Comparison of current practices for a combined management of printed circuit boards from different waste streams

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Abstract

Waste electrical and electronic equipment and end of life vehicles are two of the main sources of solid waste (after municipal solid waste), in terms of both volume and growth rate. Although they have begun to be adequately regulated worldwide, the management of printed circuit boards embedded into them still presents many challenges. One of these challenges is related to the management of automotive electronic waste. The development of the automotive industry enabled the wide application of electronics within cars. This way, the similarity with electrical and electronic equipments have increased during the last decades, especially considering the presence of printed circuit boards. In spite of these increasing similarities, the treatment of waste printed circuit boards from both electrical and electronic equipments and end-of-life vehicles still follows quite different paths. The aim of this paper is to highlight the unsustainability of their different treatment. A comparison of current practices and a quantification of potential improvements arising from a combined management of printed circuit boards are described within the paper, in terms of both volume and profit. The results demonstrate, even if only theoretically, how a change in managing waste printed circuit boards could offer interesting business opportunities.

Keywords: Waste Printed Circuit Boards; Waste Electrical and Electronic Equipment; End of Life Vehicles; Waste Management; Unsustainable Practices.

1. Introduction

Waste electrical and electronic equipment (WEEE) and end-of-life vehicles (ELVs) – the two waste streams taken into account within this paper – are among the main sources of secondary raw materials [(Eurostat, 2015a); (Eurostat, 2015b)]. Based on official estimates, their annual volumes vary –

for Europe alone – from 7 to 12 million tons [(Reuter et al., 2013); (Ongondo et al., 2011)] and from 7 to 14 million tons [(Eurostat, 2015a); (Andersen et al., 2008)] for WEEE and ELVs, respectively. Companies, for the production of new goods, already reuse about 50% of WEEE and 70% of ELV amounts [(Reuter et al., 2013); (Reuter et al., 2006)]. Because of their impressive annual growth rates worldwide [(Sansotera et al., 2013); (Li et al., 2013); (Passarini et al., 2012); (Vermeulen et al., 2011)], during the last decades many international directives and national laws have been introduced to control material flows both landfilled and illegally shipped abroad. In general terms, these regulations adopted a weight-based approach to define the average mass to be recovered from a specific waste stream. However, the application of these laws resulted in an improved recovery of basic materials (e.g., steel, aluminium, plastics, glass, and wood). In contrast, the recovery of valuable and critical materials was disadvantaged as their mass percentages within products are very low. Thus, many issues have been raised in recent years:

- Significant amounts of valuable resources continue to be landfilled (or lost) each year during the recovery process [(Reuter et al., 2013); (Rochat et al., 2007)], with huge economic and environmental losses for the worldwide society [(Despeisse et al., 2015); (Widmer et al., 2015)]. This is particularly critical as the natural environment capability to fulfil the request of materials from the market is also decreasing (Yahaya, 2012).
- Original equipment manufacturers (OEMs) started to design components and select materials embedded into products in a sustainable way only since the last decades. In the past, substitution strategies were the only adoptable choice and eco-friendly policies were not available or rarely taken into account [(Xu et al., 2014); (Kuo, 2010)]. Therefore, current reverse manufacturing processes treating old wastes (e.g. ELVs) are coping with complex products that are very difficult to recover. Subsequently, the development of sustainable reverse manufacturing processes is becoming very expensive.
- Worldwide regulations dedicated to WEEE and ELVs still do not push reverse logistics actors to invest in dedicated recovery plants for valuable materials, as their focus is on recovery plants for basic materials (e.g. steel, aluminium, plastics, glass, and wood) [(Li et al., 2014); (Chatterjee, 2012)].
- Current products and material recovery technologies are not able to reach the performance levels defined by WEEE and ELV global regulations [(Morselli et al., 2010); (Jody et al., 2009)]. More sustainable technologies are barely developed or limited to pilot scales [(Xue et al., 2013); (Zeng et al., 2013)].
- Reverse logistics chains are strongly disconnected, and collaboration among different actors is even more difficult, especially for patented components [(Hu and Wen, 2015); (Wübbeke and Heroth, 2014)]. This way, the chance to develop dedicated networks able to better intercept valuable scraps from different waste streams is limited. In some case, the scarce availability of treatable volumes does not support the development of a dedicated reverse logistics chain [(Li et al., 2014)]. This favours their illegal export in developing countries.
- Recovery plants, focused on basic materials, exploit neither flexibility nor economy of scale opportunities [(Hu and Wen, 2015); (Jody et al., 2009)]. This way, the expected augment of future waste mixes could become a big issue if not managed on time.
- Current end-of-life (EoL) business models and strategies need a strong revision in terms of sustainability performance [(Li et al., 2014); (Xu et al., 2014)]. However, both global regulations and the absence of already known best practices discourage newcomers to enter the market and already present actors to act differently.

The paper assesses these issues with a specific focus on the treatment of printed circuit boards (PCBs) from WEEE and ELVs. The evolution of vehicles of the last few years saw an incredible increase in the number of electronic devices (e.g., PCBs) within cars. Because of the huge similarities characterizing both WEEE and ELV PCBs, a different (waste stream-dependent) treatment appears rather inefficient. Starting from this point, the final aim of this work is to address the presence of potential chances to establish a combined treatment of them, whatever their source. To this aim, a comparison of current practices characterizing the two waste streams is carried out within the paper. Furthermore, a quantification of potential improvements arising from a change in the management of waste PCBs is conducted, in terms of both volume and profit. The final results, although only estimates, demonstrate how an integrated recovery of PCBs could potentially offer great advantages to public and private companies, not only in environmental terms but also in economic ones. The potential profits calculated in this work demonstrate this thesis. They are estimated to vary between 3.3 and 4.9 billion € at minimum and between 13.2 and 18.1 billion € at maximum. In addition, by considering the current evolution of transportation towards hybrid and electric technologies and auto-guided systems, these numbers are projected to further increase in the next several decades. Thus, it is of utmost importance to solve this issue before it becomes unmanageable. Only by highlighting the issue and trying to quantify it we can hope to enable a positive discussion among the experts and a change in the mental models of decision makers within international governments and industrial companies.

The paper is organized as follows:

- Section 2 presents a series of inequalities about the current management of electrical and electronic equipments and cars during their end-of-life stage, focusing on discussions in the literature, physical processes, strategies and environmental issues;
- Section 3 assesses the existing commonalities in their management practices, describing PCBs' technological issues, recovery processes, physical characteristics and existing economic models;
- Section 4 conducts an overall discussion and assessment of potential improvements offered by a combined management of waste PCBs;
- Section 5 presents some future perspectives;
- Section 6 concludes the paper.

