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Utilization of Life Cycle Assessment methodology to compare two strategies for recovery of copper from printed circuit board scrap

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ABSTRACT

Waste electrical and electronic equipment (WEE) is a source of valuable materials which poses great risks to environment and human health if improperly managed. To overcome this barrier and close the loop in a production chain, several end-of-life (EoL) strategies based on reuse, recovery and recycling are under development. Material recovery from printed circuit board (PCB) scrap may contribute to reduce the environmental impacts caused by the extraction of high-valued and/or highly toxic materials from nature. However, each recovery process itself requires resource consumption and generates some forms of impact. Given that situation, Life Cycle Assessment (LCA) methodology can aid decision-making processes on which EoL strategy to adopt. The goal of this study consisted of applying LCA methodology to evaluate and compare two processes for recovering copper from PCB scrap. Initially, a review was conducted, focusing on material recovery processes adopted as EoL options for WEEE management; several methods for copper recovery from PCB scrap were found. LCA methodology was then applied in order to evaluate and compare two of these processes. Both combine mechanical and electrochemical processing and have similar efficiency; one of them employs sulfuric acid and the other employs acqua regia (combination of nitric and chloridric acid). Evaluation of the impact categories considered in the study has shown that the process that uses acqua regia has better environmental performance. The work reported here can be seen as a starting point for more in-depth evaluations of these and other material recovery processes, especially in countries such as Brazil, where WEEE management is often neglected – in the absence of a well-structured recycling chain, it is usually disposed of in landfills. In that light, the paper presents some closing remarks and suggestions for future research.

1. Introduction

The United Nations Environment Programme (UNEP, 2009) points out that the amount of electrical and electronic equipment (EEE) placed on the market every year is increasing both in industrialized and industrializing countries. On the other hand, the available data on generation of waste electrical and electronic equipment (WEEE) is poor and insufficient. Through estimation techniques that extended known data to regional-global coverage, the UNEP report predicted an increase from 200% to 400% in the generation of WEEE in developing countries from 2010 to 2020 (UNEP, 2009).

Indeed, the increasingly rapid evolution of technology combined with a strong incentive for consumption causes rapid obsolescence of a wide array of products and, therefore, generation of WEEE at much higher volumes than other consumer goods – consumers now rarely take broken electronics to a repair shop as replacement is now often easier and cheaper than repair (Puckett and Smith, 2002). This early obsolescence makes the linear 'extraction–production–usage–disposal' chain is even more resource-intensive, increasing emissions to air, water and soil along all the phases of a product life cycle and, therefore, their impacts on environment, human health and economy.

This scenario is aggravated by the peculiarities of WEEE: it contains more than a thousand different substances (Puckett and Smith, 2002), many of which are high-valued and/or highly toxic, and thus are both great sources of precious materials and waste disposal-related issues (Li et al., 2004). The fact this waste is a

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potential source of valuable materials – to the point it has been already called an 'urban ore' – and a vehicle for hazardous substances to impact environment and human health makes it an object of several studies, focusing on its economic importance and/or its potential impact, in case it is not managed properly.

There are frequent reports of landfilling and even illegal transboundary movement of such waste, frequently followed by inadequate practices like open burning – to recover metals or to 'treat' plastic waste – and river dumping of residual fractions, like spent acid solutions used in chemical stripping operations (UNEP, 2009; BAN, 2012). Such practices cause toxic substances to be released to the air, ground and groundwater, compromising its quality and exposing population to several health risks, as shown by Zheng et al. (2008) and Sepulveda et al. (2010) in their review of informal recycling processes. Besides the environmental impacts, practices such as WEEE direct landfilling or uncontrolled burning causes the loss of valuable and/or precious materials (Hagelüken, 2006a).

Brazil, one of the developing countries analyzed in the UNEP report, does not have a complete recycling chain that includes collection, sorting, dismantling, processing and refining or disposal; there are only scarce and isolated initiatives, as described by Saavedra and Ometto (2012). Brazil's Solid Waste National Policy, which came into force in 2010, establishes the setting of reverse logistics systems for specific products, such as EEE (Brazil, 2010), and seeks to improve waste management plans and practices, so that waste that is commonly disposed on landfills will, for instance, become a source of materials, through recycling.

According to Das et al. (2009), computer waste accounts for the major portion of the total WEEE generated. The metals contained in personal computers (PC's) commonly include aluminum, antimony, arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, gallium, gold, iron, lead, manganese, mercury, palladium, platinum, selenium, silver, and zinc (Bleiwas and Kelly, 2001).

Printed Circuit Boards (PCB's) account for about 6% of the total weight of WEEE. They contain the highest precious metal values, and may be considered a high value waste (Bleiwas and Kelly, 2001; Das et al., 2009).

