.05 g/10 mL compared to 0.15 g/L for biochar) and achieved faster adsorption. However, the study did not evaluate the desorption of gold from biochar or its reusability, limiting the assessment of its practical potential. Furthermore, the energy-intensive nature of biochar production raises concerns about the overall environmental sustainability of this approach.

Another approach for the extraction of metals from e-waste was explored by Mathaiyan et al. [38]. In this study, simultaneous bioleaching and biosorption of copper from wPCBs was attempted. To that end, a phosphorylated crosslinked cellulose matrix was prepared and subsequently added to a pH 4, 7, or 10 aqueous solution, together with 10 g of wPCB fragments. These waste fragments were pretreated with 10 M sodium hydroxide for the removal of epoxy resin coverings. The reaction, which occurred for 48 h at 60 °C, yielded a copper concentration of 35.9 mg/g in the cellulose matrix at pH 7, with competition with H+ ions at lower pH and higher solubility of copper at higher pH, resulting in lower biosorption (18.4 mg/g and 12.75 mg/g, respectively). Temperature was also found to be a key factor, with reduced efficiencies at and below 45 °C (no higher than 13.34 mg/g). It was also found that the phosphorylated cellulose matrix was capable of retaining sorption ability throughout multiple sorption and desorption cycles. Specifically, after three sequential leaching experiments, copper sorption capacity was recorded to be 36.2 mg/g (with a recorded minimum of 34.7 mg/g in the second experiment). Furthermore, this study describes subsequent steps for the revalorization of this waste, specifically the synthesis of several copper-based materials through a calcination process (400 °C/700 °C, 1 h), which were then applied as working electrodes for ascorbic acid detection, exhibiting fast response times dependent on concentration. Overall, it was found that Cu(II)-P(O)-cellulose calcinated at 700 °C generally exhibited the most attractive performance, particularly regarding linear range (20–2300 µM), limit of detection (0.047 µM, surpassed mainly by catalysts with much more limited linear ranges of up to 1–100 µM), as well as accuracy and selectivity for ascorbic acid. This study stands out by exploring both a novel metal extraction method from e-waste and its upcycling into functional components in a relatively short period. The main environmental impact originating from this approach relates to the calcination processes used for the revalorization of the recovered copper and their energy intensiveness; however, these processes are not strictly part of the copper extraction strategy itself, and thus this process retains a high environmental friendliness. It is unclear, on the other hand, how hazardous the process effluents resulting from simultaneous bioleaching and biosorption are. Further studies considering phosphorylated cellulose for the adsorption of leached metals may also consider alternate, biobased approaches for metal leaching, thus combining sustainable, biobased approaches for both the leaching and recovery of metals from the lixiviant solution. As will be explored henceforth, in the subsequent sections, most studies analyzed for this review focus only on a single aspect of metal recovery efforts from e-waste—specifically, most studies' single focus remains the bioleaching process itself. As such, the integration of sequential biobased procedures for the leaching and subsequent recovery of leached metals from e-waste, and thus an integrative biobased approach for e-waste treatment, remains underexplored.

A direct comparison between these different approaches for metal extraction and recovery from e-waste is challenging despite the same waste type and target metal. However, some general observations can be made, which are transversal to most recent studies in this field. For example, the different units used to report the amount of copper recovered in both studies highlights one of the key challenges to the comparability of the results in this field, as will be further explored in the subsequent sections. Otherwise, the process described by Mathaiyan et al. [38] displays key advantages, as the energy-intensive biochar and activated carbon production processes are not required (although the upcycling of the recovered copper would also demand such energy-intensive procedures), and the leaching process occurs at neutral pH (7), rather than under acidic conditions (i.e., leaching through sulfuric acid), which could also have implications in the need, or lack thereof, for intensive effluent treatment after the e-waste processing. However, the processes described above demonstrate a waste recovery efficiency that exceeds that of key current industrial players (e.g., the pyrometallurgical–hydrometallurgical combined approach of Umicore, at 70% [40]). Further research is needed to directly compare the efficiency, cost-effectiveness, and environmental impact of these two approaches under similar conditions.

The potential toxicity of effluents generated during bioprocessing is also a crucial factor to consider, as the presence of toxic compounds and the pH of the effluent may condition its disposal/treatment and, thus, the financial sustainability of the overall process. Thus, a more comprehensive analysis of effluent composition is necessary to assess its environmental impact. This analysis should include identifying and quantifying specific contaminants, such as heavy metals, organic compounds, and residual leaching agents. For example, tailings from cyanidation processes (still common) incur concern over environmental harm. Cyanide use and discharge in some industrial activities has been tightly regulated in several regions (e.g., the European Union [41]). However, such tailings can result from processes using up to 1.0 pounds of sodium cyanide per ton of solution; meanwhile, across the studies analyzed for this review, Merli et al. [42] reached a comparatively lower biogenic cyanide production of 10 mg/L. Still, given that cyanide concentrations as low as 2.1 mg/kg of body weight could prove life-threatening for animals [43], compounds such as sodium cyanide in concentrations as low as 30 mg/m^3 could prove life-threatening for humans in as little as 1 h of exposure [44], and that the presence of cyanide in water is tightly regulated (e.g., 50 µg/L in the European Union [45]), the analysis of processual effluents remains integral for the assessment of process sustainability.

Industrial yields for rare earth metals (a set of metal elements including scandium, yttrium, and the 15 lanthanides [46]) from e-waste have reached 90% while processing large batches of waste (e.g., a pilot-level plant processing disk drives, capable of managing over one ton of disk drives per batch [47]). Thus, bioprocessing approaches using inactive biomass to enhance the extraction of metals from various sources have been found to largely meet or even exceed these current industrial yields for select metals, but the scale-up of these methodologies may yet present challenges.

2.2. *Use of Live Biomass Directly in E-Waste Bioprocessing*

Live organisms have been extensively explored for metal bioprocessing applications in various scenarios—other than bioleaching, a diversity of organisms have been found to use mechanisms such as biosorption, bioaccumulation, and bioprecipitation to remove these contaminants from the sources, with examples including sponges and algae (e.g., Pinto et al., 2023 [48]). However, most studies concerning the bioprocessing of e-waste, specifically, focus on metal bioleaching rather than the capture of the metals solubilized from e-waste sources, wherein fungi and bacteria are the more researched microorganisms. These microorganisms can induce metal bioleaching through different mechanisms, such as the secretion of cyanide (bio-cyanidation) (e.g., Thakur and Kumar, 2021 [49]) and organic and inorganic acids (acidolysis) (e.g., Arshadi et al., 2020, Arshadi et al., 2021 [50,51]) or the production of redox lixiviants such as ferric iron (oxidation) (e.g., Van Yken et al., 2020 [52]) (Figure 3). An overview of recent literature regarding the use of live biomass for the bioprocessing of metals from e-waste sources can be found in Table 2.

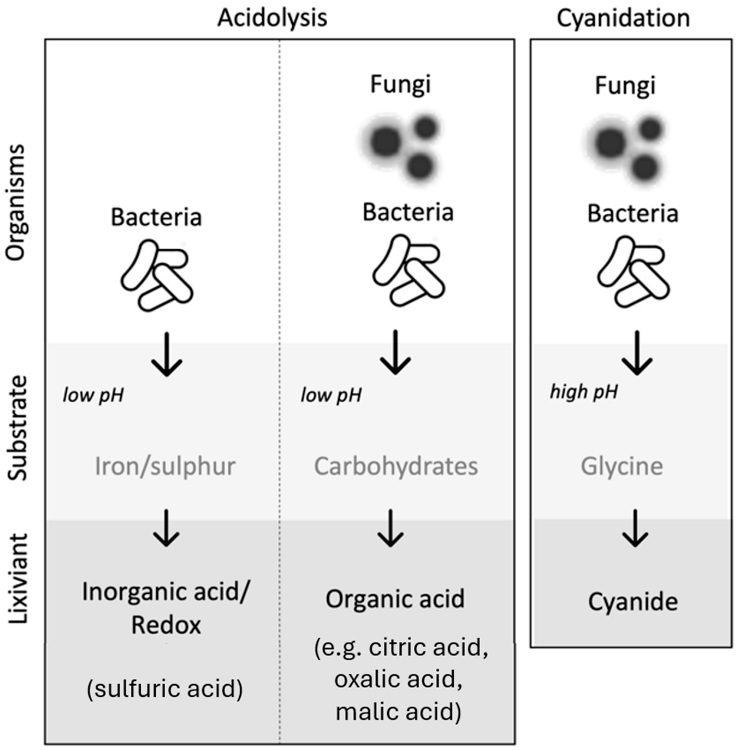

Figure 3. Main reaction types of metal bioleaching employed by fungal and bacterial species described in recent studies concerning e-waste, including examples of lixiviant compounds (as per Table 2).