2. Current inequalities in WEEE and ELV management practices

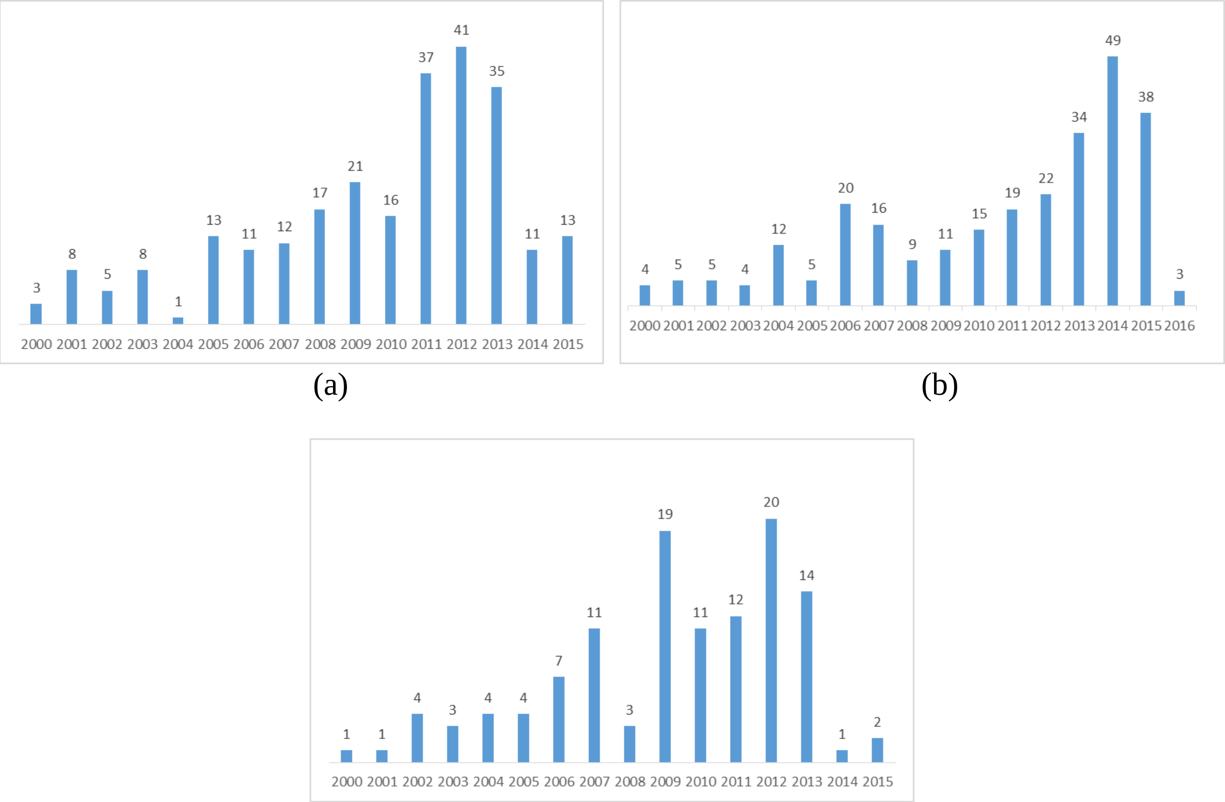
During the years, several papers (see section 2.1) have been published about the recovery of raw materials from WEEE and ELVs. The concern for ELV recovery dates back to the 1960s (Buekens and Zhou, 2014) and the reuse of scrap metals in foundries to produce new materials has existed for centuries. In contrast, WEEE recovery is a modern process dating back to only the 1990s. This means that, even though similar processes and technologies are – or seemingly should be – applied for the treatment of both the fractions, their different development paths have resulted in quite different focuses and performance (Jalkanen, 2006).

The management of PCBs is one of the significant differences in the management of these waste streams. WEEE recovery processes follow clear guidelines, procedures and responsibilities for the treatment of PCBs. In contrast, ELV recovery processes have little or no information about the treatment of PCBs, except on their hazardousness. This lack of information pushed ELV dismantlers to act autonomously. Hence, only few of them currently exploit the official WEEE recovery channels also for the treatment of automotive PCBs (Wang and Chen, 2011). No reference recovery levels exist yet and great differences among nations still exist (European Union, 2000). The aim of this section is to present the main differences between the two waste streams taken into account.

2.1. State of the art analysis on WEEE, ELVs and PCBs

A first distinction between WEEE and ELVs is the different way in which the international literature assesses their current issues. For WEEE, issues related to a sustainable management of PCBs are a common topic among the experts. Likewise, almost all papers that discuss PCBs consider WEEE as their main source [(Wang and Xu, 2015); (Yamane et al., 2011)]. For ELVs, there is a completely different trend. The issues about a sustainable management of ELVs are well assessed by the experts, as a response to even more stringent requirements. However, the focus is still on alternative – or better – ways to increase recovery percentages of the car hulk (or the remaining mass of a car after depollution and dismantling) that is currently incinerated or landfilled – the so called automotive shredder residue (ASR) [(Zorpas and Inglezakis, 2012); (Vermeulen et al., 2011)]. Therefore, the literature prefers to follow the same weight-based principle of the ELV Directive instead of focusing on a better exploitation of valuable elements embedded in ELVs. Thus, the experts rarely considered automotive PCBs and data about them are difficult to gather from official sources of scientific information.

The discovery of these literature gaps was supported by a structured literature review of articles speaking about WEEE, ELVs and waste PCBs published from 2000 to 2015 and implemented before starting to write this paper. This activity was fundamental for the identification of both industrial and regulation gaps and their commonalities. The terms “printed circuit board”, “waste printed circuit board” and their acronyms (i.e., PCB and WPCB) were linked to terms such as “waste electrical and electronic equipment”, “end of life vehicles”, and their related acronyms (i.e., WEEE and ELVs). These terms were searched for in the titles, abstracts, and keywords of top scientific journals and conference proceedings, international directives and reports, and industrial reports. The reviewed publications were gathered into two phases. First, 363 scientific and industrial documents focusing on WEEE and waste PCBs were gathered. Subsequently, 246 scientific and industrial documents focusing on ELVs were gathered. Scientific papers were selected through the most popular scientific works search engines (e.g., Google™ Scholar, Sage™, Science Direct™, Springer™, Emerald™, Scopus™, Taylor&Francis™ Online, and Wiley™ Online Libraries). After reading all of the articles, a structured literature review was performed, and the main results are summarized here. Figure 1 displays the results of the search, in terms of both the number of papers per year and publication trends. The total number of papers (637) shows the enormous attention devoted to these topics (from 2000 to 2015), especially for WEEE and ELVs (252 and 268 papers, respectively).

