Chapter 1
Life-Cycle Assessment of Metal Recovery from Electronic Waste

Marco Villares

Abstract Increasing technological development is driving the demand for metals, especially in the field of electronics. Conversely, electronic waste is a growing global waste stream which is becoming more problematic in its management. Unsafe disposal contributes to environmental pollution as well as wasting secondary resources and threatening human health, particularly in developing countries with immature waste treatment and recycling technologies. This chapter gives an outline of European regulations and an overview of the global electronic waste situation and formal and informal recycling in the developed and developing countries. Since metal concentrations in electronic waste can be even higher than in mineral ores and some metals are considered critical in supply, there is a strong incentive to recover them as a secondary resource.

Life-cycle assessment, LCA, is an analytical tool based on physical metrics of material and energy flows of the life-cycle of a product or service system used to evaluate its environmental performance. The recovery of valuable metals from electronic waste can be achieved by bioleaching, involving microorganisms working at near ambient temperatures. The possible environmental performance from a life-cycle perspective of this novel metal recovery technique is evaluated in a summarised illustrative case study applying life-cycle assessment.

Keywords Electronic waste • Life-cycle assessment • Circular economy • Metal recovery • Bioleaching • Scenario • LCA • Recycling • Secondary resource

Contents

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1.1 Introduction

Our contemporary technological civilisation requires the diverse use of many sorts of metals in order to operate. Huge quantities of bulk metals such as steel, copper and aluminium are used to produce buildings, electrical wires and aircraft respectively. With increasing economic and technological development, the unique properties of every metal have been applied to improve product performance (Reck and Graedel 2012). There is more demand for other types of metals, such as rare earth metals. These are used in novel technologies and to produce the generators and batteries for renewable transport and energy systems. For example, thin film solar cells, which are cheaper to make than single crystal silicon, need indium, gallium, selenium and tellurium, while the large magnets used in wind turbines generators require neodymium, praseodymium and dysprosium to improve their resistance to overheating (Bradshaw et al. 2013). Moreover, a shift to renewable energy systems could be far more resource intensive than the present, fossil-fuel-based system (Kleijn and Van der Voet 2010; Kleijn 2011).

With the drive for cleaner technologies, the use of more electronic information technology and the development of emerging economies, demand for all metals is on the rise. At the same time, primary extraction can rely less on high-grade easily recoverable metal ores. Moreover, mining entails environmental risks owing to the toxic chemicals involved and it is energy intensive.

Large-scale inefficient use of metals over their whole life-cycle from extraction to disposal increases metals dispersed in the environment as pollutants that disrupt the biological functions of living organisms. Metals are considered strong contributors to ecotoxic impacts as they do not degrade in the environment and in principle their presence is for ever (Van der Voet 2013). Geopolitics and concerns regarding reliable, sustainable and undistorted access to certain raw materials is of growing concern within the European Union and across the globe (European Commission 2014a).

All the above factors underline the importance of utilising sources of metals in discarded products and other waste streams. These factors in turn provide the rationale for the development of more effective and efficient techniques for recycling and metal recovery. Within the landscape sketched above, waste electrical and electronic equipment plays a significant role as a growing waste stream and carrier of diverse secondary metals. In the following sections an overview of the electronic waste topic and management and recovery techniques for the metals it contains will be given. Then the life-cycle assessment (LCA) framework will be described before presenting the summary of a case study of its application to evaluate the environmental performance of bioleaching of electronic waste for copper recovery.

1.2 Electronic Waste and Metal Recovery

1.2.1 The Growth of Electronic Waste

In the developed and developing world, rising levels of wealth coupled with shortened product life-cycles driven by fast innovation and fashion have led to a dramatic rise in the global consumption of consumer goods (Kiddee et al. 2013; Breivik et al. 2014). Economies of scale have brought down the costs of electrical and electronic equipment and made it almost accessible for all in recent decades. The total number of discarded computers and other devices that generate electronic waste strongly correlates with a country's gross domestic product (GDP), since electrical and electronic items are now required for most contemporary economies to function (Robinson 2009).

Global amounts of electronic waste are already enormous, estimated to be between 20 and 50 million tonnes per year (Ongondo et al. 2011; Baldé et al. 2015). This comes to 3–7 kg/person each year, taking the world population to be 7 billion people. In Europe alone about 12 million tonnes of waste electrical and electronic equipment (WEEE) per year are generated, with an expected increase in the coming decades at a rate of at least 4% per year (Reuter 2013). At this rate, and assuming a European population of 500 million, an average of more than 30 kg/person per year in 2020 will be generated.

This growth is taking place throughout the present linear throughput economy, based on the steps of extraction, production, distribution, consumption and disposal. This mode of operation has two important consequences. First, increased exploitation of natural resources takes place (mining of finite minerals) with more potential of future scarcity (Graedel et al. 2015). Second, the generation of larger waste streams, which have been initially dispersed in the natural environment. Such waste electrical and electronic equipment has typically gone to local landfill sites or been exported for disposal to less-developed nations with less-stringent environmental regulations.

1.2.2 Disposal of Electronic Waste

Safe disposal of electronic waste is a challenge owing to the composition of the products, typically made up of heavy metals and other chemical components threatening to human health and the environment. Electronic waste can contain more than one thousand different substances, many of them toxic, such as arsenic, cadmium, hexavalent chromium, lead, mercury, selenium, and flame retardants that create dioxins emissions when burned (Widmer et al. 2005). Research has shown that toxic metals as well as polyhalogenated organic compounds such as polychlorinated biphenyls and polybrominated diphenyl ethers can be released from e-waste in landfills (Kiddee et al. 2013).