After analyzing WEEE recycling data from Switzerland, Widmer et al. (2005) point out that pollutants and hazardous components have been on a steady decline over time, may be due to legislation and market pressures, while the metal content has remained the dominant fraction, well above 50%. Therefore, WEEE management strategies such as recycling, reuse and recovery could reduce the need for virgin materials, especially metals; consequently, energy consumption and other environmental impacts generated during material extraction might be reduced, as well as the amount of waste to be treated and properly disposed (Bleiwas and Kelly, 2001; Cui and Forssberg, 2003). However, each recovery process itself requires resource consumption and generates some forms of impact.

Many technologies and methods that combine shredding or crushing with pyrometallurgical or hydrometallurgical processes have been developed to recycle PCBs, initially for purely economic reasons, though environmental considerations have increasingly influenced end-of-life processing of electronics (Yu et al., 2009). Still, such researches are mainly focused on the recovering rates of the processes, and do not present an in-depth assessment of their environmental aspects and potential impacts. Electrochemical process, for instance, while achieving high recovery rates, are characterized by an intensive use of acid solutions which, if inappropriately discarded, may cause severe damage to the environment.

Therefore, a lack of research on the potential impacts of material recovery based on electrochemical processing was identified. In order to provide information on which WEEE management option is, environmentally speaking, the best – or least harmful – it is necessary to investigate the viability of existing assessment methods. In this sense, Life Cycle Assessment (LCA) can play an important role. The literature presents LCA as a valuable tool for comparing products and processes in terms of their environmental impacts, therefore assisting in decision-making processes.

The goal of this study consisted precisely in applying LCA methodology to evaluate and compare two electrochemical processes for recovering copper from PCB scrap: one using sulfuric acid and one using nitric and chloridric acid.

2. Background

2.1. End of life options for WEEE management – potential and barriers for material recovery

End of life (EoL) of a product can be defined as the moment in which it no longer can perform its original function. Several factors such as natural degradation over time, changes in user preferences, careful use and maintenance, among many others, affect the useful life of a product, extending it or – as it is frequently the case with electric and electronic equipment – shortening it. The definition of EoL adopted by Rose (2000) is "*the point in time when the product no longer satisfies the initial purchaser or first user*", and the researcher justifies it by pointing out that user preferences change more rapidly than the product wears out in several product categories, such as electric and electronic equipment.

EoL strategies are defined by Rose (2000) as the approaches or methods associated with dealing with a product at its end-of-life, including the activities associated with recovering value from the product, through manual labor and/or machinery.

The literature presents EoL options that basically range from recovery of the product itself or its materials to treatment and disposal. Thierry et al. (1995) present five recovery options: reuse, repair, refurbishing, remanufacturing, cannibalization and recycling, all of which help to create a closed loop materials flow (Mansour and Zarei, 2008). Indeed, material recovery is seen as an attractive EoL strategy for WEEE due to economic, environmental and social motivators. Increased reuse and recycling contribute to closing material – especially metals – loops, which could enhance the overall resource productivity, representing one of the key elements of a transition towards more sustainable production and consumption patterns (Ruhrberg, 2006). Ravi (2012) points out that a new recycled product could be produced from recovered materials and metals; they also could reach secondary markets or parts suppliers for making new products.

However, the adoption of strategies such as recovery and recycling of WEEE such as PCB scrap requires facing and overcoming some barriers. One of them is the sheer volume of such waste generated (Das et al., 2009), which poses a great problem in terms of storage/disposal space and handling.

Another barrier to PCB recycling is the complexity and variety of electric and electronic equipments, which in turn present a large quantity of components and materials. Consequently, it is difficult to present a generalized material composition for the entire waste stream; most studies examine five categories of materials: ferrous metals, non-ferrous metals, glass, plastics and other materials (Widmer et al., 2005).

Recovery processes are usually designed with focus on one or just a few materials. One of the most common approaches is to recover copper from PCB scrap (Li et al., 2004; Cui and Forssberg, 2003). According to the European Topic Centre on Resource and Waste Management (ETC/RWM, 2012), copper accounts for 7% of the total weight of WEEE, being the third largest component by weight, just below iron and steel (over 47%) and plastics (around 15%) (ETC/RWM, 2012).

Copper is the base metal in major amount in PCB scrap and, in terms of economic value, surpasses silver and is close to palladium, both precious metals.

(Hagelüken, 2006a; Park and Fray, 2009; Yu et al., 2009; Hino et al., 2009), as shown in Table 1.

PCB scrap presents a high concentration of copper in comparison with explorable ores (Veit et al., 2005). It is also worth noticing that iron, while quite closer to copper in terms of concentration, has a much lower economic value. Besides the economic benefits, there is another positive aspect: the refined copper from secondary sources is of the same quality as new metal (Van Beers et al., 2007).

