

An overview of treatment technologies of E-waste

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1.1 Introduction

During the last two decades, technological advancement has rapidly happened causing obsolete and end-of-life electronic devices to become electronic wastes (E-waste) (Kiddee et al., 2013a). For instance, lifespan of a computer has reduced from 4–6 years in 1992 to 2–3 years in 2015 (Widmer and Lombard, 2005; Yazici and Deveci, 2013; Shamim et al., 2015). In 2016, the quantity of E-waste generated globally grew up to approximately 44.7 million tons equivalent to 6.1 kg per inhabitant (Ilankoon et al., 2018). The increase rate of E-waste generation is 3%–5% per annum globally (Kumar et al., 2017; Ilankoon et al., 2018). Despite problem in terms of quantity, E-waste itself is also toxic. E-waste contains up to 1000 toxic substances (Puckett and Smith, 2002). Common toxicants found in E-waste include toxic metals and metalloids such as arsenic, barium, beryllium, cadmium, cobalt, chromium, copper, iron, lead, mercury, nickel, and zinc, and persistent organic pollutants such as dioxin, brominated flame retardants (BFRs), polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), polybrominated dibenzop-dioxins, dibenzofurans (PBDD/Fs), polychlorinated dibenzo-p-dioxins, dibenzofurans (PCDD/Fs), polyvinyl chloride (PVC), and alternative halogenated flame retardants (AHFRs). Such a variety of toxicants could cause environmental problems and harm human health unless appropriate management procedure applied.

E-waste disposal to landfills and incineration can produce significant quantities of toxicants. Hazardous substances found in landfill leachates are worsened specifically with the old landfills that were not initially designed to receive E-waste, and do not have proper liners or barriers to prevent leakage of leachates (Kiddee et al., 2014). Landfill leachates can be a source of contamination to the soil, surface water, and groundwater (Baccini et al., 1987). A number of studies (e.g., Osako et al., 2004; Danon-Schaffer et al., 2006; Spalvins et al., 2008; Odusanya et al., 2009;

Hearn et al., 2011; Kiddee et al., 2014) reported high level of heavy metals and poly-halogenated organics including polybrominated diphenyl ether (PBDE) found in landfill leachates. During incineration of E-waste, greenhouse gases, mercury, and dioxins are also released into the environment (Balde et al., 2015).

Although E-waste is simply categorized as hazardous waste, it has significant potential for value recovery. E-waste in fact consists of several valuable materials (such as iron, copper, aluminum, and plastics) as well as precious metals (such as gold, silver, platinum, and palladium). Hence, it could be said that E-waste is a feasible urban mine. It provides materials for remanufacture, refurbishment as well as recycling. For example, 11% of the global gold production (2,770 tons) came from mines in 2013 while approximately 300 tons of gold was recovered from E-waste in 2014 (USGS, 2014). The resource in E-waste is normally recycled by both formal and informal procedures. Manual E-waste recycling inescapably leads to the release of toxins and persistent organic pollutants into the environment in addition to harming the health of the recycling person due to the primitive techniques used (Wong et al., 2007). The hazards of E-waste contamination to the surrounding environment including soil, sediment, water, and air has become a serious issue in many countries such as in China (Tang et al., 2010; Wang et al., 2011; Bi et al., 2018; Wu et al., 2019), India (Ha et al., 2009), and Thailand (Muenhor et al., 2010; Kiddee and Decharat, 2018). In case of humans, exposure to toxic substances through inhalation, ingestion and dermal contact can harm the human health in both chronic and acute conditions (Julander et al., 2014). Several studies demonstrated high level of health risks in E-waste recycling sites (Ha et al., 2009; Bi et al., 2018; Kiddee and Decharat, 2018; Oguri et al., 2018; Singh et al., 2018; Wu et al., 2019). Therefore, E-waste recycling industries become increasingly aware of such danger, and start to mitigate impacts from unsafe procedures along with applying appropriate E-waste treatment technologies. Innovative technologies including pyrometallurgy, hydrometallurgy, biometallurgy, high-pressure compaction, thermal treatment, organic dissolution, thermal plasma coupled with acid leaching, substrate oxidation and bioleaching can be applied to recover the potential resources in E-waste. This chapter provides an overview of the toxicity of hazardous substances in E-waste and various E-waste treatment strategies.