Table 2. Recent advances in metal bioprocessing from e-waste sources using live biomass (NE—not explicit in literature; "-"—not explored in study; wPCB—waste printed circuit board; LEDL—light-emitting diode lamp; LCD—liquid crystal display; Au—gold; Ag—silver; Cu—copper).

Process(es)	Waste Type	Bioprocessing Agent	Pulp Density	Maximum Recovery of Select Metals			Metal Sorption in Biomass	Operational Volumes and Experimental Conditions	Valorization of Generated Products	Reference
				Au	Ag	Cu				
Bioleaching	wPCB	*Acidithiobacillus ferrooxidans* and *A. thiooxidans* consortium (2:1), *Aspergillus niger*	8%	-	-	93%	-	200 mL (flasks); Microorganism culture (9 K (bacterial) and Czapek Dox media (fungi), unspecified duration and conditions); bioleaching (10 days at 120 rpm, 308 K, pH 2)	Undiscussed	[53]
Bioleaching, Bioaccumulation	wPCB	*A. niger* and *Aspergillus* consortium	11 g in 1.6 L	56%	-	-	Biosorption only assessed at the end, not considered in the recovery calculation	1600 mL (Bioreactor); Bioleaching (38 discs 7 mm in diameter as initial inoculum, incubation up to 32 days, 19–23 °C, no agitation)	Undiscussed	[54]
Bioleaching, Biosorption, Bioaccumulation	wPCB	*Penicillium simplicissimum*	10 g/L	-	-	96.94%	Adsorbed metals considered recovered	1000 mL (Bioreactor); Microorganism culture and adaptation (10^8 spore inoculum in 1 L Bosshard medium, 1 week, 30 °C, followed by adaptation in 100 mL, 30 °C, 130 rpm to 40 g wPCB); bioleaching (8.8% (v/v) molasses, 0.29 L O_2/min for copper)	Undiscussed	[55]
Bioleaching, Biosorption, Bioaccumulation	wPCB	*A. fumigatus* A2DS	0.5%	-	-	62%	Adsorbed metals not considered in recovery calculations	100 mL (flasks); Fungi culture (7 days at 30 °C, 120 rpm); bioleaching (7 days at 20 to 40 °C, 120 rpm); centrifugation (10,000× g, 10 min)	Undiscussed	[56]
Bioleaching	wPCB	*Aspergillus* spp.	3 g/L	-	-	86%	Unexplored	500 mL (flasks); Fungi culture (sucrose medium, pH 5, 30 °C, 170 rpm); bioleaching (pH 5.7, 30 °C, 170 rpm, up to 33 days)	Undiscussed	[57]
Bioleaching	wPCB	*A. niger*	10 g/L	-	-	97%	Unexplored	100 mL (flasks); Fungi culture/adaptation (Bosshard medium, 1 mL spore inoculum 30 °C, 130 rpm); bioleaching (pH 5.15, 107 spore inoculum size, 30 °C, 130 rpm, 11.5 days)	Undiscussed	[50]
Bioleaching	wPCB	*A. niger*	2.5%	-	-	60%	Unexplored	100 mL (flasks); Fungi culture (14 days in glucose medium at 30 °C, 120 rpm, pH 6.5); bioleaching (7 days at 30 °C, 120 rpm, pH 3, 40.67 g/L $Fe_2(SO_4)_3$)	Undiscussed	[58]
Bioleaching	wPCB	*Frankia* consortium	0.2%	0.11 mg/g	0.09 mg/g	0.17 mg/g	Unexplored	Unspecified volume and architecture; fungal culture (up to 3 days, defined propionate minimal medium); bioleaching (28 °C for up to 33 days)	Undiscussed	[59]
Bioleaching	wPCB	*A. ferrooxidans*, *Acidiphilium acidophilum*	7.5 g/L	38%	46%	96%	Unexplored	500 mL (flasks); Bacterial culture (Silverman and Lundgren 9k and National Collection of Industrial Microorganisms 206 media, pH 2.5, 30 °C, 170 rpm for up to 48 h); bioleaching (pH 2.5, 30 °C, 170 rpm for up to 30 days)	Undiscussed	[60]
Bioleaching	wPCB	*A. ferrooxidans*	20 g/L	-	-	54%	Unexplored	1000 mL (Bioreactor); Bacterial culture (9 K medium, 30 °C, 130 rpm, 250 mL flasks, up to 15 g/L wPCB, and 30 °C, 1 L O_2/min, bioreactor, up to 40 g/L wPCB); bioleaching (1.5 vvm, 40 g/L ferrous sulfate, 30 °C for 9 days)	Undiscussed	[51]

Table 2. *Cont.*

Process(es)	Waste Type	Bioprocessing Agent	Pulp Density	Maximum Recovery of Select Metals			Metal Sorption in Biomass	Operational Volumes and Experimental Conditions	Valorization of Generated Products	Reference
				Au	Ag	Cu				
Bioleaching	wPCB	*A. ferrooxxidans*, *Pseudomonas aeruginosa*	1 g/L	20%	90%	-	Unexplored	500 mL (flasks); Culture (50 mL nutrient broth medium, 30 °C, 150 rpm 48 h); bioleaching (two step: growth phase—1 g/L glycine, pH 8, 20 h; bioleaching—pH 9, 30 °C, 150 rpm for 7 days)	Undiscussed	[42]
Bioleaching	wPCB	*P. balearica* SAE1	1%	-	36%	-	Unexplored	100 mL (flasks); Ferric chloride pre-treatment (50 °C, 150 rpm, 1–3 h); bioleaching (Luria broth medium, 5 g/L glycine, pH 9, 30 °C, 150 rpm, 8 days, following bacterial culture in same conditions, 2 days)	Undiscussed	[49]
Bioleaching	wPCB	*Leptospirillum ferriphilum*, *Sulfobacillus benefaciens*	1%	-	-	100%	Unexplored	100 mL (flasks) 2200 mL (Bioreactor); Bioleaching (3C medium, 3 g/L Fe(II), pH 1.2, 60 L (O_2/CO_2)/min, 650 rpm, 35 °C, for up to 25 days)	Undiscussed	[61]
Bioleaching	wPCB	*Magnetospirillum* sp., *M. gryphiswaldense*	NE	-	-	89%	Unexplored	Unspecified working volume (flasks); bacterial culture (*Magnetospirillum* growth medium or oxygen-sulfide gradient medium, up to 10 days); bioleaching (pH 6.7–7, 28 °C, 120 rpm for 12 days)	Undiscussed	[62]
Bioleaching	wPCB	*A. ferrooxidans*	7.5 g/L	-	-	100%	Unexplored	350 mL (flasks); Bioleaching (30 (*v*/*v*) fresh culture inoculum, pH 1.7, 30 °C, 130 rpm for 48 h)	Potential reuse of biomass in the process	[63]
Bioleaching	wPCB	*A. ferrooxidans*	3 g in 400 mL	-	-	80%	Unexplored	400 mL (bioreactor); Bacterial culture (900 mL 6 K medium supplemented with 30 g $FeSO_4 \cdot 7H_2O$/100 mL, pH 1.7, 30 °C, 200 rpm, 50 NL/h aeration for at least 48 h); bioleaching (bacterial culture pumping through column, 54 mL/min, pH 1.75)	Potential reuse of biomass in the process	[64]
Bioleaching	LEDL	*Bacillus Megaterium*, *A. ferrooxidans*	10 g/L	93%	91%	87%	Unexplored	50 mL (flasks); Ferric pre-treatment (9 K medium, pH 2, 29 °C, 140 rpm, followed by centrifuging at 12,000 rpm for 20 min, and application on LEDL, 20 g/L, 150 rpm, up to 10 days, followed by centrifuging, 8000 rpm 15 min and drying, 70 °C, 2 h); *Bacillus* culture (Nutrient broth medium with 10 g/L L-methionine, pH 7–11, 30 °C, 150 rpm for 24 h); bioleaching (2% (*v*/*v*) inoculum, 30 °C, 150 rpm for 4 days)	Undiscussed	[65]
Bioleaching	wPCB	*A. ferrivorans*, *A. ferrooxidans*, *P. putida* and *P. fluorescens*	1%	45%	-	96%	Unexplored	100 mL (flasks); Bacterial culture (DSM 882 medium, pH 2.5, 30 °C and King's B agar medium at 28 °C); copper bioleaching (10^6 CUF/mL inoculum, 28–30 °C, 180 rpm, up to 480 h); gold bioleaching (culture supplemented with 8 g/L glycine, 28 °C, 160 rpm for 120 h)	Undiscussed	[66]