Within electronic waste, printed circuit boards have a heterogeneous, diverse and variable composition reflecting a market with different manufacturers and varying product designs according to their application (Hall and Williams 2007). Continuing advances in functional efficiency also result in more recent products being composed of fewer but more diverse materials (Reuter 2013). Hazardous materials derive from both the non-metallic and the metallic fraction of printed circuit boards. Although many brominated flame retardants used in the polymers of printed circuit boards are toxic, and halogen-free alternatives based on phosphate or metallic compounds are viable alternatives, the application of the former is still prevalent (Hadi et al. 2015). Many of these flame retardants are persistent, dissolve readily in organic fats and have been shown to reach high residue levels in sediments or bioaccumulate in living organisms (Li and Zeng 2012). Their ultimate behaviour and fate in water and soils will depend on how the materials are treated and to what degree dispersal in the environment takes place. In the metallic fraction of printed circuit boards, lead from the soldering tin is the most toxic fraction because of its higher concentration levels. However, mercury can also be present in switches and cadmium in the pins.

1.2.3 Electronic Waste Regulations in the European Union

The disposal issue has been addressed in part by wealthier nations by focusing on more environmentally benign end-of-life options for these products such as reuse, repair, refurbishment and recycling. Switzerland has been at the forefront of collection and recycling experiences with voluntary schemes in place even before the introduction of legislation (Ongondo et al. 2011).

Some legislation at the European Union level now regulates these issues restricting the use of hazardous substances, and requiring manufacturers to take back their products, recycle them and dispose of them safely. These are the respective directives restricting the use of hazardous substances (RoHS 2 Directive 2011/65/EU) and waste electrical and electronic equipment (WEEE directive 2012/19/EU) (Ongondo et al. 2011). Member states have to directly incorporate regulations into their national legal frameworks. The directives allow each member state room for interpretation in their implementation. Legislation stems from a general policy objective to reduce waste, preferably by prevention, and its promotion as a secondary resource for reuse or recycling.

1.2.3.1 Waste Electrical and Electronic Equipment Directive

This directive seeks to prevent and minimise electronic waste by reuse, recycling and recovery. A chief role is given to manufacturers and distributors being required to cover the costs of collection, treatment, recycling and recovery of electronic waste. Producers are required to set up individual or collective schemes which will finance the collection and treatment of electronic waste using the best available methods.

1.2.3.2 Restriction of Hazardous Substances Directive

This directive falls within the broader waste electrical and electronic equipment directive and is stricter, as its objectives are to protect human/animal health and to ensure the environmentally sound recovery and disposal of electronic waste (Stewart 2012). The onus again is on manufacturers as, since July 2006, in principle no electrical and electronic equipment and spare parts on the European Union market can contain six major toxic substances. These are lead, mercury, cadmium, hexavalent chromium, polybrominated biphenyls or polybrominated diphenyl ether, with important implications for printed circuit board manufacture (Ravi 2012). However, the directive has some leeway and foresees exemptions and specifies maximum concentrations in materials and components. Manufacturers have to mark their products with the European Conformity “CE” marking, CE being an abbreviation of the French “Conformité Européenne”, and formally declare that their products are compliant with the directive. Such obligations also apply to importers and distributors. This compels manufacturers to establish as much uniformity in their products as is feasible (Stewart 2012).

1.2.3.3 Registration, Evaluation, Authorisation and Restriction of Chemicals Regulations

This legislation is based on the precautionary principle and tasks industry to take its own responsibility for the safe use of chemicals. Manufacturers are required to make exposure scenarios for their manufacturing processes and for identified uses of the substances on their own or in a preparation and for all life-cycle stages resulting from these uses. Industry is expected to manage the risks and has the burden of proof that they are acting responsibly. The objective is to support competition within the chemicals industry, while also protecting human health and the environment (Van Leeuwen 2007).

Substances of which more than one tonne is made or imported are to be registered in a database to pool information to avoid unnecessary testing by industry. Downstream and upstream information provision on health and safety, environmental risks and measures for the management of risks between manufacturers, importers, distributors and customers is mandatory. Chemicals that are carcinogenic, mutagenic or reproductive toxic substances and persistent, bioaccumulative toxic substances require previous authorisation before being put on the market. These aspects are managed by the European Chemicals Agency to ensure consistency across the European Union (Stewart 2012).

This drive for harmonisation across the European Union through these regulations has been fragmentary in its implementation due to differences in technological progress and ultimate responsibilities residing with each member state. Nevertheless, the implications for electronic waste of these directives are key defining elements regarding pre-production, production, and post-production of electronic waste streams (Hadi et al. 2015). Such standardisation and formalisation can be regarded as positive for the prospect of metal recovery from electronic waste. For example, in Germany household electronic waste is managed under a formal collection system under the responsibility of public waste-management authorities and retailers. The United Kingdom has a distributor take-back scheme and a producer-compliance scheme in place. Elsewhere in the developed world, such as the United States, municipal waste services handle the electronic waste stream. Voluntary schemes are also found there as well as in Australia and Canada, while Japan uses collection via retailers.

