Doctoral Research in Economics and Management of Technology (DREAMT) – XXX Cycle

Doctoral Dissertation

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Three essays on the identification of innovative operation strategies for the sustainable recovery of automotive electronic wastes

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Invited:

To my family

Abstract

The thesis contributes to the area of sustainable management of wasted automotive components with a twofold target: the first one deals with the quantification of potential profits coming from the recovery of materials from automotive electronic wastes. The second one deals with the identification of innovative operation strategies related to the sustainable recovery of these elements and the measurement of their effect on current economic performances of the Italian End of Life Vehicle (ELV) recovery chain through a System Dynamics (SD) model.

The management of obsolete cars has become a relevant issue during the last decades. Given the fast evolution of vehicles towards lighter, safer, more connected, electrified and auto-guided mobility concepts, automotive wastes have become even more difficult to manage for current actors involved in the official ELV recovery chain. Many regulations and directives were activated worldwide. However, current recovery technologies are still maintaining features and performances like in the sixties. This way, innovation trends representative of the automotive sector (e.g. construction logics, embedded components and materials) cannot be managed correctly by dismantlers and shredders. Lots of components and materials that could be either reused, remanufactured or recycled are still lost either in landfills or in incinerators. Trying to limit this enormous depletion of natural resources and make the actors involved in the ELV recovery chain aware of these issues, the current thesis wants to identify innovative operation strategies related to the sustainable recovery of automotive electronic equipments, one of the most valuable elements embedded in modern cars.

Keywords: end of life vehicles, automotive electronics, decision-support tool, system dynamics.

General acknowledgements

I have to thank lot of person for this thesis.

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Extended abstract

Introduction

The PhD thesis is the synthesis of a series of research activities done during the three years of study by the candidate, following a first research proposal designed in November 2015 (concluding the first year of PhD) and modified during the following two years. It is relevant to underline as this PhD was jointly directed by both the University of Bergamo and the University of Pavia. This “boundary condition” is evident also within the PhD thesis, resulting as a joined set of activities: a set of research activities in line with Business Engineering orientations (e.g. business contexts analysis, modelling and simulation of production systems) and a set of research activities in line with Economics orientations (investment analysis). This double view allowed the candidate to develop an original output, matching together Business Engineering and Economic topics and issues. Another relevant element to underline is that this PhD thesis takes the form of a collection of already published (and independent) papers. This way, the reader could find either some overlapping sections or difficulties in following the logical path of the dissertation.

Research motivation

The PhD candidate started its academic career in 2010 after its Master of Science (MsC) degree, working at Politecnico di Milano. During this period, the PhD candidate was involved in the Manufacturing Group of Prof. Taisch and Prof. Garetti, particularly in the research areas of Product Lifecycle Management (PLM) and Industrial Maintenance. In 2013, the candidate left Politecnico di Milano and started a new work at ITIA-CNR. During this period, the candidate was involved in many research activities related with Sustainable Business Models in several industries. Considering this last point, the candidate started in studying the End-of-Life management of mass electronic products from a practical view, given the presence of a dedicated pilot plant in ITIA-CNR. In 2014, the candidate left ITIA-CNR and came back to Politecnico di Milano. Considering the previous experience in ITIA-CNR, the candidate continued in studying the End-of-Life management of products, but from a theoretical view. Trying to specialize into a dedicated research area, the candidate decided to focus on a particular kind of wasted electronic products, or the ones coming from the automotive sector. Given the chance of participating in a PhD course at University of Bergamo and Pavia, in September 2014 the macro research context delegated to the PhD student has been identified in the area of PELM (Product End-of-Life Management) of wasted automotive electronics. In 2016, the candidate ended its experience at Politecnico di Milano and continued his research at University of Bergamo – CELS research group, where he is still working today.

Research activities definition

The research activity allocated to the candidate at the beginning of his PhD course consisted in comprehending how an augment of electronic components into cars could affect the End-of-Life Vehicle (ELV) recovery chain performances - particularly in economic terms – when cars would have reached the end of their useful life. Given this main aim, the candidate was asked to start in evaluating the current state of the scientific literature considering electronic components, especially the - so called – Printed Circuit Boards (PCBs), by pointing out any kind of gap. During the three years of PhD, many research activities and sub-activities were done, going to deeply assess those areas only preliminarily defined through the state of the art analysis. Considering a temporal and logical path, it is possible to synthesize those activities as it follows:

- Structured state of the art analysis of ELV, Waste from Electrical and Electronic Equipment (WEEE) and wasted PCB management practices. Trying to consider the widest sample of information about the management of wasted electronic components, several research activities (one for each of the previous three perspectives) were conducted, by exploiting different methods that interested (i) the study of wasted automotive electronic components, (ii) the study of WEEE management practices and (iii) the study of wasted PCBs. From this initial activity (reported within Sections 2 and 3), a list of theoretical gaps and a set of research questions of different nature (e.g. strategic, tactic, operative and technologic) were identified. Considering the research areas taken into account by the DREAMT PhD course, only some of these research questions were pointed out as relevant topics and developed during the following years. These topics were related to the quantification of the economic benefits and the identification of innovative operation strategies related to the recovery of automotive electronic wastes, by measuring their effects on current economic performances of ELV recovery chains.
- Field interviews of relevant actors involved in the Italian ELV recovery chain. Considering the very few amount of scientific materials speaking about automotive electronic wastes and trying to add some more information on the available one, the subsequent step was the assessment in person of the current state of the Italian ELV recovery chain. Six months were dedicated on this activity. During these months an overall amount of more than 50 companies (among car manufacturers, dismantlers, shredders, foundries, metal traders and industrial associations) were interviewed both in person and by phone. All the recovered data were of utmost importance in a twofold manner. Firstly, these data allowed me in comprehending the real state of the industrial context, by defining a list of real lacks affecting the sector. Secondly, these data were also indispensable in the calculation process.
- Data analysis and simulation model selection. Once assessed both theoretical and real lacks, the subsequent step was the analysis of all the gathered data and the definition of the technical requirements for the selection of the simulation model to be adopted within my thesis to check and quantify the effect of automotive electronics recycling activities on the overall ELV recovery chain economic performances. After a first assessment of (i) the available simulation models present in literature, (ii) their pros and cons, (iii) the key features representing the industrial context taken into account by this thesis and (iv) the simulation models adopted by the experts to study the same industrial context the decision was to consider System Dynamics (SD) as the reference method for modelling and simulating the whole ELV recovery chain.
- SD model implementation and scenarios comparison. After having selected System Dynamics as reference method, the subsequent step was the development of the SD model in a virtual environment. Starting from scratches reported in a MsC thesis of Massachusetts Institute of Technology (MIT) twenty years ago, more than six months of my PhD were dedicated to the development of the model on Vensim®, a professional software dedicated to the development and simulation of SD models. Given the high complexity of the industrial context taken into account, the SD model was splitted into several interrelated views, each of them dedicated to a particular feature and/or actor involved in the ELV recovery

chain. A particular interest was given to the economic views of the model, mapping the decisional process of two main actors of the ELV recovery chain, like dismantlers and shredders. Once all the simulation errors were identified and corrected, the obtained SD model was tested intensively, by comparing both realistic and potential scenarios. Simulation cycles were conducted by modifying input variables of the SD model and seeing the final effect on dismantlers and shredders economic performances. The range of changes of input variables were identified with the support of industrial partners. From this activity many graphs were obtained and the most relevant ones are reported within this thesis.

- SD model validation. The last activity taken into account during my PhD was the final validation of the SD model. To this aim, a group of industrial experts coming from the most relevant Italian companies involved in the ELV recovery chain were interviewed, asking for their comments on the results coming from the SD model. Once assessed some misaligned data, their comments were all positive. The SD model was judged by them as realistic and potentially useful to compare different industrial strategies in the next future. Considering the identification of the best operation strategy to manage automotive electronic wastes, they selected as more interesting (and realistic) scenario the one seeing shredders as the main responsible of the automotive electronic wastes recovery process. However, if additional studies will be done in terms of technologies and operative procedures, also the other scenario seeing dismantlers as the main responsible of the recovery process could be achievable.

Research field and objectives

Nowadays, within all the industrial contexts, companies are facing with an increasing role of sustainability both in products and processes. Sustainability is becoming a “must have” for any kind of company willing to improve its market image. However, this need for sustainable products and processes comes from an even more awareness of people (especially in Europe) about climate change issues, natural resources depletion, flora and fauna extinctions. The worldwide community started some decades ago (the most famous measure was the Kyoto protocol of 1997) in trying to modify the current way of doing of companies and the overall society by activating a series of even more restrictive environmental measures in terms of Greenhouse Gas (GHG) emissions reduction, hazardous materials bans, percentages of materials to be reused, recovered, recycled, tracking of illegal transboundary shipments of wastes and so on. However, even if relevant results were reached worldwide and tons of basic resources are currently recovered every day, not so good performances are still available for Critical Raw Materials (CRMs) or those materials (about 40 among metals, non-metals and other elements) re-known by the European Union as the most critical in terms of supply risks, technological relevance and natural availability. Within this industrial context the sustainable management of any kind of material embedded into modern products is becoming a key issue. Many experts gave their contribution in proposing innovative technologies and processes for the recovery of specific materials from ELVs, WEEE and PCBs. However, the actors involved in current waste recovery chains are still adopting technologies that were implemented more than fifty years ago. In general, there is a convincement about the non-profitability coming from the adoption of modern plants. This point, together with the high variability of materials market prices and the complete absence of any kind of financial support from national governments, are stopping any kind of evolution of the industrial context, by involuntarily favouring unsustainable behaviours. Obviously, this evolution is not free of charge (asking for relevant investments in new technologies), requires a big re-organization of the business and of the entire value chain. Again, there are also several risks to consider, both exogenous and endogenous to the industrial context. Exogenous risks are related to the high variability of materials market prices, especially precious metals ones, influencing the economic performances of companies and making them most difficult to control and forecast. Endogenous risks are related to current behaviours of companies, adopted technologies and convincements. Within this industrial context this thesis wants to consider two main objectives:

- Quantify potential economic benefits coming from the recovery of automotive electronic wastes. (Section 4).
- Identify innovative operation strategies related to wasted automotive electronics and measure their effect on current ELV recovery chains economic performances (Sections 5).

In this sense the PhD thesis lies within an innovative scenario, by coping with current ELV recovery chain's needs and demonstrating potential benefits coming from different – and more sustainable – behaviours.

Methodologies and results

The achievement of different research activities and sub-activities was reached through the adoption of several research methodologies, in particular:

- The research boundaries and theoretical gaps were assessed through a deep structured literature review implemented for each of the most relevant sources of electronic waste (e-waste) (Sections 2 and 3).
- The quantification of potential profits was implemented by the adoption of a mathematical model completely based on the most common assessment principles for industrial investments (Section 4).
- The identification of innovative operation strategies and the measurement of their effect on current ELV recovery chain economic performances were implemented through a SD model. Starting from an already existing SD model it was updated with Italian data and validated through a simulation of a realistic representation of the Italian ELV recovery chain (Section 5).

Trying to summarize, it is possible to assert what described below in terms of results and reached objectives:

- The Italian ELV recovery chain, like any other ELV recovery chain, is completely depending from materials market prices, especially from ferrous scrap prices. This way, any evolution in materials constituting a modern car different from ferrous metals could drastically reduce the profitability of the entire sector, by exposing companies to financial defaults. To this aim, it is unavoidable that more sustainable and diversified practices and technologies have to be adopted by the actors.
- Together, the mathematical and the SD model can offer a good chance to evaluate the realistic impact coming from the adoption of this new way of doing. Together, these two models offer to companies an innovative perspective on potential benefits coming from the real adoption of circular economy principles. The validation obtained from a first virtual application of the SD model to industrial cases demonstrate the high relevance at industrial lev-

el. In addition, results suggest also a comparison with current practices, by leaving space to future developments.

- The obtained results could be of interest for several kinds of actors involved in any ELV recovery chain, being them industrial or political ones. From the industrial side, results coming from the two models could be useful for both car manufacturers, car dismantlers and car shredders. Manufacturers could assess in real time the effect of a change in materials composition of their cars on the entire ELV recovery chain, by identifying most feasible alternatives. Dismantlers and shredders, instead, could do the same, but from a more restricted perspective basing on their role within the ELV recovery chain. From the political side, results could be of interest for both national/international environmental associations, national ministries of environment and supra-national entities. Like what happened in Germany and Switzerland – inspiring from the results coming from this thesis – environmental associations could push national governments in modifying their regulations according to new kinds of electronic wastes to take into account in future revisions of environmental laws. This change, if spread in many nations, could also influence a change in supra-national directives, like the European WEEE and ELV Directives in force. Once directives will oblige companies in behaving in a more sustainable way (i.e. recovering electronic products from wasted cars), a real improvement in the sustainability level of the entire European economy could happen and new business opportunities for companies could rise in the next future.

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1. Introduction

1.1. General introduction of the context

Within the current competitive world, enterprises are increasingly stressed and subjected to precise market requests. Customers are asking for even more sustainable products and processes. Products must be not only cheap and available at the right time and into the right place, but also green for having some chance to reach the success in their market. In order to maintain (or gain) competitive advantages, modern enterprises have to manage two sides of sustainability:

- They must improve the internal sustainability, reducing all the unsustainable practices.
- They must improve the external sustainability, asking for greener practices also to other actors constituting their value chain.

According to these needs, enterprises have to focus on different ways of managing their businesses, even defending their core competences. Looking to this research, the management of wasted products is becoming a relevant issue for companies and one of the main sources of green materials able to improve their overall sustainability. Within the globally scaled scenario, circular economy principles took place in the last decades (see **Figure 1**). In general terms, circular economy follows natural laws through which any element produced by a biological process can easily re-enter within it and become the starting point for new processes. The same philosophy should be followed also by human production processes, limiting as much as possible the generation of hazardous/unusable wastes. These principles pushed traditional linear consumption patterns against constraints on the availability of resources and the rising demand from the world's growing population. This unsustainable overuse of resources is influencing even more material prices, by adding uncontrollable volatility in several markets (Ellen MacArthur, 2013).

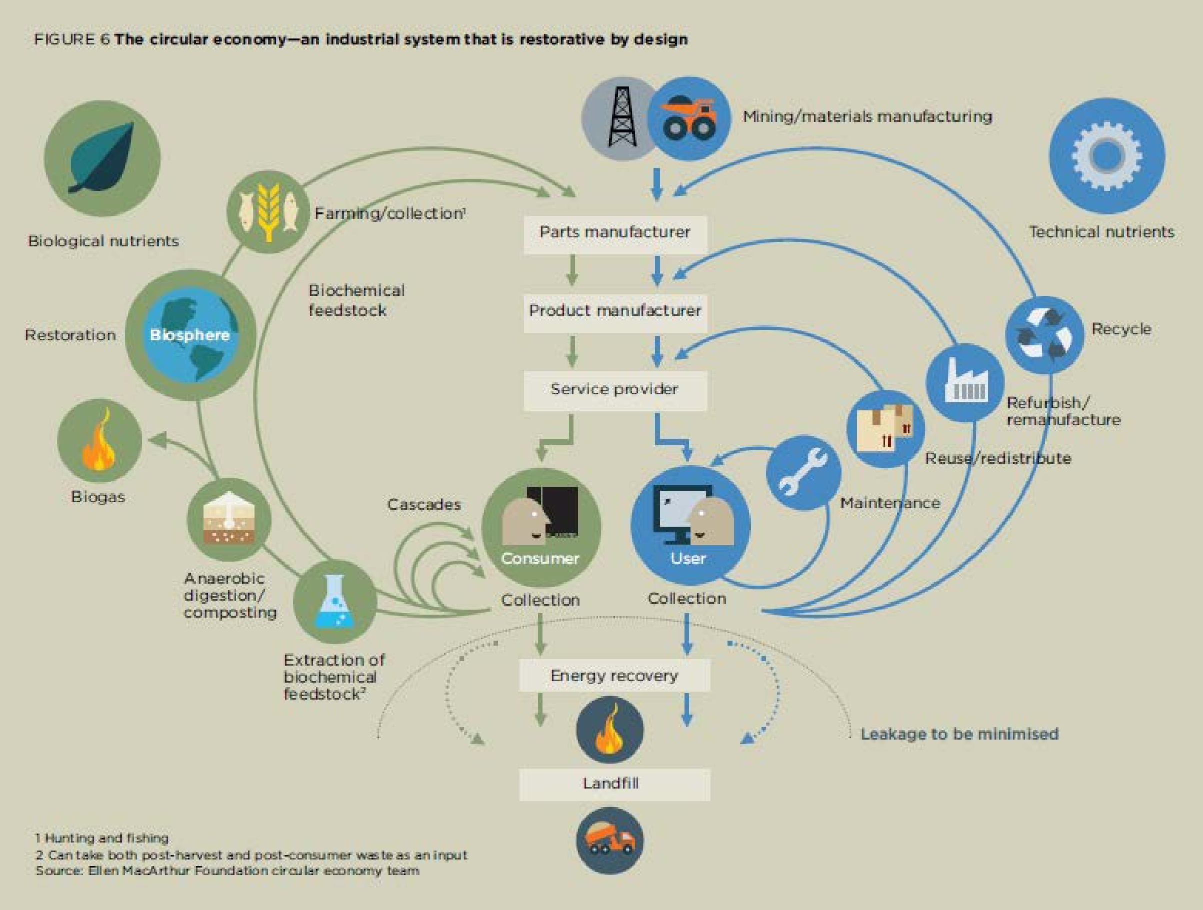

Figure 1. A systematic view of circular economy - Source: (Ellen MacArthur, 2013)

All the most advanced economies started in speaking about the importance of being circular, not only in environmental but also in economic terms. Consequently, companies are even more stressed by both requests from customers about greener products and processes and constrained by even more severe regulations supporting circularity principles. This way, the product end of life management is becoming an unavoidable key aspect. In particular, the management of the two main sources (WEEE and ELVs) of obsolete electronic products is gaining even more relevance. The objective of this thesis is to take into account one of these waste streams and demonstrating potential benefits coming from its sustainable management.

1.2. Main objectives of the work

Within the presented context, the candidate has been asked to formulate his research proposal, taking into account the expertise's of the two Universities jointly managing the PhD Course, University of Bergamo and University of Pavia. The macro research context delegated to the PhD student has been defined in September 2014. Taking advantage from a previous experience in ITIA-CNR, the macro research has been identified in the area of PELM (Product End-of-Life Management) with a particular focus on wasted automotive electronics. The thesis has a twofold target: the first deals with the quantification of potential economic benefits coming from the recovery of automotive electronic wastes while, in a complementary way, the second deals with the identification of innovative operation strategies related to the recovery of automotive electronic wastes and the measurement of their effects on current ELV recovery chain's economic performances. The two targets are interconnected, as it will be demonstrated in the thesis, even if they clearly show two different points of view.

Quantification of potential profits from the recovery of automotive wastes

The first most important issue of this thesis is the quantification of potential profits related to the recovery of automotive electronic wastes, just to offer companies a realistic idea of how much should be the effort required to upgrade their business in a more sustainable perspective. To do that, the current thesis formalizes a reference mathematical model that, starting from European data about ELV volumes and weights, materials market prices and materials characterization data of automotive electronic components is able to quantify the economic benefit offered by this kind of activities. In addition, the cost structure was taken from the literature, by considering a typical electronic recovery process and two optimized capacities, one for mobile plants and one for field ones (see Section 4 for details). In such a context, the adopted research methodology is composed by four main activities:

- Analysis of the current state of the recovery of automotive electronic wastes, provided by the analysis of the most updated scientific and industrial papers on this topic.
- Definition of the main requirements of the mathematical model, realized taking into account inputs coming from the literature analysis and field interviews (partly derived from the first results of the thesis).
- Definition of the reference mathematical model, formalized following standard investment assessment methods (e.g. NPV, IRR and PBT).
- Application of the mathematical model. The mathematical model has been applied for the evaluation of potential European profits coming from the recovery of automotive electronic wastes.