Table 2. *Cont.*

Process(es)	Waste Type	Bioprocessing Agent	Pulp Density	Maximum Recovery of Select Metals			Metal Sorption in Biomass	Operational Volumes and Experimental Conditions	Valorization of Generated Products	Reference
				Au	Ag	Cu				
Bioleaching	wPCB	*Tissierella, Acidiphilium and Leptorspirillum*-majority consortium	10 g/L	28%	<0.25%	69%	Unexplored	3 L (Bioreactor); Bacterial culture (9 K medium, pH 2, 150 rpm, 28 °C for 46 days, with re-seeding for up to 9 weeks); bioleaching (4.5 K medium, pH 2, 10% (*v*/*v*) inoculum, 30 °C, 150 rpm, 500 mL O_2/min for 5 days until desired bacterial density, and further 18 days in contact with wPCB)	Undiscussed	[67]
Bioleaching	wPCB	*L. ferriphilum*, *S. benefaciens*	1%	-	-	96%	Unexplored	2.25 L and 150 mL (stirred and bubble column bioreactors); continuous bioleaching (bubble column: 0C medium with 1 g/L Fe(II), 15 g activated charcoal, 10% (*v*/*v*) inoculum, pH 1.1, 1% CO_2 airflow 20–30 L/h, 36 °C; stirred tank: 47 mL/h feed rate, STR 48 h, pH 1.5, 600 rpm, 36 °C, 60 L O_2/h for up to 20 days)	Undiscussed	[68]
Bioleaching	wPCB	*Acidithiobacillus* spp.-dominated consortium	1%	-	-	89.09%	Unexplored	200 mL (flasks); Bacterial culture (modified 9 K medium, pH 2.0, 10% (*v*/*v*) inoculum, 30 °C, 120 rpm); bioleaching (25–40 °C, 120 rpm, up to 20 days)	Undiscussed	[69]
Bioleaching	wPCB	*A. ferrooxidans*, *L. ferrooxidans*	5%	-	-	94%	Unexplored	100 mL (flasks); Bacterial culture (DSMZ 882 medium, 120 rpm, 30 °C for 5 days); bioleaching (48 h of isolated growth, followed by insertion of wPCB and substitution of 80% of volume by new medium, followed by leaching at 30 °C, 120 rpm, pH 1.6 for 11 days)	Undiscussed	[70]
Bioleaching, Biosorption	wPCB	*Lentinus edodes*, *Pleurotus florida*, *Ganoderma lucidum*, *A. niger*, *Trametes versicolor*, *Streptomyces* spp., and *Pseudomonas* spp.	5 g/L	-	-	18%	Single Cycle sorption/desorption reported	Unspecified working volume (flasks); bioleaching (mushroom minimal fungal media, pH 6.5, 25 °C, or M9 bacterial media, pH 7.2, 27 °C, 8 days, 12 subsequent for biosorption); desorption (50 mL 0.05 M nitric acid, 160 rpm for 24 h)	Undiscussed	[71]
Bioleaching, Bioaccumulation, Bioprecipitation	wPCB	*A. ferrooxidans*, *A. thiooxidans*, *L. ferrooxidans*, *P. putida*, *Fusarium oxysporum*, *B. Cereus*	NE	50%	-	100%	-	100 mL (flasks); Bioleaching (Culture medium supplemented with 200 mM Fe(II), pH 2.5, 30 °C; secondary bioleaching—dry residues used, pH 7.3, 35 °C 150 rpm; both cultures inoculated with 107–108 cells/mL); nanoparticle synthesis (30 °C, 160 rpm for 96 h; secondary treatment at 37 °C 200 rpm for 24 h; metal recovery through centrifugation, 30 min, 14,700 rpm)	Undiscussed	[72]
Bioleaching	wPCB	Unspecified heterotrophic bacterial consortium	10 g/L	-	-	4%	Unexplored	60 mL (flasks); Microbial cultures and adaptation (nutrient medium, pH 3, 30 °C, 150 rpm for 7 days, adaptation with up to 100 mg/60 mL wPCB for 21 days); bioleaching (10% (*v*/*v*) inoculum, 30 °C, 150 rpm for 15 days)	Undiscussed	[73]

Table 2. *Cont.*

Process(es)	Waste Type	Bioprocessing Agent	Pulp Density	Maximum Recovery of Select Metals			Metal Sorption in Biomass	Operational Volumes and Experimental Conditions	Valorization of Generated Products	Reference
				Au	Ag	Cu				
Bioleaching	wPCB	*B. megaterium*, *Bacillus* sp., *B. amyloliquefaciens*, *B. safensis*, *Brevibacterium frigoritolerans*, *Chryseomicrobium amylolyticum*, *Lisynibacillus sphaericus*	10 g/L	73.6%	-	87.5%	Unexplored	100 mL (flasks); Bioleaching (2 × 10^8 CFU/mL inoculate, medium supplemented with 5 g/L glycine, pH 9, 30 °C, 150 rpm for 2 days for growth followed by 7 days with wPCB)	Undiscussed	[74]
Bioleaching	wPCB	*A. ferrooxidans*	50 g/L, 800 mL replaced by fresh medium after 48 h	-	-	NE	Unexplored	100 mL (flasks); Bioleaching (DSMZ 882 medium, pH 1.6, 30 °C, 120 rpm; growth for the first 48 h; first leaching step with wPCB, 48 h; replacement of 80% of solution with fresh medium and incubation up to 168 h)	Undiscussed	[75]
Bioleaching	wPCB	*A. aquatilis*	2%	-	-	169.45 mg/g (flask), 132.55 mg/g (reactor)	Unexplored	100 mL (flasks) and 250 mL fluidized bed bioreactor; Bacterial culture (Nutrient broth media, 28 °C, 80 rpm with acclimation up to 5% wPCB pulp density, 24 h per step); bioleaching (28 °C, 100 rpm for 96 h in flasks, 3 LPM airflow rate, 28 °C for 96 h in the bioreactor)	Undiscussed	[76]
Bioleaching	wPCB	*Bacillus* sp.	1%	75%	63%	86%	Unexplored	3 L (flask); Bacterial growth (nutrient broth supplemented with 5 g/L of amino acids, 4 × 10^8 CFU/mL inoculum, pH 9, 40 °C for 30 h); bioleaching (nutrient broth supplemented with 5 g/L of amino acids, 1 mg/L methionine and 5 g/L glycine, pH 9, 40 °C, 180 rpm)	Undiscussed	[77]
Bioleaching	wPCB	*A. niger* and *A. tubingensis* mixed culture	3 g/L	-	-	86%	Unexplored	500 mL (flasks); Fungal culture (Sucrose medium, pH 5, 30 °C, 170 rpm); bioleaching (two step: 30 °C, 170 rpm, for 72 h, followed by the addition of wPCB and 33 days of incubation)	Undiscussed	[57]
Bioleaching	LCD	Acidophilic bacterial consortium (*L. ferriphilum*, *L. ferrooxidans*, *A. ferrooxidans*, *A. ferridurans*, *A. ferriphilus*, *A. ferrivorans*, *A. ferrianus*, *S. thermosulphidooxidans*, *S. acidophilus*, *A. caldus*, *A. thiooxidans*, *A. ambivalens*, *Ferroplasma* spp.)	32.5 g/L	-	-	-	Unexplored	Flasks (volume undisclosed); Bacterial culture (acidophile basal salts and trace elements medium, 107 cells/mL inoculum, pH 1.8, 30 °C, 130 rpm); bioleaching (adaptation with up to 32.5 g/L wPCB over 14 days, 108 cells/mL inoculum; leaching under same conditions, 30 °C, 130 rpm for 14 days)	Undiscussed	[78]