1.2.4 Export and Informal Recycling of Electronic Waste

Formal recycling of electronic waste in developed countries uses a treatment chain applying four operational phases to target the diverse material fractions (Li and Zeng 2012; Ghosh et al. 2015; Hadi et al. 2015). Decontamination aims to separate as much as possible any hazardous components and fractions. Liberation involves dismantling and sorting the substances into more or less clean fractions. This depends on the design and the composition of the product and the degree of bonding of the target fractions. The recyclate is then made suitable for treatment during size reduction, termed comminution. Treatment then isolates the desired material fractions for recovery or disposal by means of chemical, metallurgical and thermal processes. However, effective reprocessing technology, which recovers the valuable materials with minimal environmental impact, is expensive (Robinson 2009).

Thus, part of the electronic waste is exported outside Europe, possibly under unethical conditions, for reuse/recycling under inadequate working and environmental conditions that also do not result in an effective recovery of the metals (Kiddee et al. 2013). Hence developing countries present a more distressing case, where regulations and cleaner methods of disposal are not in place. This raises ethical concerns of problem shifting when wealthy nations export their electronic waste to them. China and India are currently at the forefront of electronic waste treatment in unsafe conditions, followed by other African, Asian and Latin American developing countries where consumption of electronic goods is growing (Ongondo et al. 2011).

Informal recycling is quite specialised and provides an income for practitioners in dedicated workshops but it is inefficient and unsafe. Products are manually dismantled using cutting torches, hammers and chisels to break apart solder connections. More complex components are cooked on a coal or electrically heated plate to melt them, in some cases with little or no control over temperature and extraction of poisonous exhaust fumes. Much of the material is lost and workers and the environment are exposed to toxins (Ongondo et al. 2011; Li and Zeng 2012; Reuter 2013).

Contamination can happen locally in developing countries, but such problem shifting can rebound on the e-waste originators. Wealthy nations have a compelling incentive to deal with the issue, since electronic waste contaminants become omnipresent, being re-exported in food and manufactured products along global supply chains back to the developed world, constituting a global health risk (Robinson 2009).

Thus within the broader context of economic growth, increasing consumption and waste generation has resulted in circumstances that challenge the prevailing logic of business-as-usual practices. The global level of production and consumption creates large flows of both toxic and potentially valuable substances (Widmer et al. 2005). Large primary reserves of metals still exist, yet their extraction entails a high environmental burden compounded by declining ore grades of these reserves (Van der Voet 2013). Recycling can contribute to a solution by diminishing part of the increased demand for metals and the related energy and resource use for their production. Often potential ecological benefits of recycling are cancelled out if electronic waste has to be transported long distances, owing to the negative environmental effects of fossil-fuel-based transportation. However, the recycling process itself can have a lower ecological impact than landfilling of incinerated electronic waste (Robinson 2009). Recycling is one of the most immediate, tangible and low-cost investments available for decoupling economic growth from environmental degradation and escalating resource use (Reuter 2013).

1.2.5 The Circular Economy

These issues align with key aspects of the currently vaunted circular economy model (Ellen MacArthur Foundation 2013). Acknowledging the limits of linear consumption, this model proposes decoupling of the current unsustainable economy from material inputs. It also highlights potential economic opportunities arising from using resources more efficiently and effectively. Moreover, the precarious availability of primary resources exposes economies to risks of scarcity and rising costs of extraction. Benefits are to be gained from reducing waste at all points of the material chain and minimising disposal at end of life by reuse and recycling. Traditional concepts of use and ownership are questioned. The model largely foresees a partnership between consumers and producers where reusable materials are returned to the producers to close material loops. The European Union has recognised the multifaceted nature of the challenge to reform products, material and value chains, identify barriers in consumer habits, and develop new economic models and financial instruments to promote transition to a circular economy (European Commission 2014b).

1.2.6 Electronic Waste as a Secondary Metal Resource

Electronic waste has a heterogeneous, diverse and variable composition, reflecting a market with several manufacturers and product designs varying according to their application (Hall and Williams 2007). More recent products are composed of fewer but more diverse materials, reflecting advances in functional efficiency (Reuter 2013). Electronic waste has an average predominant metal content of about 60%, as illustrated in Fig. 1.1, comprising all the base, heavy and precious metals and rare earth metals. The most well-known precious metals, gold, silver, platinum and palladium, are less reactive and rarer than base metals and have high economic value. The rare earth metals are a group of 17 chemically similar metallic elements consisting of the 15 lanthanides, plus scandium and yttrium. Primary mining of rare earth metals is costly and complex because they occur associated with each other in varying ratios in minerals and ores (Binnemans et al. 2013).

Electrical and electronic equipment can sometimes contain up to 50 different kinds of metallic elements, as indicated in Table 1.1, sometimes in quite small amounts.

However, electronic waste streams have greater concentrations of metals than natural ores, which makes their recycling as secondary resources significant for both economic and environmental motivations (Zeng et al. 2012). For example, the printed circuit boards found in all electronic equipment typically contain the material concentrations shown in Table 1.2.