Identification of innovative operation strategies for the recovery of automotive electronic wastes and measurement of their effect on current ELV recovery chain economic performances

After having quantified the potential profits related to the recovery of automotive electronic wastes, the second most important issue of this thesis is the identification of innovative operation strategies related to the recovery of automotive electronic wastes and the measurement of their effect on current ELV recovery chain economic performances. The main answer to this question is the development of a SD model able to compare different behaviours of actors involved in the current ELV recovery chain and allowing to make some decisions on future operation strategies to implement. In fact, looking at the international literature on this specific topic, there are very few examples of simulation tools dedicated to the ELV management. Furthermore, none of the existing models consider automotive electronic wastes. The decision support tool for the ELV recovery chain is the second result provided in the research thesis. In such a context, the adopted research methodology is composed by six main activities:

- Analysis of the current state of the art about the adoption of SD models for the assessment of End-of-Life strategies in different industrial contexts, provided by the analysis of the most updated scientific and industrial papers on this topic.

- Assessment of the current state of the Italian ELV recovery chain, by interviewing more than 50 Italian actors among industrial companies and associations.
- Calculation of all the main constants at the base of the SD model functioning, realized by taking into account not only data coming from interviews, but also literature data and information gathered from websites.
- Development of the SD model in a virtual environment – starting from schemes reported in the original MIT thesis – and refinement of any kind of simulation error.
- Application of the SD model to the Italian ELV recovery chain. Once added data about both Italian ELV recovery chain features and scrap automotive electronics, the SD model was tested intensively and several scenarios were compared in terms of different economic performances.
- Validation of results coming from the SD model. A focus group of industrial experts, selected among the most relevant actors involved in the Italian ELV recovery chain, was involved for the final validation of results.

1.3. Description of the contents

The following two sections (Section 2 and 3) present the general research context of this thesis. More in detail, Section 2 is focused on the automotive electronics sector, going into detail about its main features and the current management of e-wastes from cars. Section 3, instead, has a wider perspective on e-wastes, going to compare the main sources of e-wastes basing on common and uncommon characteristics. These two sections allow to comprehend the current situation in terms of the management of automotive electronics and the existing differences among several kinds of electronic wastes. The literature gaps evidenced by these state of the art analyses and comparison were at the base of the entire research activity and allowed me to focus on specific issues affecting the selected research context, or the lack of information about the profitability of automotive electronics recovery processes and the absence of decision-support tools supporting the ELV recovery chain during its ordinary work. From this point, the research activity continued by following a multi-layered methodology. Section 4 takes into account the first of the two identified lacks. It is dedicated completely to the quantification of potential profits coming from the management of automotive electronic wastes. Starting from a general description of their recovery process, automotive electronic wastes are classified basing on their material contents. The matching of (i) operational costs related to the recovery process, (ii) the market value and purity levels of recovered materials and (iii) ELV generated volumes expected trends in the period 2015-2030 allowed the quantification of potential profits coming from the recovery of automotive electronic wastes in the same period. The quantified costs and revenues were of utmost importance for the final assessment of the ELV recovery chain performances through simulation. Section 5 considers the second of the previously identified lacks. It is completely dedicated to the identification of innovative operation strategies through the SD model adopted within this thesis, taken into account as a mean for the overall simulation of the Italian ELV recovery chain. Starting from an intensive period of person interviews of the main actors involved in the Italian ELV recovery chain (more than 50 companies were interviewed in six months), it was possible to identify the real lacks affecting the industrial context and compare them with the ones identified from the literature. Again, direct interviews allowed also to gather many information at the base of the SD model functioning and to define its main requirements. These data were of utmost importance to update the original SD model. The lacking data that neither literature nor industrial actors were not able to identify were maintained the same like in the original SD model. Once assessed all the needed information, the SD model simulation started and several scenarios were compared and assessed, both under historical and equilibrium conditions. Finally, results coming from simulations were validated by a selected group of experts pertaining to relevant industrial companies of the Italian ELV recovery chain.

According to what previously described, the thesis is structured as follows (see **Figure 2**):
- Section 1 (the present chapter) introduces the research questions and methodologies.
- Section 2 illustrates a generic literature review about the automotive electronics context.
- Section 3 compares the current management of wasted automotive electronic components with the other main sources of e-waste under different perspectives, showing current similarities and inequalities.
- Section 4 describes the mathematical model at the base of the quantification of potential profits coming from the sustainable management of wasted automotive electronics, giving some important information about volumes, trends and profits.
- Section 5 presents the SD model adopted in this study. After a general explanation of the procedures implemented to gather all the needed information for enabling the full potential of the SD model, several scenarios were assessed and their performances were compared virtually.
- Section 6 concludes the thesis, summarizes results and defines future research streams.

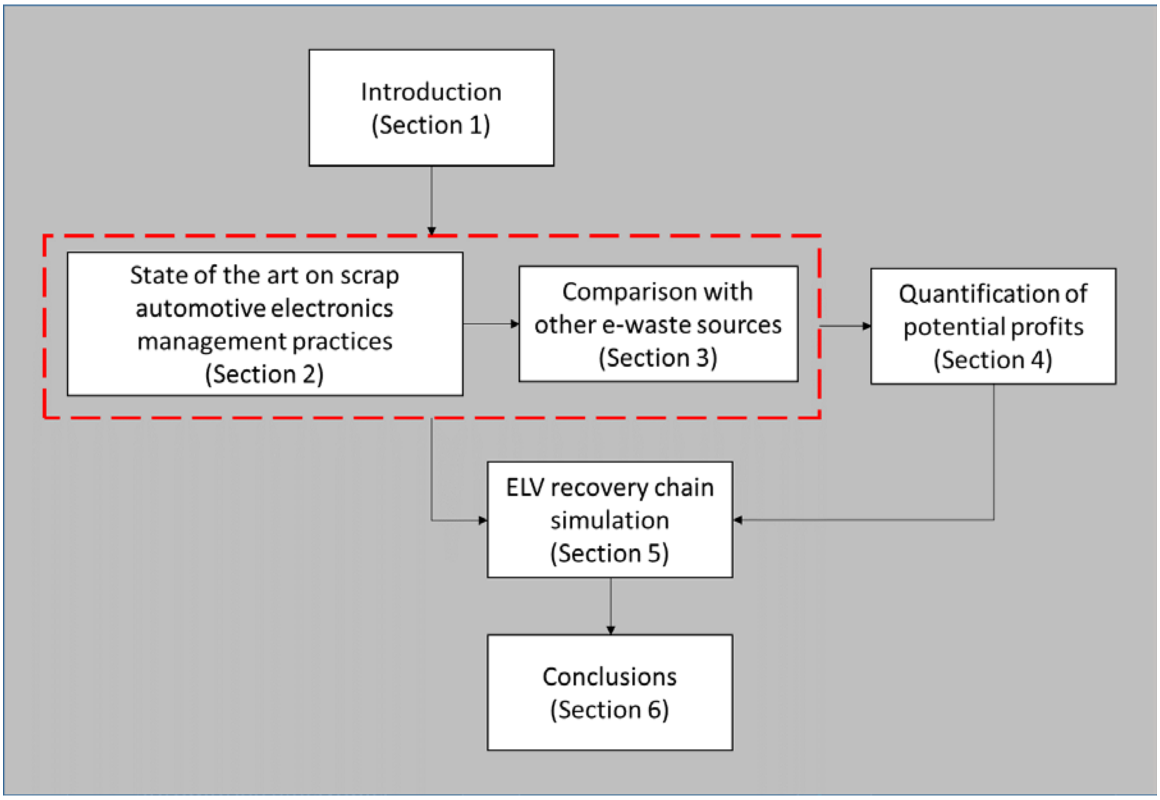

Figure 2. Overall structure of the thesis

1.4. Acknowledgements of co-authors

The present thesis is based on three papers I wrote together with these authors: Federica Cucchiella, Idiano D'Adamo and Sergio Terzi.

First of all, I have to thank Sergio Terzi, who supported me during the writing of this thesis and all of the papers constituting it.

Secondly, I have to thank Idiano D'Adamo, who shared with me the experiences at ITIA-CNR and at Politecnico di Milano, supported me during my PhD and wrote (and continues to write) with me lots of valuable papers in international journals.

Finally, I have to thank Federica Cucchiella, who supported me and Idiano during the writing of several papers in international journals.

1.5. My Contribution in each paper constituting this thesis

Paper #1: P. Rosa, S. Terzi (2016) "Comparison of current practices for a combined management of printed circuit boards from different waste streams" Journal of Cleaner Production, Vol. 137, pp. 300-312. **Contribution:** (i) gathering of information from several sources (e.g. scientific community, industrial context, supranational organizations), (ii) definition of the main dimensions of comparison, (iii) classification of works according to these dimensions and (iv) discussion of the main findings.

Paper #2: F. Cucchiella, I. D'Adamo, P. Rosa, S. Terzi (2016) "Automotive printed circuit boards recycling: an economic analysis" Journal of Cleaner Production, Vol. 121, pp. 130-141. **Contribution:** (i) general literature review of the ELV recovery chain, (ii) gathering of data from shared industrial database and (iii) clustering and refining of data for the subsequent economic assessment.

Paper #3: P. Rosa, S. Terzi (2017) "Improving end of life vehicle's management practices: An economic assessment through system dynamics" Journal of Cleaner Production, under review. **Contribution:** (i) general literature review of papers adopting System Dynamics (SD) as research methodology, (ii) interview of industrial actors involved in the Italian ELV recovery chain, (iii) gathering data for the SD model from several sources (e.g. scientific papers, industrial reports, supranational analyses) (iv) replication of the SD model in a virtual environment, (v) updating of the SD model with Italian data, (vi) simulation of the virtual Italian ELV recovery chain and (vii) analysis of different scenarios.

PhD candidate references

International journals

1. F. Cucchiella, I. D'Adamo, M. Gastaldi, P. Rosa, S. Terzi "A joined management of different e-waste sub-streams: a profitability assessment of twenty-four wasted printed circuit boards" Waste Management, under review.
2. A.K. Awasthi, F. Cucchiella, I. D'Adamo, J. Li, P. Rosa, S. Terzi, X. Zeng "Modelling the correlations of e-waste quantity with economic increase" Science of the Total Environment, accepted for publication.
3. P. Rosa, S. Terzi "Improving end of life vehicle's management practices: An economic assessment through system dynamics" Journal of Cleaner Production, under review.
4. I. D'Adamo, M. Miliacca, P. Rosa "Economic Feasibility for Recycling of Waste Crystalline Silicon Photo-voltaic Modules" Hindawi International Journal of Photoenergy, Vol. 2017.
5. F. Cucchiella, I. D'Adamo, M. Gastaldi, S.C.L. Koh, P. Rosa "A comparison of environmental and energetic performance of European countries: A sustainability index" Renewable and Sustainable Energy Reviews, Vol. 78, pp. 401-413. October 2017
6. P. Rosa, S. Terzi "Comparison of current practices for a combined management of printed circuit boards from different waste streams" Journal of Cleaner Production, Vol. 137, pp. 300-312. November 2016
7. I. D'Adamo, P. Rosa "Remanufacturing in industry: advices from the field" International Journal of Advanced Manufacturing Technologies, Vol. 86 (9-12), pp. 2575-2584. October 2016
8. F. Cucchiella, I. D'Adamo, S.C.L. Koh, P. Rosa "A profitability assessment of European recycling processes treating printed circuit boards from waste electrical and electronic equipments" Renewable and Sustainable Energy Reviews, Vol. 64, pp. 749-760. October 2016
9. I. D'Adamo, P. Rosa "Current state of renewable energies performances in the European Union: A new reference framework" Energy Conversion and Management, Vol. 121, pp. 84-92. August 2016
10. F. Cucchiella, I. D'Adamo, P. Rosa "Urban waste to energy (WTE) plants: A social analysis" JP Journal of Heat and Mass Transfer, Vol. 13(3), pp. 421-444. August 2016
11. I. D'Adamo, P. Rosa, S. Terzi "Challenges in waste electrical and electronic equipment management: A profitability assessment in three European countries" Sustainability (Switzerland), Vol. 8(7), N.633. July 2016
12. F. Cucchiella, I. D'Adamo, P. Rosa, S. Terzi "Automotive printed circuit boards recycling: an economic analysis" Journal of Cleaner Production, Vol. 121, pp. 130-141. May 2016
13. F. Cucchiella, I. D'Adamo, P. Rosa, S. Terzi "Scrap automotive electronics: A mini review of current management practices" Waste Management & Research, Vol. 34(1), pp. 3-10. January 2016
14. F. Cucchiella, I. D'Adamo, P. Rosa "Industrial Photovoltaic Systems: An Economic Analysis in Non-Subsidized Electricity Markets" Energies, Vol. 8(11), pp. 12865-12880. November 2015
15. F. Cucchiella, I. D'Adamo, S.C.L. Koh, P. Rosa "Recycling of WEEEs: an economic assessment of current and future e-waste streams" Renewable and Sustainable Energy Reviews, Vol. 51, pp. 263-272. June 2015
16. G. Copani, P. Rosa "DEMAT: Sustainability Assessment of New Flexibility-oriented Business Models in the Machine Tools Industry". International Journal of Computer Integrated Manufacturing, Vol. 28(4), pp. 408-417. April 2015
17. F. Cucchiella, I. D'Adamo, P. Rosa "End-of-Life of used Photovoltaic Modules: a financial analysis" Renewable and Sustainable Energy Reviews, Vol. 47, pp. 552-561. July 2015
18. P. Rosa, S. Terzi "Proposal of a Global Product Development Strategy Assessment Tool and its Application in the Italian Industrial Context". International Journal of Product Lifecycle Management, Vol. 7(4), pp. 266-291. December 2014

International conferences

1. R. Luglietti, P. Rosa, A. Pastore, S. Terzi, M. Taisch "Life Cycle Assessment Tool Implemented in Household Refrigeration Industry: A Magnetic Cooling Prototype Development" Procedia Manufacturing, Vol. 8, pp 231-238. March 2017
2. D. Cerri, R. Luglietti, P. Rosa, S. Terzi "Lifecycle Optimization in the Refrigeration Industry: A literature review". Proceedings of ICE/ITMC 2015 – IEEE International Conference on Engineering, Technology and Innovation/International Technology Management Conference, N. 7438650. IEEE. Belfast – United Kingdom. March 2016
3. P. Rosa, S. Terzi "Waste Electrical and Electronic Equipments versus End of Life Vehicles: a state of the art analysis and quantification of potential profits" Procedia CIRP, Vol. 48, pp. 502-507. Berlin-Germany. May 2016
4. R. Luglietti, P. Rosa, S. Terzi, M. Taisch "Life Cycle Assessment Tool in Product Development: Environmental Requirements in Decision Making Process" Procedia CIRP, Vol. 40, pp. 202-208. Ho-Chi-Minh City – Vietnam. September 2015
5. D. Cerri, R. Luglietti, P. Rosa, S. Terzi "Lifecycle optimization in the refrigeration industry: A Decision-Support Simulation Toolbox (DSST)" Procedia CIRP, Vol. 48, pp. 277-282. Berlin-Germany. May 2016

2. Literature review on scrap automotive electronics management practices

Section 2 wants to describe in a general term the overall research context (see ***Figure 3****), trying to open a discussion on the topics taken into account within this thesis. A mini review of the state of the art on automotive electronic components' recovery practices is presented here. In addition, a set of benefits coming from more sustainable practices are listed and commented by the author. This section reports partially what can be found within the paper titled "Scrap automotive electronics: A mini review of current management practices", edited by Waste Management & Research (see Section 7 for details).*

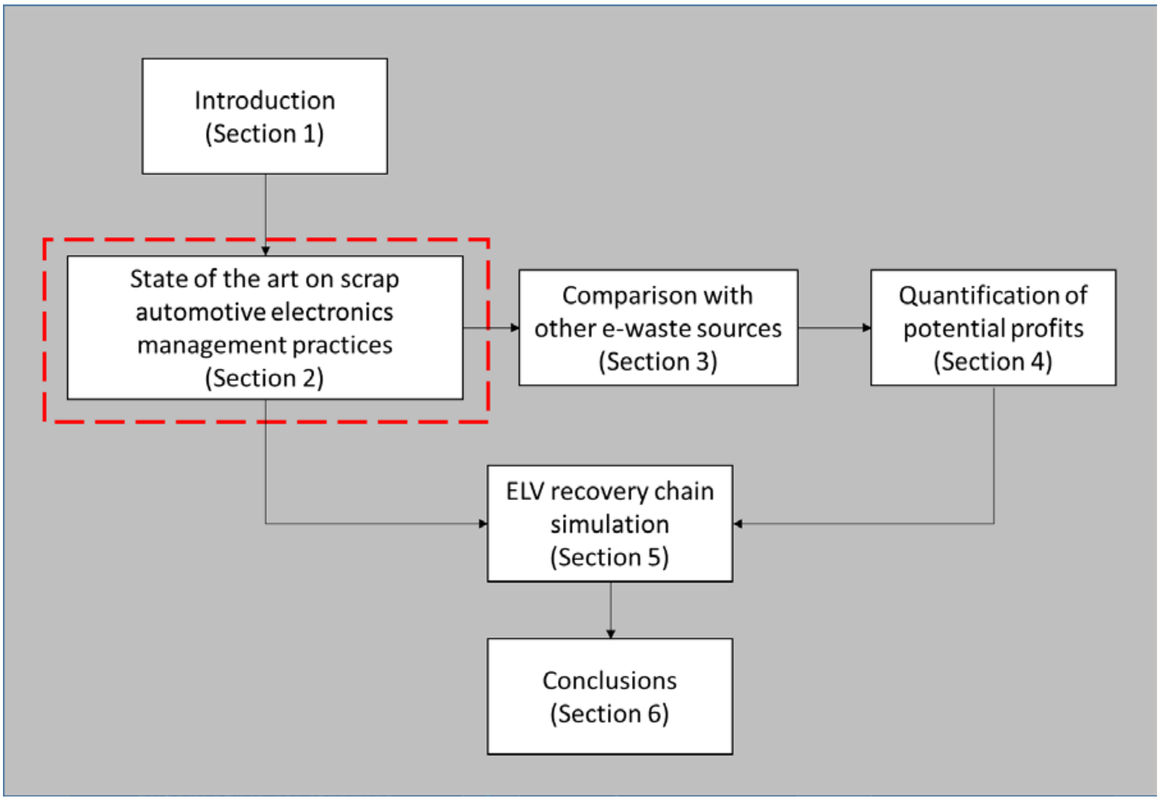

Figure 3. Relation of Section 2 with the overall structure

2.1. Introduction

The automotive sector is one of the most important sources of waste, not only because of precious metals and critical materials embedded into end-of-life vehicles (ELVs) ((Berzi et al., 2013); (Uan et al., 2007); (Reuter et al., 2013)), but also in terms of volumes (in Europe, the total ELVs annual generation is expected to reach 15.4 million tonnes in 2015 and 19.5 million tonnes in 2020) ((Sakai et al., 2014); (Tian and Chen, 2014); (Zorpas and Inglezakis, 2012)). To this aim, basic guidelines for the reuse, recovery and recycling of ELVs were established all over the world in the last decades ((European Union, 2000); (European Union, 2005); (European Union, 2013)). These directives try to regulate and control the management of ELVs for a correct and sustainable dismantling and recovery of secondary resources

((Demirel et al., 2016)). Lots of articles analysed and compared different ELV directives and recovery systems around the world ((Sakai et al., 2014); (Zhao and Chen, 2011)), by enhancing weaknesses and strengths and proposing interesting amendments. However, the recycling of automotive electronics (e.g. electronic control units (ECUs)), together with its environmental impacts, does not appears to have been adequately assessed ((Wang and Chen, 2012); (J. Wang and Chen, 2013a)). The aim of this section is the assessment of existing lacks in EoL management practices of wasted automotive electronic components, trying to define future innovative research streams allowing experts (both politicians and industrials) to better focus their efforts in the sustainable management of valuable wastes, by setting up optimised regulations and more efficient reverse logistics chains.