Table 2. *Cont.*

Process(es)	Waste Type	Bioprocessing Agent	Pulp Density	Maximum Recovery of Select Metals			Metal Sorption in Biomass	Operational Volumes and Experimental Conditions	Valorization of Generated Products	Reference
				Au	Ag	Cu				
Bioleaching, Biosorption	wPCB	*A. ferrooxidans*, *L. ferrooxidans* and *L. ferriphilum* consortium (encapsulated in alginate beads, bioleaching); *Saccharomyces Cerevisiae* (biosorption)	8%	98% selective biosorption	-	100%	Single Cycle sorption reported	0.5 to 2 L working volume (2-chamber bioreactor); Bacterial culture (M16 medium, supplemented with 25% $FeSO_4 \cdot 7H_2O$ (w/v), pH 1.2; encapsulation in alginate-based beads); bioleaching (1 vvm air inflow, 31 °C, pH 1.5 (leaching) or 1.7 (bio-oxidation), 700 rpm (leaching) or 600 rpm (bio-oxidation)); gold biosorption (10% (w/v) yeast inoculum dispersed by ultrasonic bath, 60 s, on aqua-regia dissolved metal mixture, incubated 15 min at 150 rpm, followed by centrifugation, 4430× g)	Undiscussed	[79]

As per Table 2, recent investigations have employed both bacterial and fungal biomass, with bacterial-based approaches representing the bulk of bioleaching research. Of those, 16 studies used chemolithotrophic bacteria (redox bioleaching), whereas a relative minority used heterotrophic organic acid producing bacteria for the solubilization of metals from e-waste sources. Fungal-based bioleaching has been relatively less explored overall. Some advantages of the use of fungal species, both unicellular and filamentous, include high tolerance to the presence of several metals, even when compared to bacterial species [80], the plasticity to stand a wide range of pH, and the relatively higher production and secretion of metabolites, some of which are used in bioprocessing and in potential valorization processes [81]. The milder acidic conditions associated with fungal metal bioprocessing in comparison, for example, with many bacterial-based processes, especially those dependent on chemolithotrophic bacteria, may also contribute to reducing the potential costs associated with effluent treatments (pH as low as 2, but generally over 5, compared to pH as low as 1 for the latter). As per Table 2, common genera of fungi investigated for metal bioprocessing, be it through leaching, sequestration, or precipitation of metals in their surroundings, include *Penicillium* and *Aspergillus*.

Over the past years, both bench- and bioreactor-scale studies have been conducted using fungal species, as shown in Table 2 (Operational Volumes and Experimental Conditions). While these studies have demonstrated the potential of fungi for metal bioleaching from e-waste, they have also revealed challenges related to long incubation times, the need for carbon supplementation, and the potential for the biocrystallization of metals. For example, Arshadi et al. [50] achieved high bioleaching efficiencies for copper and nickel (97% and 74%, respectively) from ground computer PCBs using *Aspergillus niger*. However, the process required relatively long incubation times post inoculation for optimal efficiency (up to 4.5 days) and faced challenges with metal solubilization and biocrystallization due to reactor size and incubation period, potentially hindering metal recovery. Specifically, metal solubilization fluctuated significantly throughout the 19-day incubation period. Copper recovery, specifically, varied from a minimum of slightly over 60% on the 12th day of incubation to close to 100% on the 2nd and 16th day. This fluctuation was attributed to the biocrystallization of the solubilized metals through their reduction to their elemental forms, inducing their (bio)precipitation, resulting in lower concentrations of metals in the solution. Thus, although *A. niger* has demonstrated high leaching efficiency for copper and nickel under these conditions, these biocrystallization events may hinder efforts towards the recovery of the leached metals after the leaching process. Thus, knowledge of processes affecting metal solubilization after their bioleaching, such as biosorption, bioaccumulation, and bioprecipitation, is also crucial to understanding the impact of these challenges on further development. However, these considerations were out of the scope of the study, a knowledge gap evident across most studies analyzed in this review.

Different bioreactor architectures may also help reduce this problem by enhancing the mass and heat transfer associated with e-waste loading and filamentous fungal biomass, as demonstrated by Nili et al. [55]. Using a bubble column reactor, the aeration rate was found to be a key factor in order to counter the increased medium viscosity that results from increased pulp densities and the use of filamentous fungi (in this study, *Penicillium simplicissimum*). High bioleaching efficiencies of 96.94% and 71.51% for copper and nickel, respectively, were recorded in an experiment using molasses as a cost-effective alternative to sucrose and glucose. Carbon sources had different effects on the bioleaching of copper and nickel, being one of the most significant parameters for the bioleaching of nickel, but not for copper. This is due to the different complexation extent between the organic acids produced and the different metals—a difference in nutrition may induce the release of different organic acids and thus result in variable extraction rates for different metals, which could pose both an opportunity and a challenge in the treatment of metal mixtures. Specifically, oxalic acid was identified as the main organic acid for the purposes of nickel bioleaching; however, molasses concentration was also found to correlate negatively with this organic acid's secretion. As such, this study necessitated the optimization of molasses concentration. The pH also affected these two metals' leaching differently, with slightly more acidic conditions promoting nickel leaching, whereas copper leaching was optimal at more alkaline conditions. The presence of nickel also had a negative effect on the biosorption efficiency of copper in fungal biomass. For copper bioleaching, optimal conditions were 8.8% (v/v) of molasses and an aeration rate of 0.29 L/min; 1.9% (v/v) and 0.37 L/min for optimal nickel extraction. The lengthy operational times remain a downside, however, especially if considering the pre-processing spore and adaptation phases (1 week for the initial culture, followed by an acclimation to the waste), possibly contributing to higher operational costs. However, despite advantages such as high metal tolerance, higher biomass generation than bacterial species, and the resulting versatility of biomass valorization, the application of fungi in e-waste bioprocessing is currently limited by the slow rate of organic acid production and the need for higher carbon supplementation, both potentially contributing to incurred expenses. This may be the reason why, overall, fungal species are less investigated for e-waste bioprocessing compared to bacterial species.

Like fungi, bacterial species' strategies for metal bioprocessing are vast, including bioleaching through environment acidification, biosorption, bioaccumulation, biosurfactant production, and bioprecipitation. Some bacterial genera, such as *Acidithiobacillus*, thrive in high metal environments, thus having acquired an evolutionary advantage in resistance to these often highly toxic contaminants [82]. Bacterial species have another advantage over fungi, in that some bacterial-mediated bioprocessing processes, especially concerning bioleaching, seem to occur at faster rates, allowing for a quicker process in obtaining satisfactory yields. This can be observed in Table 2, and a direct comparison between these two types of microorganisms has been recently conducted regarding copper bioleaching using bacterial *A. thiooxidans* and *A. ferrooxidans* (in a 1:2 consortium), compared to *Aspergillus niger* [53]. Furthermore, the necessity for carbon supplementation when using live bacterial biomass may also be reduced compared to fungi, potentially contributing to the reduced operational costs for this industrial activity. As such, studies directly comparing bacterial- and fungal-based approaches are essential. Such is the case with Abhilash et al. [53], testing the copper bioleaching from tested wPCBs using both a bacterial consortium (*A. thiooxidans* and *A. ferrooxidans*) and *Aspergillus niger*. The bacterial consortium achieved significantly higher copper solubilization (93%) compared to the fungus (66%) under similar conditions (pulp density, 120 rpm shaking at 308 K for 10 days), which can be attributed to the bacteria's ability to create a stronger acidic environment (despite a similar initial pH of 2). The study also investigated the effects of pulp density, temperature, and pH on bioleaching efficiency, observing a general trend of increased metal solubilization at lower pH and higher temperatures. Importantly, the authors explored the kinetics of copper bioleaching, finding that it followed a chemical reaction control model (i.e., the rate-limiting step is the chemical reaction at the surface of the waste particles, explained by $kt = 1 - (1 - x)^{\frac{1}{3}}$). In regard to downsides, the proposed approach highlights a potential environmental concern: the use of a solvent-based solubilization method for metal recovery employing potentially hazardous extractants (e.g., 5-nonyl-2-hydroxy-benzaldoxime, an environmentally hazardous and potentially acutely toxic/irritant solvent [83]). This may raise questions about the overall sustainability of the proposed process and emphasizes the need for further research into more environmentally friendly metal recovery methods. On the other hand, in the case of *A. niger* bioleaching, it is important to highlight that increasing pH to 4 resulted in a slight, single-digit decrease in *A. niger*'s leaching of most tested metals, except for aluminum, whose efficiency increased by 2% to 39%. These modest decreases underline how metal leaching efficiency in single-step processes and environmental sustainability could be factored into the optimization of such processes.