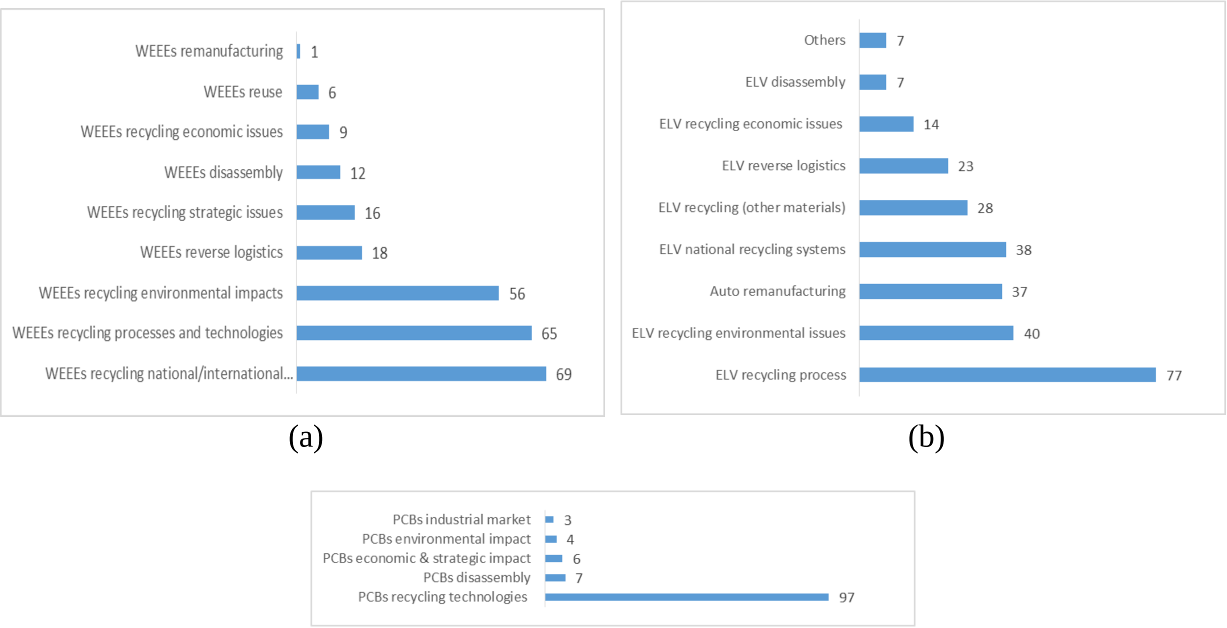


(a) (b)

(c)

Figure 1: Historical numbers of published papers – (a) WEEE, (b) ELVs, (c) PCBs

The papers included 392 publications in scientific journals with an impact factor, 63 in scientific journals without an impact factor, 86 in scientific conference proceedings, 59 scientific reports, 15 book chapters, and 25 industrial reports. There are several perspectives from which WEEE, ELVs, and PCBs were addressed. In macro terms, national and international policies are the most discussed topic for WEEE. Again, recycling processes are the most discussed topic for ELVs. Finally, recycling technologies are the prevailing topic for PCBs, as shown in Figure 2.



(a) (b)

(c)

Figure 2: Main topics of published papers – (a) WEEE, (b) ELVs, (c) PCBs

The analysis highlighted the multidisciplinary aspect of the research. For this reason, the journals pertain to various research fields and scientific areas. Subsequently, the literature analysis classified the papers by detailed topics (the related graphs are not reported here because of the purpose of the paper). However, it is only important to evidence that, in almost all the cases, works did not look either automotive PCB recovery or WEEE PCB business models. Furthermore, none of the works considered automotive PCB recovery business models at all. This situation underlines the scarce interest of the international literature regarding automotive PCBs (and EoL business models in general), although many experts (e.g., [(Sakai et al., 2014); (Zorpas and Inglezakis, 2012)] emphasize the need to address these two elements to achieve a more sustainable use of resources.

2.2. Physical treatments of WEEE and ELVs

A second difference characterizing WEEE and ELVs is their collection and recovery process. The collection of waste from mass electronics, household appliances, and automotive channels follows different logics and paths that vary from country to country, even within Europe [(Ongondo et al., 2011); (Sthiannopkao and Wong, 2013)]. WEEE is collected by both national (e.g., municipal collection sites) and commercial channels (e.g., mass electronics and household appliance producers, distributors, or dealers), as shown in Figure 3. In Europe, the amount of collected waste is managed by several authorized actors (consortia) - supported by a public waste management authority - who transfer the collected volumes and take responsibility for their regular treatment. Thus, the whole recovery chain (clearly defined within the WEEE Directive [(European Union, 2012); (UNEP-DTIE-IETC, 2012)] is mapped and controlled by a unique institutional actor.

The ELV management process is slightly different [(Zhao and Chen, 2011); (Blume and Walther, 2013)]. Cars can reach their end of life into two ways. If they are new models reaching the end-of-life (EoL) stage because of a severe accident (so-called premature ELVs), they are directly managed by insurance companies, body shops, or auto wreckers. Instead, if cars are old models reaching the EoL stage because of obsolescence (so-called natural ELVs), they can be managed by private owners, car dealers, insurance companies, or auction houses, as shown in Figure 4. Then, a network of authorized dismantlers buys the wasted cars (from one or more of the previous actors) and proceeds with their disposal, following specific procedures (clearly defined within the ELV Directive) for the correct recovery of several components. However, there is neither a predefined recovery chain structure nor the presence of institutional actors responsible for its coordination.

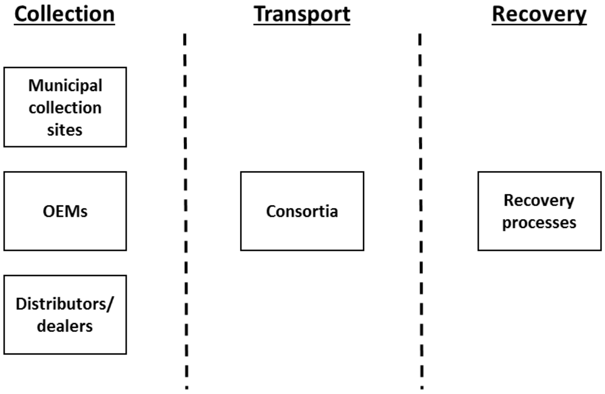


Figure 3: The current collection process of WEEE – Adapted from (UNEP-DTIE-IETC, 2012)

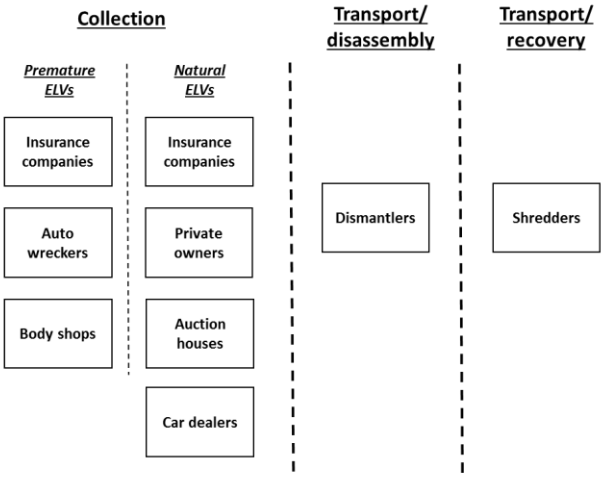


Figure 4: The current collection process of ELVs – Adapted from (Blume and Walther, 2013)