The demand for copper and zinc is anticipated to rise (Ilyas and Lee 2014a). Furthermore, rare earth metals and platinum group metals are subject to high supply risk, being critical raw materials for renewable energy production and batteries (Moss et al. 2013; European Commission 2014a, c). The major economic driver for e-waste recycling is the recovery of the precious metals, followed by copper and zinc (Lee and Pandey 2012). Therefore it is advantageous to implement effective methods to recover these useful secondary source metals to enhance resource utilisation instead of discarding them. However, adequate collection, sorting and

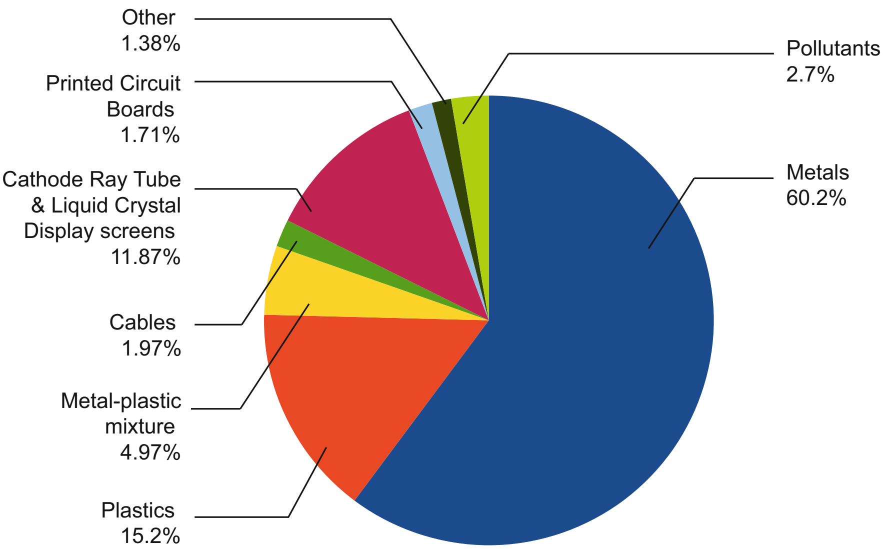

Fig. 1.1 Typical electronic waste material fractions (Adapted from Widmer et al. 2005), showing the proportionally high metal content of 60%

pre-processing and the long lifetimes of certain product groups of electronic waste pose a significant challenge for taking advantage of this dispersed secondary material source (Moss et al. 2013).

1.2.7 Metal Recovery Techniques

Traditional methods of extraction of metals from ores have been applied for thousands of years by mankind. Pyrometallurgy involves applying heat and high temperatures, while hydrometallurgy uses chemical solutions. Pyrometallurgical techniques apply roasting to convert compounds just below their melting points or smelting, which totally melts the ore into two liquid layers, one containing the metals and the other the waste rock. Hydrometallurgy involves the dissolving of compounds from an ore by an aqueous solvent otherwise known as leaching. In general, an oxidative leaching process is required for the extraction of the targeted base and precious metals. For example copper can be extracted using lixiviants such as sulphuric acid, aqua regia, which is a mixture of nitric acid and hydrochloric acid, ammonia, and hydrogen peroxide combined with acids (Tuncuk et al. 2012). Precious metals can be extracted using cyanide, aqua regia, thiourea and thiosulfate (Cui and Zhang 2008).

Further metal extraction can be done by electrowinning, where, after an electric current is passed through the solution, the metal ions are deposited onto an electrode

Table 1.1 Overview of the applications of metals in electrical and electronic equipment (Reuter 2013). The ubiquity and diversity of their applications is shown, highlighting the sensitivity to metal supply of these now indispensable products

Metal	Application in electrical and electronic equipment
Ferrous metal	Casings, as major element in magnets, magnetic coils
Aluminium	Casings, partly cables
Magnesium	Casings, body of cameras
Copper	Cables, connectors
Gold	Contacts, transistors, diodes, switches, transistors, integrated circuits
Palladium, platinum, rhodium	Capacitors, connectors, contacts, transistors, diodes, soldering
Silver	Lead-free soldering, capacitors, contacts, batteries, radio frequency identification chips, photovoltaic cells
Antimony	Alloying element, additive for flame retardants, soldering element, semi-conductor technology and photocells, additive in cathode ray tube glass
Gallium	Semiconductors, laser diodes, light emitting diodes, photo detectors, photovoltaic cells, integrated switches
Germanium	Photovoltaic cells, glass fibre, optical glasses glass fibre, semi-conductive chips
Indium	Flat panel screens, thin-film-photovoltaic cells, semi- conductors, light emitting diodes
Cobalt	Lithium-ion and nickel metal hydride batteries, magnets
Rare earth metals, neodymium, dysprosium, scandium, lanthanum and yttrium	Magnets, compact florescent bulbs, phosphors, fuel cells, nickel metal hydride batteries
Tantalum	Capacitors
Beryllium	Beryllium-copper-alloys, beryllium oxide-ceramics, metallic beryllium
Tellurium	Thin film photovoltaic cells, photoreceptors, photoelectrical devices
Tungsten	Tungsten carbide, electrodes, cables and electrical components, additives in cathode ray tube glass
Niobium	Niobium-steel alloys, super alloys magnets, capacitors
Tin	Lead-free soldering, liquid crystal displays, photovoltaic cells, miniaturized capacitors

(Charles et al. 2014). Other metal-extraction methods include adsorption on activated carbon, ion exchange, precipitation, cementation and solvent extraction, used in conjunction with electrowinning. The selected method depends on the leaching reagent system metal concentrations and the presence of impurities (Tuncuk et al. 2012).

Table 1.2 Comparison of typical copper and gold concentrations in natural ores and printed circuit boards (Erüst et al. 2013). The concentrations in electronic waste exceed by far those of mineral ores, providing a clear rationale for their recycling from this waste stream

Metal	Concentration in natural ore	Concentration in printed circuit boards	Magnitude increase of concentration in printed circuit boards
Copper, Cu	~5–10 kg/ton (~0.5–1%)	~200 kg/ton (~20%)	20 to 40-fold
Gold, Au	~1–10 g/ton (~0.0001–0.001%)	~250 g/ton (~0.025%)	25 to 250-fold

Some of these processes, such as pyrometallurgy, have been successfully applied to electronic waste to recover valuable metals by firms such as Umicore in Belgium and Outotec in South Korea (Van der Voet 2013). However, the nature of these traditional processes can have important environmental impacts. Their high-temperature furnaces generate polluting chemicals, such as dioxins, and are energy intensive.