Just as with fungal bioleaching, although most recent approaches still use small-scale flasks under stirring, some studies exploring alternative bioreactor architectures have been conducted, as is the case with Arshadi et al. [51]. The exposure of wPCB fragments to *A. ferrooxidans* in a stirred tank bioreactor was found to result in simultaneous solubilization of up to 54%, 75%, and 55% for copper, nickel, and iron, respectively (and up to 100% for each, separately, at different times during the 20-day leaching period). In this study, medium viscosity was once again a considerable factor, mainly caused by increased pulp densities; conversely, and as observed by Nili et al. [55], discussed previously, increased aeration rates (in this case, of up to 3 vvm) were found to have positive effects on mass transfer, compensating for increased medium viscosity (as well as compensating for lower concentrations of iron sulphate necessary for the production of the acidic environment). However, the increase in the aeration rate may also result in potential negative effects on biomass viability and growth, attributed to increased cellular attrition, highlighting the importance of careful control and monitoring of bioreactor conditions. Another key finding was the differential stability of metals in the bioleached liquor, an occurrence similar to the biocrystallization of copper and nickel reported by Arshadi et al. [50]. In the present study, copper was found to be more stable in its ionic form than nickel and iron, with higher precipitation rates [51]: while iron and nickel extraction fluctuated between 0 and 100% and 0 and 96% throughout the 21-day bioleaching period, copper extraction demonstrated a more direct correlation with time, up to the 13th day, subsequently stabilizing above 80%. This differential stability could pose challenges for automation and the recovery of specific metals from mixed e-waste streams. Furthermore, a decline in leaching efficiency after 20 days of incubation was observed, likely due to the precipitation of iron as jarosite, a frequent problem in bioleaching that can hinder the diffusion of leaching agents and impede metal extraction. This was reflected in the variation of metal contents in the e-waste sample before and after the experiment, with aluminum, copper, nickel, titanium, and zinc decreasing from 4.27, 44.58, 1.21, 0.29, and 4.54% (w/w) to 0.6, 20, 0.4, 0.27, and 0.6%, compared to iron, whose concentration in the bioleached residue was reported to increase (from 7.50% to close to 15%). This limitation underscores the need for strategies to minimize events such as jarosite formation to improve the efficiency and sustainability of bioleaching processes and prevent contamination of the processual effluents.

Multi-step bioleaching processes involving sequential leaching stages, with retrieval of bioleaching liquor and replenishment of culture medium, may represent a potential strategy able to prevent or minimize the formation of precipitates like jarosite. Such is the case described in Jagannath et al. [84], wherein a sequential 5-step bioleaching process was attempted using a pulse plate bioreactor for the *Acetinobacter* sp.-mediated bioleaching of copper from mobile phone wPCBs. This process achieved 63.5% metal solubilization efficiency overall (compared to 23% in a single batch experiment). In this approach, it was mainly pulsing frequency, rather than aeration rate, that was used to make up for increases in medium viscosity induced by e-waste loading, facilitating mass transfer up to a frequency of 0.2 s^{-1}, above which detrimental effects on bioleaching efficiency were observed, likely due to a decrease in cell viability. Furthermore, the study revealed significant metal adsorption onto the bacterial cell walls, which could affect both metal recovery and biomass valorization. The main downside of this approach is that the pulsed plate bioreactor design requires continuous energy input for aeration and pulsing, potentially increasing operational costs. Additionally, the potentially high alkalinity of the generated effluents (pH 12 was stated as the ideal condition for bioleaching with this species) raises concerns about the environmental impact and the need for additional treatment steps. This problem is compounded by the substantial amounts of medium required for the process—to obtain the 65.3% efficiency stated above, six sequential batches of 1.5 L each are required.

Beyond optimizing bioreactor configurations and process parameters, genetic engineering and microbial consortia offer promising opportunities for enhancing e-waste bioprocessing. Genetic engineering could improve microbial tolerance to toxic e-waste components, increase metal-specific bioleaching capabilities, or enhance biofilm formation for better cell adhesion to e-waste particles [85]. Microbial consortia, comprising different species with complementary functions, could enable the simultaneous bioprocessing of various e-waste constituents. However, more investigations are needed to assess the viability of such approaches. In Nasiri et al. [73], two naturally soured consortia (one obtained from metal-rich iron ore mine tailings and another from surface soil) were used in a bioprocessing approach for wPCB particles, but presented low prospects. The bacterial consortium sourced from the surface soil achieved a higher efficiency in copper solubilization than the one obtained from the metal-rich iron ore mine tailings, but less total solubilization of metals overall (4%, compared to 4.7%). However, this bioleaching efficiency remained low (4% and 3% of copper over 15 days of incubation for the surface soil and mine tailing microorganisms, respectively) compared with other studies (Table 2). These recoveries were recorded at a commonly used 10 g/L (or 1%) solid–liquid ratio/e-waste pulp density. Of note is also that a decrease in pulp density (to 5 g/L) did not yield higher recovery percentages for total metals (3.6% and 1.8% for microorganisms isolated from surface soil and mine tailings, respectively), which could suggest a limited contact between the microorganisms and the added e-waste, and challenges related to mass transfer at such low concentrations. On the other hand, in line with most research, recovery rates also decreased when pulp densities were raised to 15 g/L (3.3% and 3.7% when using the surface soil and mine tailing microorganisms, respectively). These low efficiencies, attributable to low medium acidity and potential e-waste toxicity on the microorganisms used (denoted by decreases in optical density as wPCB concentration increased during a previous adaptation phase) were, however, accompanied by mass loss and apparent surface degradation of the inserted waste particles. This suggests that the microorganisms used have the capacity for the deterioration of the non-metal constituents of the used waste. This, in turn, could indicate this consortium's potential for a more integrated approach capable of targeting multiple e-waste constituents if the metal leaching capabilities can be improved, warranting further exploration. In addition, the study did not explore the mechanisms by which the consortium interacts with and removes non-metallic e-waste constituents. Further research is needed to elucidate these mechanisms and assess the potential for the simultaneous bioprocessing of metals and plastics.

In sum, overall, fungal-based bioleaching processes mainly depend on producing weak organic acids, requiring longer operational times to obtain satisfactory yields, which often fall short of those from bacterial-based approaches, as per Table 2. Although the generation of higher amounts of biomass may represent an advantage from a potential biomass valorization standpoint, the nutritional supplementation needed to support it may be the main cost associated with these processes. However, alternative feeds, such as chickpea, lentils, and split pea-based formulations (without any supplementation with further carbon sources), for example, were found to be reliable alternatives to the media traditionally used for fungal growth of *Penicillium* sp. and *Aspergillus* sp. [86,87]. Going forward, such studies could be conducted using products with low shelf-life that are more likely to become food waste. Additionally, the ability to degrade plastic constituents and use them as a carbon source may help mitigate this, while also further remediating complex e-waste sources. However, to that end, more explorations of fungi in e-waste plastic biodegradation (e.g., Ferreira-Filipe et al., 2024 [88]) are needed.