Important distinctions can also be found in recovery processes. A representation of a typical WEEE recovery process is shown in Figure 5, under the form of an IDEF0 model. In general, consumer and industrial WEEE is collected and directly transferred to authorized treatment facilities. Here, depending on its type, WEEE is disassembled up to divide both valuable and hazardous components, which are then stored and transferred to dedicated recovery plants. The remaining WEEE wreck is directly shredded and separated onsite to recover basic materials (e.g., construction metals, plastics, wood, glass, paper, and concrete) [(Chatterjee, 2012); (Dalrymple et al., 2007)]. Because PCBs are one of the most valuable components embedded in WEEE, they are separated from the waste product during disassembly, classified into one of three categories (i.e., high, medium, or low value cores), stored, and transferred to dedicated plants (Reuter et al., 2013). These plants, specialized in PCB treatment, take PCBs from tier-one recyclers, characterize them (based on the expected gold content), and either pay back the related value of gold embedded in the wastes or refine a set of selected materials by sending them back to the owners, and requesting a recovery fee (Rochat et al., 2007).

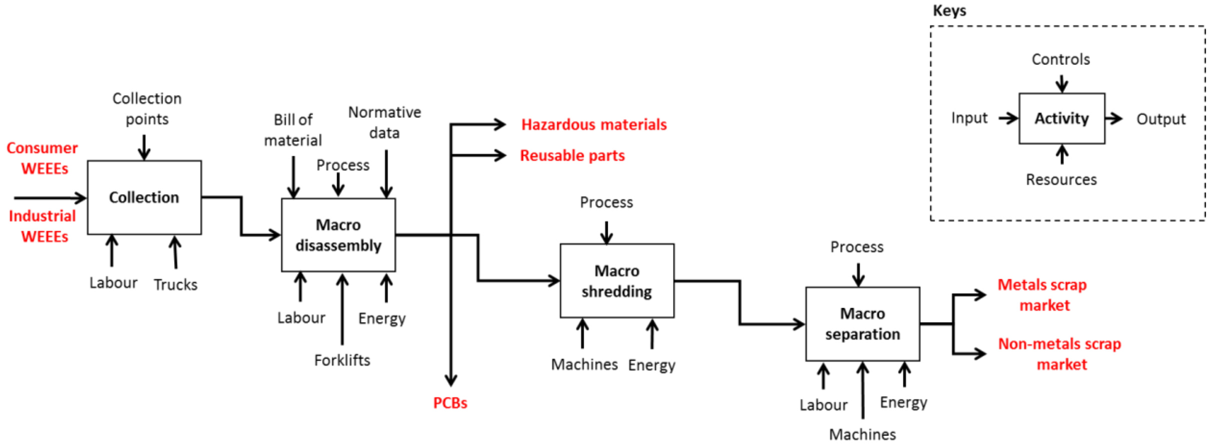


Figure 5: The current recovery process of WEEE – Adapted from (Dalrymple et al., 2007)

For the automotive sector, a typical ELV recovery process is reported in Figure 6, under the form of an IDEF0 model. Regardless of the ELV type (both premature and natural), they are collected and deleted from the public register, and the main hazardous components (e.g., batteries, fuel, and filters) are removed. Subsequently, most of the valuable parts (e.g., catalyst, engine, and some mechatronic components) are disassembled (if in good conditions and with a market request), and reused as spare parts in secondary markets. The car hulk is then crushed and fragmented into small scraps. Subsequently, the scraps are separated by exploiting their physical characteristics (e.g., density, weight, and magnetism) to obtain uniform groups of materials. In general, the metal part is directly reintroduced into the automotive supply chain (as input material for foundries). The non-metal part (generally named Automotive Shredder Residue - ASR) is currently landfilled or used as fuel for energy generation [(Hu and Wen, 2015); (Ni and Chen, 2015)]. Even if information about the point at which PCBs leave the ELV recovery process are available [(Kumar and Sutherland, 2009); (Williams et al., 2007)], information about their final destination are difficult to gather. However, based on some papers discussing this issue [(Wang and Chen, 2012); (Wang and Chen, 2011)], it is possible to confirm that, if not disassembled for direct reuse, automotive PCBs are crushed together with the car hulk, becoming an irrelevant percentage of both metal and non-metal fractions (Cucchiella et al., 2016a). This lack of information could be explained by the lifecycle of a car, which is estimated by experts [(Fiore et al., 2012); (Vermeulen et al., 2011); (Mazzanti and Zoboli, 2006)] to be approximately 10–15 years before becoming an ELV. Therefore, ELVs currently taken into account by the literature are cars from the 1990s or perhaps the early 2000s. Thus, these cars contain very limited amounts of electronic components.

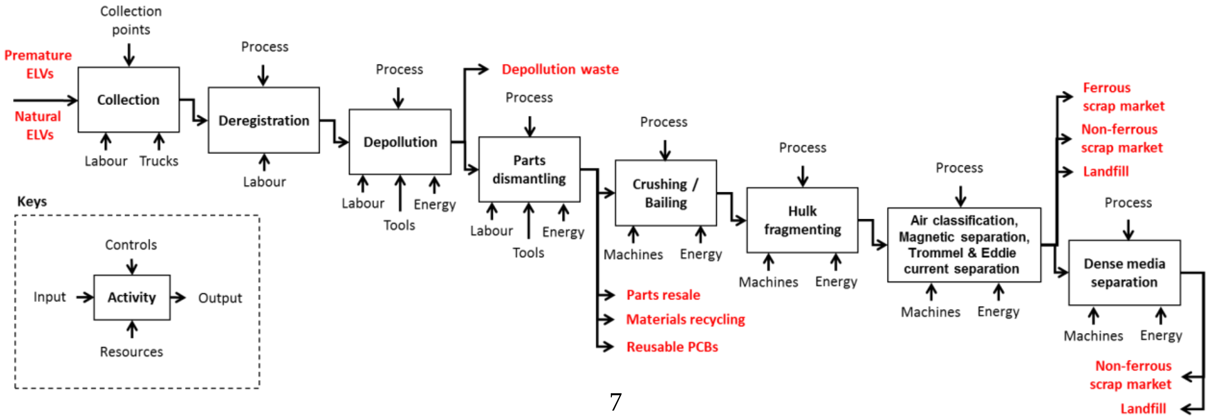


Figure 6: The current recovery process of ELVs – Adapted from (Vermeulen et al., 2011)

2.3. Common end-of-life strategies for WEEE and ELVs

Another difference between WEEE and ELVs is their main EoL strategy. Recycling is the preferred strategy for the management of WEEE components [(Zeng et al., 2015); (Baldé et al., 2014)] and remanufacturing is the most common one for the recovery of ELV components, particularly in the USA [(Freiberger et al., 2012); (Gerrard and Kandlikar, 2007)]. However, the presence of different strategies has to be related to the intrinsic value of the cores. The components embedded in WEEE are generally low or medium value elements, and their remanufacturing would not allow recyclers to recover the sustained costs (D'Adamo and Rosa, 2016). In contrast, automotive components (especially the mechatronic ones) have a very high value (because of their complexity), and the demand from the secondary market is well developed (Kripli et al., 2010). Therefore, remanufacturing costs are completely covered by revenues from their resale, guaranteeing good profits to all of the actors involved in these reverse logistics chains.