Biohydrometallurgy, a potentially more environmentally friendly alternative, involves using microbes with the natural capability to extract metals for their own metabolic functions. Certain strains of microorganism, such as bacteria, archaea and fungi, can survive in environments with high metal concentrations, whereby the metals are leached to solution after using them as an energy source (Mishra and Rhee 2014). For example, acidophilic iron and sulphur oxidizing bacteria can directly and indirectly extract bivalent metals to solution (Cui and Zhang 2008; Petersen 2010; Erüst et al. 2013). Heterotrophic bacteria that require an organic carbon source for growth, such as *Pseudonomonas putida*, produce cyanide, which can be used to extract precious metals (Işıldar et al. 2015). The microorganisms suitable for metal bioleaching with respect to electronic waste still remain inadequately characterised and limited literature exists on their possible mode of interaction and extent of leaching (Ilyas and Lee 2014a). To optimise biohydrometallurgical processes, important parameters need to be controlled, such as pH, temperature, growth media composition, oxygen and carbon dioxide content (Watling 2006). Other parameters such as toxic elements present in the electronic waste and acid/base consumption are also relevant (Erüst et al. 2013). Current investigations to optimise applications of bioleaching for metal recovery from electronic waste are in their early stages at the level of laboratory studies, mainly using shake flasks. The process purportedly has potential to afford environmental benignity, operational flexibility, and lower costs with less energy consumption than the traditional methods (Ilyas and Lee 2014b). An early assessment of its potential environmental performance from a life-cycle perspective can provide insights for further development.

1.3 Life-Cycle Assessment

A tool for the evaluation of the environmental impacts of a technology is life-cycle assessment. It has its origins in energy analysis, which led to the first chiefly comparative life-cycle studies of consumer products in the 1960s and 1970s. Since these first studies on diapers and drink containers, the methodology has been further developed, standardised (International Organisation for Standardization (ISO) 2006) and applied to assess a wide range of services and products such as waste incineration, building materials, military systems, and tourism (Guinée et al. 2011).

1.3.1 Life-Cycle Assessment Methodology

Life-cycle assessment is a comprehensive analytical tool based on physical metrics of material and energy flows of the life-cycle of a product or service system, principally applied to improve sustainability performance (Rebitzer et al. 2004; International Organisation for Standardization (ISO) 2006). The boundary of analysis is typically from cradle to grave, taking into account extraction, production, distribution, consumption and disposal. All upstream and downstream processes should be considered, aiming to identify their related potential environmental burdens and avoid problem shifting between life-cycle phases, between regions or between environmental problems (Guinée et al. 2002; Finnveden et al. 2009). It goes beyond the typical sole focus on a production site and the specific manufacturing process or processes involved there (Castro-Molinare et al. 2014).

The method assumes that the product system is in a steady state to quantify its associated environmental interventions and their impacts. These impacts can be climate change, acidification, eutrophication, stratospheric ozone depletion and resource depletion, and/or others chosen, depending on the application. Other types of impacts of financial, political, social or other nature are not dealt with by the methodology (Guinée and Heijungs 2005). Apart from the identification of processes where environmental performance can be improved, known as hotspots, Fig. 1.2 shows that LCA can also be applied for product development, strategic planning, policy making and marketing (Guinée et al. 2002).

The product or service system is composed of unit processes, connected by material, energy, product, waste and service flows shown in Fig. 1.3. It is in turn embedded in an economic subsystem that is made up of the following main activities: mining of raw materials; production of materials, products and energy; use and maintenance of products; waste treatment and processing of discarded products; and transport (Udo de Haes and Heijungs 2009).

The product system can be considered as two subsystems, a foreground system governed by internal factors and a background system by external ones. The foreground system includes those processes “whose selection or mode of operation

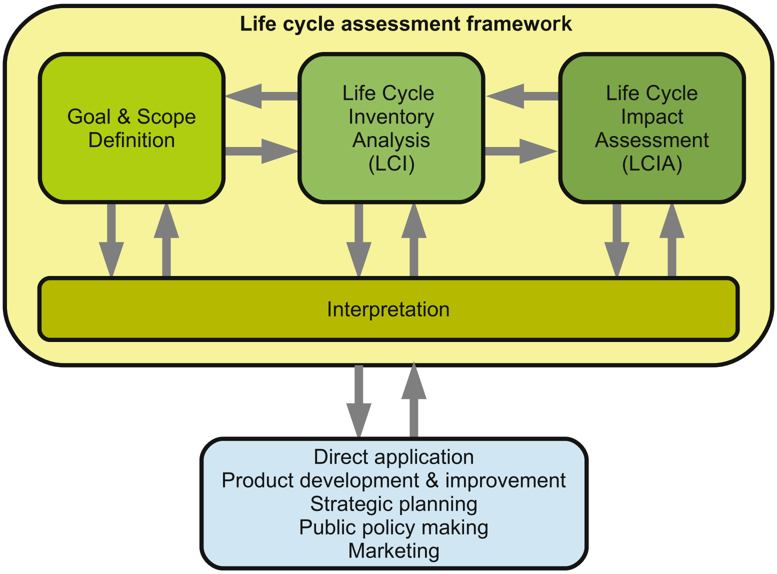

Fig. 1.2 Life-cycle-assessment-framework diagram showing the four phases of goal and scope definition, life-cycle inventory analysis, life-cycle impact assessment and interpretation, as well as possible applications of the method (Guinée and Heijungs 2005)

