



Bioleaching of electronic waste using acidophilic sulfur oxidising bacteria



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ABSTRACT

Bioleaching of copper-rich electronic waste material was carried out using mesophilic *Acidithiobacillus thiooxidans*. The leaching behaviour of copper was investigated using three methods: abiotic chemical leaching using inorganic sulphuric acid, indirect leaching using bacterially generated sulphuric acid and by direct leaching using the acidophilic bacteria. The yield of bacterially generated sulphuric acid used for both indirect and direct leaching was 14.9 g/dm³ grown in a medium containing 25 g/dm³ of elemental sulphur and basalt salts for 14 days at 30 °C. This acid was diluted to achieve various pHs for the leaching tests. The variables tested were solution pH, temperature, time, pulp density and copper concentration in the waste. The results indicated that copper dissolution is influenced by passivation and galvanic coupling, both of which reduced the Cu yield and resulted in slower leaching. Increasing the acid concentration, copper concentration in the waste, higher temperature and prolonged leaching favoured higher yields and higher copper selectivity. e-waste toxicity had little effect on direct bioleaching at the pulp density used (10 g/dm³). However the growth medium containing partially oxidised sulphide compounds promoted copper surface passivation resulting in lower Cu recovery (60%) relative to abiotic leaching (98%).

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1. Introduction

The continual and unprecedented consumer demand for the latest gadgets and devices has led to a technological boom that has brought huge technical benefits to society; creating jobs, wealth and generally increasing living standards across the globe. However, in the process large amounts of electric and electronic wastes (e-wastes) have been generated. The problem with e-waste is its growing volume, its toxicity and its content of valuable resources (e.g., gold, copper) which are lost when e-waste is disposed (Zheng et al., 2013). The challenge in managing e-waste will be in developing sustainable recycling technologies that are able to address the volume and complexity of this waste using cost effective and ecologically sensitive methods.

Many articles are widely available in literature giving reviews of the metallurgical recovery of metals from electronic wastes, discussing in detail the pyrometallurgical, hydrometallurgical and bioprocessing of electronic wastes (Cui and Zhang, 2008; Lee and Pandey, 2012; Tuncuk et al., 2012) and hybrid technologies

combining both hydrometallurgical and bioprocessing methods (Pant et al., 2012). It is apparent that despite significant work that has been achieved in this area, these processes continue to be challenged by economic, technical and ecological issues.

Although the technologies for recycling of relatively pure forms of metals including steel, aluminum, copper, zinc, lead and nickel are well established, recycling metals from e-wastes is greatly more challenging. There is often reluctance by smelters and scrap metal buyers to recycle e-waste. This is attributed to complexity, composition and the presence of toxic materials in the waste. Only a limited number of thermal processes are available for formal processing of the metallic components of e-waste worldwide. This include Aurubis smelter in Germany (Kahhat and Williams, 2009), Noranda copper smelting in Quebec, Canada and Ronnskar smelter in Sweden (Cui and Zhang, 2008). These pyrometallurgical processes are challenged in upgrading the final metal products because of the nature of the metallic feedstock, which in e-wastes consist of both pure metals and alloys. Metals that are in their pure forms are easily processed by melting in smelters (Reck and Graedel, 2012). However the stability of the alloys and their separation behavior during melting makes the recovery of the pure metals by thermochemical methods more energy intensive and relatively difficult or impossible (Nakajima et al., 2010). Upgrading of the final metals often necessitates the subsequent use of hydrometallurgical

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processing (Cui and Zhang, 2008). In addition smelting of electronic waste can generate dioxin from the plastic components of e-wastes (Kahhat and Williams, 2009). Heavy investments by thermal processes are therefore required to manage gaseous emissions resulting from e-waste re-processing. Unfortunately an alternative route for processing the metallic components of e-waste is through informal or un-regulated processes. Often conducted in open burning sites (Chi et al., 2011; Labunsk et al., 2013; Tsydenova and Bengtsson, 2011) these emit large quantities of toxic heavy metals (Hg, Zn) and polychlorinated dibenzodioxins and dibenzofurans resulting in both unprecedented and long term ecological and health problems (Cui and Zhang, 2008; Kahhat and Williams, 2009; Tsydenova and Bengtsson, 2011).