2.2. A conceptualization of the topic

A product reaches the end of its useful life for a series of technological reasons – as obsolescence or deterioration, and for changes in consumer needs ((Garetti et al., 2012); (Rosa and Terzi, 2014)). Recycling is one of the EoL strategies capable of recovering a great part of materials embedded in these products ((Cucchiella et al., 2014c); (Ijomah et al., 1999)). When recycling is applied to one of the most important sources of waste, as in case of ELVs, it automatically acquires a central role in sustainable development terms at all levels, from manufacturers to suppliers, from local to national and international economies ((Simic and Dimitrijevic, 2012); (Simic, 2015)). The trend followed by car manufacturers in the last decades, trying to reduce fuel consumption and CO_2 emissions, is to lighten vehicles through a massive use of high-resistance steels, aluminium alloys and different types of plastic compounds ((Raugei et al., 2014)). In parallel, modern cars are becoming even more similar to big e-products, with great amounts of embedded electronic systems able to control almost all the vehicle's functions and virtual connections among cars and the surrounding environment ((Kohlmeyer, 2012); (Wang and Chen, 2012)). This caused a drastic increase in the production of printed circuit boards (PCBs) for automotive purposes ((Li et al., 2014b); (Reuter et al., 2013)). ECUs are among the most valuable electronic devices embedded into modern vehicles. They are able to perform the reading of signals coming from sensors embedded in a car, and control the behaviour of many sub-systems, as engine, air conditioning system, infotainment system, safety devices, etc. (National Instruments, 2009). The current amount of electronic systems is impressive, both in numbers and in impact on costs. In fact, a modern medium-sized car can embed up to 15 electronic systems on average ((Freiberger et al., 2012); (Kripli et al., 2010)) and luxury cars can reach up to 50 among microcomputers and electronic components (Wang and Chen, 2011). Furthermore, a statistic of the Bayerische Motoren Werke Corporation shows that, generally, these systems can account for more than 30% of total vehicle cost, reaching more than 50% in luxury cars (J. Wang and Chen, 2013a). These last data alone provide evidence of the importance of recovery of the embedded value in these components.

2.3. Research methodology and framework of analysis

In a systematic review of the literature, current findings are usually discussed in relation to a particular research question ((Achillas et al., 2013); (Cucchiella and D'Adamo, 2013)), which in this article is represented by ECU recycling. Scientific articles, published from 2000 up to

2014, provided by the most popular academic search engines (e.g. Google Scholar, SAGE, Science Direct, Springer, Taylor&Francis Online and Wiley Online Libraries) have been evaluated. The keyword 'recycling' was combined with different terms (automotive, automotive electronics, electronic control unit, end of life vehicles) and researched in titles, abstracts and keywords of scientific articles ((Hiratsuka et al., 2014); (Sakai et al., 2014); (Vermeulen et al., 2011); (Wang and Chen, 2011); (Xu et al., 2014)).

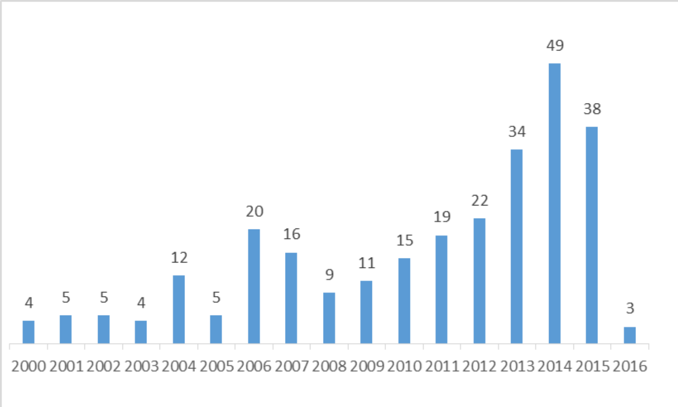

Figure 4. Historical series of published papers on ELVs

Figure 4 displays results of the search process, in terms of number of articles per year, and publications trend. The total amount of articles (142) reveals the relevant attention devoted to this topic (from 2000 up to end 2014) by the experts, especially in 2014 and at the beginning of 2015. A total of 92 articles were published in scientific journals with impact factor, 15 in scientific journals without impact factor, 22 in proceedings of scientific conferences, six scientific reports, five book chapters and two industrial reports. The nationality of the articles' first author indicates China as the major contributor, with 21 articles (14.8%), followed by Germany (10.6%), Italy (9.2%), USA (7.0%) and UK (6.3%) (**Figure 5**). Chinese experts seem to be the most involved in this research field, despite the infancy of the Chinese ELV recycling chain (Zhao and Chen, 2011).

There are several perspectives from which ELV recycling was approached. In macro topic terms, the strategic and economic perspective is the most discussed in literature (26.8%), followed by national recycling systems (13.4%), automotive shredder residue (ASR) treatment and recycling processes (12.0% each), ELV remanufacturing (10.0%), national policies and environmental issues (8.5% each), ELV recycling – other materials (5.6%), ELV recycling – electronics (2.1%) and ELV volumes predictions (1.4%) (**Figure 6**).

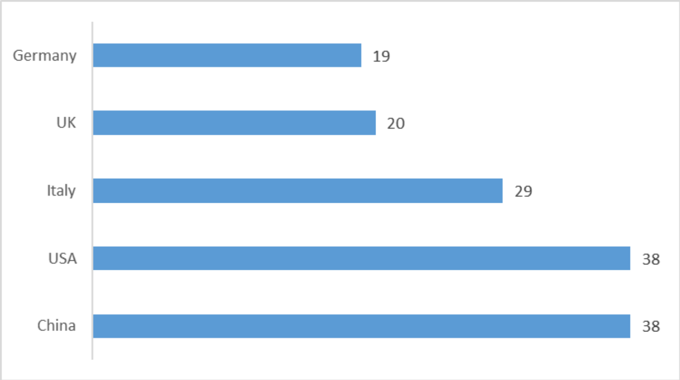

Figure 5. Top five publishing countries

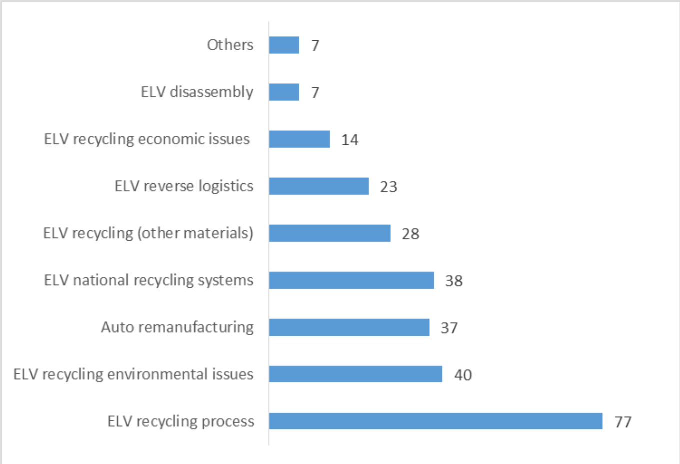

Figure 6. ELV recycling macro research areas

The analysis highlighted a multi-disciplinary topic. For this reason, journals pertain to various research fields and scientific areas. This situation underlines the scarce interest of the international literature about automotive electronics recycling. In fact, only some authors ((Berzi et al., 2013); (Gerrard and Kandlikar, 2007); (Sakai et al., 2014); (Vermeulen et al., 2011); (Zorpas and Inglezakis, 2012)) consider this type of wastes as critical for reaching the updated European ELV directive recovery and recycling levels. This way, the adoption of dedicated recycling processes within the current reverse logistic chain was not supported at all.

2.4. Results

The literature review allowed analysis of several aspects of the disposal of ECUs and a list of perspectives can be highlighted:

- ELV policies;
- ECUs EoL strategies;
- ECUs recycling environmental benefits;
- ECUs recycling technological benefits;
- ECUs recycling economic benefits.

2.4.1. ELV policies

The management of ELVs around the world widely differs by a series of variables (e.g. regulations, social structure, political system, economic development) and it can be roughly divided into several geographic macro areas, as Europe, Asia and rest of the world.

ELV policies in Europe. The European Union (EU) was one of the first super-national entities to establish a dedicated ELV Directive in 2000 ((European Union, 2000); (European Union, 2005); (European Union, 2013)). According to this regulation, each EU member state must ensure that all ELVs generated within its national borders will be treated by authorised treatment facilities (ATFs) for a correct recycling; furthermore, a series of reuse/recovery/recycling rates were set for EU car manufacturers ((Blume and Walther, 2013); (Ferrão et al., 2006); (Lu et al., 2014); (Mazzanti and Zoboli, 2006)). However, this directive does not specify how to manage each component of a car. Given the above, EU car manufacturers launched, many years ago, dedicated remanufacturing processes able to manage valuable components (e.g. engines, alternators, transmissions, etc.). In some nations, like in Germany, the remanufacturing industry reached high performances, being able reasonably to reuse or deal with about 90% of the parts dismantled from ELVs ((Wang and Chen, 2011); (Zorpas and Inglezakis, 2012)). However, the ECUs remanufacturing sub-sector remains even very limited, with the presence of only a few independent actors ((Kripli et al., 2010); (Peters et al., 2014)). In parallel to the ECUs management, the resolution of other important issues (e.g. illegal exports control, dedicated informatics tools development, ATFs role and relationship with car manufacturers, standardisation of design for reuse, remanufacturing and recycling (Df3R) rules) could support the improvement of the entire ELVs management sector ((Gerrard and Kandlikar, 2007); (van Schaik and Reuter, 2004)). Europe is one of the highest producers of ELVs, and the literature estimates an annual generation of 14 million tonnes per year since 2015 ((Hiratsuka et al., 2014); (Sakai et al., 2014)). It is clear that the correct management of this amount of waste could offer great profitability chances to European car remanufacturing and recycling chains, allowing it easily to reach new ELV directive's targets (Kunze, 2012).

ELV policies in Asia. From the Asian side, Japan was the first nation to activate ELV legislation in 2005. Its structure is similar to the EU Directive, and virtually covers all vehicle's categories and components (ECUs included). A well-structured reuse/remanufacturing industry was established since some years by Japanese car manufacturers ((Hiratsuka et al., 2014); (Sakai et al., 2007)). In South Korea, the national ELV Directive is coupled with the one related to waste from electric and electronic equipment (WEEE) management. Even in this case, the structure is similar to the EU Directive, but tyres, batteries and air bags are not considered within the automotive components to be recycled (Sakai et al., 2014). In Taiwan, since 1994, ELV recycling has gradually become systematic. In 2004, the Taiwan Environmental Protection Administration defined a series of ELV guidelines, inserted in the 'Waste Disposal Act', trying to regulate the overall recycling chain. Since then, the recycling channels, processing equipment and techniques for ELVs in Taiwan have gradually become established. However, recycling rates and the ASR management remain lower than in developed countries ((Che et al., 2011); (Cheng et al., 2012)). Finally, China developed an ELV recycling regulation in 2001, enforced in 2006 with a set of more stringent constraints for the management of automotive scraps, expected to reach 13 million tonnes per year by 2020 (Zorpas and Inglezakis, 2012). However, these limits are even lower than the European ones.

ELV policies in rest of the world. From the American side, USA is one of the most important nations (together with Canada, Australia and New Zealand) without any form of a federal law on the treatment of ELVs, even if their car market is the largest in the world ((Lu et al., 2014); (Sakai et al., 2014); (Zorpas and Inglezakis, 2012)). Fortunately, USA car manufacturers autonomously have shared this responsibility since 1992 and, currently, are able to reach a ELVs recycling rate of about 95% (according to the vehicle's weight). Furthermore, car parts remanufacturing (ECUs included) is a common activity, allowing customers to buy recovered/refurbished spare parts directly from ATFs ((Keivanpour et al., 2013b); (L. Wang and Chen, 2013)).

2.4.2. ECU EoL strategies

In terms of EoL strategies, several articles ((Freiberger et al., 2011); (Reuter et al., 2006); (Steinhilper et al., 2006); (Tian and Chen, 2014); (L. Wang and Chen, 2013)) analyse 'where electronic systems end when cars are dismantled'. It is possible to highlight that the destiny of an ECU completely depends on both its market request and structural/functional conditions.

- If ECUs have a high market request and are in a good structural condition, during the dismantling of a car, ATF operators disassemble the specific electronic component (usually, a sub-part of more complex mechatronic systems) and, after a cleaning phase, test the element, by verifying its functional conditions. If the test is positive, the electronic component is re-assembled into the main system and, then, is sold as a 'guaranteed used part' in the spare part market by independent companies or directly by original equipment manufacturers (OEMs). In this case, the environmental impact is almost inexistent if compared with a new product, the value recovery is maximised and the operational costs are very limited ((Kripli et al., 2010); (Tian and Chen, 2014); (Xue et al., 2012a)).
- If ECUs have a high market request and are in good structural conditions, but do not pass the functional conditions test, they are remanufactured (Subramoniam et al., 2013). During remanufacturing, the product is repaired, re-tested and reassembled. Then, it is sold as an 'as good as new' product in the spare part market by independent companies or directly by OEMs. Their cost can vary from 50% up to 70% of the corresponding new component, with an energy and resource saving of about 90% when compared with a new part ((Kripli et al., 2010); (Tian and Chen, 2014); (Xue et al., 2012a)).
- If ECUs do not have a market request and/or are not in good structural conditions, these electronic components are not disassembled at all. This way, they are subsequently crushed and shredded within the car hulk, ending in the ASR fraction (Wang and Chen, 2011). Then, ASR is incinerated to recover the embedded energy or, in the worst case, landfilled ((McKenna et al., 2013); (Viganò et al., 2010)).

Curiously, these three processes act without any assessment of the potential embedded value (in terms of materials content) of treated ECUs by recyclers ((Garcia et al., 2015); (Go et al., 2011); (Park et al., 2014); (Xu et al., 2014)). This means that some high valuable ECUs in bad structural conditions (and almost all of the low value ECUs) are incinerated or landfilled (as part of ASR fraction), instead of following a more accurate recycling process. This way, a great profit loss for recyclers and waste of valuable resources are lost during thermal and chemical processes. Unfortunately, this lack of any kind of assessment in ECUs recycling is a common issue among nations ((Che et al., 2011); (Lu et al., 2014); (Sakai et al., 2014); (Zhao and Chen, 2011)).

2.4.3. ECUs recycling environmental benefits

With the rapid growth of vehicle population and electronic control components used in automotive, the energy consumption and environmental emissions of automotive electronic control components reached 20,306,000 tonnes of standard coal equivalent (SCE) in 2007, that is an impressive consumption of energy, and related toxic environmental emissions (Wang et al., 2012). As ECUs are a particular type of PCBs, similar environmental impacts as the ones widely discussed in literature for these elements are expected. Hence, the presence of tin and lead in solder and components, copper in contacts and circuits, iron and nickel in components and flame-retardant chemicals (e.g. polychlorinated biphenyls) represent the main sources of environmental pollution coming from ECUs ((He and Xu, 2014); (Wang and Chen, 2012)). For example, about 50 g of lead per vehicle can be extracted from medium-sized cars (Wang and Chen, 2011). Avoiding the dispelling of lead (but also copper and nickel) during the ELVs recycling process could allow the reduction of heavy metals contamination in water and soil, and the release of hazardous substances in the environment, especially in countries where environmental protection measures are still under development ((Mancini et al., 2014); (Zhao and Chen, 2011)). To this aim, the consideration of environmental impacts caused by the increasing use of electronic components in modern cars during the design phase could allow car manufacturers to improve the sustainability of future private transportation systems ((Ni and Chen, 2014); (Sakai et al., 2014)).

2.4.4. ECUs recycling technological benefits

The recycling of ECUs can also offer (even if indirectly) a series of technological benefits. In fact, (Clappier et al., 2014) highlight that an increase in the recovery of scarce raw materials is needed, in order to reach 2015 EU recovery and recycling targets (95% of ELV weight on average). However, up to now, international research did not look into this particular kind of recovery, but strongly focused on the improvement of the ASR management, which is the only part of ELVs commonly considered as the most promising resource to exploit for the increase of recycling performances, mainly because of its huge amount ((Taylor et al., 2013); (Vermeulen et al., 2011); (Zhang and Chen, 2014)). To this aim, experts established two research lines: (i) intensive dismantling, involving the separation and collection of materials at the dismantling stage and (ii) post-shredder treatments involving the collection of materials from ASR fraction, after the shredding stage ((Lu et al., 2014); (Sakai et al., 2014)). Hence, experts preferred to think about new processes, either before or after the ELV shredding phase. From one side, the intensive dismantling seems to be the most promising strategy because, this way, different components could be correctly transferred to a specific recovery and recycling chain, by reducing the overall generation of ASR as well as its hazardousness (Sakai et al., 2014). However, especially in developed countries, this could imply an increase in dismantling labour costs (Go et al., 2011). From an opposite side, post-shredder treatments have to cope with a big issue, represented by the management of a highly heterogeneous waste stream (Taylor et al., 2013). However, even intensive dismantling is not immune from problems. In fact, as vehicle material's composition changes, higher dismantling/recovery rates are needed to ensure economic viability of the recycling infrastructure ((Ferrão et al., 2006); (Garcia et al., 2015)). Furthermore, even in the case of significantly higher rates of dismantling and plastics recovery, the amount of shredder residue per vehicle will continue to rise (Raboni et al., 2015). Hence, in order to reach new targets, a higher efficiency in ASR recovery is needed, in addition to material recycling of collectable components and metals (Sakai et al., 2014). This leaves open the doors to innovative ELV dismantling techniques or strategies for the recovery/recycling of parts/materials prior to shredding (Golinska et al., 2014). As summarised by (Kohlmeyer, 2012), one way to improve the ASR recycling could be the correct management of a vehicle's increasing computerisation before it becomes an issue in the near future (e.g. in hybrid, electric and hydrogen cars). In fact, the use of rare metals and hazardous substances for computer-related components have further made ASR recycling difficult (Sakai et al., 2014) and the presence of metal–plastic composites and flame-retardant chemicals (e.g. polychlorinated biphenyls) in vehicle's electronic components require high temperatures to separate and recover valuable materials (Lee et al., 2015). These factors are considered significant obstacles to polymer processing and, so, to the improvement of recycling performances ((Gerrard and Kandlikar, 2007); (Tian and Chen, 2014)).

2.4.5. ECUs recycling economic benefits

The economic benefits coming from the reuse and recycle of ECUs have already been proposed in literature. For example, (Wang and Chen, 2011) indicate a series of revenue sources coming from ELVs recovery.

- High value (high demand), undamaged, recovered reusable components.
- High value secondary raw materials with high purity levels.
- Energy recovered and sold from incineration of the ASR's light fraction.

Basing on the EoL strategies, hierarchy, reuse and remanufacturing are preferable to recycling and energy recovery because they allow keeping a higher percentage of the embedded value already present in wasted products with a lower consumption of energy and natural resources ((Kripli et al., 2010); (Wang and Chen, 2012); (L. Wang and Chen, 2013)). However, given the reduced amounts of reused and/or remanufactured parts (almost from 20% to 30% of cores follow this process ((Ferrão and Amaral, 2006); (Hiratsuka et al., 2014); (Morselli et al., 2010)), recycling is one of the most common ways to recover materials and, even if in small percentage, the embedded value. In fact, the correct recycling of automotive electronic components could allow ATFs to improve their performances without increasing labour costs, and reducing disposal and plant's maintenance costs ((Gerrard and Kandlikar, 2007); (Tian and Chen, 2014)). Furthermore, the reverse logistics flows could be optimised, by reducing transportation costs (Demirel et al., 2016). However, given a series of factors like ((Clappier et al., 2014); (Garcia et al., 2015); (Schmid et al., 2013); (Xu et al., 2014)):

- Wide amounts of ECUs models and alternatives;
- Lack of precise data in literature;
- Absence of specific regulations;
- Inexistence of proper informatics decision-support tools,

An estimation of potential amounts of ECUs coming from ELVs (Vermeulen et al., 2011) and, subsequently, the amount of valuable resources potentially reusable as secondary raw materials, is not so simple to do.