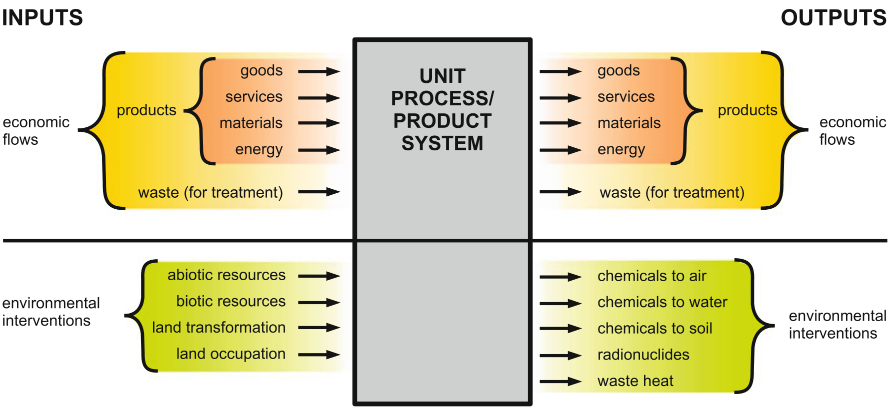

Fig. 1.3 Schematic illustration of a unit process in life-cycle assessment showing economic and environmental inflows balanced with the corresponding economic and environmental outflows (Adapted from Guinée and Heijungs 2005)

is affected directly by decisions based on or inspired by the study". The background system then consists of all other processes which interact with the foreground system "by supplying or receiving material and energy through a market" (Höjer et al. 2008).

The life-cycle assessment approach itself can be subdivided into two types: attributional and consequential LCA (Finnveden et al. 2009). An attributional LCA has a descriptive focus, depicting the environmental impact of all flows that are attributed to a certain amount of functional unit. A consequential LCA is change-oriented and estimates the system-wide change in environmental impact resulting from a modification of the amount of functional unit produced in response to possible different decisions. In attributional LCA, scaling the results linearly is possible, while consequential LCA focuses on marginal changes and the results therefore depend on the magnitude of change (Rebitzer et al. 2004).

The methodology has been standardised by the International Standards Association (ISO 14040) into a framework of four interdependent phases (Guinée et al. 2002; Reap et al. 2008). Firstly in the g*oal and scope* phase the ambit of the analysis, as well as establishing in what way it will be applied, is designated. The system boundaries, function, functional unit and reference flow of the product system under study are set. The system boundary marks the limit defined between the environment and the physical economy within which the product system lies. All flows have their origin and end in the environment. The flows bridging this boundary will be environmental interventions, namely extraction of resources, emissions to the environment and the use of land (Guinée and Heijungs 2005). In general, when human intervention takes place this can be regarded as an economic process and not part of the natural environment. Thus controlled landfills, waste-water treatment, agriculture and forestry and mining tips are typically not included in the natural environment (Guinée et al. 2002). The functional unit is based on the function provided by the product or service, and need not be a mass-based quantity of material (Guinée and Heijungs 2005). The functional unit indicates the quantity of the function under consideration in the life-cycle assessment. The functional unit allows different systems to be considered functionally equivalent and allows reference flows to be set for each of them, thus enabling comparison of their environmental performance.

In the second phase of *life-cycle-inventory analysis* the flow of material and energy into, through, and from a product system is collated and quantified conforming to the definition of the goal and scope. This requires the determination of the system boundary, the representation of the system of unit processes with a flowchart, data gathering of unit process data and the performance of necessary allocations for any multifunctional unit processes.

In the third phase, *life-cycle-impact assessment,* the inventory data is converted into potential environmental impact estimates by means of a two-step process of classification and characterisation using cause-effect models. The outcomes can be further normalised, grouped and weighted into aggregated indicators reflecting consensual value preferences.

In the final phase of *life-cycle-interpretation* the results are evaluated for consistency and completeness and analysed for robustness. Conclusions are drawn and recommendations based upon the inventory and impact assessment data can be made. Typically iteration between life-cycle interpretation and the other life-cycle-assessment phases is often carried out.

1.3.2 Life-Cycle Assessment Applied to Emerging Technologies

Applying life-cycle assessment effectively ex post has its own implicit demands and these are magnified in ex ante applications when more uncertainties are involved. The potential of applying life-cycle assessment ex ante is recognised but also its limitations. The standard problems of ex post life-cycle assessment involving system boundaries, scaling issues, data availability and uncertainty are magnified when applied ex ante owing to unknown future technologies at the industrial scale and larger data gaps that further increase uncertainty (Hetherington et al. 2013). Problems can emerge from three areas: difficulties in defining the goal and scope of the life-cycle assessment at such an early developmental stage; uncertainty involving the data which may be of poor quality, resulting in dubious potential environmental impacts; and the establishment of an accurate level of confidence in data interpretation (Cinelli et al. 2014). There is a mismatch between the capability of the tools for analysis available and the fact that typically 80% of the environmental impacts are determined at the early development stage (Tischner et al. 2000). Differences between laboratory systems and industrial processes are crucial to data validity. At the lab scale, yields are typically lower than at industrial level, where efficiency gains have been integrated (Frischknecht et al. 2009). Scaling up can uncover by-products which require addressing by allocation. Also small variations in lab measurements or from model simulations may be amplified to large data errors. The impacts of specialised equipment and instrumentation may also be underestimated. Lab experiments are typically done in batches and are less efficient than typical continuous industrial-scale processes. Methods for tackling scaling issues have been proposed in studies of scaling behaviour of furnaces and heat pumps (Caduff et al. 2014) or a more general prognostication approach and a method with guidelines for going from pilot plants to industrial scale (Shibasaki et al. 2007).