1.2 Types of contaminants in E-waste

E-waste is a complex mixture of many materials that contain up to 1,000 toxic substances (Puckett and Smith, 2002). E-waste is classified as a hazardous waste because it is composed of toxic substances such as antimony, arsenic, barium, cadmium, chromium, lead, manganese, mercury, indium, selenium, brominated flame retardants, polyaromatic hydrocarbons, polybrominated diphenyl ethers, and polychlorinated biphenyls. Distinct from other categorization, E-waste also has significant potential for value recovery. E-waste can be composed of several valuable materials (such as aluminum, copper, iron, and plastics) and precious metals (such as gold, silver, platinum, and palladium) (Table 1.1).

Table 1.1 Toxic substances and precious metals associated with E-waste and their health impacts.

| Substances | Precious metals | Component of electrical and electronic equipment | Effects |
|----------------|-----------------|--|--|
| Aluminum (Al) | ✓ | Printed wiring board, cathode ray tubes, computer chips, hard drives, central processing unit, mobile phones, and connectors | Skeletal development and metabolism, neurotoxicity, and fetal toxicity |
| Antimony (Sb) | | Housing, printed wiring board, mobile phones, and cathode ray tubes | Respiratory problem, associated with lead intoxication, with symptoms including headache, abdominal pain, constipation, colitis, distaste for food, loss of appetite, small mouth ulcers with salivation, dizziness, loss of weight, albuminuria, and glycosuria |
| Arsenic (As) | | Printed wiring board and mobile phones | Skin alterations, decreased nerve conduction, increased risk of diabetes and of cancer (skin and other tissues) |
| Barium (Ba) | | Cathode ray tubes, mobile phones, and fluorescent lamps | Low blood potassium, cardiac arrhythmias, respiratory failure, gastrointestinal dysfunction, paralysis, muscle twitching, and elevated blood pressure |
| Beryllium (Be) | | Power supply boxes, computers, x-ray machines, mobile phones, and ceramic components of electronics | Affect organs such as the liver, kidneys, heart, nervous system, and the lymphatic system, may develop beryllium sensitization or chronic beryllium disease |
| Bismuth (Bi) | | Printed wiring board | Nephropathy, encephalopathy, osteoarthropathy, gingivitis, stomatitis, and colitis |
| Cadmium (Cd) | | Switches, springs, connectors, printed circuit boards, batteries, infrared detectors, semiconductor chips, ink or toner photocopying machines, cathode ray tubes, housing, and mobile phones | Long-term cumulative poison, bone disease, affecting the kidneys, reproductive damage, and lung emphysema |

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Table 1.1 (Continued)

| Substances | Precious metals | Component of electrical and electronic equipment | Effects |
|-----------------|-----------------|---|---|
| Cerium (Ce) | | Catalyst, fuel additive, and optical polish | Toxicological effects on aquatic-terrestrial organisms, itching, sensitivity to heat, and skin lesions |
| Chromium (Cr) | | Anticorrosion coatings, decorative hardener, data tapes, floppy disks, mobile phones, and housing | DNA damage, lung cancer, human carcinogens, impacts on neonates, reproductive, and endocrine functions |
| Cobalt (Co) | | Printed wiring board, cathode ray tubes, housing, hard drive, and mobile phones | Human osteoblast and osteoclast proliferation and function |
| Copper (Cu) | ✓ | Printed wiring board, cathode ray tubes, computer chips, central processing unit, heat sinks, cables, and mobile phones | Liver damage |
| Dysprosium (Dy) | | Lasers and magnets | Nausea, headache, and paresthesia |
| Gold (Au) | ✓ | Printed wiring board, computer chips, central processing unit, mobile phones, and connectors/fingers | |
| Holmium (Ho) | | Lasers, magnets and optics | Stimulate metabolism |
| Indium (In) | | Printed wiring board | Lung damage |
| Iron (Fe) | ✓ | Printed wiring board, cathode ray tubes, mobile phones, and housing | Liver damage |
| Lanthanum (La) | | Batteries, catalyst, lenses, and cathode ray tubes | Pneumoconiosis |
| Lead (Pb) | | Printed circuit boards, glass in cathode ray tubes, light bulbs, televisions, mobile phones, solder, and batteries | Kidney failure, central and peripheral nervous systems, damage to blood and reproductive systems, anemia, and chronic neurotoxicity |
| Lithium (Li) | | Batteries | Cause nausea, diarrhea, dizziness, muscle weakness, fatigue, and a dazed feeling |