2.4. Environmental impacts and illegal flows of WEEE and ELVs

The final distinctions between WEEE and ELVs are their impact on the environment and their illegal flows. From the environmental point of view, several works [(Lecler et al., 2015); (Wang et al., 2015); (Kiddee et al., 2013)] analyse WEEE and PCBs. These papers showed as the overall impact of WEEE and PCBs on the environment (and human health) is caused by the treatment of considerable amounts of flame retardants [(Hadi et al., 2013a); (Duan et al., 2011)] and different types of plastics, especially polybrominated diphenyl ethers (PBDEs) [(Chancerel and Rotter, 2009); (Dimitrakakis et al., 2009)]. In contrast, ELV environmental impacts comes from both the type of metallurgical processes applied for the recovery of basic metals and recovery processes for the treatment and incineration of ASRs [(Vermeulen et al., 2011); (Viganò et al., 2010)].

Looking at illegal shipments, there are clear distinctions characterizing both WEEE and ELVs [(Fischer et al., 2012); (Reichel et al., 2012)]. The volumes and final destinations are very different. For WEEE, illegal flows are approximately 50% of the total volumes generated worldwide each year. This means that by considering a global annual amount from 30 to 50 million tons of WEEE [(Wang and Gaustad, 2012); (Xue et al., 2012)], illegal shipments account for 15–25 million tons. Furthermore, their final destinations are well known by the experts [(Li et al., 2013); (Huang et al., 2009)] and are represented by several developing countries (e.g., China, India, Pakistan, and Nigeria). In contrast, illegal shipments of ELVs represent a limited issue, quantified by some authors as approximately two million units per year in Europe and approximately one million units in China [(Hiratsuka et al., 2014); (Li et al., 2014)]. The final destinations of ELVs generated within Europe are eastern European and extra-European countries.

3. Existing commonalities in WEEE and ELV management practices

The presence of PCBs and the ratio between their value and the overall value of the whole product allows combining WEEE and ELVs under the same umbrella. PCBs represent the most complex, hazardous, and valuable component of both electrical and electronic equipments and cars. They can contain more than 60 elements (on average), including heavy metals (such as Lead (Pb), Chromium (Cr), Cadmium (Cd), Mercury (Hg), and Arsenic (As)) and toxic organic substances (such as brominated flame retardants, polycyclic aromatic hydrocarbons, and dechlorane plus) [(Song et al., 2013); (Zhou et al., 2013); (Wang and Gaustad, 2012)]. Although WEEE is the main source of waste PCBs due to its greater volumes, the electronic components in cars have increased in quantity and value in the last several decades and are now used for the management of almost all vehicle's functionalities [(Gerrard and Kandlikar, 2007); (Kim et al., 2004)]. Undoubtedly, this trend contributed to the increasing volume of PCBs produced annually and therefore the increasing number of

PCBs to be dismantled. However, PCBs from WEEE and ELVs, although very similar, continue to be treated and regulated in two completely different ways. The aim of this section is to demonstrate these commonalities from several points of view.

3.1. *Main issues in waste PCB recovery processes*

Issues related to the management of waste PCBs are well described by the literature and several works have already been written (Ghosh et al., 2015). First, even though PCBs are known to be the most important component in e-wastes (and among one of the most important components in cars), there are no explicit regulations concerning their treatment. For example, European directives consider PCBs to be hazardous components (such as batteries, air bags, condensers, fuels, and filters) that must be treated separately from the main recycling process of e-wastes and ELVs, but there are no details about specific recovery levels that have to be reached by authorized centres [(European Union, 2012); (European Union, 2011); (European Union, 2000)]. Second, the physical characteristics of PCBs (e.g., materials layering, component miniaturization, and current safety regulations) limit the opportunity to recover 100% of materials, and a significant part of them is unintentionally lost during mechanical treatments, heating phases, or chemical reactions [(Chatterjee, 2012); (Castro et al., 2004)]. Third, common technologies used for PCB treatment are taken from the mining sector (Yahaya, 2012). Therefore, their focus is quantity (and not quality) optimisation, with final recovery rates barely exceeding 20%–30% of materials entering the recovery process (Reuter et al., 2013). Fourth, current regulations do not impose any limitation to PCBs exports from one nation to another. Therefore, local resources that could be potentially maintained within national borders (with positive effects for the local economy) are transferred abroad, implicitly denying any sort of new entrepreneurial initiatives in this context.

3.2. *The recovery process of waste PCBs*

Recent works (Johansson and Luttropp, 2009) have verified that scrap automotive PCBs are, in effect, very similar to PCBs from e-wastes. Consequently, it is possible to use the same technological process for their recovery [(Wang and Chen, 2013); (Wang and Chen, 2012); (Cucchiella et al., 2016b)]. In general, PCB recovery processes can be described as the sum of five main phases that, starting from waste PCBs, allow the recovery of several (almost pure) raw materials. These phases are as follows: pretreatment, disassembly, shredding, separation and refining [(Yahaya, 2012); (Yu et al., 2009)], as shown in the IDEF0 model in Figure 7.

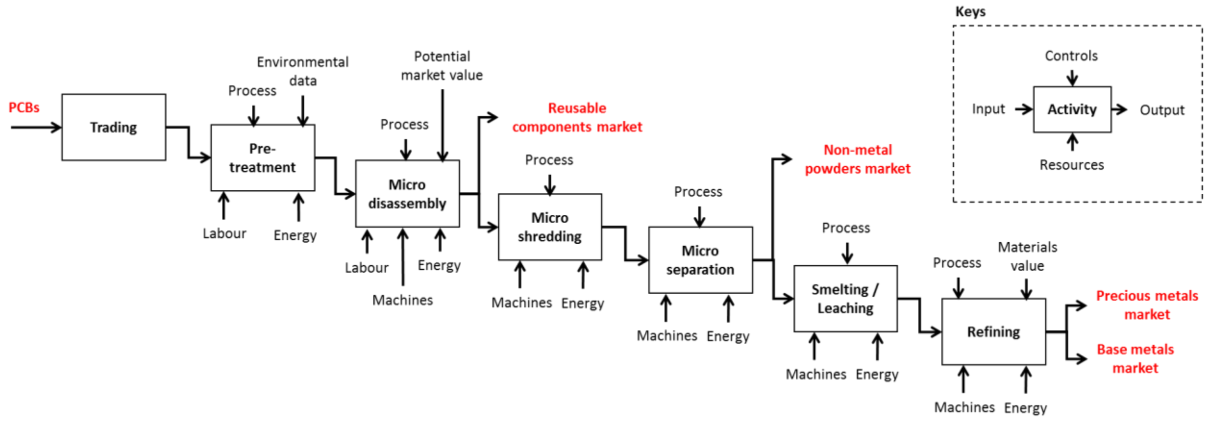


Figure 7: A traditional waste PCB recovery process – Adapted from (Li et al., 2004)