It is important to point out that the term ‘plastics’ in Table 1 actually corresponds to several kinds of different plastics found in PCB scrap, that cannot be joint together during recovery processes, or separated in an economically feasible way. In the case of a complex feed material such as WEEE, mechanical separation of plastics may increase the loss of precious metals, so impurities like plastics, aluminum and iron can be acceptable to boost overall precious metals recovery and generated value, according to Hagelüken (2006b). While it is not possible to recover material from plastic fractions, they can be used as a feedstock substitute for coke (Hagelüken, 2006b).

2.2. Copper recovery processes

According to Yu et al. (2009) and Youssef et al. (2012), PCB recycling can be broadly divided into two major steps. The first one involves a comminution process, in which PCB scrap is shredded and/or crushed in a hammer mill for homogenization and particle size reduction (Li et al., 2004; Veit et al., 2005; Long et al., 2010); crushing can also be performed in a wet crushing equipment, such as hammer mill with water medium, which can help control the fugitive odors and dust emissions, while avoiding excessive temperature in parts of the mill during the process (Duan et al., 2009).

Comminution is followed by separation of metals from non-metals, in order to provide appropriate conditions for further processing (Youssef et al., 2012); this separation process contributes to upgrade the desirable material content (Yu et al., 2009). Several methods can be employed for separating non-metals from metals; some of them, such as incineration and acid bathing, may cause release of hazardous substances. On the other hand, despite being energy intensive, physical separation techniques are able to produce streams of metals and nonmetals, while being comparatively safe and eco-friendly to operate (Youssef et al., 2012). Examples of these techniques include:

- Screening: separates the material according only to particle size; often used in material recovery in order to prepare a uniformly sized feed (Cui and Forssberg, 2003; Veit, 2005; Long et al., 2010). According to Cui and Forssberg (2003), screening allows to upgrade metals contents, once these materials differ from non-metals in terms of particle size and shape properties.
- Shape separation: these techniques are based on the fact that the dynamic behavior of a particle is influenced by its shape (Furuuchi and Gotoh, 1992). In recycling processes, shape separation can be performed on an inclined conveyor and inclined vibrating plate (Koyanaka et al., 1997).
- Magnetic separation: for the recovery of ferromagnetic elements, producing magnetic and nonmagnetic fractions (Cui and Forssberg, 2003; Veit et al., 2006; Park and Fray, 2009). According to Veit et al. (2006), iron is the main element retained in the separators, once it is the main magnetic element found in the highest concentration in PCB scrap.
- Electric conductivity-based separation: for separating materials of different electric conductivities; Cui and Forssberg (2003) present three typical techniques: Eddy current, for separating non-ferrous metals from non-metals; Corona electrostatic separation, for separating metals from non-metals; and triboelectric process for separating non-conductors.
- Density-based separation: employed to separate heavier materials (e.g., metals) from lighter ones (e.g., non-metals); jigging, an example of such methods, allows sorting of small pieces of metals; however, WEEE is a heterogenous waste stream, and that can make it difficult to operate a jigging process (Cui and Forssberg, 2003).

The second step in PCB recycling processes is comprised of further material separation and processing; the goal is to separate the target metals from other metals (Yu et al., 2009; Youssef et al., 2012). Two processes are adequate for obtaining high purity metals as output according to Youssef et al. (2012): pyrometallurgical and hydrometallurgical processes.

- Pyrometallurgical processing: is the traditional PCB recycling method, with wide industrial utilization in leading global smelter companies. However, investments for smelters require large upfront investments, and facilities are unavoidably large scale (Yu et al., 2009). Furthermore, pyrometallurgy requires major amounts of energy and can form uncontrolled harmful products (Youssef et al., 2012), while achieving only a partial separation of metals. For example, smelters recover precious metals, copper and other metals, but not aluminum or iron, so combination with hydrometallurgical techniques and/or electrochemical processing is therefore necessary (Yu et al., 2009).

Table 1
Approximated mass content and economic value of materials found in PCB scrap.

Material	Hagelüken (2006a)		Park and Fray (2009)		Yu et al. (2009)		Hino et al. (2009)	
	Mass content (%)	Value (%)	Mass content (%)	Value (%)	Mass content (%)	Mass value (%)	Mass content (%)	Value (%)
Plastics	23	–	–	–	–	–	31.8	–
Copper (Cu)	20	14	16	9.7	9.7	4.8	14.6	–
Glass	18	–	–	–				–
Iron (Fe)	7	–	5	0.1	9.2	0.51	4.79	–
Aluminum (Al)	5	1	5	1.1	5.8	1.35		–
Tin (Sn)	2	–	3	4.5	2.15	3.84	5.62	–
Lead (Pb)	1.5	–	2	0.5	2.24	0.37	2.96	–
Nickel (Ni)	1		1	2.4	0.69	0.99	1.65	–
Silver (Ag)	0.100	5	0.100	4.6	0.06	2.6	0.045	–
Gold (Au)	0.025	65	0.025	65.4	0.023	77.17	0.0205	–
Palladium (Pd)	0.011	15	0.010	11.4	0.01	8.38	0.022	–
Others	22	–	–	–	–	–	38	–