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Table 1.1 (Continued)

| Substances | Precious metals | Component of electrical and electronic equipment | Effects |
|----------------|-----------------|---|--|
| Manganese (Mn) | | Printed wiring board, housing, mobile phones, and cathode ray tubes | Respiratory symptomatology and a neuropsychology |
| Mercury(Hg) | | Thermostats, sensors, monitors, cells, printed circuit boards, housing, batteries, and cold cathode fluorescent lamps | Chronic damage to brain, liver damage, cause damage to the central and peripheral nervous systems as well as the fetus, neurobehavioral development of children (methyl mercury), anemia, kidney damage, and chronic neurotoxicity |
| Nickel (Ni) | ✓ | Batteries, printed wiring board, housing, mobile phones, and cathode ray tubes | Lung cancer, cardiovascular disease, neurological deficits, developmental deficits in childhood, and high blood pressure |
| Palladium (Pd) | ✓ | Hard drives, circuit board components (capacitors), mobile phones, and printed wiring board | Skin and eye irritations |
| Platinum (Pt) | ✓ | Hard drives and circuit board components | Respiratory effect |
| Scandium (Sc) | | Lasers, lighting, and aerospace | Respiratory effect |
| Selenium (Se) | | Rectifiers, mobile phones, and printed wiring board | Hair loss, nail brittleness, cardiovascular, renal, and neurological abnormalities |
| Silver (Ag) | ✓ | Printed wiring board, computer chips, keyboard membranes, mobile phones, and capacitors | Induction of genes associated with cell cycle progression and DNA damage |
| Tantalum (Ta) | ✓ | Printed wiring board, capacitors, and power supply | Low toxicity; irritation to eye, skin, mucous membranes, and upper respiratory tract; coughing |
| Thulium (Tm) | | Lasers and X-rays | Stimulates metabolism |
| Tin (Sn) | ✓ | Printed wiring board, cathode ray tubes, solder, liquid-crystal display screens, and computer chips | Adverse gastrointestinal effects |

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Table 1.1 (Continued)

| Substances | Precious metals | Component of electrical and electronic equipment | Effects |
|--|-----------------|---|--|
| Vanadium (V) | ✓ | Cathode ray tubes and mobile phones | Neurological disorders and cardiovascular diseases |
| Yttrium (Y) | | Lasers and superconductors | Causes shortness of breath, coughing, chest pain, and cyanosis |
| Zinc (Zn) | | Cathode ray tubes, printed wiring board, mobile phones, batteries, and metal coatings | Increased risk of copper deficiency (anemia and neurological abnormalities) |
| Brominated flame retardants (BFRs) | | Fire retardants for electronic equipment, plastic casings of computers, cables, as dielectric fluids in capacitors and transformers, lubricants and coolants in generators, fluorescent lighting, ceiling fans, dishwashers, electric motors, components such as connectors and mobile phones | Bioaccumulation in the environment, neurotoxicity, long-term exposure can lead to impaired learning and memory functions, interfere with thyroid and estrogen hormone systems, exposure in the womb has been linked to behavioral problems |
| Dioxin-like polychlorinated biphenyls (DL-PCB) | | Released as a combustion by product but also found in dielectric fluids, lubricants and coolants in generators, capacitors and transformers, fluorescent lighting, ceiling fans, dishwashers, and electric motors | Tumor promoters in tissues, such as liver; different congeners may alter different pathways, such as the induction of oxidative stress and/or inhibition of apoptosis |
| Polyaromatic hydrocarbons (PAH) | | Released as combustion by product | Carcinogenicity, mutagenicity, and teratogenicity |
| Polybrominated diphenyl ethers (PBDEs) | | Fire retardants for electronic equipment | Reproductive development, neurobehavioral development, thyroid function, and hormonal effect levels in animals |
| Polychlorinated biphenyls (PCBs) | | Dielectric fluids, lubricants, and coolants in generators, capacitors and transformers, fluorescent lighting, ceiling fans, dishwashers, and electric motors | Carcinogenicity, liver, thyroid, immune function, reproduction, and neurobehavioral development |

(Continued)

Table 1.1 (Continued)

| Substances | Precious metals | Component of electrical and electronic equipment | Effects |
|--|-----------------|--|---|
| Polychlorinated dibenzodioxins and dibenzofurans (PCDD/Fs) Polyvinyl chloride (PVC) | | Released as combustion by product, significant bioaccumulation related to lipid solubility, interaction with the aryl hydrocarbon receptor (AhR) Insulation on wires and cables | Reproductive and neurobehavioral development Immune development and carcinogenicity Incineration of PVC produces chlorinated dioxins and furans, which are highly persistent in the environment and toxic even in very low concentrations |