The pretreatment phase prepares PCBs for further recovery. During this phase, PCBs are eventually extracted from their cases and cleaned. The subsequent disassembly phase is responsible for extracting the toxic components present on the main board (e.g., condensers or mini-batteries). Generally, they are disassembled and destined to specific processes for the recovery of hazardous materials. The shredding phase acts a mechanical transformation on products. Here, waste PCBs are crushed into micro pieces to become a uniform powder through a series of dedicated machines (e.g., shredders and grinders) (Duan et al., 2009). Then, the separation phase characterizes the powders basing on their composition by distinguishing metals from non-metals [(Xue et al., 2013); (Guo et al., 2011)]. For this phase, several physical principles of materials (e.g., density, weight, and magnetism) are exploited. Currently, the non-metal fraction is sent to landfills. However, there are interesting works studying alternative (and valuable) ways to reuse them for different purposes [(Hadi et al., 2013b); (Guo et al., 2010)]. Finally, the metal powders are subjected to further chemical transformation during the refining phase. Then, the obtained materials are directly reused for the production of new goods. Their purity level differs from one material to another [(Wang and Gaustad, 2012); (Graedel et al., 2011)]. The refining process can be based on different technologies (e.g., pyrolysis, pyrometallurgy, hydrometallurgy, biometallurgy, or a mix of them) [(Ghosh et al., 2015); (Mankhand et al., 2013)]. In general, hydrometallurgy is considered as the best refining process because of its high level of sustainability in comparison with other methods [(Behnamfard et al., 2013); (Birloaga et al., 2013); (Yang et al., 2011)]. Although biometallurgy could be even better than hydrometallurgy, there is currently no information about its use at industrial scale [(Zhu et al., 2011); (Liang et al., 2010)].

3.3. Materials characterization of waste PCBs

Before the treatment of any type of waste PCBs, the materials undergo a characterization phase (see section 2.2). This phase approximately defines the set of materials embedded in a certain amount of PCBs by chemically analysing a sample of them. This process accomplishes the following: (i) it determines the presence of valuable materials (in order to classify PCBs as high, medium or low-grade ones) and (ii) it defines the expected revenues from their recovery. From a WEEE perspective, information about the material characterization of PCBs are widely available in the literature (e.g., [(Wang and Gaustad, 2012); (Ongondo et al., 2011)], and similar works). Considerable numbers of studies were performed during the last several decades, and currently significant data are available by reading the annual reports released by dedicated international organizations (e.g., (Reuter et al., 2013)). For example, the European Union classifies WEEE into ten categories (see (European Union, 2012) for details) depending on their reference typology. To prove the existence of commonalities with automotive PCBs, four out of the ten WEEE categories were selected as reference samples based on their relevance (approximately 93.4%) to the overall amount of WEEE, in terms of both volume and materials content. These are as follows:

- Type 1 WEEE, or big household appliances (e.g., fridges, washing machines, and air conditioners);
- Type 2 WEEE, or small household appliances (e.g., microwave ovens and vacuum cleaners);
- Type 3 WEEE, or IT and telecommunication equipments (e.g., PCs, tablets, notebooks, and smartphones);
- Type 4 WEEE, or consumer equipments (e.g., TVs, monitors, stereos, and cameras).

The literature has already classified the type of PCBs embedded into these categories [(Reuter et al., 2013); (Kasper et al., 2011)]. Type 1 and Type 2 WEEE are known to embed low-grade PCBs. In contrast, Type 3 and Type 4 WEEE can embed medium or high-grade PCBs. Table 1 reports a short list of materials embedded into each of the four PCBs classes.

Table 1: Valuable materials embedded into electrical and electronic equipment PCBs – source: (Reuter et al., 2013)

Materials	*Type 1 PCBs (%)*	*Type 2 PCBs (%)*	*Type 3 PCBs (%)*	*Type 4 PCBs (%)*
Silver (Ag)	0.01	0.02	0.17	0.08
Gold (Au) (*)	0.003	0.002	0.04	0.01
Copper (Cu)	13.0	11.0	20.0	17.3

(*) For example, 0.003% of Au is equal to 3 ppm or 3 grams of Au in 1 ton of waste PCBs

From an ELV perspective, the lack of information within the scientific literature is considerable. Hence, a characterization of PCBs was implemented in a different way. Data about the materials characterization of automotive PCBs were directly gathered from an official industrial source, the IMDS database (Gerrard and Kandlikar, 2007). IMDS is a materials data management system used by automotive original equipment manufacturers (OEMs). Designed by Audi, BMW, Daimler, HP, Ford, Opel, Porsche, VW, and Volvo, IMDS was then adopted by other car manufacturers, becoming a global standard used by almost all of the automotive OEMs worldwide. In total, data related to almost 500 different automotive PCBs were extracted from the IMDS database. Subsequently, these data were categorized into four typologies, basing on the weights distribution (divided into quartiles). The four resulting groups are represented as followed:

- Small PCBs, from 0.2 grams up to 8.7 grams;
- Medium to small PCBs, from 8.8 grams up to 52.9 grams;
- Medium to large PCBs, from 53.0 grams up to 134.2 grams;
- Large PCBs, from 134.3 grams up to 477.9 grams.

This choice was purely objective and derives from the fact that waste automotive PCBs differ significantly in size, shape, and composition depending on their functionality (Wang and Chen, 2013). Hence, a subdivision such as the one followed for WEEE PCBs was considered to not be representative. Table 2 reports a short list of materials embedded in the four PCBs categories.

Table 2: Valuable materials embedded into automotive PCBs – source: (Cucchiella et al., 2016b)

Materials	*Small PCBs (%)*	*Medium-small PCBs (%)*	*Medium-large PCBs (%)*	*Large PCBs (%)*
Silver (Ag)	0.09	0	0	0
Gold (Au) (*)	0.42	0.20	0.24	0.09
Copper (Cu)	18.84	24.19	14.52	16.30
Tantalum (Ta)	0.08	0	0	0

(*) For example, 0.42% of Au is equal to 4200 ppm or 4200 grams of Au in 1 ton of waste PCBs

By comparing Table 1 and Table 2, it is possible to confirm that the material compositions of WEEE PCBs and automotive PCBs are not significantly different. Instead, what clearly differs is the amount of materials (especially precious metals), with a large impact on the overall profitability of any recovery process (Wang and Gaustad, 2012).