2.5. Discussion

In a world where the correct management of sustainable products and processes (and related lifecycles) is increasing in its importance, the automotive sector plays a relevant role. The European Union tried over the last two decades to develop a circular economy based on the exploitation of resources recovered by wastes (Cucchiella et al., 2015b). However, the previous literature review highlighted as current ELV directives, depending on weighted-based principles, do not adequately take into account the management of EoL automotive electronic components. To this aim, the article offers interesting findings in technological, environmental and economic benefits that could be potentially achieved through the correct recycling of ECUs. In fact, their management could allow the reduction of both the quantity of wastes ending up in landfills and the use of additional resources during the production of new cars. This way, a considerable reduction of the overall automotive supply chain environmental impact could be reached. Furthermore, many new work positions could be opened, reducing the post-crisis effects on employment rates. Finally, all these benefits could be achieved with a very low political effort in terms of regulation changes. However, the current vehicle recycling system technological level, together with the absence of a clear design for dismantling/ recycling standards, does not easily comply with new recycling rates imposed by the EU ELV Directive for 2015 (95% by an average weight per vehicle and year as reference reuse and recovery rate, of which 10% coming from energy recovery of non-recyclable materials). Over the years, many experts have found agreement in saying that the easiest way to improve current recycling rates is, from one side, to focus on a better management of the so-called ASR fraction and, from another side, to improve the dismantling phase performances by exploiting information communication technologies (ICTs) and databases. The article follows both these two visions, going to assess the embedded value in electronic components, a kind of systems that, especially in newest cars, seems to acquire even more importance in the management of all the vehicle's main functions. Currently, this kind of subcomponent (if not remanufactured) end directly in the ASR fraction, without any type of control. Instead, the literature confirms that their value is comparable with the one embedded in medium grade waste PCBs, present in many types of WEEEs (Birloaga et al., 2014). Given the increasing use of electronic subsystems in the automotive sector, the right management of these new types of e-wastes could represent a profitable business, both for automotive recyclers and car manufacturers. Aiming to fulfil the gaps presented in the literature, a series of important studies should be implemented. First of all, the scientific literature focusing on EoL automotive electronics management should increase in number, trying to better present the issues from different views. For example, more focus on the potential profitability coming from the recovery of this type of e-waste would support the implementation of dedicated and flexible recovery centres (both mobile and field ones), where different types of automotive electronic components could be correctly managed in the same place (something similar was already studied for WEEEs (Cucchiella et al., 2015b)). In addition, profitability could be guaranteed also through a dedicated set of national laws (or changes in the current ELV Directive) and fees to be paid in case of noncompliance of correct EoL processes. In this direction, a quality-based (instead of a weight-based) policy on the recovery of resources should be followed. Also ICTs supporting the EoL decision-making process should be better integrated with the existing informative networks, already present inside a company. This need is evident in literature, where the non-cooperation between car manufacturers and OEMs and the rest of the reverse logistics chain is clear. Finally, some studies analysing possible issues arising from the recovery of materials from future ELVs are needed. For example, the dismantling process of lithium-ion batteries embedded into electric vehicles has already been considered by many authors as a primary issue to be solved before these cars will reach their EoL.

2.6. Description of the original contribution of the three papers

Paper #1 original contribution: the original contribution given by the second paper embedded in this thesis lies in several elements. Firstly, it is one of the few papers comparing directly different sources of e-wastes. Secondly, all these waste sources are assessed for the first time under several perspectives, evidencing current similarities and inequalities in their management. Finally, the comparison aims to support the reader (being it either a manager of a company or a general reader) in comprehending the potentials coming from a sustainable management of these wastes and giving it all of the numbers to quantify them. Some of the numerical data come from the second paper embedded within this thesis.

Paper #2 original contribution: the original contribution given by the third paper embedded in this thesis is even stronger than the previous one. For the first time in literature, it was possible to quantify the potential profit coming from the sustainable management of automotive electronic wastes. No other experts before assessed this important dimension. To do that, several information sources were put together trying to gather all the needed data. The literature assessment done in previous works allowed us to comprehend the current state of the recovery process, available technologies and the cost structure related to them. Furthermore, several authors gave us reference parameters to set up a hypothetical recovery process. Again, the literature offered us the chance to gather information also in terms of ELV materials, volumes and future trends. Once assessed all these data, revenues were calculated basing on information coming from an official automotive database containing lots of information about materials contained in each component constituting a car. Together, costs, revenues and volumes allowed us to identify potential performances coming from this kind of activities.

Paper #3 original contribution: the original contribution given by the fourth paper embedded in this thesis lies in the middle between the previous two papers. The adoption of a SD model for the assessment of economic potentials related to the ELV recovery chain is not an original contribution. This work was already done twenty years ago by a student of the Massachusetts Institute of Technology (MIT), trying to assess the evolution of the North American ELV recovery chain. Instead, the original contribution in this case lies in the re-adoption of the same SD model, in the same industrial context, but in another nation (Italy), in another period of time (2015 instead of 1995) and for the assessment of the influence of a specific activity on the overall performance of the ELV recovery chain. Results coming from the simulation of several business scenarios demonstrate what evidenced only numerically by the previous mathematical model.

2.7. Objectives of the work

The main objectives of this thesis are essentially three. Firstly, this work wants to assess the current state of the literature about e-waste management practices and, more into detail, about the management of e-wastes from the automotive sector. This assessment aims to put the light on current sustainability issues affecting the ELV recovery chain, trying to propose a potential solution to some of them. Secondly, this work wants to quantify the potential benefits coming from this kind new practices, especially in economic terms. This is a fundamental step for enabling the introduction of these new practices in real industrial contexts. Finally, this work wants to assess the effect coming from the introduction of these new practices on current economic performances of two of the most important actors involved in any kind of ELV recovery chain, or dismantlers and shredders. This last assessment can allow to comprehend what should be the best way of doing and who should be the best user of the presented solution.

2.8. Short summary of the three papers

Paper #1 summary: 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.

Paper #2 summary: End of Life Vehicles (ELVs), together with Waste from Electric and Electronic Equipments (WEEEs), are renown as an important source of secondary raw materials. Since many years, their recovery allowed the restoring of great amounts of metals for new cars' production. However, the management of electronic systems embedded into ELVs is yet rarely considered by the scientific literature. The purpose of the paper is trying to fill in this gap through the proposition of an innovative economic model able to identify the presence of profitability within the recovery process of automotive Waste Printed Circuit Boards (WPCBs). Net Present Value (NPV) and Discounted Payback Time (DPBT) will be used to demonstrate the validity of investments in this type of plants. Furthermore, a sensitivity analysis on a set of critical variables (plant saturation level, gold (Au) content, Au market price, Au final purity level, WPCBs purchasing cost and opportunity cost) will be conducted for the evaluation of the impact of significant variations on results. Finally, the matching of predicted European ELVs volumes (during the period 2015 - 2030) and NPVs coming from the economic model will quantify the potential advantages coming from the implementation of this new kind of circular economy.

Paper #3 summary: End-of-Life Vehicles (ELVs), together with Waste from Electrical and Electronic Equipments (WEEE), are one of the most valuable sources of secondary raw materials. Their reuse for producing new goods is a well-known topic in the literature. However, End-of-Life (EoL) strategies implemented by companies remained the same since the last century, completely based on materials market prices. Progressively, this way of doing exposed the entire ELV recovery chain to a series of unwanted market risks, like unpredictable fluctuations of material market prices, uncontrolled illegal transfer of vehicles among nations and the unavoidable evolution of the material's mix in ELVs. The purpose of this paper is to apply an already existing model based on the system dynamics methodology to the Italian context, trying to assess current economic performances and discuss some possible future evolutions offered by the recovery of automotive electronic components. Considering the current volumes of ELVs in Europe (estimated in 7-14 million tons per year) and their annual growth rates (estimated in 2%-8%) potential economic performances could be very high. Results demonstrate the potential impact on both profits and the overall sustainability of the entire ELV recovery chain, leaving space to interesting evolutions of the industrial context.

3. Paper #1: Comparison of current practices for a combined management of printed circuit boards from different waste streams

Section 3 compares in a non-conventional way the two main sources of electronic waste, or WEEE and ELV, trying to define and evidence both current inequalities and commonalities. Taking advantage from what presented in the previous chapter, Section 3 expands the point of view of the reader, making him more aware about the overall reference context (see ***Figure 7****). This section reports integrally what can be found within the paper titled "Comparison of current practices for a combined management of printed circuit boards from different waste streams", edited by Journal of Cleaner Production (see Section 7 for details).*

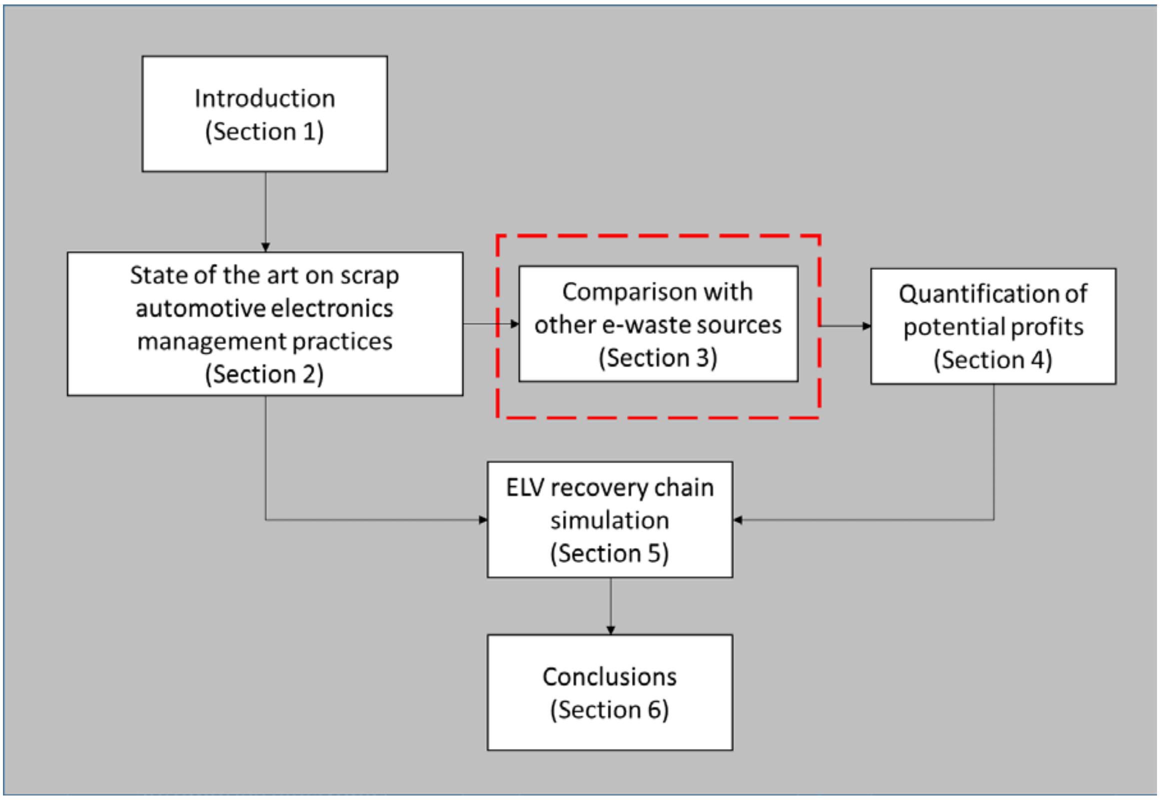

Figure 7. Relation of Section 3 with the overall structure

3.1. Introduction

The previous literature review evidenced some important aspects related to the current management of scrap automotive electronics (especially ECUs), by putting the light on potential benefits that could be achieved through more sustainable practices. However, automotive electronics is not the only source of e-waste. More specifically, the entire literature does not consider at all automotive electronics as a source of e-waste. So, any tentative to gather information on this topic only basing on the scientific literature does not have success in reaching a relevant mass of papers demonstrating the importance of this issue. Considering this point, for the first time the authors tried to compare WEEE – or the commonly re-known main (and only) source of e-waste – with ELVs – and, more specifically, with automotive electronics – under different points of view. This way of doing allowed us to put in evidence similarities and inequalities characterizing the two e-waste streams, by assessing the existence of potential ways to fill in some of the existing sustainability issues. The present paper assesses these issues with a specific focus on the treatment of printed circuit boards (PCBs) from both 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 wasted PCBs, 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 estimated, 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 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.

3.2. 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.

3.2.1. 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 8**, **Figure 9** and **Figure 10** display 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).

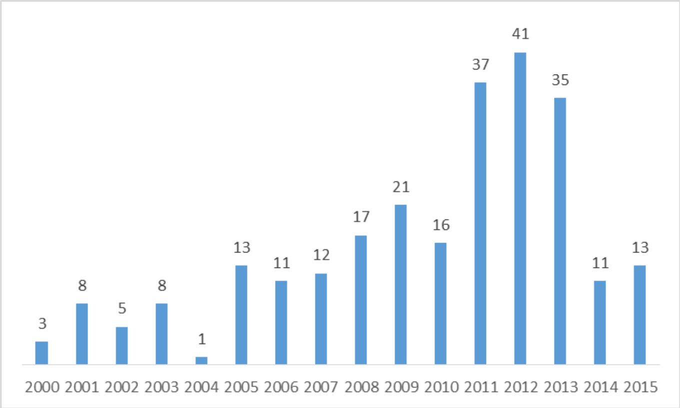

Figure 8. Historical numbers of published papers – WEEE

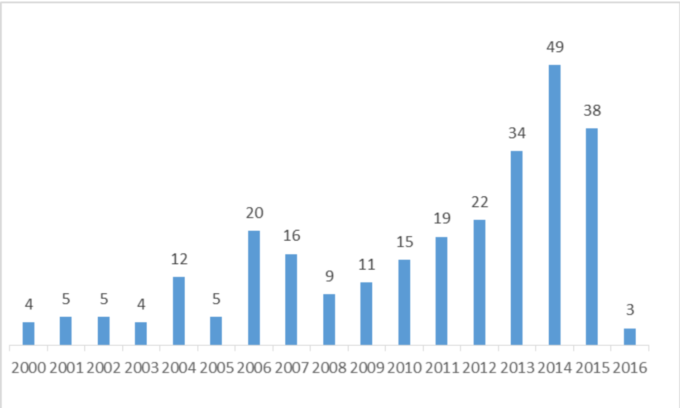

Figure 9. Historical numbers of published papers – ELVs

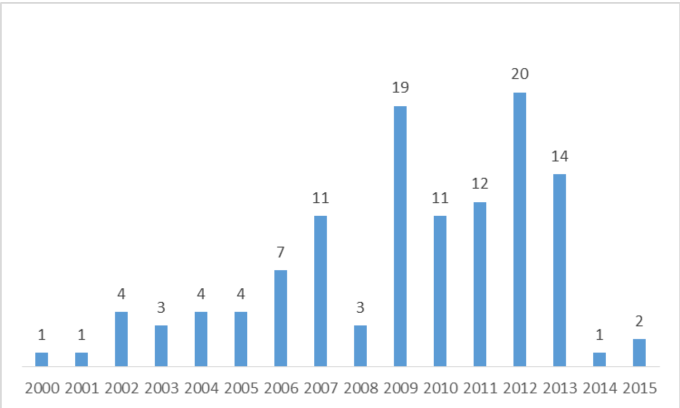

Figure 10. Historical numbers of published papers – PCBs

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 (**Figure 11**). Again, recycling processes are the most discussed topic for ELVs (**Figure 12**). Finally, recycling technologies are the prevailing topic for PCBs, as shown in **Figure 13**.

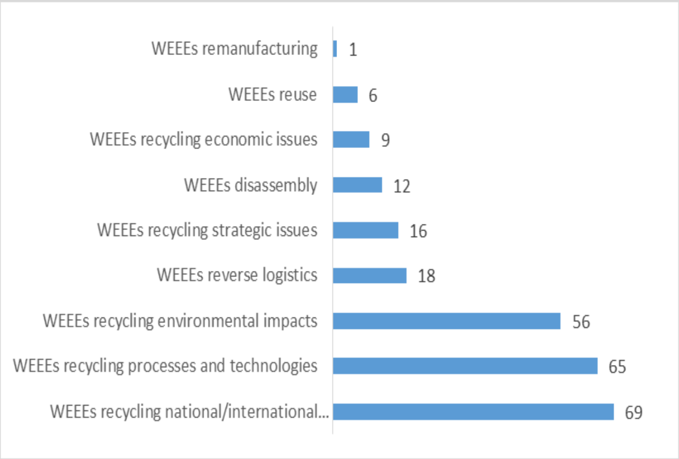

Figure 11. Main topics of published papers – WEEE

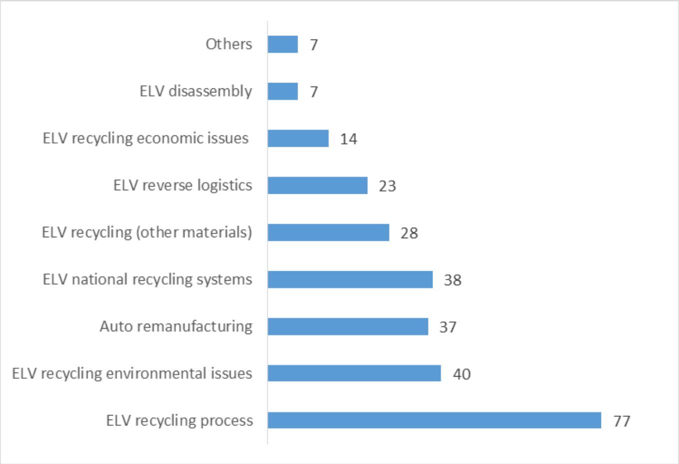

Figure 12. Main topics of published papers – ELVs

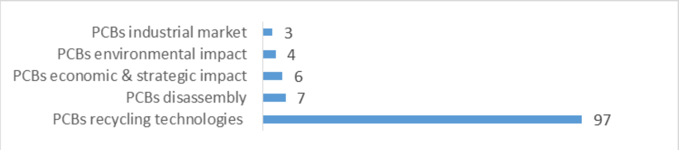

Figure 13. Main topics of published papers – PCBs

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.

3.2.2. 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 14**. 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 15**. 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.