Source: Slikkerveer, A., deWolf, F.A., 1989. Pharmacokinetics and toxicity of bismuth compounds. *Med. Toxicol. Adverse Drug. Exp.* 4 (5), 303–323; Cooper, R.G., Harrison, A.P., 2009. The exposure to and health effects of antimony. *Indian J. Occup. Environ. Med.* 13 (1), 3–10; Frazzoli, C., Orisakwe, O.E., Dragone, R., Mantovani, A., 2010. Diagnostic health risk assessment of electronic waste on the general population in developing countries' scenarios. *Environ. Impact Assess. Rev.* 30, 388–399; Andrew, R.E., Shah, K.M., Wilkinson, J.M., Gartland, A., 2011. Effects of cobalt and chromium ions at clinically equivalent concentrations after metal-on-metal hip replacement on human osteoblasts and osteoclasts: implications for skeletal health. *Bone* 49 (4), 717–723; Kiddee, P., Naidu, R., Wong, M.H., 2013a. Electronic waste management approaches: an overview. *Waste Manag.* 33, 1237–1250; Grant, K., Goldizen, F.C., Sly, P.D., Brune, M., Neira, M., van den Beg, M., et al., 2013. Health consequences of exposure to e-waste: a systematic review. *Lancet Global Health*, 1 (6), e350–e361; Rim, K.T., Koo, K.H., Park, J.S., 2013. Toxicological evaluations of rare earths and their health impacts to workers: a literature review. *Saf. Health Work.* 4 (1), 12–26; Needhidasan, S., Samuel, M., Chidambaram, R., 2014. Electronic waste – and emerging threat to the environment of urban India. *J. Environ. Health Sci. Eng.* 12, 36, Song, Q., Li, J., 2014. A systematic review of the human body burden of e-waste exposure in China. *Environ. Int.* 68, 82–93; Dahle, J.T., Arai, Y., 2015. Environmental geochemistry of cerium: applications and toxicology of cerium oxide nanoparticles. *Int. J. Environ. Res. Public Health* 12, 1253–1278; Tansel, B., 2017. From electronic consumer products to e-wastes; Global outlook, waste quantities, recycling challenges. *Environ. Int.*, 98, 35–45; Chen, Y., Chen, M., Li, Y., Wang, B., Chen, S., Xu, Z., 2018. Impact of technological innovation and regulation development on e-waste toxicity: a case study of waste mobile phones. *Sci. Rep.*, 8, 7100; Ilankoon, I.M.S.K., Ghorbani, Y., Chong, M.N., Herath, G., Moyo, T., Petersen, J., 2018. E-waste in the international context: a review of trade flows, regulations, hazards, waste management strategies and technologies for value recovery. *Waste Manage.* 82, 258–275.

The materials and their concentrations in E-waste depend mainly on the type of manufacturer, type of devices, model, technology, and age of the equipment (Stenvall et al., 2013). Broadly, the materials in E-waste are categorized into four groups, namely (1) ferrous metals (38.1%), (2) nonferrous metals (16.5%), (3) plastics (26.5%), and (4) others (18.9%) (Bigum et al., 2012). The metals in E-waste are found mostly in their native metallic form, and some are in alloys of multiple elements embedded in nonmetallic components (Tuncuk et al., 2012; Sun et al., 2015). Further, E-waste components are grouped based on heavy metal contents as rare earth metals, precious metals, hazardous heavy metals, and halogenated compounds (Fig. 1.1).

E-waste contains 70% of the hazardous waste in heavy metals (Li et al, 2008). These hazardous pollutants comprise only about 2.7% of the total but causes a variety of health hazards due to environmental contamination. The hazardous heavy metals enter into the biological system through soil, water, and air. The health effects of E-waste derived heavy metals and other compounds are listed in Table 1.1.

1.3 Treatment strategies of E-waste

1.3.1 Recycling

Growing quantities of E-waste pose a major challenge for its recycling efforts. E-waste recycling can be intended as part of the “formal” or “informal” economic sector. Commonly, E-waste is exported from developed to developing countries.