Hospido et al. (2009) have recommended an approach with the following five guidelines for ex ante life-cycle assessment. The life-cycle assessment should be forward looking and descriptive, termed “prospective attributional LCA”. The functional unit should be a physical quantity or have an economic dimension. Scenarios should be applied to define a relevant future state. System boundaries can exclude unit processes that are not affected by the novel process. The foreground system should be modelled with specific data, while the background system should use average data.

The combinatorial and aggregative nature of the analysis brings with it data uncertainty at the laboratory level, after scaling up, and of the life-cycle assessment modelling itself, that seriously brings into question the validity of the results. Such limitations should be recognised and made explicit to all stakeholders involved (Hetherington et al. 2013). This does not necessarily negate the value of exploratory studies which can be later added to and corrected cumulatively by further research.

A second-order analysis can involve uncertainty analysis and validity of data at all levels to determine the general confidence in the final outcomes of the proposed research. At the life-cycle assessment level, statistical estimates of uncertainty can also be applied as outlined in the following section.

1.3.3 Uncertainty Issues in Life-Cycle Assessment

The practice of life-cycle assessment deals with and delivers its quantified information as point values, rarely reflecting variability or a spread of possible values (Henriksson et al. 2013). Recognition of the existence of such variabilities means acknowledging uncertainty. Uncertainty can be subdivided into two types which are not mutually exclusive. Epistemic uncertainty relates to the incompleteness of knowledge, while stochastic uncertainty pertains to the inherent randomness of the natural world. Stochastic uncertainty can relate to spatial or temporal variability (Clavreul et al. 2012). Epistemic uncertainty loops back to uncertainty itself, as the estimation of uncertainty is in itself a source of uncertainty (Björklund 2002).

The results of a life-cycle assessment can be uncertain owing to data variability, error in measurements, incorrect estimations, modelling assumptions, outdated data, unrepresentative data and data gaps (Finnveden et al. 2009). The cumulative nature of the methodology means that uncertainty can build up and combine to become manifest at all levels of life-cycle assessment. For example the data inputs can be inaccurate, missing, outdated or unrepresentative. Averaging is often done without acknowledging the spatial and temporal heterogeneity of the data source. Since life-cycle assessment is a tool founded on quantification, uncertainty is present at the data inventory level of the unit processes and also in the characterisation models, weighting factors and resulting potential impacts. The generation and use of more precise data is one way to tackle this type of uncertainty (Wender et al. 2014).

The simplifications implicit in life-cycle assessment are also a source of uncertainties. Its broad, non dynamic life-cycle perspective does not account for localised or temporal effects. For instance these are not aligned with the temporal frames of characterisation models with different time horizons (Guo and Murphy 2012). Its modelling assumes simple linear scaling of economic and environmental processes and its result should always be termed “potential” impacts, not specified in space and time. Standardisation of the methodology has been a way of curtailing arbitrariness requiring transparency about the choices made relating to goal and scope, system boundaries, the functional unit and allocation methods (Guinée et al. 2002).

Given all the aforementioned, the aggregative nature of the tool means that uncertainty can accumulate and combine within the modelling perceivable at all levels of life-cycle assessment. A taxonomy proposed by Huijbregts (1998) linking all of the above back to life-cycle assessment defined three groups of uncertainties:

- parameter uncertainties referring to the uncertainty in values because of inherent variability, measurement imprecision or lack of data, for example;
- scenario uncertainties owing to the necessary choices made to build scenarios; and
- model uncertainties owing to the mathematical models underlying life-cycle assessment calculations.

Finnveden et al. (2009) reinterpret these more broadly, referring to the sources of a life-cycle assessment where uncertainty may arise, namely data, choices and relations. Efforts can be made to improve the quality of the data (better measurements and models) and the choices made (stakeholder discussions to reach consensus on uncertainty) but this is often impractical in a short time frame. Thus the trend is to incorporate uncertainty into life-cycle assessment using probability and statistical methods.

Ways that uncertainty has been acknowledged involve replication of life-cycle assessment using scenarios where assumptions are changed one at a time to compare different outcomes (Clavreul et al. 2012), exploring and reporting the sensitivity of the outcomes to changes, and applying and incorporating statistical uncertainty analysis (Henriksson et al. 2013). Statistical techniques have been incorporated to evaluate the quality of the data of the life-cycle inventory. The uncertainty of the inventory data is represented by six characteristics (reliability, completeness, temporal, geographical and technological correlation and sample size). Each characteristic is divided into five levels with a score from 1 to 5. An uncertainty factor in terms of contribution to the square of the geometric standard deviation is given to each score of the six characteristics. By representing the data-quality-indicator value by a 'default' log normal distribution, this approach translates the data-quality indicators into probability distributions (Guo and Murphy 2012). Then the method stochastically propagates the probability distributions using random sampling such as Monte Carlo analysis (Clavreul et al. 2012).