Bacterial-based bioleaching, on the other hand, has been the main type of study conducted on the bioprocessing of e-waste metals within this review's target timeframe owing to their greater versatility and generally higher yields (Table 2). However, like fungal species, direct bacterial-based bioleaching may also incur concern over the toxicity of e-waste in the microorganisms used, potentially disrupting their growth and metabolic activity and thus having a negative effect on yields. Thus, increasing resistance to toxic compounds often found in e-waste is essential to increase both yields and pulp densities, thereby increasing the waste processing capacity of these approaches. Another concern more pressing with bacteria than fungi is the hazardousness of the processual effluents. *Acidithiobacillus* is, as per Table 2, a popular bacterial genus, given its ability to generate strong acidic environments, often enabling higher yields. However, the resulting processual effluents may also be considerably more acidic and iron-rich, potentially having a negative effect on the environmental sustainability of the process and incurring additional treatment and safety-related costs. Nevertheless, given their simpler genetic nature, higher yields, lower nutritional requirements, faster processing times, and bio-lixiviant variety (Figure 3), bacterial-based approaches seem to be currently regarded as the ones with the most potential overall. Both types of biomass have the potential to be used in processes with high metal yields, although bacterial-based approaches consistently retain the upper hand, whereas the use of fungal biomass results in comparatively lower leaching rates. However, given the wide variety of waste sources and bioprocessing approaches, comparison between the two types of biomass remains difficult outside of studies where both are tested. Compared to industrially applied techniques, as exemplified in the previous chapter, the use of live biomass with lower pulp densities has the potential to match and exceed industrial yields. Increasing pulp density may exhaust the lixiviant, inhibit biomass activity due to increased medium toxicity, and result in higher medium viscosity, thus remaining one of the biggest challenges in the field. Other challenges faced by both types of biomass include sterilization and other steps to avoid contamination and later effluent treatments, contributing to additional environmental footprint and costs. Additionally, as per Table 2, studies and discussions concerning the valorization of the used/generated biomass after its use for bioprocessing are uncommon. The use of fungi as a source of plant cell wall-degrading enzymes, for example, with the intent of bioenergy production, has been largely explored (e.g., Monclaro et al., 2022 [89]). However, the occurrence of bioaccumulation and biosorption of metals during bioprocessing and their impact on metal recovery have seldom been discussed thus far, and their potential effects on biomass valorization have not been ascertained. Bioaccumulation of leached metals, rather than just their biosorption on the surface of microorganisms through weak bonds, may also further frustrate valorization efforts [90]. One possibility to minimize this problem is that of a spent medium strategy, wherein microorganisms are grown with the intent of producing metabolites for bioleaching, but removed from it before waste treatment, to be discussed in Section 2.3 (Figure 4 presents a comparison of these approaches).

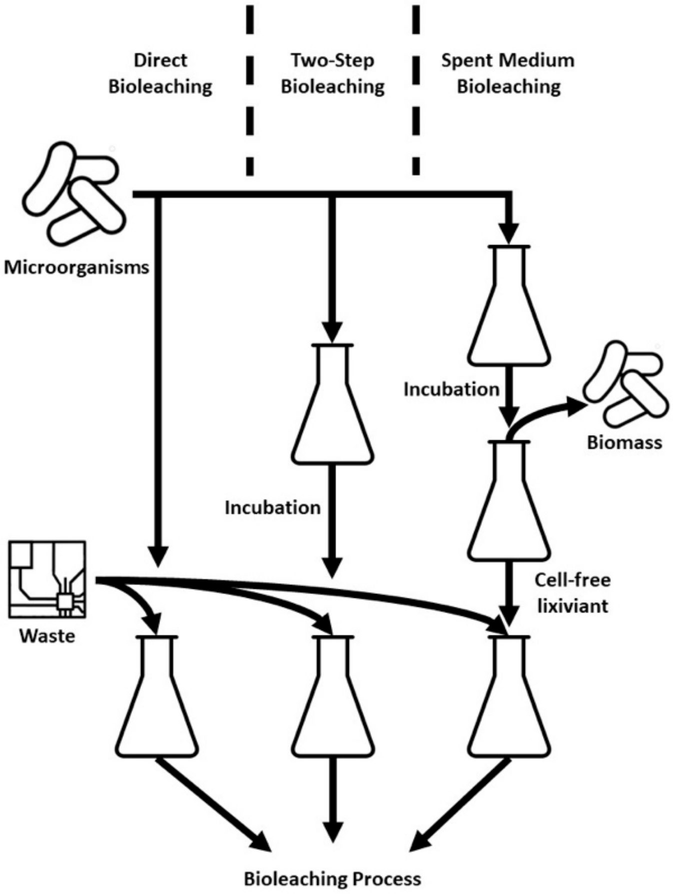

Figure 4. Comparison between direct bioleaching, two-step bioleaching, and spent medium bioleaching approaches regarding the inclusion and removal of biomass and waste in/from the leaching solution at different stages during the process.

2.3. Use of Biogenic Lixiviants in E-Waste Bioprocessing

Biogenic compounds are naturally occurring substances originating from living organisms, waste streams, or biological processes (e.g., biological synthesis or conversion) and may be used for e-waste bioprocessing. Much like the approaches described in the previous section, such processes make use of microorganisms to produce the lixiviants for the bioleaching process, with a key difference being that the biomass used for their production is separated prior to their use. The compounds produced may be purified and commercialized before use (e.g., commercialized hydrogen peroxide [91] or used directly for the bioprocessing experiments [92]. In this sense, biogenic lixiviants used in recent investigations fit into two main categories, spent (culture) medium (Figure 4) or (purified) biogenic compounds (e.g., organic acids, bio-cyanide). These studies are explored in Table 3.

Table 3. Recent advances in metal bioprocessing from e-waste sources using biogenic lixiviants (NE—not explicit in literature; "-"—not explored in study; wPCB—waste printed circuit board; LEDL—light-emitting diode lamp; Au—gold; Ag—silver; Cu—copper).

Process(es)	Waste Type	Bioprocessing Agent	Pulp Density	Maximum Recovery of Select Metals			Operational Volumes and Experimental Conditions	Valorization of Generated Products	Reference
				Au	Ag	Cu			
Bioleaching	wPCB	Biogenic thiosulphate, ammonia	5 g/L	65%	-	-	Unspecified working volume (flasks); bio-thiosulphate production (30 °C, 140 rpm, over 7 days); bioleaching (160 rpm, up to 48 h); centrifugation (10,000 rpm, 4 °C. 10 min)	Undiscussed	[93]
Bioleaching	wPCB	Glucose oxidase-based bio-Fenton reaction	1 g/L	-	-	100%	500 mL (flasks); Bioleaching (up to 144 h, 335 rpm at 30 °C, with 20 mM D-Glucose, pH 3.5, 300 U/L glucose oxidase, 10 mM Fe^{2+})	Undiscussed	[94]
Bioleaching	LCD	*Aspergillus niger* biometabolite solution	10 g/L	-	-	-	100 mL (flasks); Lixiviant solution preparation (1 mL spore inoculum size, 30 °C, 130 rpm, undisclosed duration); thermal treatment (1100 °C for 70 min, cooling to 500 °C and reheating to 700 °C); bioleaching (2 days, 70 °C)	Undiscussed	[95]
Bioleaching	wPCB	Biogenic hydrogen sulfide gas	1%	-	-	99%	0.46 L (Fluidized bed reactor); leaching (597 days, using either lactate or glycerol as electron donor, pH 4.5, 35 °C)	Undiscussed	[96]
Bioleaching	wPCB	Biogenic lixiviant solution (including citric acid, produced by *Yarrowia lipolytica)*	10 g/L and 50 g/L	-	-	-	Flasks (undisclosed volume); lixiviant production (9 days at 140 rpm and 30 °C); bioleaching (Up to 9 days, pH 7, 80 g/L carbon source, at 60 °C and 140 rpm)	Undiscussed	[97]
Bioleaching	wPCB	Biogenic lixiviant solutions (produced by *Acidithiobacillus thiooxidans* and *Acidithiobacillus ferrooxidans*)	10%	-	-	100%	250 mL (flasks); lixiviant solution preparation (0 K medium supplemented with elemental sulfur at 10 g/L, 2.5 × 10^8 cells/mL inoculum (*A. thiooxidans*)), 9K medium supplemented with 44.2 g/L $FeSO_4 \cdot 7H_2O$, 30 °C, 3.0 × 10^8 cells/mL inoculum (*A. ferrooxidans*); bioleaching experiment (40 °C, pH 1.1, 220 rpm)	Undiscussed	[98]
Bioleaching	wPCB	Ferric sulphate solution (bioregenerated by *Acidithiobacillus ferrooxidans* and *Leptospirillum ferrooxidans)*	10%	-	-	99%	170 mL (Flooded packed bed bioreactor); bioleaching (40 g/L Fe(III), 60 °C and 0.5 L/h O_2 for 24 h)	Biomass recycling discussed, unexplored	[99]
Bioleaching	LCD	Biogenic lixiviant solution (*Aspergillus niger* spent medium)	10 g/L	-	-	-	100 mL (flasks); Bioleaching solution preparation (1 mL spore inoculum incubated in Czapek–Dox medium, 14 days at 130 rpm, 30 °C); bioleaching (100 mL spent medium, 160 rpm, 70 °C for 29 h).	Undiscussed	[92]
Bioleaching	wPCB	*A. acidophilum* NCIM 5344	NE	-	-	100%	120 mL (flasks); Bio-lixiviant preparation 9 K medium, pH 3.5, 150 rpm, 30 °C for 10 days (spent medium, supplemented with 15 mL H_2O_2, 30 °C, 150 rpm for 4 h, 60 °C for 2.5 h)	Undiscussed	[100]