- Hydrometallurgical processing: comprises a series of acidic or caustic leaches of solid materials with a range of reagents, and the recovery of metals from these leachates, as described by Yu et al. (2009). According to those authors, such processes are easier to operate, when compared to pyrometallurgical processes, and usually require a smaller scale and therefore less investment. However, they may pose a considerable environmental impact due to the toxicity of the reagents used and the large amount of by-products generated, which requires additional investment on waste and water treatment (Yu et al., 2009). On the other hand, such processes can provide controlled environment, good recovery, high selectivity, high purity output, which allows the recovered metal to be sold without any further processing (Li et al., 2004; Youssef et al., 2012). For these reasons, Youssef et al. (2012) propose them as the most beneficial processes for separating the target materials.

According to Li et al. (2004), different hydrometallurgical processes are used, depending on the substrate material. For nonmetallic substrates, metals are recovered from substrates by the process of leaching in the resulting solution; for metallic substrates, electrochemical processing is used to recover metals (Veit et al., 2006).

The reviewed literature presents some other methods for PCB recycling. Pyrolysis (heating without oxygen), substitutes shredding/crushing by decomposing the organic material contained in PCB scrap to low molecular products (liquids or gases), which can be used as fuels or chemical feedstocks (Zhou and Qiu, 2010; Hall and Williams, 2007; Guo et al., 2010). The pyrolysis residues can then undergo further processing which, as reported by Long et al. (2010), can include crushing and gravity separation.

Xie et al. (2009) studied a copper and iron recovery process consisting in applying ultrasound on a previously acidified PCB waste sludge, which is then pressed by a filter. Xiu and Zhang (2010) employed a technique based on the use of supercritical methanol (supercritical substance is the substance whose temperature and pressure are over its critical points): waste PCBs were comminuted and mixed with methanol in a reactor where temperature and pressure were raised below critical points for methanol; the liquid–solid mixture was then filtered, and the main elements separated in the solid portion were copper, iron, tin, lead and zinc (Xiu and Zhang, 2010).

Of all the reviewed references, the only recovery process widely performed on industrial scale is the one reported by Hagelüken (2006a,b), which is comprised of manual separation according to PCB components, comminution by shredding and separation of metals on an integrated metals refinery and smelter. All the other processes are performed mainly on laboratory scale.

2.3. Life cycle thinking and Life Cycle Assessment

Heiskanen (2002) suggests that the optimization of product systems, or the physical lifecycles of products, should occur across the individual organizations taking part in them. In effect, an ever-increasing number of organizations have been adopting the concept of product life cycle as a response to public pressure and/or legal provisions, in an attempt to move from a linear economy to a cyclic one.

Pigosso et al. (2010) define life cycle thinking as the integration of life cycle perspectives into the overall strategy, planning and decision-making processes of an organization, taking into account economic, social and environmental aspects. According to the authors, the introduction of this concept requires efforts towards increasing efficiency on activities such as product design, manufacturing processes, usage and selection and adoption of EoL strategies. It is also necessary to decide which strategy implies less environmental impacts, as well as which processes are the most efficient for the implementation of the chosen strategy, and that can be accomplished by properly performing a Life Cycle Assessment.

Life Cycle Assessment (LCA) is a method for assessing environmental impacts of a product system along its life cycle, based on the evaluation of its set of inputs and outputs – called life cycle inventory (LCI) – and their associated environmental impacts (ISO, 1997, 2006).

Wenzel et al. (1997) identify four main applications of LCA, one of which is identifying improvement potentials, including comparing alternative solutions, and thus assisting in decision making during process and product design. LCA may be used to compare several aspects of EoL management. For instance, Barba-Gutiérrrez et al. (2008) evaluated different waste scenarios for some EE products focusing on the environmental impact of their transportation during take back, in order to provide information and tools for decision-making on a new reverse logistic network.

However, of special interest in this text is how LCA methodology provides a means to determine which EoL processing of WEEE is the best – or, as Goggin and Browne (2000) put it, the least harmful – option, under an environmental perspective.

Using literature data, Bigum et al. (2012) applied LCA to determine the environmental impacts associated to the process of recycling and recovery of metals from high-grade WEEE. The product system considered starts after collection, when a manual sorting is performed, followed by shredding, air/magnetic/Eddy current/optical sorting and refining stages for a specific metal to be recovered, namely precious metals such as gold, silver and palladium.

LCA-based studies have been conducted to compare certain aspects of different copper recovery processes. Johansson and Björklund (2010) applied LCA to determine whether a proposed prestep for copper removal may be environmentally beneficial compared to a standard shredding process. Improvements in the efficiency of the recovery system help lowering energy consumption, and therefore alterations in global warming potential due to lower emissions during energy production. Li and Guan (2009) compare copper recovery from copper slag and copper ore, the latter being less resource-intensive and impacting.