3.4. Economic models supporting the management of WEEE and ELVs

Another important topic linking WEEE and ELVs is the scarcity of economic models supporting their management. Furthermore, the existing models suffer of a set of challenges that impede their practical application by industrial actors. First, models assessing the profitability of recycling plants are focused on a particular phase of the process (Ghosh et al., 2015) and, apart from some exceptions such as (Cucchiella et al., 2016b), the whole waste PCB recycling process is never taken into account. Even if this research method can leave more space for different technological configurations of a recycling plant, it undoubtedly influences the overall economic result given by the proposed model. Therefore, from a practical point of view, this can offer limited support to the industrial actors when they have to decide whether to invest in this type of plants. Second, there is a lack of standards for material composition of waste PCBs taken into account by the experts (Wang and Gaustad, 2012), and a direct comparison of works is not always possible. Because of the role of waste PCB material characterization (see section 3.3) in defining the profitability of the entire recycling process, it is important to correctly characterize wastes to maintain the reliability of results (Cucchiella et al., 2015). Third, the existing economic models present limited application fields (Wang and Xu, 2015). Current studies almost completely focus on PCBs from a particular set of WEEE, or those known by the experts as the most profitable. Hence, because of the lack of data about waste PCBs from ELVs, this topic has rarely been considered (IMDS, 2015).

4. Discussion and assessment of potential improvements

This section discusses what could be the main results of a unified management of waste PCBs from both WEEE and ELVs. This means a quantification of potential volumes and profits and the analysis of their expected trends within the next 15 years. Toward this aim, their calculation procedure was taken from (Cucchiella et al., 2016b).

For WEEE, the overall expected volumes generated from 2015 to 2030 were the first acquired data. These data, together with the related trends, were gathered directly both from Eurostat (regarding 2012 collected volumes in Europe) and the literature (regarding both the ratio between WEEE and PCB mass (estimated to be approximately 3%–6% [(Wang and Gaustad, 2012); (Chatterjee, 2012)]) and the expected growth rate of 3%–5% (Reuter et al., 2013) per year for each category. Then, it was possible to predict (with logical approximations) the expected profits within a min–max range from a correct management of the expected amounts of PCBs. These profits were gathered by multiplying the average weight of each material – compared with the overall PCB average mass – by its unit profit (€/kg) obtained by considering single material market prices, a set of costs characterizing a reference PCB recovery process, and a purity level equal to that required by the market for virgin resources. Table 3 reports the main data derived from the calculation procedure.

Table 3: Estimates of PCB volumes in Europe from WEEE – Sources: ((Eurostat, 2015b); (Reuter et al., 2013); original analysis)

	2015	2020	2030
EU total WEEE expected annual generation (Mtons)	3.73	4.32	5.81
EU total PCB expected annual generation (ktons)	186.50	216.00	290.50
EU total PCB expected NPV – min values (M€)	2399	2781	3737
EU total PCB expected NPV – max values (M€)	4784	5546	7453

For ELVs, the quantification process was more complex. Data about the overall amount of expected volumes generated from 2015 to 2030 – and related trends – were gathered directly from the literature [(Eurostat, 2015a); (Andersen et al., 2008)]. Then, ELV volumes were separated into premature and natural ones. Because of their general condition at the EoL stage, premature ELVs, representing almost 20% of the total volumes generated annually [(Ferrão and Amaral, 2006); (Zhou and Dai, 2012)] were hypothesised to be completely recovered. Instead, natural ELVs representing 80% of the total amount of annual ELV volumes [(Hiratsuka et al., 2014); (Morselli et al., 2010)] were hypothesised to be partially remanufactured. This assumption reduced annual ELV volumes, accountable by the experts in approximately 20%–30% of the overall amount of ELVs [(Hiratsuka et al., 2014); (Wang and Chen, 2012)]. Once the average mass of an ELV was defined, the initial number of vehicles reaching the end of their life was translated into million tons potentially treated and then divided between premature and natural ELVs. The next step was the definition of the average PCB mass (in percentage) out of the total ELV mass, starting from IMDS data. Once defined both the average ELV and PCB masses, a ratio was established (estimated to be approximately 0.1%–0.7% [(Zorpas and Inglezakis, 2012); (Che et al., 2011)] and directly used to quantify the annual generated volumes of PCBs from ELVs. Finally, it was possible to predict (with logical approximations) the expected profits within a min–max range from the correct management of these amounts of PCBs. This last phase followed the same principle previously described for PCBs from WEEE. Table 4 reports the main data derived from the calculation procedure.

Table 4: Estimates of PCB volumes in Europe from ELVs – Sources: ((Andersen et al., 2008); (Ferrão and Amaral, 2006); (Hiratsuka et al., 2014); (Vermeulen et al., 2011); original analysis)

	2015	2020	2030
EU total ELVs expected annual generation (Mtons)	15.43	16.94	19.49
EU total PCB expected annual generation (ktons)	16.97	18.63	21.44
EU total PCB expected NPV – min values (M€)	891	978	1125
EU total PCB expected NPV – max values (M€)	8412	9235	10,628

Considering the data reported in Table 3 and Table 4 together, it is possible to have a picture, even if only hypothetical, of the potential dimension of the overall PCB recovery market in Europe. Looking at volumes, the amounts are measurable in kilotons per year. Even if the hypothesized volumes of PCBs from WEEE are an order of magnitude greater than the ones from ELVs, by considering data reported in Table 1 and Table 2, ELV PCBs could be more profitable on average given their higher content of precious metals. This vision supports the unsustainability of the weight-based principles followed by current European directives. The same result can be deduced by considering the potential profits. Maximum values could be achieved only through a combined recovery of PCBs from both WEEE and ELVs. They are estimated to be distributed between 3.3 and 4.9 billion € at minimum and between 13.2 and 18.1 billion € at maximum. An important part of these profits comes from ELV PCBs, accounting for 23% - 27% at minimum and for 59% - 64% at maximum. These numbers, even if theoretical, demonstrate the extreme importance of the combined management of PCBs and the potential economic impact achievable in the near future (or that is currently lost, looking at only the 2015 data).

In addition, by considering the current evolutions of transportation towards hybrid and electric technologies and auto-guided systems, the use of electronics within cars is expected to further increase in the next several decades (Despeisse et al., 2015). These new types of cars embed high quantities of electric and electronic equipments, with a huge exploitation of valuable and critical materials (e.g., precious group metals (PGMs) in PCBs, rare earth elements (REEs) in electric motors and batteries, and aluminium and magnesium in frames) [(Yano et al., 2015); (Gaustad et al., 2012)]. Therefore, once these cars will reach their end of life, they could become a very important source of materials.

Many authors have already started to study this phenomenon [(Richa et al., 2014); (Xie et al., 2014)], and some companies have implemented the first examples of dedicated recovery plants (especially for batteries) (Tytgat, 2013). However, as in other industrial fields, recovery targets are still very limited, and international regulations have not yet started to regulate them (Hiratsuka et al., 2014). Considering this additional trend, data estimated in this work could be lower than the real ones. Without any doubt, this market sector could become an interesting business for many companies involved in closed-loop supply chains.