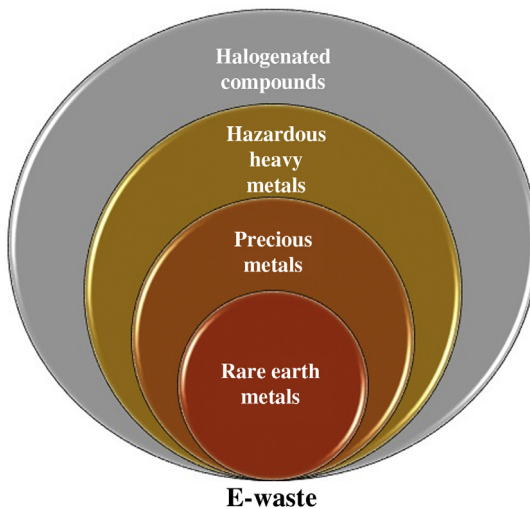


Figure 1.1 Constituents of E-waste grouped based on heavy metals and other elements/compounds.

It is evaluated that 23% of E-waste produced in developed countries is transferred to developing countries (Balde et al., 2015). Particularly, because of the limited policies, precaution, legislation and enforcement, imported E-wastes and second-hand electronic equipment have led to human hazard and ecological system problems in many countries. The primitive techniques including, stripping of metals in open-pit acid baths, chipping, and melting plastics without suitable ventilation, burning cables in open air, disposing unsalvageable matters in the sites and dismantling devices are used at informal E-waste recycling sites (Wong et al., 2007). Numerous studies have shown that the contamination of toxic substances from E-waste recycling sites release directly into the surrounding environment, and impact human health (Ha et al., 2009; Tang et al., 2010; Muenhor et al., 2010; Wang et al., 2011; Bi et al., 2018; Kiddee and Decharat, 2018; Wu et al., 2019). Ha et al. (2009) studied the contamination of trace elements at E-waste recycling sites. They found that the concentrations of Cu, Zn, Ag, Cd, In, Sn, Sb, Hg, Pb, and Bi were higher in soil from E-waste recycling sites compared to reference sites. Tang et al. (2010) investigated the contamination of heavy metal(loid)s (Cu, Cr, Cd, Pb, Zn, Hg, and As) and persistent organic compounds (PAH and PCBs) in soil at E-waste recycling sites. The results found that most heavy metals exceeded the Dutch optimum values. Total PAHs in soil reached 1,231.2 g/kg, and comparatively higher PAHs concentrations were found in soils taken from simple household. Total PCBs were measurable in all samples with concentrations ranging from 52.0 to 5789.5 g/kg. Kiddee and Decharat (2018) investigated the concentrations of lead and cadmium in soil and in the blood of workers at E-waste recycling facilities and assessed the health risks. The results found that the concentration of Pb in soil reached up to 2,866.97 mg/kg. The greatest hazard quotient (HQ) of lead exceeded 1.74, signifying that high exposure of Pb posed a health risk to workers in E-waste recycling sites. Wu et al. (2019) studied the contamination of metals, PBDEs, and AHFRs in the vicinity of an abandoned E-waste recycling site and translocation of the pollutants in rice plants cultivated at the nearby paddy field. They found that the rice plants could effectively absorb some metals such as Mo, Cr, and Mn (bioconcentration factor > 1) from soil and translocate them to the leaves. The health risk from rice consumption was high principally because of Sb and Sn (HQ > 20). Thus, many results support that E-waste recycling sites still act as the pollution source to the ecological systems and adversely affect the human health.

1.3.2 Landfill disposal

E-waste disposal in landfills leads to resource loss and can facilitate the contamination of the groundwater and soil via leaching that has the potential to pose negative impact on the surrounding environment. For instance, many developed countries [such as Belgium (17 kilo ton), Denmark (3 kilo ton), France (62 kilo ton), Germany (114 kilo ton), Great Britain (394 kilo ton), Italy (61 kilo ton), Spain (45 kilo ton), Sweden (12 kilo ton), and Switzerland (10 kilo ton)] threw away E-waste in mixed residual waste to landfills (Balde et al., 2015). Toxic substances and putrescible materials in landfills are decomposed and transferred by water to