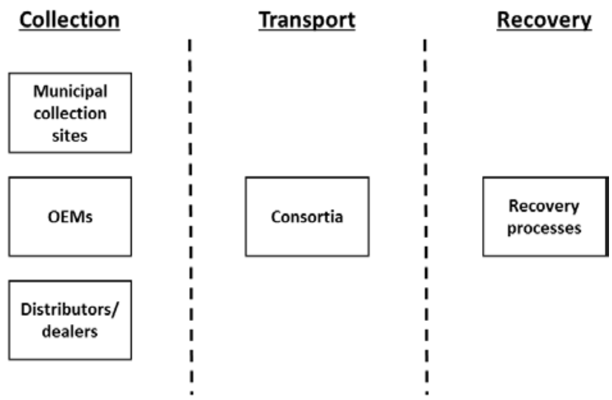

Figure 14. The current collection process of WEEE – Adapted from (UNEP-DTIE-IETC, 2012)

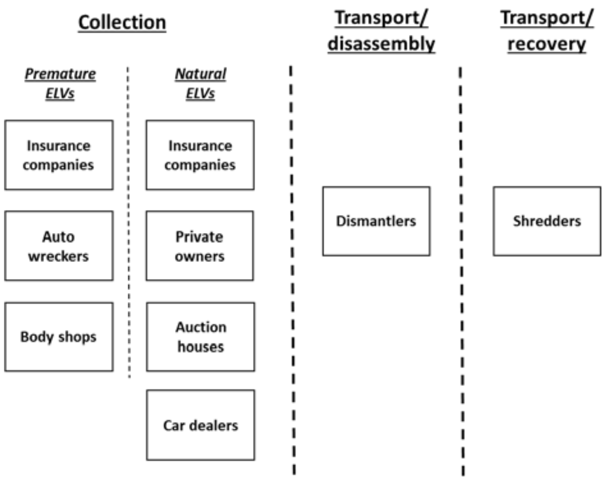

Figure 15. The current collection process of ELVs – Adapted from (Blume and Walther, 2013)

Important distinctions can also be found in recovery processes. A representation of a typical WEEE recovery process is shown in **Figure 16**, 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).

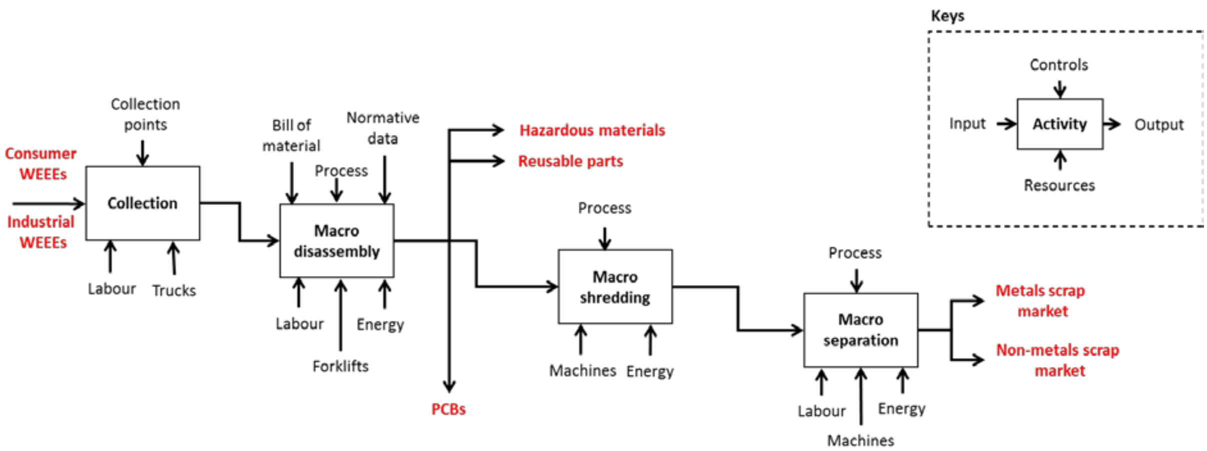

Figure 16. The current recovery process of WEEE – Adapted from (Dalrymple et al., 2007)

For the automotive sector, a typical ELV recovery process is reported in **Figure 17**, 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, 2014)]. Even if information about the point at which PCBs leave the ELV recovery process are available [(Kumar and Sutherland, 2009); (Williams et al., 2006)], 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.

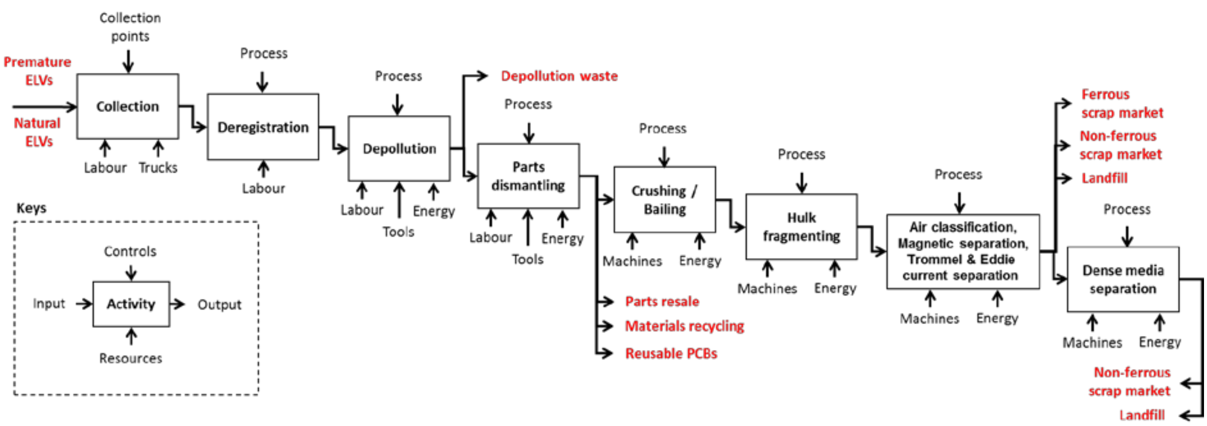

Figure 17. The current recovery process of ELVs – Adapted from (Vermeulen et al., 2011)

3.2.3. 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 [(Steinhilper 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.

3.2.4. 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., 2012b)], 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., 2014a)]. The final destinations of ELVs generated within Europe are eastern European and extra-European countries.

3.3. 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.

3.3.1. 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.

3.3.2. 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 [(L. 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 18**.

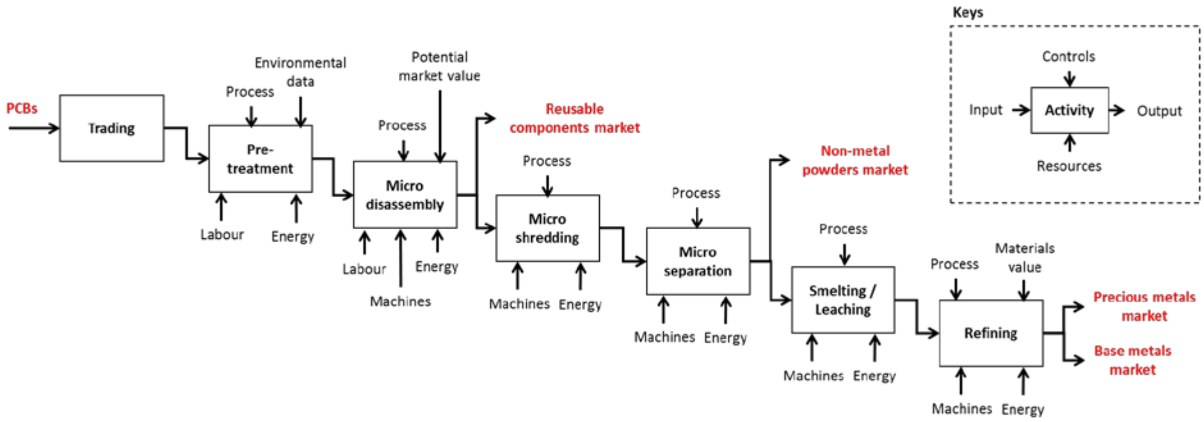

Figure 18. A traditional waste PCB recovery process – Adapted from (Li et al., 2004)

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)].

3.3.3. 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 (L. 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).

3.3.4. 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., 2015b). 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).

3.4. 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 – Source: (Eurostat, 2015a); (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, 2015b); (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 – Source: (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.

3.5. 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.

3.6. 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.

Post-publication potential improvements

The limited number of words imposed by the journal format did not allowed to better describe the WEEE sector (e.g. in terms of industries, components and key figures) and insert some additional tables better summarizing the main differences and commonalities between ELVs and WEEE. Again, an analytical model for the assessment of PCB volumes describing the variables and their interrelationships could have been embedded within the paper. This way, the formulation problem could have favoured the uncertainty modelling, by enabling the introduction of stochastic parameters and make possible the business risk assessment. Finally, a wider description of existing standards recycling/recovery processes could give a more concrete picture of the current situation.

4. Paper #2: Automotive printed circuit boards recycling: an economic analysis

*Section 4 tries to quantify in terms of potential profits the economic benefits coming from the recovery of materials from scrap automotive electronic components. In a logical term, Section 4 should be put between Section 2 and Section 3. In fact, some of the final tables presented here were already presented in Section 3 (see **Figure 19**). This causes some irregularities in the linear consecution of works. However, the author preferred to position his work to better follow the logical consecution offered by the two research questions. This section reports integrally what can be found within the paper titled "Automotive printed circuit boards recycling: an economic analysis", edited by Journal of Cleaner Production (see Section 7 for details).*

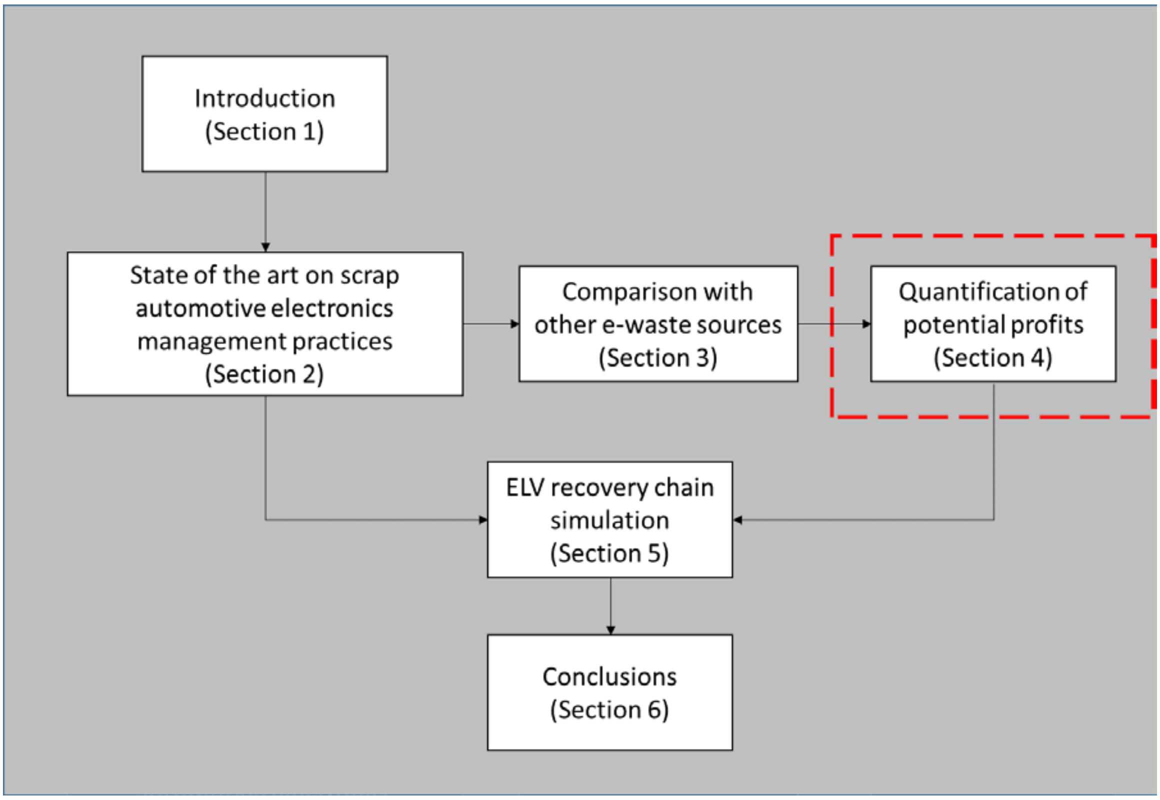

Figure 19. Relation of Section 4 with the overall structure

4.1. Introduction

The previous paper discussed about the presence of similarities and inequalities in the management of different sources of e-waste, by defining new areas of improvement and giving some numbers about their potential profits. However, these profits derive from the application of a dedicated mathematical model developed by the authors during the years and presented in several papers. Considering the only automotive sector, this paper presents an application of this model, adequately configured for the management of automotive electronic components. This mathematical model is one of the only methods present in literature able to give some estimations in terms of profits coming from the recovery of wasted automotive electronics, or one of the most important sources of value present in ELVs. Within the automotive sector, the use of PCBs inside a car for the management of almost all the functionalities of a vehicle drastically increased in the last decades ((Kim et al., 2014)). Hence, this trend undoubtedly contributed in increasing volumes of PCBs produced and, so, to the overall amounts of wasted PCBs dismantled. In fact, the automotive sector, together with the mass electronics sector, is one of the most important sources of waste, both in volumes ((Zorpas and Inglezakis, 2012); (Cucchiella et al., 2014a); (Sakai et al., 2014); (Tian and Chen, 2014)) and in materials content terms ((Berzi et al., 2013); (Uan et al., 2007)). For this reason, basic guidelines for the reuse, recovery and recycling of ELVs were established allover the world in the last decades. Within the scientific literature, lots of papers analysed and compared different ELV directives and national recovery systems ((Sakai et al., 2014); (Zhao and Chen, 2011)). However, only in very few cases the attention was focused on automotive electronics recovery (e.g. (Wang and Chen, 2011)). Considering this literature gap, the main objective of this paper is threefold. Firstly, the paper wants to describe a mathematical model able to assess the potential profitability characterizing all the phases of a typical waste PCBs recovery process. Secondly, the potential profitability coming for different types of plants (e.g. field and mobile ones) will be calculated, by giving a practical demonstration of the validity of results. Finally, the potential profitability will be linked with expected future ELV volumes for an estimation of the expected market dimension.

4.2. Research framework

ECUs are among the most valuable electronic devices embedded in modern vehicles. They are able to perform the reading of signals coming from sensors embedded in a car, and control the behaviour of many sub-systems, as engine, air conditioning system, infotainment system, safety devices, etc. ((National Instruments, 2009)). The current amount of electronic systems is impressive, both in numbers and in impact on costs. In fact, a modern medium-sized car can embed up to 15 electronic systems on average ((Kripli et al., 2010); (Freiberger et al., 2012)) and luxury cars can reach up to 50 among microcomputers and electronic components ((Wang and Chen, 2011)). Furthermore, a statistic of the Bayerische Motoren Werke Corporation shows that, generally, these systems can account for more than 30% of total vehicle cost, reaching more than 50% in luxury cars ((J. Wang and Chen, 2013b)). These last data alone allow to evidence how much important is the recovery of the embedded value in these components. However, current ELV directives (based on weighting principles) seems to do not adequately take into account the management of these types of e-wastes ((Cucchiella et al., 2016a)). Hence, there are no benefits for the actors involved in the automotive reverse logistic chain to invest in dedicated recovery centres ((Cucchiella et al., 2015b)).

4.2.1. Automotive PCBs characterization

Before the treatment of any kind of WPCBs amounts, there is a materials' characterization phase. This means a definition of the set of materials embedded in a certain amount of WPCBs, by chemically analysing a sample of them. This is a relevant phase because it allows to: (i) comprehend the presence (or not) of valuable materials (this way a WPCB is classified as high, medium or low grade waste), and (ii) define the expected revenues coming from their recovery. In literature, the common ways to characterize WPCBs are essentially two: (i) considering already available data coming from other papers or intra-governmental reports ((Reuter et al., 2013)), and (ii) implementing dedicated laboratory tests ((Wang and Gaustad, 2012)). The first one is the most common in papers focused on the economic sustainability of PCBs recycling processes. The second one is common when environmental sustainability is the main focus. Given both the clear focus of this paper on the economic side of sustainability, and the lack of existing data about automotive PCBs composition, the approach selected by the authors was the exploitation of existing data coming from industrial database. This explains the decision to consider IMDS as a relevant source of data from which starting with the economic assessment. IMDS is a materials data management system used in the automotive sector. Designed by Audi, BMW, Daimler, HP, Ford, Opel, Porsche, VW, and Volvo, IMDS was then adopted by other car manufacturers, so becoming a global standard used by almost all the automotive Original Equipment Manufacturers (OEMs) worldwide. Data related to 500 different automotive electronic devices were extracted and, subsequently, categorized into four typologies basing on their weights distribution (divided into quartiles). The four resulting groups are represented by:

- Small WPCBs, going from 0.2 g up to 8.7 g;
- Medium-small WPCBs, going from 8.8 g up to 52.9 g;
- Medium-big WPCBs, going from 53.0 g up to 134.2 g;
- Big WPCBs, going from 134.3 g up to 477.9 g.

This choice was purely objective and derives from the fact that waste automotive PCBs are very different in terms of size, shape and composition, depending on their functionality ((J. Wang and Chen, 2013a)). Hence, a subdivision like the one commonly done for WPCBs coming from WEEEs (or high, medium and low grade waste) was considered as not representative.

4.2.2. WPCBs recycling processes

Starting from the main assumption that scrap automotive electronic devices are, in effect, WPCBs, consequently it is possible to consider the same technological process followed for the recycling of WPCBs coming from WEEEs ((J. Wang and Chen, 2013a), (Wang and Chen, 2012); (Cucchiella et al., 2016a)). Hence, the recycling process can be seen as the sum of three main phases that, starting from WPCBs, are able to obtain as final output a set of (almost pure) raw materials. These phases can be distinguished in: dismantling, pretreatment and refining ((Sohaili et al., 2012); (Yu et al., 2009)) – see **Figure 20**. During disassembly, both the casings embedding PCBs and toxic components present on the main board are separated. Toxic components (e.g. condensers or batteries) are disassembled and destined to specific treatments for hazardous materials. Instead, casings (generally, Aluminium (Al)-made elements) can be directly sold to smelters, becoming an additional source of revenues for recyclers. The pretreatment process is implemented through a series of dedicated machines, or shredders, grinders and separators (based on several physical principles). During pretreatment, WPCBs are crushed into micro pieces up to become a uniform powder, through the use of shredders and grinders. After this phase, powders are separated basing on their composition, by distinguishing metal from non-metal powders ((Zeng et al., 2015); (Li and Xu, 2010)). Nowadays, these last ones are destined to landfills, however there are interesting works studying alternative (and valuable) ways to reuse them for different purposes ((Li et al., 2012); (Hadi et al., 2013b)). Finally, metal powders are refined, up to obtain almost pure secondary resources (the purity level differs from on material to another ((Wang and Gaustad, 2012)) directly reusable for the production of new goods. The refining process can be based on different technologies (e.g. pyrolysis, pyrometallurgy, hydrometallurgy, biometallurgy).