Such studies indicated that it might be possible to use LCA methodology to compare copper recovery through electrochemical processes in terms of their environmental burdens. Besides being applicable for the processes under study, LCA is also regarded as an acceptable and reliable scientific tool and for these reasons it was used in the work reported here.

3. Methodology

3.1. The system under study

The recovery process chosen for this study is a combination of mechanical and electrochemical processing that reaches a recovering rate of 99% (Veit, 2005; Veit et al., 2006). This choice is justified by the fact that, while this process is highly efficient in terms of material recovery, it requires the use of acid solutions and therefore has potential for severe environmental impacts. It is comprised of the following steps:

- comminution: in this step, PCB samples are comminuted on a cutting mill until they are smaller than 1 mm;
- magnetic separation: the magnetic fraction of the PCB scrap is separated from non-magnetic materials in order to minimize

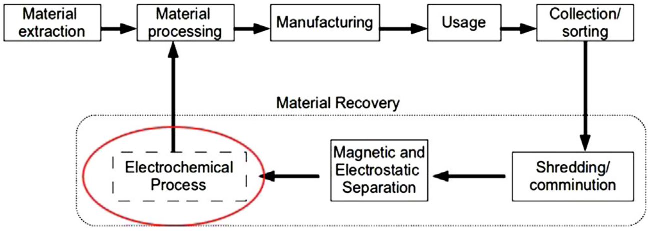

Fig. 1. Life cycle of PCB including the system considered in the study.

impurities in the final solution; the magnetic fraction content is 2.25% wt (in weight), and is composed by more than 40% of iron, as well as nickel, copper and lead. The equipment presents a material loss of 12.5% wt.

- electrostatic separation: separates the conductive fraction (10.53% wt) from the rest of the sample; copper accounts for 51.55% of that fraction, while tin, lead and aluminum are found in lower concentrations.
- electrochemical process: the material obtained from the previous steps is dissolved by an acid solution and used in an electrolysis for copper production by electro refining. Initial concentration of copper in the solution for the electrochemical process was 5.155 g/L decaying to 0.05155 g/L. Efficiency of 99% was adopted for both processes, because that was the peak value, reached after 60 min into the process.

Two different acid solutions were analyzed in Veit (2005) and Veit et al. (2006): (1) acqua regia (combination of nitric and chloridric acids, in a 1:3 ratio) and (2) sulfuric acid. When using LCA methodology to compare systems, it is possible to rule out common life cycle steps, therefore only the electrochemical processes were evaluated; that implied the assessment of only the life cycle inventory data related to the acids used for the solutions.

When applying the LCA methodology, the first step is to provide goal and scope definitions for the study. The following elements were defined:

- *Product system*: the product system considered in the study is presented in Fig. 2, identifying the EoL strategy for the life cycle of PCB's. The only unit process that differs from one life cycle to the other is the electrochemical process, which was highlighted on Fig. 1 and further detailed in Fig. 2. Comminution and magnetic/electrostatic separation are mechanical processing and common to both systems under study.

Avoided burdens were considered in the study inasmuch as the copper contained in PCB scrap is intended to be recycled back into new PCB's.

- *System function*: recovery of copper from PCB scrap;
- *Functional unity*: recovery of 102 g of copper;
- *Reference product corresponding to the functional unity*: considering that the goal of the assessment was to compare the electrochemical processes, the adopted reference product was 200 g of PCB shredded in particles less than 1 mm and concentrated by magnetic and electrostatic separation ('pre-processed PCB'). It is important to stand out that, in order to obtain this quantity of PCB after mechanical processing, 1.898 kg of PCB, free of hazardous components, must be provided to the system.
- *Allocation procedure*: initially, the allocation procedure was intended to be performed only on the electrochemical obtention, due to deposition of other materials such as tin and lead in the cathode. However, the fractions of these materials were too small to be representative, so allocation was not performed.
- *Environment impacts, resource consumption, assessment and interpretation*: the assessment was performed according to the Danish method for Environmental Design of Industrial Products (EDIP). The EDIP method was developed over a 5 year period from 1991 to 1996 by a team comprising major Danish companies within the electro-mechanical industry, the Confederation of Danish Industries, the Institute for Product Development, and the Danish Environmental Protection Agency. The EDIP method is in compliance with the methodological requirements of the ISO 14040, 14041, 14042 and 14043 standards, and addresses all impact categories quantitatively, including environmental impacts and resource consumption (Wenzel and Alting, 1999); although data of EDIP are regional, the method has global validity.

Table 2
Inputs and outputs for process 1 (acqua regia).