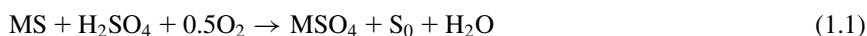
percolate through the soil as landfill leachate. Leachate can contain high levels of dissolved and suspended organic substances, inorganic compounds, and heavy metals. Nevertheless, the concentration of hazardous substances from leachate depends on the waste characteristics and stages of waste decomposition in a particular landfill (Qasim and Chiang, 1994). Hazardous substances found in landfill leachates are worsened specially with the old landfills that were not primarily designed to receive E-waste and had no appropriate liners or no barriers to prevent leakage of leachates (Kiddee et al., 2014). Several researches studied the leaching of components that evolve from E-waste. Toxicity characteristic leaching procedure (TCLP) test, simulated landfill, and realistic landfill monitoring are the techniques to investigate the leachate quality. Jang and Townsend (2003) compared leachates from eleven Florida landfills with TCLP test to define Pb leachability of printed circuit boards from computers and cathode ray tubes from computers and televisions. They found that the concentration of Pb in landfill leachates ranged from <0.04 to 0.07 mg/L while Pb in TCLP extracts were detected from 0.53 to 5.0 mg/L in printed circuit boards and 1.7 to 6.0 mg/L in cathode ray tubes. Leachability by TCLP technique demonstrated higher chemical concentration than monitoring in realistic landfill. Spalvins et al. (2008) used the simulated landfills to assess the impact of E-waste disposal on Pb leachability from electronic devices including computers, keyboards, mouse devices, smoke detectors, monitors, cell phones, and cell phone batteries. They found that Pb concentrations (7–66 µg/L) in the simulated landfill column containing E-waste was higher than the concentration of Pb in the simulated landfill column without E-waste (<2–54 µg/L). Kiddee et al. (2013b) investigated bioavailable metal(oid)s including Al, As, Ba, Be, Cd, Co, Cr, Cu, Fe, Ni, Pb, Sb, V, Zn, and polybrominated diphenyl ethers (PBDEs) in leachates from three different landfill columns that contained intact or broken E-waste under conditions that simulate landfills in terms of waste components and methods of disposal of E-wastes, and with realistic rainfall. They found that the average concentrations of Al, Ba, Be, Cd, Co, Cr, Cu, Ni, Pb, Sb, and V in leachates from the column that contained broken E-waste elements were significantly higher than the column without E-waste. Both Spalvins et al. (2008) and Kiddee et al. (2013b) confirmed that the Pb concentrations in the columns containing E-waste were higher than those in the simulated landfill columns without E-waste. Danon-Schaffer et al. (2006) studied PBDEs in landfill leachate and potential for transfer from E-waste. They found significant difference in PBDE levels in the leachate over 5 years period. Kiddee et al. (2014) investigated 14 metals and metalloids, including Al, As, Ba, Be, Cd, Co, Cr, Cu, Fe, Ni, Pb, Sb, V, Zn, and PBDEs in leachates from four selected Australian landfills receiving E-waste. They found a range of total PBDEs values from 2.13 to 59.75 ng/L in the leachates. Thus, several researches confirmed the high concentration of heavy metals and PBDEs in landfills receiving E-waste.

1.3.3 Biological treatment

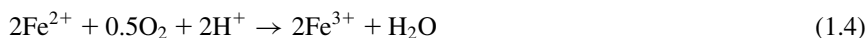
Metal recovery by biological processes has emerged as an alternative technology today especially in metals like Au and Cu. Biometallurgical processes utilize

microorganisms that convert metals into soluble salts in aqueous media from ores/concentrates/wastes. Biometallurgy is a complete blending of biotechnology and metallurgy. Scientifically, the physicochemical process is a branch of hydrometallurgy that involves the use of microorganisms to generate chemical oxidants. It has been an established process for the extraction of metal(loid)s including Cu, Au, Co, Ni, Zn, As, Mo, Cd and U (Watling, 2014). More than 15% of the total annual Cu and 5% of Au along with a small fraction of Ni and Zn are produced using biometallurgical processes (Johnson, 2014; Schlesinger et al., 2011). Both prokaryotic and eukaryotic microbes interact with metal species for structural and/or catalytic functions. The basis of biometallurgy is the interaction between metals and microbes, which aids in the recovery of metals. It was reported that mesophilic, moderately thermophilic and extremely thermophilic bacteria and archaea were involved in biometallurgy (Norris, 2007; Schippers, 2007; Vera et al., 2013). Acidophilic bacteria, cyanogenic heterotrophs, and/or acid-producing heterotrophs were used in biometallurgy to selectively recover metals from waste streams. Biometallurgy is broadly divided into two types, namely bioleaching and biosorption, on the ground of mobilization and immobilization of metals due to biological activities.