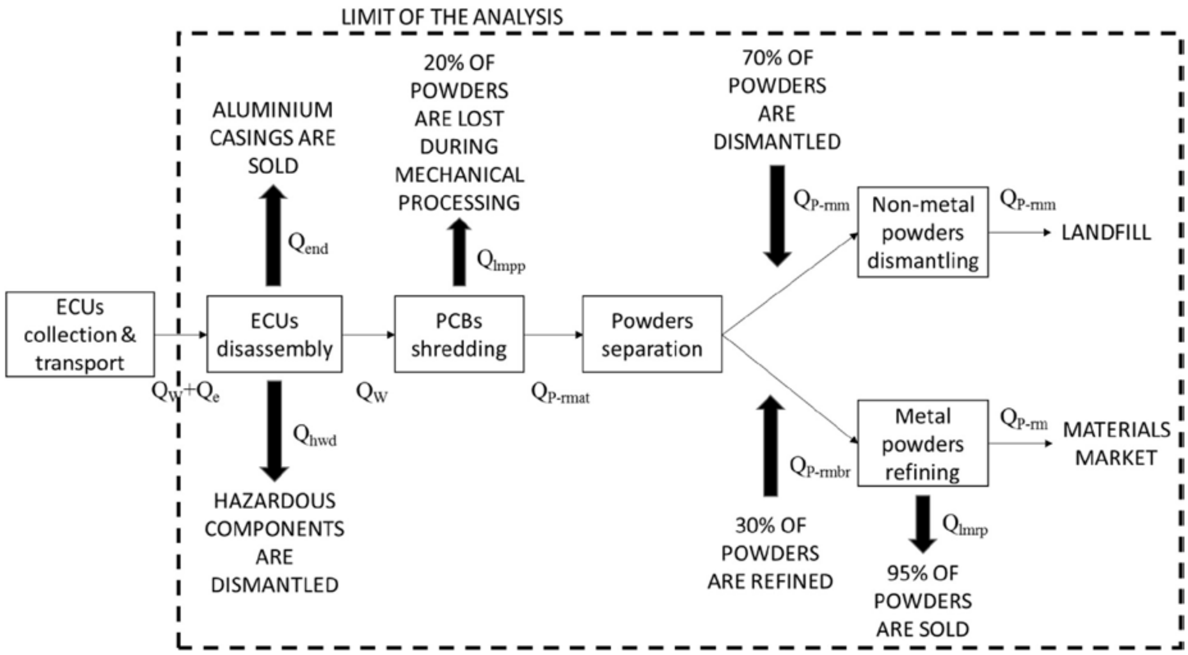

Figure 20. A traditional WPCBs recycling process (Source (Reuter et al., 2013))

In this work, hydrometallurgy is considered as the only refining process because the literature ((Behnamfard et al., 2013); (Birloaga et al., 2013)) commonly agrees on its higher sustainability, if compared to other methods. Although biometallurgy could be even better than hydrometallurgy, currently there are no information about its use at industrial scale ((Zhu et al., 2011); (Liang et al., 2010)). The economic model that will be proposed in Section 3 has a high level of detail. Hence, it is of utmost importance to take a look on material flows (and related nomenclature) to better comprehend its logic – see **Figure 20**. The recycling process starts from the entire ECU (composed by a casing (Q_e) and a PCB (Q_w)). As described before, ECUs are disassembled up to extract PCBs (Q_w) from casings (Q_{end}) and eliminate hazardous components (transferred to dedicated recovery processes e Q_{hwd}). Then, they are reduced into powders and part of them remains trapped into shredder's/grinder's/conveyors mechanisms (Q_{lmpp}). The remaining part ($Q_{p\text{-}rmat}$) is, then, separated into metal ($Q_{p\text{-}rmbr}$) and non-metal ($Q_{p\text{-}rnm}$) powders. The first one is directly refined up to obtain almost pure materials ($Q_{p\text{-}rm}$). For this reason, this part is the one giving an idea of the potentially reachable profitability characterizing input elements. However, during refining a little percentage of materials is lost, because of chemical reactions (Q_{lmrp}). The second one (mainly composed by inert materials, like plastics and ceramics) is currently landfilled ($Q_{p\text{-}rnm}$). In this work, this last part was defined as the difference between the overall WPCB weight and the sum of the metal powders embedded on it.

$$Q_W = p_h * n_h * n_d \quad (1)$$
$$Q_e = Q_w * p_e/(1 - p_e) \quad (2)$$
$$Q_{hwd} = Q_e * p_{ed} \quad (3)$$
$$Q_{end} = Q_e - Q_{hwd} \quad (4)$$
$$Q_{lmpp} = lm_{pp} * Q_w \quad (5)$$
$$Q_{P-rmat} = Q_W - Q_{lmpp} \quad (6)$$
$$Q_{P-rnm} = Q_{P-rmat} * p_{rnm} \quad (7)$$
$$Q_{P-rmbr} = Q_{P-rmat} - Q_{P-rnm} \quad (8)$$
$$Q_{lmrp} = lm_{rp} * Q_{P-rmbr} \quad (9)$$
$$Q_{P-rm} = Q_{P-rmbr} - Q_{lmrp} \quad (10)$$
$$Q_{P-rmj} = Q_{P-rm} * p_{rmj} * m_u * \left(1 \Big/ \sum_{j=1}^{n_{rm}} p_{rmj} * m_u \right) \qquad \forall j = 1 \ldots n_{rm} \quad (11)$$
$$Q_{P-hrmj} = \sum_{j=1}^{n_{hrm}} Q_{P-hrmj} \quad (12)$$
$$Q_{P-srmj} = Q_{P-rmj} - Q_{P-hrmj} \quad (13)$$
$$N_W = Q_W/W_w \quad (14)$$

Table 5. Reference nomenclature of the process

Nomenclature			
j:	recycled metal	prnm:	% of not metals in recycled materials
lmpp:	lost materials in pretreatment process	Qe:	quantity of envelope
lmrp:	lost materials in refinement process	Qend:	quantity of not dangerous envelope
mu:	1 kg of WPCB	Qhwd:	quantity of hazardous waste (disassembly)
nd:	number of days	Qlmpp:	quantity of lost materials (pretreatment)
nh:	number of hours	QP-hrm,j:	quantity of hazardous recycled metal
nhrm:	number of hazardous recycled metal	Qlmrp:	quantity of lost materials (refinement)
nnrm:	number of non-recycled metals	QP-rm:	quantity of powders (recycled metals)
nrm:	number of recycled metals	QP-rmat:	quantity of powders (recycled materials)
Nw:	number of WPCBs	QP-rm,j:	quantity of powders (recycled metal j)
pe:	% of envelope	QP-rmbr:	quantity of powders (before refinement)
ped:	% of “dangerous” envelope	QP-rnm:	quantity of powders (recycled non-metals)
ph:	hourly productivity	QP-srm,j:	quantity of selling recycled metal
phwd:	% of hazardous waste (disassembly)	QW:	quantity of WPCBs
prm,j:	% of metal j in 1 kg of WPCB	Ww:	weight of WPCB

4.2.3. Recycling plant sizing

After having defined the typical phases constituting a WPCBs recycling process, the next step is the plant's capacity sizing. Given the features of a recycling process (very similar to a productive plant), the sizing activity does not takes into account only the expected level of service ((Cucchiella et al., 2015a); (Pan et al., 2015)), but the required hourly productivity. Hence, by considering the reference values reported in literature ((Zeng et al., 2015); (Li and Xu, 2010)) this parameter was defined to be equal to 0.125 t/h and 0.3 t/h (for mobile and field plants, respectively). These flows are materials (under the form of powders) flowing out from the pretreatment phase. Furthermore, by considering a working period of 240 days and 8 working hours per day (according to equation (1)), these are the overall resulting values:

- 240 t powders/year (mobile plant);
- 576 t powders/year (field plant).

These two configurations of a plant are proposed together because, within the EU-28, there are very different distributions of e-wastes from one country to another and within the same country. In some cases, a field plant is useful to recover great amounts of wastes from a specific location. In other cases, a mobile plant able to be transferred from one location to another is preferable to guarantee always a correct saturation of the plant ((Zeng et al., 2015); (Cucchiella et al., 2014b)). However, these generic dimensions present a common limit. In fact, they are related to mono-core plants (able to treat only one type of e-wastes), with a very low flexibility level.

4.3. Research methodology

Given the literature gaps reported in Section 2.1 and the detailed description of the research framework in Section 2.2, it is now possible to better comprehend the economic model at the base of this paper. Firstly, an analysis of the current economic models available in literature (and related to e-waste recycling processes) will be presented. Secondly, the basic pillars (Discounted Cash Flow (DCF) method and reference financial indexes e NPV and DPBT) taken into account for the profitability assessment will be assessed. Finally, the economic model will be described into detail, together with economic and technical input.

4.3.1. Current economic models

The main features (see Section 1) characterizing almost all of the current economic models focused on e-waste recycling processes can be listed in three points: (i) the focus on a particular phase of the process (Ghosh et al., 2015), (ii) the absence of standards in material composition of WPCBs taken into account (Wang and Gaustad, 2012), and (iii) the limited set of application fields (Wang and Xu, 2015). The focus on a particular phase of the process, even if can leave more space to different technological configurations of a recycling plant, influences the overall economic result given by the proposed model. So, from a practical point of view this can offer a limited support to industrial actors when they have to decide to invest (or not) in this type of plants. WPCBs materials composition is the most important variable influencing the profitability of the entire recycling process, as already underlined in Section 2.2. Hence, it is important to correctly characterize WPCBs to maintain the reliability of results. Finally, current studies are almost completely focused on PCBs coming from a particular set of WEEEs, or the ones re-known by the experts as the most profitable to be recovered. Hence, because of the lack in data about WPCBs from automotive scraps, this topic was rarely considered (IMDS, 2015). In practice, the previous three lacks generated a particular kind of papers, whose goals can be briefly synthetized here:

- A costs comparison of different PCBs dismantling processes (e.g. manual versus mechanical techniques) (Zeng et al., 2013);

- A costs comparison of different PCBs disassembly processes (e.g. on a specific product) (Fan et al., 2013);
- A cost comparison of different PCBs shredding þ separation processes (e.g. different technologies and plants dimensions) ((Zeng et al., 2015); (Li and Xu, 2010); (Xue et al., 2013));
- A cost comparison of different powders refining processes (e.g. through hydrometallurgical technologies) (Kamberović et al., 2011);
- An evaluation of theoretical economic models for PCBs recycling (Niu et al., 2007);
- An assessment of potential revenues coming from the recovery of entire PCBs (Wang and Gaustad, 2012).

Table 6 reports a list of economic indexes currently used within these papers. Given the current state of literature about economic models related to e-waste recycling processes, it is now possible to evidence the main differences between the proposed model and the existing ones.

Table 6. Current economic indexes used in literature

Plant size	Index	Value	Reference
0.5 kt WPCBs	Total cost	25,000 $ (manual) 50,000 $ (mechanical)	(Zeng et al., 2013)
10 kt WPCBs	Total cost	350,000 $ (mechanical) 400,000 $ (manual)	(Zeng et al., 2013)
Not specified	Net profit	1.61 € (per notebook)	(Fan et al., 2013)
0.1 t WPCBs/h	Net Profit	600 RMB	(Niu et al., 2007)
0.2 t WPCBs/h	Net Profit	1300 RMB	(Niu et al., 2007)
0.125 t WPCBs/h	Gross Profit	-83 $/t (field plant) 14 $/t (mobile plant)	(Zeng et al., 2015)
0.3 t WPCBs/h	Gross Profit	129 $/t (manual - automatic line) 256 $/t (automatic line)	(Li and Xu, 2010)
50 kg of WEEE per batch	Total revenues Payback Time	62,000 $/y (200 ppm Au) 161,000 $/y (1000 ppm Au) Not feasible (200 ppm Au) 3 y (1000 ppm Au)	(Kamberović et al., 2011)
100 kg of WEEE per batch	Total revenues Payback Time	99,000 $/y (200 ppm Au) 339,000 $/y (1000 ppm Au) Not feasible (200 ppm Au) 1 y (1000 ppm Au)	(Kamberović et al., 2011)
Not specified	Payback Time Internal rate of return	2.5 y 43%	(Xue et al., 2013)
1 t WPCBs	Potential revenues	21,500 $/t (baseline scenario) 3800-52,700 $/t (alternative scenario)	(Wang and Gaustad, 2012)

4.3.2. Discounted Cash Flow method

DCF is a well-known economic assessment method estimating the attractiveness of an investment opportunity. The standard practice is to define a vision of future events precise enough to be captured in a DCF analysis (Courtney et al., 1997). The reliability of this approach is guaranteed also by the European Commission, proposing it as reference method for the evaluation and comparison of investments (European Union regional policy, 2008). The main points characterizing the DCF method are the following:

- Only cash inflows and outflows are considered within the analysis;
- The determination of investment's cash flows is based on the incremental approach;
- The aggregation of occurring cash flows during different years requires the adoption of an appropriate discount rate.

A critical point of this method is that its reliability completely depends by the level of confidence of future cash flows.

4.3.3. Profitability indexes

Several economic indexes can be chosen to represent profitability, as evidenced in Section 1. However, net and gross profit seems to be the most common among the experts. The problem is that they do not analyse all the lifetime of an investment, but only a predefined period. Hence, the authors decided to consider other kind of indexes, as Net Present Value (NPV), Discounted Payback Time (DPBT) and Internal Rate of Return (IRR) ((Cucchiella et al., 2015a); (Chiaroni et al., 2014); (Larsson et al., 2015); (Weigel et al., 2016); (Cucchiella et al., 2015c)):

- NPV is defined as the sum of present values of individual cash flows;
- DPBT represents the number of years needed to balance cumulative discounted cash flows and initial investment;
- IRR identifies the discount rate at which the present value of all future cash flows will balance the initial investment.

However, among these three indexes only NPV and DPBT were selected, because of the poor relevance of criticisms related to them. In fact, NPV does not consider the size of the plant and DPBT ignores both instant and value of cash flows. However, these indexes provide a single result. Instead, IRR can cause conflicting answers (multiple IRR can occur) when compared to NPV in mutually exclusive investments (Brealey et al., 2011).

4.3.4. The economic model

The profitability of a recycling plant is influenced by two main variables, or materials embedded into WPCBs and plant capacity. For this reason, the set of selected scenarios evaluated in this paper are eight. They are obtained by a combination between the four WPCBs groups (Small WPCBs, Medium-small WPCBs; Medium-big WPCBs and Big WPCBs) and the two sizes of the plants (240 t/y and 576 t/y) – please see Section 2.2 for details. The economic model considered within the paper can be described with the following equations:

$$NPV = \sum_{t=0}^{n} C_t/(1+r)^t = \sum_{t=0}^{n}(I_t - O_t)/(1+r)^t \tag{15}$$

$$\sum_{t=0}^{DPBT} C_t/(1+r)^t = 0 \tag{16}$$

$$I_t = \sum_{j=1}^{n_{rm}} Q_{P-srm,j} * pl_{rm} * pr_{rm,j,t} \qquad \forall t = 1 \dots n \tag{17}$$

$$0_t = C_{lcs,t}^{2°s} + C_{lis,t}^{2°s} + C_{lcs,t}^{3°s} + C_{lcs,t}^{3°s} + C_{o,t}^{1°s} + C_{o,t}^{2°s} + C_{o,t}^{3°s} + C_{tr,t} + C_{tax,t} \quad \forall t = 1 \dots n \tag{18}$$

$$C_{inv}^{2°s} = C_{inv}^{u,2°s} * Q_W \tag{19}$$

$$C_{lcs,t}^{2°s} = C_{inv}^{2°s}/n_{debt} \qquad \forall t = 0 \dots n_{debt} - 1 \tag{20}$$

$$C_{lis,t}^{2°s} = \left(C_{inv}^{2°s} - C_{lcs,t}^{2°s}\right) * r_d \qquad \forall t = 0 \dots n_{debt} - 1 \tag{21}$$

$$C_{inv}^{3°s} = C_{inv}^{u,3°s} * Q_{P-rmbr} \tag{22}$$

$$C_{lcs,t}^{3°s} = C_{inv}^{3°s}/n_{debt} \qquad \forall t = 0 \dots n_{debt} - 1 \tag{23}$$

$$C_{lis,t}^{3°s} = \left(C_{inv}^{3°s} - C_{lcs,t}^{3s}\right) * r_d \qquad \forall t = 0 \dots n_{debt} - 1 \tag{24}$$

$$C_{o,t}^{1°s} = C_{a,t}^{1°s} + C_{d,t}^{2°s} + C_{l,t}^{1°s} \tag{25}$$

$$C_{a,t}^{1°s} = C_a^u * Q_W \tag{26}$$

$$C_{a,t+1}^{1°s} = C_{a,t}^{1°s} * (1 + inf) \qquad \forall t = 1 \dots n \tag{27}$$

$$C_{d,t}^{1°s} = C_d^u * Q_{hwd} \tag{28}$$

$$C_{d,t+1}^{1°s} = C_{d,t}^{1°s} * (1 + inf) \qquad \forall t = 1 \dots n \tag{29}$$

$$C_{l,t}^{1°s} = C_l^u * n_d * n_{op}^{1°s} \tag{30}$$

$$C_{l,t+1}^{1°s} = C_{l,t}^{1°s} * (1 + inf) \qquad \forall t = 1 \dots n \tag{31}$$

$$C_{o,t}^{2°s} = C_{cm,t}^{2°s} + C_{e,t}^{2°s} + C_{i,t}^{2°s} + C_{l,t}^{2°s} + C_{m,t}^{2°s} \tag{32}$$

$$C_{cm,t}^{2°s} = C_{cm}^u * Q_{P-rnm} \tag{33}$$

$$C_{cm,t+1}^{2°s} = C_{cm,t}^{2°s} * (1 + inf) \qquad \forall t = 1 \dots n \tag{34}$$

$$C_{e,t}^{2°s} = C_e^u * (e_u^{2°s}/p_h) * Q_W \tag{35}$$

$$C_{e,t+1}^{2°s} = C_{e,t}^{2°s} * (1 + \text{inf}) \qquad \forall t = 1 \dots n \qquad (36)$$

$$C_{i,t}^{2°s} = p_i * C_{inv}^{2°s} \qquad (37)$$

$$C_{i,t+1}^{2°s} = C_{i,t}^{2°s} * (1 + \text{inf}) \qquad \forall t = 1 \dots n \qquad (38)$$

$$C_{l,t}^{2°s} = C_l^u * n_d * n_{op}^{2°s} \qquad (39)$$

$$C_{l,t+1}^{2°s} = C_{l,t}^{2°s} * (1 + \text{inf}) \qquad \forall t = 1 \dots n \qquad (40)$$

$$C_{m,t}^{2°s} = p_m^{2°s} * C_{inv}^{2°s} \qquad (41)$$

$$C_{m,t+1}^{2°s} = C_{m,t}^{2°s} * (1 + \text{inf}) \qquad \forall t = 1 \dots n \qquad (42)$$

$$C_{o,t}^{3°s} = C_{d,t}^{3°s} + C_{e,t}^{3°s} + C_{i,t}^{3°s} + C_{l,t}^{3°s} + C_{m,t}^{3°s} + C_{rem,t}^{3°s} \qquad (43)$$

$$C_{d,t}^{3°s} = C_{cm,t}^{3°s} * Q_{P-hrm} \qquad (44)$$

$$C_{d,t+1}^{3°s} = C_{d,t}^{3°s} * (1 + \text{inf}) \qquad \forall t = 1 \dots n \qquad (45)$$

$$C_{e,t}^{3°s} = C_e^u * e_u^{3°s} * Q_{P-rmbr} \qquad (46)$$

$$C_{e,t+1}^{3°s} = C_{e,t}^{3°s} * (1 + \text{inf}) \qquad \forall t = 1 \dots n \qquad (47)$$

$$C_{i,t}^{3°s} = p_i * C_{inv}^{3°s} \qquad (48)$$

$$C_{i,t+1}^{3°s} = C_{i,t}^{3°s} * (1 + \text{inf}) \qquad \forall t = 1 \dots n \qquad (49)$$

$$C_{l,t}^{3°s} = C_l^u * n_d * n_{op}^{3°s} \qquad (50)$$

$$C_{l,t+1}^{3°s} = C_{l,t}^{3°s} * (1 + \text{inf}) \qquad \forall t = 1 \dots n \qquad (51)$$

$$C_{m,t}^{3°s} = p_m^{3°s} * C_{inv}^{3°s} \qquad (52)$$

$$C_{m,t+1}^{3°s} = C_{m,t}^{3°s} * (1 + \text{inf}) \qquad \forall t = 1 \dots n \qquad (53)$$

$$C_{rem,t}^{3°s} = C_{rem}^u * Q_{P-rmbr} \qquad (54)$$

$$C_{rem,t+1}^{3°s} = C_{rem,t}^{3°s} * (1 + \text{inf}) \qquad \forall t = 1 \dots n \qquad (55)$$

$$C_{tr} = C_{tr}^u * (Q_W + Q_e) * d_{tf} \qquad (56)$$

$$C_{tr,t+1} = C_{tr,t} * (1 + inf) \qquad \forall t = 1 \dots n \qquad (57)$$

$$C_{tax,t} = ebt_t * C^u_{tax} \qquad \forall t = 1 \dots n \qquad (58)$$

4.3.5. Economic and technical input

Economic and technical inputs are proposed in **Table 8**. The mobile plant investment cost is evaluated almost 639 kV, while the fixed plant assumed to be almost 1533 kV ((Zeng et al., 2015); (Li and Xu, 2010); (Kamberović et al., 2011); (Cucchiella et al., 2014c)). This difference evidences the presence of an economy of scale of about 29% and the investment cost is covered by third party funds. The recovered materials evaluation occurs in function of market prices historical trend per a defined period of time. By taking as reference the March 2014-March 2015 period, monthly observations were gathered from the most relevant websites dedicated on raw materials exchanges. Initial assumptions were taken from scientific literature. However, with the aim to better explain the effects of changes in values of relevant variables, a sensitivity analysis will be proposed in the next Section 5. After having defined the economic model structure (and related input values), all the financial indexes useful for the assessment of the investment will be estimated in Section 4.