Future research trends could take into account several perspectives related to the combined management of different waste streams. First, an assessment of the technological requirements for the real implementation of the ideas presented within this paper is mandatory. The theoretical purity levels assumed within the previous calculations must be assessed in practice for a correct quantification of profits. Depending on the achieved purity level, there could be important increases or decreases in profits, based on the volumes taken into account. Hence, technological requirements have a direct impact in terms of operational costs and, therefore, on final profits. Second, the assessment of potential environmental impacts related to this new type of processes has to be quantified and compared to current practices for the definition of new environmental guidelines and performance indexes. Obviously, this phase can only be performed once the technological requirements are completely consolidated. However, thanks to both the known high content of critical and precious metals in waste PCBs and the sustainability level reached by some innovative recovery technologies (e.g., hydrometallurgy and biometallurgy), the potential environmental impacts related to these new recycling processes is expected to be very positive. Third, the literature should better support the ELV recycling chain (at least, as already performed for WEEE) through the definition of potential solutions of the current practical issues. For example, a good improvement in process engineering could be achieved through the implementation of dedicated simulation models and decision-support tools able to consider the perspective of different actors involved in the automotive recycling chain (e.g., automakers, dismantlers, and shredders), and to assess not only the environmental impacts but also the economic and social ones. This could favour the definition of innovative reverse logistics chains and business models for the recovery and reuse of sub-components and materials from EoL management processes. Subsequently, best practices and innovative businesses could also arise in this industrial context. However, the scarcity of ELVs (and, therefore, of materials) to be recovered in several European nations due to both illegal shipments and the increased usage of non-metals inside cars requires a change in the mentality of the automotive recycling chain. This change must go towards a quality-based recycling approach focused on the reuse and recovery of the most profitable components and materials.

5. Future perspectives

All of the information presented within this paper demonstrate that a combined management of waste PCBs from different waste sources is possible and it could become a relevant business for companies that are already involved in reverse logistics chains and newcomers. Potentially high gatherable profits and treatable volumes could support a series of positive aspects from a business perspective:

- The interest of OEMs in this new business area and therefore a better integration of design for reuse, remanufacture, and recycling (DF3R) strategies within their new product development (NPD) processes could favour a positive increase in future products sustainability levels. Indirectly, this integration could also improve the recovery performances of current treatment technologies by shifting their focus to quality-based approaches.
- The increase in potential profits could limit technological investment efforts required for both current and new industrial actors by favouring the advent of new plants and the rise of dedicated small and medium enterprises (SMEs) within Europe, potentially based on new types of business models and EoL management strategies.
- The aggregation of different reverse logistics chains could favour the transfer of good practices and management structures to other contexts, widening the waste sources taken into account and counterbalancing the treatment of non-profitable cores through the exploitation of flexibility and economies of scale from the management of a different mix of wastes.

Furthermore, from a governmental perspective, the integrated management of similar wastes could allow a set of additional improvements:

- The combined management of wastes could allow the recovery of greater amounts of materials that currently end up in landfills.

- The regulation simplification and integration, although continuing to distinguish between the two waste streams, could allow a uniform recovery of similar components, reducing the unsustainable effects of current practices and therefore improving the overall sustainability of EoL management processes and reverse logistics chains.

The implementation of circular economies within several markets is a topic that is debated by experts worldwide. One way to create these economies, from the personal judgement of the authors, is represented by a combined management of similar waste streams and/or sub-components embedded in wasted products. Printed circuit boards from WEEE and ELVs are a good example toward this direction. Only through a description of practical implications and a quantification of potential benefits coming from these innovative business strategies there could be the chance to support the change of current mental models of private companies and modify the decision process of both national governments and international organizations, by improving the overall sustainability level of the global industrial context.

6. Conclusions

The current evolution towards an even more connected and digitized world is changing people’s lifestyle. Each year, many products become obsolete at an even faster rate, increasing the already high quantity of wastes waiting for a sustainable treatment. Because of the role of electronics within these flows, a significant part of these wastes is constituted by PCBs. Unfortunately, both the current recovery methods and regulations do not always make their correct recovery possible. Without any doubt, something must change if we want to produce advanced technological products also in the near future. One method could be the combined management of waste PCBs from different waste streams, such as WEEE and ELVs. The paper assesses as the current management of ELVs and WEEE presents many differences (e.g., discussions in literature, operational processes, strategies, and environmental issues) both related to sector’s features and dedicated regulations. However, there are also similarities partly ignored (or not adequately assessed) neither by scientific experts nor by companies. The comparison of these two waste streams demonstrates as some of these differences are both unmotivated and unsustainable, hiding some interesting business opportunities to companies. The waste PCB management issue is an interesting example from this point of view. PCBs coming from WEEE are recovered in a proper way by dedicated plants, even if current materials recovery rates do not allow to stop (or, at least, limit) the extraction of virgin resources – especially true in terms of precious and critical materials. This is partly due to the low technological development characterizing the overall materials recovery chain. However, also the type of the international regulations introduced during the last decades pushed companies to behave in a certain way. The adoption of weighted-based approaches resulted in the only improvement of basic materials recovery (e.g., steel, aluminium, plastics, glass, and wood), available in high quantities within cars or electrical and electronic products. The consequence was the very low (or inexistent) recovery of valuable and critical materials, because of their infinitesimal presence within some specific components. In contrast to what happens for WEEE PCBs, PCBs from ELVs are not regulated at all, continuing to be improperly landfilled. Considering that, especially in the last decade, vehicles started to embed an increasing number of electronic devices (usually valuable ones) and these devices are similar to the ones embedded in WEEE, a different treatment appears to be rather inefficient. Starting from this point, the paper demonstrates the presence of potential chances to improve the sustainability of waste PCB management processes, whatever their source. These potential improvements are also quantified, in terms of both volume and profit. The obtained results reported in the last part of the work make a good picture of the current situation, even if only at European level. Starting from the official estimates about ELV and WEEE generated volumes, from 2015 to 2030 waste PCBs are expected to go from 204 ktons to 312 ktons (ELV PCBs account for 8% of volumes, on average). If these volumes could be managed in a correct way, the management of waste

PCBs could offer interesting profits to companies. Considering the official PCB materials characterizations and weights (and related materials market prices) these profits are estimated within the paper and quantified from 3.3 billion € in 2015 to 4.9 billion € in 2030 at minimum and from 13.2 billion € in 2015 to 18.1 billion € in 2030 at maximum. ELV PCBs could have a relevant role on these expected values, accounting for 23% - 27% at minimum and for 59% - 64% at maximum on the overall profits. These numbers, even if only theoretical and reachable through optimized processes, can offer a good idea of what could happen in the waste PCB recovery chain only through the exploitation of a current regulatory lack. The reported percentages demonstrate the potential role played by ELV PCBs. Even if they could account for the only 8% in terms of volume, ELV PCBs show interesting overall profits (accounting for 25% - 62% of the total, on average). Paradoxically, this means that the type of PCBs currently not regulated by the governments and rarely recovered by the industrial actors could represent the main source of value for the entire PCB recovery chain in the next future. Data referred to 2015 can also be considered as a good proxy of the current losses caused by an unsustainable recovery of waste PCBs.

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