Bioleaching is a method tested for the recovery of many base and precious metals. There are two leaching mechanisms for increasing the release rate of metals. Firstly, microorganisms can directly oxidize minerals and solubilize metals in a direct action mechanism, as the below for sulphidic minerals (MS) [Eqs. (1.1) and (1.2)] (Suzuki, 2001):



The other mode of leaching mechanism is an indirect action mechanism where the microorganisms generate the oxidizing agent (Fe^{3+} from Fe^{2+}). The reactions for indirect action mechanisms are as follows [Eqs. (1.3) and (1.4)]:



Biosorption is a process where both living and nonliving organisms can be used for releasing metals from substrates. It is a passive physicochemical interaction between the organism's surface and the metallic ion in the solution. The microorganisms used for biosorption include algae (Greene et al., 1986; Romera et al., 2006; Vilar et al., 2007), bacteria (De Vargas et al., 2004; Yong et al., 2002), yeasts (Bakkaloglu et al., 1998; Wang and Chen, 2006), and fungi (Nui and Volesky, 1999) which are known to accumulate heavy metals and precious metals actively. Bioaccumulation is a particular type of active biosorption in which microorganisms accumulate heavy metals within the living biomass (Fomina and Gadd, 2014). Immobilization of metals can be done by complexation with intracellular proteins

(metallothionein and phytochelatins) (Harms et al., 2011). The advantages of bio-sorption are low operating cost, minimization of the volume of chemicals and/or biological sludge, and high efficiency in detoxifying the effluents (Figueira et al., 2000; Mack et al., 2007; Pethkar and Paknikar, 1998). Biometallurgy has both advantages and disadvantages from other conventional processes. The advantages are low operating costs, reduction in chemical usage, easier manageability of waste effluents, and environmental friendliness. However, the major issues related to biometallurgy are it is a slow process and not fully developed for waste/ore of high metal complexity.

A limited number of studies have explained the extraction of metals from E-waste with mesophilic chemolithotrophic, acidophilic moderately thermophilic, and cyanogenic bacteria. Chen et al. (2015) investigated a one-step bioleaching process using a column for the extraction of Cu with a pulp density of 24.8% of the printed circuit board (PCB) using *Acidithiobacillus ferrooxidans* for 28 days. After 28 days, Cu extraction was 94.8% of the initial Cu concentration. The results (Chen et al., 2015) indicated that the efficiency of bioleaching decreased by precipitation of jarosite at the surface of the material, and the precipitation could be prevented by adding diluted sulfuric acid. Wang et al. (2018) carried out a two-step leaching process by using steel pickling waste liquor as the leaching agent and used a domesticated microbe *A. ferrooxidans*. This study (Wang et al., 2018) found that an optimum pulp density of 60 g/L and pH 0.5–1.0 were effective in recovering almost 100% of Cu and 51.94% of Fe. The rate of Cu leaching was controlled by external diffusion rather than internal diffusion (Wang et al., 2018). Similarly, Ilyas et al. (2007) applied moderately thermophilic acidophilic bacterial strains of chemolithotrophic and heterotrophic consortia for column bioleaching of metals from scrap electrical and electronic equipment. It was found that the leaching efficiency of Zn, Al, Cu, and Ni was 80%, 64%, 86% and 74%, respectively, after a preleaching period of 27 days followed by a bioleaching period of 280 days (Ilyas et al., 2007). In subsequent experiments, the authors observed 91%, 95%, 96%, and 94% efficiency for Al, Cu, Zn, and Ni recovery, respectively, at a pulp density of 10% (w/v) using a moderately thermophilic adapted culture of *Sulfobacillus thermosulfidooxidans* (Ilyas and Lee, 2014). The bioleaching trials were enriched with air (25% O₂ and 0.03% CO₂) and 2.5% (w/v) biogenic S₀ at 45°C (Ilyas and Lee, 2014). Because of the higher bioavailability of the biogenic S₀, the acidophiles oxidized the substrate at a faster rate by which increased the efficiency of bioleaching (Ilyas and Lee, 2014). Recently, Rizki et al. (2019) reported an alternative approach for Au recycling from E-waste where they investigated the potential utility of microbiologically mediated thiourea leaching (TU-bioleaching) for Au recovery. The degree of thiourea tolerance for different Fe-oxidizing bacteria/archaea was in the range of 5–100 mM, and *Acidiplasma* sp. displayed a vigorous Fe-oxidation when thiourea and PCB coexisted as substrates (Rizki et al., 2019). The biogenic Fe³⁺ regeneration due to microbial action enhanced the steady and continuous dissolution of Au (98% Au released at 1 mM Fe³⁺ and 10 mM thiourea) (Rizki et al., 2019). Pradhan and Kumar (2012) investigated a two-step bioleaching of metals from E-waste using cyanogenic bacterial strains (*Chromobacterium violaceum*, *Pseudomonas*