4.4. Results

The economic evaluation of a project allows the easing of its application in a real context, where profitability is verified. In fact, as in waste recycling processes, this could represent not only an environmental protection action, but also an economic opportunity. As already presented in Section 3, eight scenarios are analysed in this work, and is clear that the financial feasibility is always verified (**Table 10**).

Table 7. Reference nomenclature of the model

Nomenclature			
C_a:	acquisition cost of WPCBs	C^u_{tr}:	unitary transportation cost of the plant
C^u_a:	unitary acquisition cost of WPCB	d_{tf}:	distances of transportation of the plant
C_{cm}:	conferred material cost	DPBT:	discounted payback time
C^u_{cm}:	unitary conferred material cost	$e^{2°s}_u$:	energy power (pretreatment)
C_d:	disposal cost	$e^{3°s}_u$:	energy power (refinement)
C^u_d:	unitary disposal cost	ebt:	earnings before taxes
C_e:	electric power cost	I_t:	discounted cash inflows
C^u_e:	unitary electric power cost	inf:	rate of inflation
C_i:	insurance cost	n:	lifetime of investment
C_{inv}:	total investment cost	n_{debt}:	period of loan
$C^{u,2°s}_{inv}$:	unitary investment cost (pretreatment)	$n^{1°s}_{op}$	number of operators (disassembly)
$C^{u,3°s}_{inv}$:	unitary investment cost (refinement)	$n^{2°s}_{op}$	number of operators (pretreatment)
C_l:	labour cost	$n^{3°s}_{op}$	number of operators (refinement)
C^u_l:	unitary labour cost	NPV:	net present value
C_{lcs}:	loan capital share cost	O_t:	discounted cash outflows
C_{lis}:	loan interest share cost	p_i:	% of insurance cost

C_m:	maintenance cost	$p_m^{2°s}$:	% of maintenance cost (pretreatment)
C_o:	operational cost	$p_m^{3°s}$:	% of maintenance cost (refinement)
C_{rem}:	reactant materials cost	pl_{rm}:	purity level of recycled metal
C_{rem}^{u}:	unitary reactant materials cost	pr_{rm}:	price of recycled metal (average value)
C_t:	discounted cash flow	p_{rm}^{sd}:	price of recycled metal (standard deviation)
C_{tax}:	taxes	r:	opportunity cost
C_{tax}^{u}:	unitary taxes	r_d:	interest rate on loan
C_{tr}:	transportation cost of the plant	t:	time of the cash flow
1°s: "disassembly" step ; 2°s: "pretreatment" step; 3°s: "refinement" step			

Table 8. Economic and technical input

Variable	Value	Reference	Variable	Value	Reference
C_a^u	1195 €/t	(Zeng et al., 2015)	$n_{op}^{1°s}$	1[i] 2[ii]	(Zeng et al., 2013)
C_{cm}^u	90 €/t	(Cucchiella et al., 2015c)	$n_{op}^{2°s}$	2[i] 3[ii]	(Zeng et al., 2013)
C_d^u	325 €/t	(Zeng et al., 2015)	$n_{op}^{3°s}$	2[i] 3[ii]	(Zeng et al., 2013)
C_e^u	0.11 €/kWh	(Zeng et al., 2015)	n_{rm}	Table 3	(IMDS, 2015)
$C_{inv}^{u,2°s}$	913 €/t[i] 646 €/t[ii]	((Zeng et al., 2015); (Li and Xu, 2010))	n_{rnm}	Table 3	(IMDS, 2015)
$C_{inv}^{u,3°s}$	3860 €/t[i] 2740 €/t[ii]	((Kamberović et al., 2011); (Cucchiella et al., 2014c))	p_e	70%	(IMDS, 2015)
C_l^u	150 €/d	(Ardente et al., 2014)	p_{ed}	5%	(IMDS, 2015)
C_{rem}^u	830 €/t	(Cucchiella et al., 2014c)	p_h	0.125 t/h[i] 0.3 t/h[ii]	((Zeng et al., 2015); (Li and Xu, 2010))
C_{tax}^u	36%	(Cucchiella et al., 2014c)	p_i	2%	(Cucchiella et al., 2015c)
C_{tr}^u	0.34 €/(km*t)	(Zhao and Chen, 2011)	$p_m^{2°s}$	25%	(Copani and Rosa, 2014)
$e_u^{2°s}$	50 kW[i] 141 kW[ii]	(Zeng et al., 2015)	$p_m^{3°s}$	5%	(Kamberović et al., 2011)
$e_u^{3°s}$	3900 kWh/t[i] 9500 kWh/t[ii]	(Cucchiella et al., 2014c)	p_{rnm}	Table 3	(IMDS, 2015)
d_{tf}	200 km[i] 0 km[ii]	(Cucchiella et al., 2014c)	p_{rm}	Table 3	(IMDS, 2015)
inf	2%	(Cucchiella et al., 2015c)	pl_{rm}	95%	(Reuter et al., 2013)
lm_{pp}	20%	(Reuter et al., 2013)	pr_{rm}	Table 3	((LME, 2014); (Metalprices, 2014); (Infomine, 2014))
lm_{rp}	5%	(Reuter et al., 2013)	r	5%	(Cucchiella et al., 2015c)
n	5y[i] 10 y[ii]	(Li and Xu, 2010)	r_d	4%	(Cucchiella et al., 2015c)
n_d	240 d	(Li and Xu, 2010)	W_w	3.73 g[a] 26.66 g[b] 94.99 g[c] 209.09 g[d]	(Reuter et al., 2013)
n_{debt}	5y	(Cucchiella et al., 2015c)			
n_h	8h	(Li and Xu, 2010)			
n_{hrm}	Table 3	(IMDS, 2015)			

i = mobile plant; ii = field plant.
a = Small WPCBs; b = Medium-small WPCBs; c = Medium-big WPCBs; d = Big WPCBs (average value).
Potential revenues derived by Q_{end} are not considered.
Conversion factor: 1 $ = 0.93 €

Furthermore, it is also important to underline the relevance of results. In fact, DPBT is equal to one year. This means that cash flows allow the re-entering from the investment already at the end of the first year of activity. This result is confirmed also by the work of (Kamberović et al., 2011), even if the author analysed the only metals refining phase and 1000 ppm of Au in WPCBs on average. Instead, NPV varies basing on both plant capacity and WPCBs types. From one side, NPVs reach their maximum value (495,726 V/t) with a field plant and Small WPCBs (presenting 4200 ppm of Au). From the other side, NPVs reach their minimum value (52,495 V/t) with a mobile plant and Big WPCBs (presenting 900 ppm of Au). A direct comparison with existing literature is not possible due to absence of data related to economic performances on WPCBs recycling. However, it is possible to highlight that these values are higher than the ones obtained by (Zeng et al., 2015) who considered a field and a mobile plant with a capacity of 0.125 t/h. in this case gross profits where equal to – 83 $/t and 14 $/t (see section 3.1). Furthermore, these values are also different from the ones presented by Li and Xu (2010) who considered a field plant with a capacity of 0.3 t/h and a gross profit equal to 256 $/t (see section 3.1). However, both the authors considered only the pretreatment phase and only one type of WPCBs presenting an Au content of about 5 ppm. Consequently, their revenues come mainly from the recovery of Cu. Field plants present a longer lifecycle than mobile plants (10 years out of 5 years). This aspect, starting from equal gross profits, explains the reaching of greater NPVs. Furthermore, from one side economies of scale are exploited. Instead, from the other side, the higher number of recycled WPCBs allows to recover a higher amount of Au (**Table 11**).

Table 9. Characterization of materials embedded into ECUs

Materials	Small WPCBs	Medium-small WPCBs	Medium-big WPCBs	Big WPCBs	Generic WPCBs	Generic WPCBs
n_{rm}	p_{rm} (%)	p_{rm} (%)	p_{rm} (%)	p_{rm} (%)	p_{rm} (€/kg)	p_{rm}^{sd} (€/kg)
Revenues						
Silver (Ag)	0.09	0	0	0	480	45
Gold (Au) (*)	0.42	0.20	0.24	0.09	32,500	4500
Copper (Cu)	18.84	24.19	14.52	16.30	5.13	1.2
Iron (Fe)	0.18	0.17	0.19	0.10	0.064	0.02
Nickel (Ni)	0.69	0.43	1.13	0.89	12	1.2
Tin (Sn)	1.81	1.46	1.23	1.56	16	2.1
Tantalum (Ta)	0.08	0	0	0	148	25
Lead (Pb)	0.71	1.13	0.40	0.27	2.1	0.4
Costs						
n_{rnm}	p_{nrm} (%)	p_{nrm} (%)	p_{nrm} (%)	p_{nrm} (%)		
Epoxy resin	15.81	3.73	12.73	13.60		
Glass fibre	19.34	35.56	35.53	36.72		
Others (**)	2.63	2.63	1.23	1.47		
Delta mat (***)	39.40	30.50	32.80	29.00		

(*) 0.42% of Au is equal to 4200 ppm, or 4200 g of Au in 1 ton of WPCBs.

(**) Others are all the materials (metals and non-metals) cited by the IMDS database, but not considered in this work.

(***) Delta mat (metals and non-metals) is the difference between the overall mass of a WPCB and the sum of all the considered materials embedded in a WPCB. It represents the amount of materials not considered by the IMDS database.

Table 10. Economic indexes – baseline scenario

Index	Small WPCBs	Medium-small WPCBs	Medium-big WPCBs	Big WPCBs
Mobile plant (240 tons of powders/year)				
DPBT (y)	1	1	1	1
NPV (K€)	66,304	30,966	36,639	12,599
NPV/Q_W (€/t)	276,267	129,026	152,662	152,495
Field plant (576 tons of powders/year)				
DPBT (y)	1	1	1	1
NPV (K€)	285,538	134,271	158,562	55,656
NPV/Q_W (€/t)	495,726	233,110	275,280	96,626

However, as explained in other papers ((Zeng et al., 2015); (Kamberović et al., 2011)) mobile facilities applications represent an ideal solution for small countries or cities and, at the same time, they play an important role in collecting wastes. A scenario with a mobile plant treating Small WPCBs presents a higher NPV (66,304 kV) than a scenario with a field plant treating Big WPCBs (55,656 kV). This means that the percentage of Au embedded into WPCBs has greater effect than dimensions. However, it is of utmost importance to consider that these results were obtained by hypothesizing a full saturation of plants. To this aim, in the next section of the paper, a sensitivity analysis will be implemented on both non-saturated plants. Instead, there are no market data related to WPCBs original applications and this pushed the authors to consider mono-core plants. However, future research objectives are the evaluation of multi-cores plants, able to treat all types of WPCBs independently from their dimensions and primary industrial application. To this aim, being the Au content a critical variable, during the sensitivity analysis also this aspect will be considered. Furthermore, scientific literature and past pilot plants experiences evidenced as recycling plants flexibility plays a relevant role (Rocchetti et al., 2013). To this aim, potential revenues coming from single waste streams can be used as a benchmarking factor (Cucchiella et al., 2015b).

Table 11. Number of WPCBs treated and quantities of recycled gold

Index	Small WPCBs	Medium-small WPCBs	Medium-big WPCBs	Big WPCBs
Mobile plant (240 tons of powders/year)				
N_W (1000*unit)	64,343	9002	2527	1148
$Q_{P-rm,Au}$ (kg)	791	380	448	167
Field plant (576 tons of powders/year)				
N_W (1000*unit)	154,424	21,605	6064	2755
$Q_{P-rm,Au}$ (kg)	1898	913	1075	400

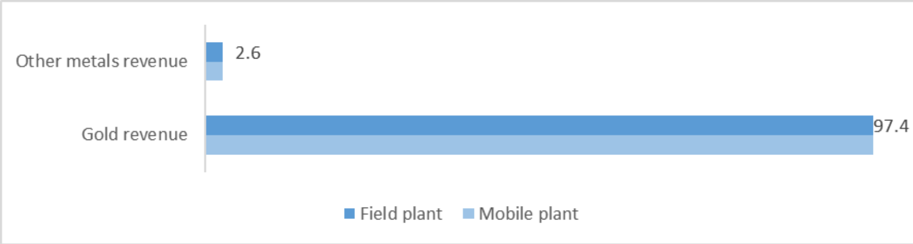

Figure 21. Plant's revenues distribution – average values

Finally, for what concerns the Au relevance among revenues items, data showed in **Figure 21** are significant: by considering the four types of WPCBs these are equal to 97.7% on average both in mobile and field plants. Another paper fixed the incidence of Au on potential revenues in 71%, by considering a value of about 15,200 $/t, but with a range going from a minimum level of 2500 $/t up to a maximum level of 40,000 $/t. Clearly, the type of WPCBs considered influenced these results. This is why the authors of this paper decided to take into account data coming from the IMDS database, allowing the management of a more significant sample. Instead, the costs distribution analysis shows as the operational costs are equal to 95.1% for a field plant and 89.1% for a mobile plant. These results are coherent with respect of what proposed by other works ((Zeng et al., 2015); (Li and Xu, 2010); (Kamberović et al., 2011); (Cucchiella et al., 2015c)). The most relevant item is represented by WPCBs purchasing both for field and mobile plants (47,5% and 37.5%, respectively). This value is followed by labour costs (19.9% and 23.6%, respectively). Finally, transport costs are equal to 7.1% in the mobile plant (**Figure 22**). In order to strengthen the obtained results, a sensitivity analysis oriented to alternative scenarios (if compared to what presented before) is implemented in the next section.

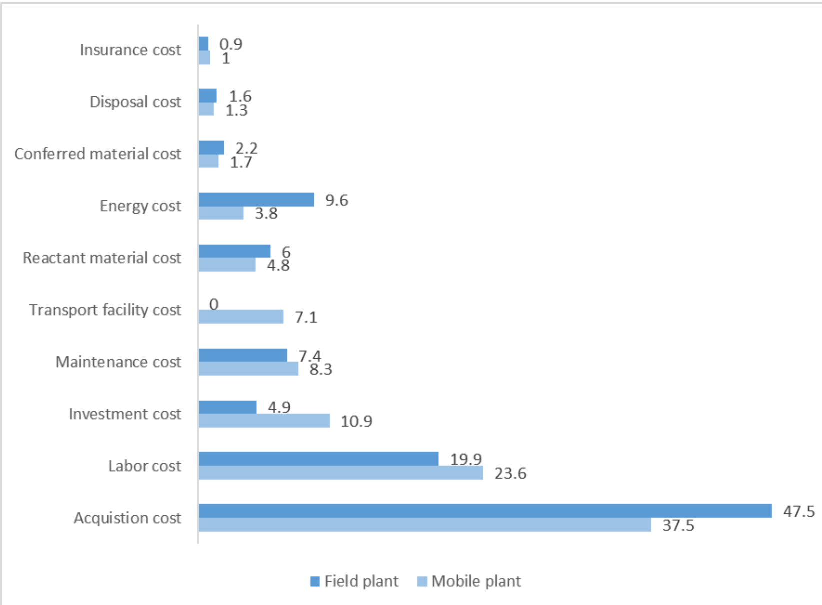

Figure 22. Plant's costs distribution – average values

4.5. Sensitivity analysis

NPV results are based on assumptions of a set of input variables. However, compared to the baseline scenario, the critical variables can record changes with respect to initial estimations (Cucchiella et al., 2015c). Basing on what obtained in Section 4, critical variables are the ones that, more than others, have an influence on revenues and costs. From the revenues point of view, it has been demonstrated that they mainly depend on the recovery of Au. Hence, three are the variables determining results: (i) the Au content, as percentage of the WPCB total weight (it was already analysed, in fact four categories of WPCBs were evaluated in this paper); (ii) the Au market price, varying from an optimistic and pessimistic scenario where the assumed value is equal to 37,000 V/ kg and 28,000 V/kg (this range is equal to its standard deviation) – see **Table 10**; (iii) the final purity level, varying from 60% up to 90%, under the common hypothesis of 95% of the literature. From the costs point of view, the most relevant item is represented by WPCBs purchasing cost. As done for revenues, also in this case an optimistic and pessimistic scenarios are assessed, where costs vary from 1000 V/t up to 1400 V/t (or an offset of about 200 V/t from the base value). Furthermore, in accord to what previously presented, is important to evaluate what happens when plants are not fully saturated. In this case, investment costs are unchanged, but operational incomes will vary. In particular, a lower amount of WPCBs in input represents a lower hourly productivity. To this aim, five pessimistic scenarios are assessed, with saturation levels going from 50% up to 90%.

Table 12. Sensitivity analysis – mobile plant

Variable	Value	Small WPCBs		Medium-small WPCBs		Medium-big WPCBs		Big WPCBs	
		NPV (k€)	Δ%	NPV (k€)	Δ%	NPV (k€)	Δ%	NPV (k€)	Δ%
pr_{Au} (€/kg)	37,000	75,670	14.1	35,472	14.6	41,944	14.5	14,571	15.7
	28,000	56,938	-14.1	26,460	-14.5	31,334	-14.5	10,627	-15.7
pl_{Au} (%)	90	62,744	-5.4	29,254	-5.5	34,622	-5.5	11,253	-10.7
	80	55,624	-16.1	25,828	-16.6	30,589	-16.5	9907	-21.4
	70	48,503	-26.8	22,403	-27.7	26,556	-27.5	8562	-32.0
	60	41,383	-37.6	18,977	-38.7	22,523	-38.5	7216	-42.7
C_a^u (€/t)	1400	66,160	-0.2	30,822	-0.5	36,495	-0.4	5870	-53.4
	1000	66,441	0.2	31,104	0.4	36,776	0.4	12,736	1.1
Q_W (t)	216	59,588	-10.1	27,784	-10.3	32,889	-10.2	12,376	-1.8
	192	52,872	-20.3	24,601	-20.6	29,140	-20.5	10,888	-13.6
	168	52,872	-30.4	21,419	-30.8	25,390	-30.7	9399	-25.4
	144	39,439	-40.5	18,237	-41.1	21,640	-40.9	7911	-37.2
	120	32,723	-50.6	15,054	-51.4	17,891	-51.2	6422	-49.0
r (%)	4	68,180	2.8	31,844	2.8	37,677	2.8	12,957	2.8
	6	64,508	-2.7	30,126	-2.7	35,646	-2.7	12,256	-2.7

Table 13. Sensitivity analysis – field plant

Variable	Value	Small WPCBs		Medium-small WPCBs		Medium-big WPCBs		Big WPCBs	
		NPV (k€)	Δ%	NPV (k€)	Δ%	NPV (k€)	Δ%	NPV (k€)	Δ%
pr_{Au} (€/kg)	37,000	325,629	14.0	153,558	14.4	181,271	14.3	64,099	15.2
	28,000	245,447	-14.0	114,984	-14.4	135,852	-14.3	47,214	-15.2
pl_{Au} (%)	90	270,299	-5.3	126,940	-5.5	149,929	-5.4	52,447	-5.8
	80	239,821	-16.0	112,277	-16.4	132,665	-16.3	46,029	-17.3
	70	209,342	-26.7	97,615	-27.3	115,401	-27.2	39,611	-28.8
	60	178,864	-37.4	82,952	-38.2	98,137	-38.1	33,193	-40.4
C_a^u (€/t)	1400	284,892	-0.2	133,625	-0.5	157,915	-0.4	55,010	-1.2
	1000	286,153	0.2	134,886	0.5	159,177	0.4	56,271	1.1
Q_W (t)	518	256,545	-10.2	120,510	-10.2	142,330	-10.2	49,812	-10.5
	461	228,053	-20.1	106,987	-20.3	126,378	-20.3	44,068	-20.8
	403	199,060	-30.3	93,226	-30.6	110,147	-30.5	38,223	-31.3
	346	170,567	-40.3	79,702	-40.6	94,195	-40.6	32,479	-41.6
	288	141,575	-50.4	65,941	-50.9	77,964	-50.8	26,634	-52.1
r (%)	4	299,933	5.0	141,044	5.0	166,556	5.0	58,465	5.0
	6	272,161	-4.7	127,978	-4.7	151,132	-4.7	53,047	-4.7

For example, considering the mobile plant, 90% of 240 t/h is equal to 216 t/h (**Table 12**). Instead, by considering the field plant, 90% of 576 t/h is equal to 518 t/h (**Table 13**). Finally, the last variable considered is the opportunity cost, able to evaluate the money value in different periods. This is a key parameter of the DCF method. Even in this case, an optimistic and pessimistic scenarios are assessed, with values varying from 4% up to 6%. The financial profitability is verified in all the one hundred twenty alternative scenarios. WPCBs purchasing and opportunity costs variations are not so relevant. Instead, the most significant offsets come from a possible low level of wastes in input. In fact, whereas both mobile and field plants have to work with a saturation of 50% (or treating 120 kt and 288 kt, respectively), NPVs are reduced of about 50%. Plants flexibility could allow the treatment of other types of wastes, but this could reduce expected profits. Basing on what evidenced by the literature, WPCBs are valuable components embedded into WEEEs. The IMDS demonstrated as automotive WPCBs are even valuable items, and their Au content could be higher than in WEEEs. Furthermore, the wide Au market price variation could determine critical offsets. However, results e even in less advantageous situations e offer relevant economic opportunities. Hence, further additional costs needed to obtain a higher Au purity level are easily compensated. Results proposed by this paper clearly define the sustainability of these recycling plants from an economic perspective. A global overview on the economic impact related to the recovery of these wastes within the European market is described in the next section.