Inputs	Quantity
Chloridric acid (HCl)	450 ml or 531 g[a]
Nitric acid (HNO_3)	150 ml or 226.5 g[b]
H_2O for dilution	19400 ml
Pre-processed PCB	200 g
Outputs	**Quantity**
Copper	102 g
Residue (HCl + HNO_3 + H_2O + PCB)	20 l
Other materials (metals)	4.26 g

[a] d (HCl) = 1.18 g/cm^3.
[b] d (HNO_3) = 1.51 g/cm^3 at 20°C.

In order to apply the EDIP method, energy consumption, as well as the heat generated after samples dissolution were assumed to be equal for both processes.

As for the life cycle impact assessment phase, the study evaluated the following impact categories, as defined by the EDIP method: global warming potential (GWP); stratospheric ozone depletion (SOD); acidification (AEP); aquatic eutrophication (AE); terrestrial eutrophication (TE); renewable non-renewable resources consumption; and renewable energy consumption. The study also adopted the equivalency and normalization factors recommended by the EDIP method. The time scales for global warming potential and for ozone depletion were, respectively, 100 years and infinity, as presented in Wenzel et al. (1997).

The impact potentials and resources consumption were normalized into person equivalents (PE), which correspond to the community average impact per person, as defined in Wenzel et al. (1997).

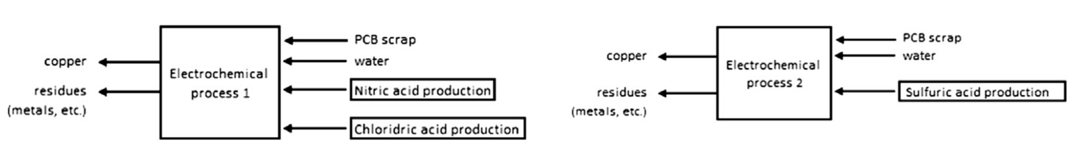

Fig. 2. Electrochemical processes and their inputs and outputs.

Table 3
Inputs and outputs for process 2 (sulfuric acid).

Inputs	Quantity
Sulfuric acid (H_2SO_4)	2000 ml or 3671.2 g[a]
H_2O for dilution	18,000 ml
Comminuted PCB	200 g
Outputs	**Quantity**
Copper	102 g
Residue (H_2SO_4 + H_2O + PCB)	20 l
Other materials (metals)	2.72 g

[a] d (H_2SO_4) = 1.84 g/cm^3, liquid.

The normalization values used for the impact categories and resource consumptions were the ones adopted by the EDIP method. In the case of environmental exchanges which apply on a global scale, the EDIP method uses the total global impact as the normalization reference; global impacts are also used for non-renewable resources (Wenzel et al., 1997), so the units of the normalization values have global scale.

Data related to renewable resources and energy consumptions were not normalized.

4. Data inventory

The inventory for inputs and outputs was comprised of secondary data, based on the quantities used by Veit (2005); the author employed 10 g of PCB for 100 ml of sulfuric acid or 30 ml of acqua regia, then added water to complete 1000 ml. Once the reference product adopted in this assessment was 200 g of pre-processed PCB, the inputs to be considered in the analysis were obtained by multiplying the necessary volumes of the acids and the final solution by 20.

Tables 2 and 3 present, respectively, the values for the inputs and outputs of the acqua regia-based process and the sulfuric acid-based process.

Life cycle inventories data concerning H_2SO_4, HNO_3 and HCl come from databases (PE International GmbH, 2007) and are shown in an aggregated way in Tables 4 and 5.

5. Results and discussion

Fig. 3 summarizes the findings related to the impact categories of each process, after normalization.

As shown, acidification is by far the biggest issue presented by these processes; this finding was already expected because of the nature of the substances used in both processes, and is in line with the statement of Yu et al. (2009) on the potential toxicity of hydrometallurgical processes. The sulfuric acid-based process presented the most significant potential impact, which is 1.26 times higher than the process that uses acqua regia.

Table 4
Life Cycle Inventory (LCI) data for the electrochemical process using aqua regia solution (process 1), considering the functional unit.

HNO_3 emissions to air	Value
Nitrous oxide	0.768 × 10^{-3} kg
HCl emissions to air	**Value**
Carbon dioxide	0.125 kg
Methane	0.460 × 10^{-3} kg
Nitrogen oxides	0.164 × 10^{-3} kg
Sulfur dioxide	0.085 × 10^{-3} kg

Source: PE International GmbH (2007).

Table 5
Life Cycle Inventory (LCI) data for the electrochemical process using sulfuric acid solution (process 2), considering the functional unit.

H_2SO_4 emissions to air	Value
Carbon dioxide	0.878 kg
Carbon monoxide	0.488 × 10^{-3} kg
Methane	3.965 × 10^{-3} kg
Nitrogen oxides	1.215 × 10^{-3} kg
Sulphur dioxide	19.079 × 10^{-3} kg

Source: PE International GmbH (2007).