Ultimately it is important to recognise that the outcome of a life-cycle assessment is not an absolute result. The value of the tool lies in its application to comprehensively compare systems, and its outcomes are useful in a relative sense in spite of the uncertainties. Moreover, it may also be argued that the shortcomings and incompleteness of life-cycle assessment instil a rigorous approach, compelling the practitioner to maintain focus and remain alert.

1.4 Illustrative Case Study Summary: Copper Recovery from Electronic Waste Using Bioleaching

Few life-cycle assessments have been carried out on metal-recovery techniques applied to electronic waste as summarised by Table 1.3.

The life-cycle assessment of metal recovery from printed circuit boards using pyrometallurgical techniques by Bigum et al. (2012) has shown that it has an improved environmental performance and also a greater yield than virgin mining. Such recycling and recovery is advised in view of resource criticality and a transition to a circular economy. If a biotechnological route can be shown to be competitive and environmentally benign relative to the existing pyrometallurgical method and other traditional techniques, this technology can be regarded as an improvement of methods of metal recovery applied to electronic waste treatment.

Bioprocesses for the recovery of metals from electronic waste are described elsewhere in this book. An illustrative case study by the author applied life-cycle assessment to a novel laboratory bioleaching process for metal recovery from printed circuit boards to determine its potential environmental performance (Villares et al. 2016). Using the laboratory process as a foundation, the potential environmental impacts of a plausible industrial scale version of the process were estimated for comparison with an established pyrometallurgical technology. To stimulate new paths of enquiry and to guide further development of the technology,

Table 1.3 Overview of life-cycle assessments carried out on metal recovery from electronic waste

Metal recovery technique	Remarks	References
Pyrometallurgical	Life-cycle assessment of the recovery of aluminium, copper, gold, iron, nickel, palladium and silver from high-grade electronic waste modelled on the Boliden smelter refinery at Rönnskär, Sweden.	Bigum et al. (2012)
Pyrometallurgical	Life-cycle assessment to quantify the environmental impacts of recovery of 17 metal products of the Umicore integrated precious metals smelter-refinery in Hoboken, Belgium using detailed industry data.	Stamp et al. (2013)
Hydrometallurgical	Life-cycle assessment of sulphuric acid leaching and selective precipitation for yttrium, zinc, copper, lithium, and cobalt from fluorescent lamps, cathode ray tubes, Li-ion accumulators and printed circuit boards.	Rocchetti et al. (2013)
Hydrometallurgical	Using literature data, a comparison of environmental performance of two processes for recovering copper from printed circuit boards, one using sulphuric acid and one using a mix of nitric and hydrochloric acid.	Rubin et al. (2014)
Hydrometallurgical	Life-cycle assessment of printed-circuit-board recycling chain in China for recovery of lead, zinc, copper, gold, palladium and silver using mechanical beneficiation, acid leaching and electrolysis.	Xue et al. (2015)

the scaled up novel bioleaching process was embedded in a larger product system of upstream and downstream processes and life-cycle assessment was applied.

Potential environmental hotspots were identified in the energy and material inputs for the bioleaching unit process, particularly the air input, and solvents for copper recovery. A pre-treatment stage of shredding of printed circuit boards contributed relatively marginally to potential environmental impacts. Bioleaching itself contributed to more than 50% of potential environmental impacts of eutrophication and acidification potential, photochemical oxidation, climate change and three toxicity categories: human, terrestrial and aquatic freshwater toxicity. Solvent extraction and electrowinning for recovery of elemental copper contributed around 80% to the potential depletion of abiotic resources and stratospheric ozone. The comparison with the existing pyrometallurgical technology returned an inferior environmental performance even after simulating a further optimisation by increasing the amount of printed circuit boards treated, all else remaining the same.

The estimations and uncertainties around the environmental performance of the scaled up bioleaching product system of the case study mean that it cannot be considered a definitive result. Nonetheless, Villares et al. (2016) propose that the insights gained can guide the further development of the bioleaching technique and contribute to developing secondary metal recovery from electronic waste in an environmentally responsible manner. Indeed, such broadening of the research domain of the novel bioleaching process the by study spurred thinking regarding its possible future optimisation. This includes further exploration of bioleaching mechanisms, adaptation of the microorganisms and exploring more effective naturally occurring bacterial consortia, improving the efficiency of the bioreactors, recovering and recycling process water and using waste products, such as biogenic sulphur as nutrient inputs.

1.5 Conclusion

The need for addressing the growing waste stream of electronic waste to effectively recover secondary materials taking advantage of the benefits of circularity and recycling have been discussed in this chapter. Life-cycle assessment can assist in evaluating future metal recovery alternatives but its application requires a well bounded and defined product or service system. At an early development stage, life-cycle assessment of not yet existent technologies that are sketchily defined do not provide the same type of results. However, the fact that it does provide useful insights when confronting such uncertainties obliges the early application of life-cycle assessment to be regarded as an instrument aiding in the making of a plausible mock up of the potential future technology and its possible future context. The case study serves to illustrate the benefits of applying life-cycle assessment to evaluate an emerging technology to guide its further development accounting for possible environmental impacts. The procedure should be reiterated shifting from the estimates of the early stage to newly acquired real data from the further development of the novel process. These later iterations of life-cycle assessment, refining previous work in later stages of technological development, should result in more confidence in the environmental performance results.

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Case Study Supplementary Information For extended information on the case study see master thesis with appendices available for download: http://repository.tudelft.nl/view/ir/uuid:ad116c32-ea7c-40eb-955a-ba96d62ac5c8/

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