aeruginosa, and *Pseudomonas fluorescens*). The authors used both single and consortial cyanogenic bacteria to mobilize metals from E-waste with different efficiencies. At an E-waste concentration of 1% (w/v), *C. violaceum* alone, and a consortium of *C. violaceum* and *P. aeruginosa* exhibited the maximum metal mobilization (Pradhan and Kumar, 2012). While *C. violaceum* leached around 79%, 69%, 46%, 9%, and 7% of Cu, Au, Zn, Fe, and Ag, respectively, the mixture of *C. violaceum* and *P. aeruginosa* leached 83%, 73%, 49%, 13%, and 8% of the above elements (Pradhan and Kumar, 2012). Likewise, Faraji et al. (2018) performed fungal bioleaching of PCBs using *Aspergillus niger*. After 21 days of bioleaching, around 97%, 44%, and 64% of Zn, Ni, and Cu, respectively, were recovered based on acidolysis and complexolysis dissolution mechanisms of organic acids, and a two-step bioleaching process was more promising than one-step or spent-medium bioleaching processes (Faraji et al., 2018).

Several reports are available on the scaling up of bioleaching of E-waste for recovery of base, precious and rare earth metals, and methods are improving day by day. A variety of microbes can participate in the E-waste bioleaching process, and a huge potential exists for discovering new microorganisms for industrial applications. However, still there are many challenges to transfer laboratory-scale experiments to pilot plant testing and commercial plant design. This is mainly due to the heterogeneity of E-waste, toxic effects to microbes, environmental factors, and optimum conditions for the production of microbial biomass.

1.3.4 Advanced methods

Among the advanced E-waste treatment methods, high-pressure compaction, cement solidification, thermal treatment, organic dissolution, simple acid leaching, plasma-coupled acid leaching and substrate oxidation are the most prominent ones. In an attempt to compare between the high-pressure compaction and cement solidification methods, Niu and Li (2007) found that the former method although being able to reduce the volume of printed wire boards (PWBs) significantly, the impact resistance of the compacts was not long-lasting under environmental conditions. The latter method on the other hand was able to lock heavy metal Pb strongly in the solidified materials (using Portland cement and slag cement) that lasted long even under harsh environmental conditions (Niu and Li, 2007). Vehlow et al. (2000) suggested that pretreated electrical and electronic waste (free of metallic components) could be added with municipal solid waste (MSW) followed by their thermal treatment at high temperature in order to reduce the volume of the final bottom ash. The authors (Vehlow et al., 2000) found that the cocombustion of the mixed E-waste and MSW substrate did not significantly alter the emission levels of polychlorinated dibenzo-p-dioxins (PCDD) and dibenzofurans (PCDF) during the heat treatment. However, this method of E-waste treatment can be highly energy-intensive, and requires adequate facilities to contain the emitted gaseous substances that are highly toxic to the human health. On the other hand, methods such as organic dissolution can be more energy saving than the thermal incineration method. For example, Verma et al. (2017) reported that dimethylacetamide

(DMAc) dissolved PCBs more effectively than dimethylformamide (DMF), and the optimum conditions for PCB dissolution was under DMAc at 140°C with a PCB:DMAc ratio (w/v) of 3:10. Rath et al. (2012) suggested that thermal plasma treatment followed by mineral acid leaching could be an environmentally benign method for recovering metals from E-waste. This combined method achieved Cu, Ni, and Co leaching from E-waste with >90% efficiency with HCl at room temperature, and thus was highly energy efficient (Rath et al., 2012). Similarly, the grinding and shredding of E-waste could be avoided, and thus energy could be saved by employing a substrate oxidation method to the waste. For example, ammonium persulfate oxidation was found effective to recover Au from the superficial coating of E-waste with an estimated 98% recovery (Alzate et al., 2016). Additionally, recent years have seen significant advancement in the implementation of biological treatment strategies of E-waste (as described in Section 1.3.3). The biological treatment methods are considered as the most environmentally friendly processes but might lack in achieving high resource recovery performance.

1.4 Conclusions

The currently available E-waste treatment strategies have several limitations. Many of these processes demonstrated successful performance under laboratory conditions but failed to perform under real environmental conditions at large commercial scale. The selection of a particular E-waste treatment method involves several considerations including the energy consumption, availability of facilities, risk of secondary pollution, and scalability of the method. Therefore, continuous research effort is a need of the hour in order to develop region specific new E-waste treatment methods.

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