In this scenario, if one plans to employ hydrometallurgical processes that depend on acid solutions, a proper after-use management of these solutions becomes mandatory, in order to reduce their acidification potential. Both processes might be followed by the neutralization of the acid solutions by sodium hydroxide (NaOH) or calcium oxide (CaO); however, despite being popular, neutralization implies the generation of huge amounts of sludge (Li et al., 2012). Acids such as H_2SO_4 can be recovered by other methods: evaporation, distillation, solvent extraction and diffusion dialysis, among others (Li et al., 2012; Jeong et al., 2005). Solvent extraction processes can be used to recover sulfuric acid from bleed streams, eliminating the need to neutralize these effluents (Gottliebsen et al., 2000). Diffusion dialysis, a membrane separation process, can also be considered, once it has been employed to recover H_2SO_4 (Li et al., 2012), HCl (Xu et al., 2009) and HNO_3 (Lan et al., 2009).

As for the remaining impact categories, there were not significant results, and the possible causes are low or inexistent emissions – or occasional consumption, in posterior stages of the product chain – of substances such as:

- carbon monoxide, carbon dioxide, methane and halogenated organic compounds such as CFC's (*Global Warming Potential*);
- halogenated organic compounds (*Stratospheric Ozone Depletion*);
- nitrogen and phosphorus-based substances, such as nitrates, nitrites and phosphates (*Aquatic and Terrestrial Eutrophication).*

Consumption of renewable resources and renewable energy is illustrated on Fig. 4.

Once again, it is clearly noticeable that the consumption of a few specific resources – air and water – pose problems on this end-of-life strategy. Overall, the process that utilizes sulfuric acid is more resource-intensive than the process that utilizes acqua regia.

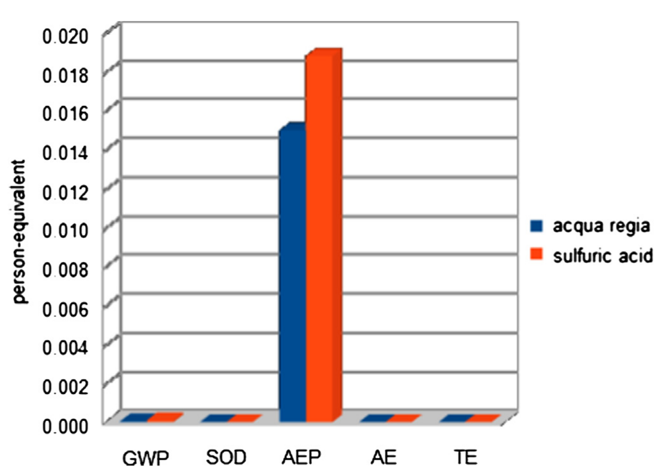

Fig. 3. Comparison of impact categories.

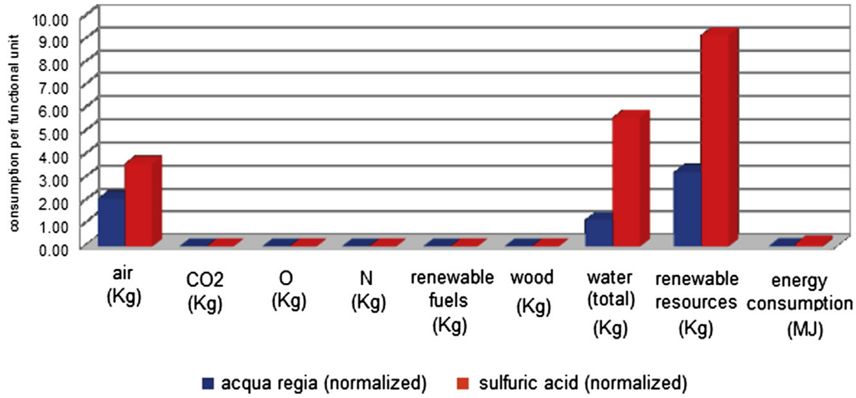

Fig. 4. Consumption of renewable resources and renewable energy.

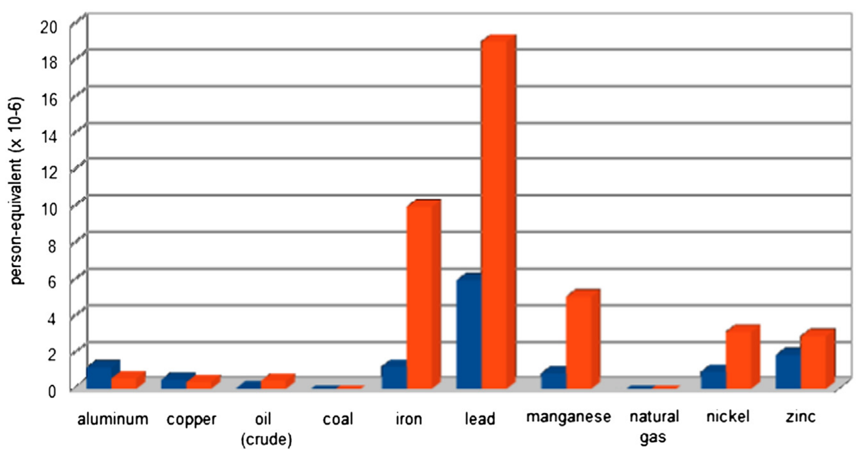

Fig. 5. Non-renewable resources consumption.