A case study by Parsa et al. [92] exemplifies the use of spent culture medium for metal bioleaching. The authors employed *Aspergillus niger* to produce an organic acid mixture containing, among others, oxalic acid, whose production peaked at 14 days of incubation. This mixture was then separated from the biomass and used to leach aluminum, arsenic, strontium, and indium from the LCDs of several manufacturers. This study demonstrated the potential for recovering multiple metals from e-waste using a biogenic lixiviant while also investigating the leaching kinetics for each metal under the tested parameters, revealing that different metals followed different kinetic models. Briefly, aluminum leaching initially followed a mixed mechanism model, likely involving interfacial transfer and diffusion ($kt = \frac{1}{3}ln(1-x) + [(1-x)^{-\frac{1}{3}} - 1]$, where k represents the reaction rate constant (day^{-1}), t the bioleaching time (day), while x represents the extracted fraction), followed by a solid product layer diffusion model after 12 h ($k(t-t_1) = 1 - \frac{2}{3}\left(\frac{x-x_1}{1-x_1}\right) - \left(\frac{1-x}{1-x_1}\right)^{\frac{2}{3}}$, with t_1 representing the time, in days, of the change in mechanism, and x_1 the extracted fraction at t_1); this suggests that in an initial stage, the bioleaching of this metal depends mainly on reactions of the organic acids on the e-waste powder surface. Arsenic extraction was rapid within the first minutes, then slowed in the next 12 h, fitting a mixed mechanism model dominated by interfacial transfer and diffusion through the product layer ($kt = \frac{1}{3}ln(1-x) + [(1-x)^{-\frac{1}{3}} - 1]$ and $k(t-t_1) = -ln\left(\frac{1-x}{1-x_1}\right)$ during the first and second phase, respectively). Indium bioleaching also followed this model, with potential limitations due to calcium oxalate precipitation. Finally, strontium leaching initially fits the mixed mechanism model, transitioning to a liquid boundary layer diffusion model ($k(t-t_1) = 1 - \left(\frac{1-x}{1-x_1}\right)^{\frac{2}{3}}$), and ultimately experiencing limitations due to oxalate complexation and precipitation. These findings highlight the complexity of metal bioleaching from complex sources and the need for tailored approaches for different target metals. While the study achieved promising metal solubilization rates (81.4% for aluminum, 69.1% for arsenic, 60.0% for indium, and 33.3% for strontium), it did not address the recovery of solubilized metals or the potential reuse of fungal biomass. Furthermore, the bioleaching solution's high initial acidity (pH 1.9) raises concerns about its potential environmental impact and the need for further downstream treatment.

Several investigations have been conducted on biogenic compounds, such as lactic acid, hydrogen peroxide, thiosulphate, and iron sulphate, rather than complex spent media retrieved after microbial culture (for example, Pourhoussein and Mousavi, and Kremser et al. [93,101], among others, described in Table 3). Generally, these consist in effective processes with high metal solubilization rates, with simple workflows, but all sporting the same concerns as discussed before, mainly in the way of energy expenditure due to temperature maintenance and stirring, as well as potential costs relating to effluent treatment. The inclusion of potentially hazardous compounds in the processes, such as ammonia and hydrogen peroxide, may also incur concerns over reagent transport and storage, as well as worker safety—concerns applicable, for example, to the approach detailed in Cecchi et al. [91]. In this study, up to 236.908, 17.112, 0.321, 0.072, and 0.002 mg of copper, nickel, zinc, lead, and gold were recovered per gram of e-waste used in the experiment, using 2.0 M of lactic acid at a 1:1 molar ratio with hydrogen peroxide. Increased reagent concentrations, on the other hand, failed to yield higher recoveries in the 12 h leaching period. Specifically, the main disadvantages of this approach relate to the relatively high molarity of hydrogen peroxide needed for gold solubilization. In this study, the molar ratio of hydrogen peroxide to lactic acid for the complete peeling of gold from wPCB fragments over 12 h was at least 1:1 (lower ratios negatively impacted efficiency, whereas higher ratios had a negligible positive impact). This could result in a considerable amount of hydrogen peroxide in the process, at potentially high temperatures and shaking speeds (although said temperatures were not disclosed in the study), given that concentrations of the lactic acid used could reach as high as 6 M (minimum of 1.0 M). Additionally, the reagents used were not produced locally (but purchased from Carlo Erba Reagents GmbH), so the valorization of food waste and biomass as an integral part of this e-waste treatment approach was not explored, which could be considered in further investigations. Instead, the transportation issue is highlighted: the producer of this reagent is in Germany, whereas the authors are affiliated with Italian and Canadian institutes. It follows that in the absence of local production solutions, the carbon footprint of this process would increase, as would any safety concerns and costs associated with the transportation of reagents.

Biogenic compounds have also been explored in conjunction with biochar. Kadivar et al. [36] (detailed in Table 1) describe the use of oak tree biochar, combined with biogenic sulfuric acid produced by *Acidithiobacillus thiooxidans* and chemical sulfuric acid in bioleaching experiments, achieving solubilization of up 98% and 82% of copper and nickel. Copper solubilization was negatively affected by pulp density (amount of acid being a limiting factor), as well as by the depolymerization of wPCB constituents (resulting in medium alkalinization), demonstrating that the increase in pulp density also represents a challenge in biomass-free methods. Biochar concentration also contributed to this, due to the plentiful presence of carboxyl groups in its surface, affecting overall pH. Although a high pH may inhibit the bioleaching of certain metals, it may, on the other hand, provide suitable conditions for the leached metals' biosorption. Given the calculation of copper recovery in Kadivar et al. [36], the biosorption potential was disregarded while considering only the solubilized metals, and higher biochar concentrations were linked to a loss in solubilization efficacy. However, the use of biochar for the biosorption of the solubilized metals towards their separation from the bioleaching liquor and later retrieval is an approach that should be investigated. This negative effect on the concentration of solubilized metals was not observed for nickel—regardless of the biochar concentration tested, the bioleached metals in solution remained relatively constant at each tested pulp density, suggesting that the only limiting factor is the bio-lixiviant and that the biosorption of nickel on this biochar may occur mainly at even higher pH levels. Given the observed adverse effect biochar had on copper solubilization, however, the optimal conditions were estimated to be 1.6 g/L of biochar and 16 g/L pulp density. Regarding chemical leaching with sulfuric acid, it was found that leaching efficiencies for both metals were lower than when using the biogenic lixiviant, despite the similar properties of the two lixiviants (98% to 72% of copper and 82% to 18% of nickel). Furthermore, Kadivar et al. [36] report that the inclusion of the optimal concentration of 1.6 g/L of biochar led to increases in solubilization rates for both metals in both lixiviants. When using biogenic sulfuric acid, increases in metal solubilization were observed (93% to 98% and 79% to 82% of copper and nickel in biogenic sulfuric acid and 63% to 72% and 16% to 18% in chemical sulfuric acid, respectively). While displaying good solubilization rates of the target metals in a short period using a lixiviant with less environmental impact, the main downside of this approach may include the generation of a highly acidic effluent (unspecified) with the potential presence of other metals as well as other constituents of the ground wPCB.

Biogenic lixiviants allow for the leaching of metals with increased process sustainability compared to their traditional counterparts. Given the variety of reagents that can be produced, an equal variety of different strategies can be tailored for the extraction of specific metals. Furthermore, these approaches provide the added benefit of not having to consider the response/inhibition of active biomass. Thus, yield-wise, the use of biogenic reagents allows for high metal bioleaching yields in most recent explorations (Table 3), generally meeting the yields stated by industry actors, as previously indicated (although in the pre-scale-up stage). This, together with reports that biogenic lixiviants demonstrate similar/higher efficiencies in metal solubilization, underlines the potential these alternatives to traditional chemical lixiviants have, pending scale-up experiments. On the other hand, the absence of live biomass in the medium also means that the lixiviant would not be continuously replenished. Further, the hazardousness of generated effluents could also present a challenge to their application: while biogenic lixiviants could allow for the creation of an environment more conducive to metal bioleaching, with biomass viability not being a factor to consider, these more hazardous operational conditions (e.g., more extreme pH, higher concentrations of toxic compounds) could necessitate higher downstream costs to remedy.