Interest, on the other hand in using hydrometallurgical methods to process e-waste has increased because of its simplicity, the exactness of the technology and the requirement of lower capital investment and operating costs in comparison to pyrometallurgical methods (Oishi et al., 2007). Hydrometallurgical methods involve the dissolution of the metallic fractions of e-wastes in either acidic or alkali solutions, the nature of which has been driven by the recovery of precious metals (Cui and Zhang, 2008). Leaching reagents that have been used include cyanide, thiourea, thiosulfate and halide solutions (Cui and Zhang, 2008; Tuncuk et al., 2012; Zhang et al., 2012). Leaching is followed by conventional metal recovery methods including precipitation, solvent extraction, adsorption and ion-exchange and electrowinning (Coman et al., 2013; Cui and Zhang, 2008; Rimaszeki et al., 2012; Robotin et al., 2012). Although the technical feasibility of using hydrometallurgical routes have been proven, the economics of processing and the environmental impact, in particular of using toxic reagents such as cyanide and thiourea, continue to be an issue (Cui and Zhang, 2008; Tuncuk et al., 2012).

In the light of these concerns, the search for simple, cost effective and environmentally friendly method for reclaiming value added materials and energy from e-waste is an urgent goal.

Bioleaching is a promising technology that uses naturally occurring biological micro-organisms and their metabolic products in extracting valuable metals from the waste such spent catalyst and electronic waste (Brandl et al., 2008; Ivanus, 2010; Mishra et al., 2008; Valix et al., 2001). The wide range of waste treated by bioleaching is a clear indication of the ease of implementation of the process (Mishra et al., 2009). The increase in interest in the use of biohydrometallurgical route for re-processing wastes is driven by the fact that this method is environmentally sound with a huge potential to lower operational cost and energy requirements. Chemolithoautotrophs bacteria (e.g., *Acidithiobacillus ferrooxidans* and *Acidithiobacillus thiooxidans*), which uses CO₂ as carbon source and inorganic compounds (Fe²⁺, reduced S) as an energy source, has been the most widely considered group of microorganism in terms of bioleaching applications due to their ability to facilitate metal dissolution through a series of biooxidation and bioleaching reactions (Brandl et al., 2001). Other organisms including thermophiles *Sulfobacillus thermosulfidooxidans* and *Bacillus stearothermophilus* and *Metallosphaera sedula* and heterotrophic fungi including *Aspergillus niger* and *Penicillium Simplicissimum* and *Cyanobacterium violaceum* have also been used to effectively dissolve various metallic fractions from e-wastes (Brandl et al., 2001, 2008; Ilyas et al., 2010).

The numerous studies that have been devoted in the use of microorganisms in leaching e-wastes (Brandl et al., 2001; Ilyas et al., 2010; Pradhan et al., 2010) primarily feature metal recoveries and the efficacy of the various organisms in the mobilisation of the metals, however very few studies examined the mechanisms that influences the metal solubilisation. Biological leaching is thought to occur by abiotic or chemical dissolution

Table 1

Element composition of the Cu rich waste.

Metals	Al	Cu	Fe	Mg	Pb	Sn	Zn
(wt%)	0.25	86.63	0.063	0.028	0.026	0.029	0.167

where protons carry out metal solubilisation by biooxidation and leaching reaction. However secondary reactions including metal adsorption, precipitation and passivation can hamper metal solubilisation (Valix et al., 2001) resulting in both poor metal yields and slow leaching rates. The complex composition of the wastes and the components of growth media for organisms can make the leaching of e-wastes prone to these reactions (Sasaki, 2011). The objective of this paper was to establish the factors that influence copper mobilisation bioleached indirectly and directly from e-waste.

2. Material and methods

2.1. e-waste material

Ground electronic wastes (copper-rich) were obtained from Total Union PCB Recycle in Hong Kong. Metallic components of the PCB were separated by crushing, milling and magnetic separation. The waste from the recycling company was used as received with minor size separation. After undergoing sieving, waste fraction with a particle size range of 40–104 µm was collected and used throughout all experiments. The elemental composition of this waste is summarised in Table 1. A photograph of the waste is shown in Fig. 1. As shown the waste consists primarily of metallic components consisting of copper with minor quantities of plastic.

2.2. Bioleaching

A pure culture microorganism, *Acidithiobacillus thiooxidans* (ATCC8085) was used in this study. The acidophilic bacteria was grown in 25 g of elemental S per litre of basalt salts at 30 °C for 14 days, after which the acid was harvested and used for in-direct bioleaching tests. This yielded 14.9 g/dm³ of sulphuric acid. For direct leaching, sterilised waste was added directly to the growing bacteria. The in-direct bioleaching and abiotic leaching with sulphuric acid tests were carried out in a series of temperature controlled 50 ml Variomag batch reactors at temperature from 30 to 90 °C with a pulp density of 10–100 g/dm³ for periods of 0.5–24 h. Metal recoveries were estimated from the ratio of dissolved metals to the original metal content of the waste. Because of the heterogeneity of the waste, the original metal content the waste was determined by summing the dissolved metal and metal present from the leaching residue. The metal content in the residues



Fig. 1. Photograph of the Cu-rich waste.

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