The low or non-representative results for consumption of CO_2, O and N may be related to the equally non-representative results of some impact categories, as previously discussed.

Fig. 5 presents findings on non-renewable resources consumption.

First of all, it was perceived that the non-renewable resources consumed by the processes are mostly metals, as seen on Fig. 5. The process that utilizes sulfuric acid is surpassed by the acqua regia-based process only in the consumption of aluminum ($\sim 2\times$) and copper ($\sim 1.3\times$). Otherwise, its consumption of other materials was always higher - and, in the case of manganese ($\sim 6.2\times$), iron ($\sim 8.2\times$) and lead ($\sim 3.2\times$), much higher. Therefore, it becomes clear, from these results, that the electrochemical process that uses acqua regia has the best performance.

The use of sulfuric acid implies high consumption of metals such as lead and iron probably due to the fact that it is obtained from sulfur dioxide, which in turn can be produced by roasting metal ores that are compounds of sulfur – some important examples are precisely lead, zinc, iron and nickel sulfides; wherever metals such as these are processed, the resulting sulfur dioxide can be converted on the site to sulfur trioxide, SO_3, and thence to sulfuric acid (Britannica Academic Edition, 2012).

Overall, the acqua regia-based process performed better in this evaluation, but that does not exempt it from being compared to other methods that do not rely on the use of dangerous substances which may pose additional environmental risks themselves.

As for LCA uncertainty issues, Ross et al. (2002) warn that there are two main sources of uncertainty that are able to significantly compromise the reliability of LCA results: poor data quality and the exclusion of site-specific data from the inventory, the former being a practical problem encountered during the inventory phase of an LCA, and the latter a constraint of the methodology itself. The research reported here, as previously mentioned, utilized data from LCA software databases, so data quality-related uncertainty can be traced to problems during the elaboration and updating of such databases. The exclusion of site-specific data represents a limitating factor to the interpretation of the results, especially in the case of renewable resources and energy consumption: such results are based on the assessment of inventories that do not necessarily reflect the reality of the region or country where the assessment is being performed. For instance, characteristics of energy mixes vary from region to region: some are more dependent on coal, while others rely on hydroelectric generation.

Despite these issues, the LCA approach can provide a valuable insight on the strengths and weaknesses of design options, assisting the decision making in the crucial phase of product and process planning.

6. Conclusions

Currently, proper WEEE management still is often neglected. As discussed earlier, the inadequate management practices applied to this waste stream contrast with its high generation rates, and that may be traced to a lack of incentives or pressures to implement material recovery systems that are simultaneously profitable, environmentally efficient and socially equitable.

The adequate setting of recycling systems, as the ones envisioned in the Brazilian solid waste policy, requires gathering and analysis of information on several aspects of material recovery in order to help not only the design, but also the operation and maintenance of such systems in the country. And, in the absence of a consolidated structure, the most basic choices are yet to be made – for instance, decisions on which collection strategy or even which process or combination of processes should be adopted, among all the alternatives previously mentioned.

In this sense, the review of currently available methods for material recovery revealed a large number of alternatives, and helped shed light on this activity as an effective WEEE management, while the comparison between acid solutions employed on electrochemical processing demonstrated the importance of LCA as a useful tool in decision making.

7. Perspectives and recommendations

Further work is planned in order to extend this research to the complete end-of-life processing of PCB scrap. The authors of this paper also understand that using LCA methodology to evaluate and compare material recovering processes represents a starting point for more in-depth evaluations of these processes, as well as a stimulus to conduct other kinds of investigations.

It is necessary, for instance, to assess issues regarding treatment of the effluents generated by each process, as well as the economic aspects associated to them. In that sense, suggestions for future research include:

- utilization of LCA methodology to compare hydrometallurgical and pyrometallurgical processes;
- proposal and comparative assessment of recycling and reuse options for the acid solutions after the electrochemical process;
- evaluation and comparison of material recovery processes in terms of economic, environmental and social benefits and impacts.

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Glossary

AE: aquatic eutrophication
AEP: acidification
EoL: end-of-life
EDIP: Environmental Design of Industrial Products
EEE: electrical and electronic equipment
GWP: global warming potential
LCA: Life Cycle Assessment
PCB: printed circuit board
RoHS: restriction of hazardous substances
SOD: stratospheric ozone depletion
TE: terrestrial eutrophication
WEEE: waste electrical and electronic equipment