Various biogenic compounds have been tested for the remediation of common contaminants, mainly metals, found in e-waste. Beyond the strategies presented in this section, the use of enzymes, too, has been attempted with a high degree of success for the recovery of precious metals from non-e-waste-derived substrates [102,103], and the field of e-waste bioprocessing and bio-recycling may benefit from the application of these approaches for e-waste sources.

2.4. E-Waste Bioprocessing—Comparison of Approaches for Metal Recovery

Comparing these three main approaches to e-waste metal bioleaching, some of the advantages and disadvantages of each approach become apparent. Specifically, it is possible to observe how the increased toxicity of the medium may affect the activity of live biomass, with Table 2 displaying lower pulp densities or yields and higher operational times for the selected metals compared to Tables 1 and 3. However, most studies do not overtly consider the possibility that some of the leached metals may have been captured by the biomass itself, which could cast doubt over the exactness of the presented efficiencies—due to the potential biomass capture of metals, the detection of metal concentrations in solution may underestimate the extent of the bioleaching, whereas the variation in mass of the inserted waste may lead to an overestimation of the amount of metals in solution. Biomass-bound metals would likely necessitate additional procedures for recovery, potentially compromising biomass valorization efforts. These factors would also translate to obtained revenue, compounding lower pulp densities compared to current industrial approaches, effectively meaning lower metal recovery yields from lower waste volumes that can be treated in a single process.

On the other hand, the self-regeneration of the bio-lixiviant medium may constitute the biggest advantage of single- and multi-step bioleaching approaches using live biomass—whereas reagents without live biomass are eventually exhausted, requiring continuous replenishment to retain operational capacity, the use of live biomass allows for continuous bio-lixiviant production (as long as microbial activity is not prohibitively inhibited). The studies included in this review, including those using different bioreactor architectures, focused mostly on batch experiments, which are more prone to medium toxification over time. Continuous-feed bioreactors may be a solution for such situations, allowing for a continuous replenishment of either biogenic reagents or substrates, allowing for continuous microbial activity [104].

Generically, the use of batch systems may be more advantageous in the short term and bench scale due to lower operational costs and maintenance, as well as increased process simplicity and a higher degree of control over the process conditions [105]. However, the depletion of the medium and accumulation of toxicants, be it metals or toxic metabolites, require operational downtime for retrieval, cleanup, and process restart purposes. At the expense of higher operational and maintenance costs, continuous bioreactors would allow for extended operational times, and thus increased productivity, which could then be reflected in revenue. However, one major disadvantage is the potential for contamination, which, together with the architectural complexity, may constitute a negative incentive for their research compared to batch conditions. Continuous reactors, especially those with more complex architectures, may also suffer problems due to increased medium viscosity arising from biomass growth (i.e., clogging). These drawbacks are likely the reason for the much more limited number of studies conducted in continuous settings in recent years (e.g., Hubau et al., 2020 [68]).

The development of new gene editing tools and strategies may still play an essential role [106], especially in the case of heterotrophic bacteria, generally easier to engineer, in what could be an advantage compared to their chemolithotrophic counterparts (some species, such as *Acidithiobacillus ferrooxidans*, have resisted genetic engineering efforts; however, gene a knockdown approach that increases recombination efficiency in this species has recently been reported [107]). Specifically, the engineering of microorganisms with upregulated production of lixiviants, as well as better anti-metal toxicity mechanisms, could prove invaluable towards the industrial application of live biomass in e-waste bioprocessing. For instance, *E. coli* has been genetically engineered towards the upregulation of lactic acid production, increasing yields of the desired organic acid to 93% of the theoretical maximum production [108]. Although not explored in relation to e-waste, genetic engineering has also been used to facilitate *quorum sensing* among bacterial species to allow better biofilm formation, which could enhance bacterial communities' ability to adhere to and degrade e-waste fragments during live biomass bioprocessing [85]. In this sense, the use of heterotrophic bacteria may present a unique advantage in the e-waste metal bioleaching field, given the increased modularity towards the selective extraction of metals, while also reducing the hazardousness and potential effluent treatment costs compared to chemolithotrophic microorganisms. Additionally, such approaches may contribute towards complementing another general downside of e-waste bioleaching, the relatively low metal solubilization efficiencies when accounting for process duration (i.e., long operational times). The industrial viability of these approaches will, going forward, be dependent on overcoming this limiting factor, requiring a deep understanding of the reaction mechanisms. A relative minority of the studies incorporated in this review had a dedicated kinetics assessment of the leaching processes, and thus, going forward, kinetic studies need to be given more importance.

Additionally, the sustainability of the overall process is deeply related to the biomass used in the process, production and chemical requirements, the intensiveness of metal recovery strategies, effluent treatment, and its scale. However, the variability in e-waste samples, bioprocessing strategies, experimental conditions, target metals, and data reporting formats (e.g., units) across available (few) studies significantly hinders the comparability of results and the evaluation of bioprocessing efficiencies. In addition, such studies often fall short in some aspects of e-waste bioprocessing (e.g., concentration of remaining metals, pH and redox potentials, composition of the effluent), compromising the discussion regarding the sustainability of the tested process.

3. Conclusions and Future Perspectives

E-waste is a high-value waste stream expected to grow significantly as social dependence on various means of electronic technology intensifies. The thorough recycling and valorization of this type of waste and its constituents is a challenge that conventional methodologies have been unable to address, with recycling efforts focusing on recovering the more precious constituents.

Although biobased treatments may be used for both the dissolution and subsequent recovery of metals through adsorption and precipitation, the current literature poorly describes the exploration of stepwise approaches employing these biobased strategies in sequence. This integrative potential is one of the main advantages of these biotechnological approaches and may be key to overcoming the operational time disadvantage compared to traditional approaches [109]. Many recent studies have corroborated the ability of several biobased solutions to recover rare earth elements from non-e-waste sources such as wastewater, for example. Macro- and microalgae, fungi, and bacteria, as well as a number of food-waste-derived solutions, biochar approaches, and protein-mediated approaches, have been demonstrated to act as reliable biosorbents and bioprecipitators of various metals. However, the application of these approaches to solubilized metals obtained directly from e-waste sources such as wPCB or LCD panels, rather than wastewater, mine tailings, or artificially crafted metal solutions, is underexplored in comparison. The impact of other contaminants, such as brominated compounds, on microbial growth and viability may not be accurately assessed through studies using less complex mixtures. Thus, going forward, the study of sequential biobased steps for the recycling of e-waste and recovery of CRMs will be necessary to demonstrate the potential and increased sustainability of these alternative approaches.

Although several of the discussed approaches have demonstrated encouraging efficiencies in removing target contaminants, a decline in effectiveness is generally observed upon scale-up. Biobased methodologies, especially those making use of live biomass, have been described as being particularly sensitive to the physical and chemical conditions of their surroundings, and as such, scale-up becomes one of the major challenges facing these potential alternatives to traditional treatments, with several bioreactor architectures explored in pursuit of higher efficiencies. Nevertheless, energy costs related to the thorough homogenization of the medium and other operational and structural costs are some of the main hurdles to overcome. The pursuit of higher efficiencies and volumes also highlights the need for an intimate understanding of bioleaching reaction kinetics—only a minority of the studies explored in this review did such explorations, harming comparability across different investigations. Differences/lack of harmonization in data presentation (i.e., characterization of substrates, presentation of methodologies, calculation of yields, and unit presentation/unit harmonization) can also contribute towards this difficulty in comparability.

Overall, the field of bioprocessing of e-waste, despite considerable progress and metal extraction yields that rival those of more established pilot and industrial-scale techniques, still faces major challenges, which need to be overcome before implementing such approaches can be considered viable at industrial scales. It is important to make efforts to integrate different bioprocessing strategies and to design stepwise, integrative processes that are as independent of conventional techniques as possible for both the recovery of precious elements and the breakdown of other constituents that may pose a threat to the environment, while also exploring potential biotechnological applications of by-products generated.

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