4.6. Discussion

The aim of this section is to support the quantification of potential revenues coming from the correct management of e-wastes coming from automotive scraps and try to analyse their expected trends in the next 15 years. To do that, the first data required was the overall amount of expected ELVs generated from 2015 up to 2030. These data, together with related trends, were gathered directly from the literature ((Eurostat, 2015b); (Andersen et al., 2008)). The selection of this source, instead of others, was given by the fact that this is the official estimation of ELVs volumes also considered by the EU commission during the implementation of the current ELV Directive. The second step was the distinction between two ELV categories, or premature and natural ELVs. Premature ELVs, from one side, are almost new cars reaching their end of life prematurely, generally because of a serious accident. Representing almost 20% of total ELVs volumes generated annually ((Ferrão and Amaral, 2006); (Hiratsuka et al., 2014); (Zhou and Dai, 2012)), these cars are almost destroyed and there are very few chances to recover some components. Hence, the hypothesis done by the authors within this paper was the recycling of the total amount of volumes. Natural ELVs, from the opposite side, are cars reaching their end of life naturally, generally for ageing reasons, and represent the 80% of the total amount of annual ELVs volumes ((Ferrão and Amaral, 2006); (Hiratsuka et al., 2014); (Morselli et al., 2010); (Kanari et al., 2003)). This way, they represent a good source of second-hand spare parts or for remanufacturing scopes (remanufactured volumes are estimated in almost 20% - 30% of the overall volume of ELVs). However, for the purpose of this paper, expected remanufactured volumes are not considered in calculations. So, the initial amount of ELVs in terms of number of vehicles was translated in terms of million tons to be potentially treated and, then, divided between premature and natural ELVs amounts. The average ELV mass was defined in about 1.16 tons ((Zorpas and Inglezakis, 2012); (Andersen et al., 2008); (Vermeulen et al., 2011)). Third step was the definition of the average PCBs mass (in percentage) out of the total ELV mass. This index was defined through a series of phases. A direct observation of about 500 automotive PCBs mass coming from the IMDS database shown that, basing on their function, weights can vary from 0.2 g (e.g. in door controls or cooling fans) up to almost 500 g (e.g. in ECUs, instrument panels or navigation and entertainment systems). The mean weight was defined in 85 g per WPCB. Given that, in a medium car, there are 15 mechatronic components on average (Kripli et al., 2010) and that each of them embeds at least one PCB (Freiberger et al., 2012), this indicates a total weight, in terms of electronic components, of about 1.275 kg per car. Given an average ELV mass of about 1.16 tons ((Zorpas and Inglezakis, 2012); (Andersen et al., 2008); (Vermeulen et al., 2011)), the ratio was established in 0.11%, on average. This ratio was used to quantify the annual generation of WPCBs coming from both premature and natural ELVs. Finally, it was possible to predict (with logical approximations) the expected revenues (in a min-max range) coming from the correct management of these amounts of automotive WPCBs. The following **Table 14** reports all these data.

Table 14. Estimates of generated ECUs volumes in EU25 from natural and premature ELVs – Source: (Eurostat, 2015b); (Ferrão and Amaral, 2006); (Hiratsuka et al., 2014); (Vermeulen et al., 2011); self-made analysis.

	2015	2020	2030
EU ELV projected number (Mvehicles)	13.3	14.6	16.8
EU ELV expected mean weight (tons)	1.16	1.16	1.16
EU premature ELVs annual generation (Mtons)	3.09	3.39	3.90
EU natural ELVs annual generation (Mtons)	12.34	13.55	15.59
EU total ELVs annual generation (Mtons)	15.43	16.94	19.49
EU WPCBs expected mean weight (kg)	1.275	1.275	1.275
EU premature WPCBs annual generation (ktons)	3.39	3.73	4.29
EU natural WPCBs annual generation (ktons)	13.58	14.90	17.15
EU total WPCBs annual generation (ktons)	16.97	18.63	21.44
EU total WPCBs expected NPVs - min values (M€)	891	978	1125
EU total WPCBs expected NPVs - max values (M€)	8412	9235	10628

For example, the calculation of the results reported in **Table 14** and related to 2015 was obtained as follows:

13.3 [Mvehicles]*1.16 [t/vehicle] = 15.43 [Mtons]
15.43 [Mtons]*20% = 3.09 [Mtons from premature ELVs]
15.43 [Mtons]*80% = 12.34 [Mtons from premature ELVs]
(3.09 [Mtons]*0:11%)+(12.34 [Mtons]*0:11%)= 3.39 [Ktons]+13.58 [Ktons]= 16.97 [Ktons]
16.97 [Ktons]*52,495 €/tons = 891 M€
16.97 [Ktons]*495,726 €/tons = 8412 M€

Table 14, in the last two rows, reports the potential dimension of the automotive WPCBs recycling market. Values are impressive, going from 891 million € up to 1125 billion € as minimum values, and refer to the base scenario presented in Section 4. Maximum levels are even more interesting, going from 8412 billion € up to 10628 billion €. These numbers, even if theoretical, demonstrate the utmost importance of automotive WPCBs management and the amount of profits that could be potentially achieved. Trying to strengthen the reliability of these estimates, the application of more accurate prediction models (e.g. Artificial Neural Networks (ANN)) ((Chau and Muttil, 2007); (Muttil and Chau, 2007)) could be an option when complex issues on sustainable topics has to be solved ((Zhao et al., 2006); (Wu and Chau, 2006); (Xie et al., 2006); (Muttil and Chau, 2006)). Without any doubt, this market could become an interesting business for many companies involved in closed-loop supply chains.

4.7. Conclusions

End of Life Vehicles are one of the most important sources of secondary raw materials. However, studies demonstrating the embedded value in their electronic systems are quite rare. Even if, from one side, this issue gives space to a potentially new research stream, from another side it represents a big limitation in terms of available data. Hence, this paper suffers of a lack of proved information and the entire economic and sensitivity analysis are done starting from both expected values and data coming from similar sectors. Interesting improvements of this work could be the assessment of environmental impacts of the recycling process, analysis of different configurations of closed-loop supply chains, and definition of corrective policies to current ELV directives. Future directions in this research stream are expected to be the assessment of recycling issues coming from the treatment of hybrid and full-electric cars or auto-guided vehicles. The intention of the paper was not only to partially fill in this literature gap, but also propose something of interesting from a practical point of view. Hence an economic model evaluating potential revenues and costs coming from the recycling of scrap automotive electronics was implemented and described into detail. The obtained indexes (e.g. NPV and DPBT) demonstrated the validity of investments in two different types of plants (mobile and field ones) and for all the four types of WPCBs considered. Economic values obtained from the model are so high, and different from common values available in literature, because of the relevant presence of Au in automotive WPCBs. A sensitivity analysis done on critical variables (e.g. plant saturation level, Au content, Au market price, Au final purity level, WPCBs purchasing cost and opportunity cost) allowed to test the robustness of theoretical evaluations. Finally, the matching of predicted ELVs volumes in the next fifteen years and expected NPV allowed to define the potential dimension of a market dedicated to the treatment of these wastes. These results could be useful for all the actors involved, with different roles, in closed-loop supply chains, as governments, recyclers, OEMs, consumers and other stakeholders, constituting the starting point for the definition of a new kind of circular economy.

Post-publication potential improvements

Considering the point of view of a novice reader, the previous paper is not simple to comprehend. The high level of detail presented asks to a generic reader a high level of knowledge in the field that, usually, is hard to reach when we are coming from other research areas, especially from theoretical ones. This way, the structure of the paper appears to be a little confusing and the final objective not always clear. Also in this case (like in the previous section), the availability of additional space for putting other tables and graphs could have supported a better exposition of the topic, together with more details on the backgrounding methods adopted during calculations. For example, some additional details could have been given about the prediction model used to provide future estimations of ELV volumes and conditions of applicability of alternative ones. Without any doubt, several other deep learning models could be tested and may bring further interesting results.

5. Paper #3: Improving end of life vehicle's management practices: An economic assessment through system dynamics

*Section 5 tries to expand what presented in Section 4, with a broader view. Here, the focus is not only the overall economic benefit coming from the recovery of scrap automotive electronic components, but the impact of these additional processes on the current economic performance obtained by two of the main actors involved in ELV recovery chains. In a logical term, Section 5 takes advantage of what quantified in Section 4 and assesses through simulation the impact of these activities (see **Figure 23**). Some graphs at the end of Section 5 allow to evidence how much could be those differences. This section reports integrally what can be found within the paper titled "Improving end of life vehicle's management practices: An economic assessment through system dynamics", edited by Journal of Cleaner Production (see Section 7 for details).*

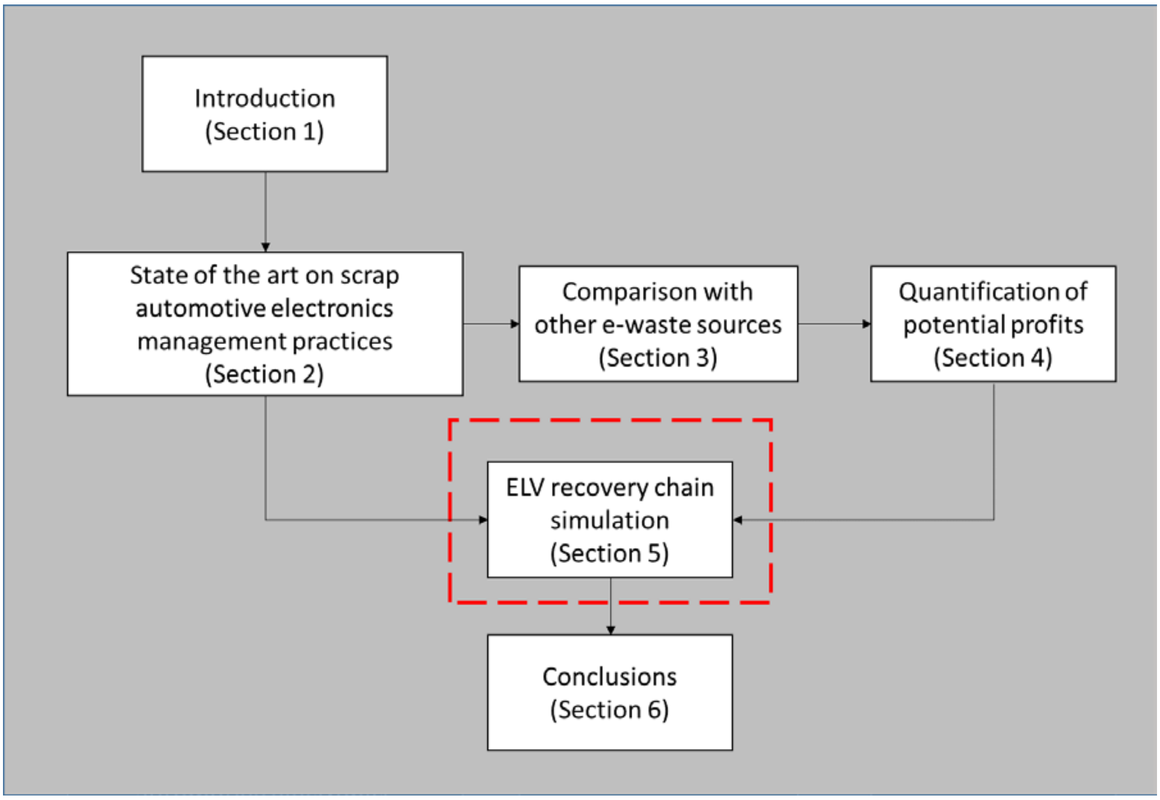

Figure 23. Relation of Section 5 with the overall structure

5.1. Introduction

The previous two papers evidenced the importance of a correct recovery of wasted automotive electronic components, both in qualitative and quantitative terms. They underlined current lacks in literature and unsustainable practices commonly followed by ELV recovery chains. However, all the quantifications were done without considering the current mechanisms driving any kind of ELV recovery chains, especially in economic terms. Even if the ELV recovery chain is an already established industry in Europe, its management strategies remained the same since many decades, completely based on raw materials market prices (especially related to ferrous metals) ((Kumar and Yamaoka, 2007); (Amaral et al., 2006)). In the last decades, this way of doing exposed national ELV recovery chains to several market risks, like unpredictable fluctuations of raw materials market prices, uncontrolled illegal transfer of vehicles among nations and the unavoidable evolution of ELVs' materials mix ((Jain et al., 2015)). Trying to control – and partially limit – this exposition to materials market risks, the experts proposed several interesting works about new methods for predicting ELV composition and materials flows ((Soo et al., 2015)), comparing different waste management strategies ((Asif et al., 2010)), mapping ELV reverse logistic chains ((Demirel et al., 2016)) and assessing the effect of new environmental policies ((Chen et al., 2015); (Wang et al., 2014)). This way, many simulation approaches (e.g. agent-based simulation, discrete event analysis and cost-benefit analysis) were adopted. Even if several contributions were published on these themes, the literature gap is still present, as underlined in a previous work (e.g. (Rosa and Terzi, 2016)). Generally, all of these contributions were focused on either a specific part of the ELV recovery chain or a specific issue affecting this industrial context, without having an overall perspective on the ELV recovery chain health and sustainability needs. During the last decades, these issues were partially assessed by other authors ((Keivanpour et al., 2017); (Keivanpour et al., 2013a); (Lehr et al., 2013); (Kibira and Jain, 2011); (Gu and Gao, 2011); (Coates and Rahimifard, 2008); (Vlachos et al., 2007); (Huang and Wang, 2007); (Amaral et al., 2006)). However, all these works refer to the methodology presented by ((Zamudio-Ramirez, 1996)), but trying to propose new elements useful in the current context. Only the original work assessed completely a national ELV recovery chain, trying to comprehend the set of internal drivers influencing economic performances of the involved actors. Considering all these elements, the main aim of this paper is to adopt the original SD model of (Zamudio-Ramirez, 1996) – once updated and upgraded – to the Italian context, by assessing and comparing current economic performances of the Italian ELV recovery chain to the ones potentially reachable through the management of automotive electronics.

5.2. Research framework

In general terms, ELVs are those vehicles reaching the end of their useful life because of either an extreme accident or obsolescence (e.g. (Vermeulen et al., 2011)). The first ones are commonly called premature ELVs. The second ones are commonly called natural ELVs. However, whatever their origin, they end all to be managed by the same reverse logistic chain, being it legal or not. Within the whole paper only the official ELV recovery chain will be considered. In the following sections a generic description of the ELV recovery chain logic, the management of automotive electronic components and the relation between SD models and ELV management will be presented.

5.2.1. A generic ELV recovery chain

A typical ELV recovery process is reported in **Figure 24**, under the form of an IDEF0 model. ELVs 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, ferrous metals (about 65% of the average mass) ((Hu and Wen, 2015)) are directly reintroduced into the automotive supply chain (as input material for foundries). Non-metals (generally named Automotive Shredder Residue (ASR) and constituting about the 25% of the average mass) are currently landfilled or used as fuel for energy generation ((Ni and Chen, 2014)). Finally, non-ferrous metals (about the 5% of the average mass) – depending on setup parameters of the specific treatment plant – becomes impurities of both the ferrous and non-metal fractions.

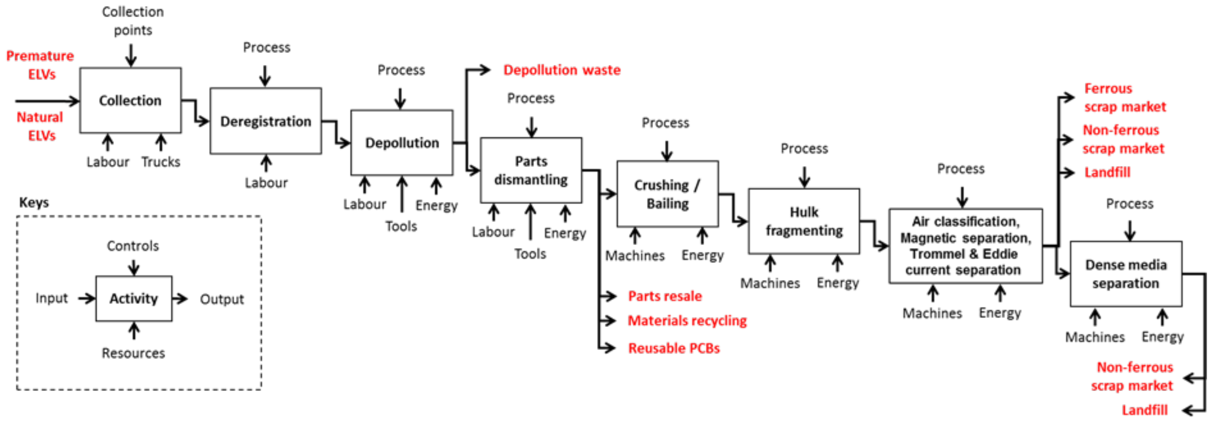

Figure 24. The current recovery process of ELVs – (Adapted from (Vermeulen et al., 2011))

However, the entire recovery process is based on technologies developed more than fifty years ago. The experts proposed many innovative procedures during the last decades [(Tian et al., 2015)]. However, the focus is still on alternative – or better – ways to increase recovery percentages of the automotive shredder residue (ASR) – or the remaining mass of a car after shredding – that is currently incinerated or landfilled [(Zorpas and Inglezakis, 2012); (Vermeulen et al., 2011)].

5.2.2. Scrap automotive electronics management

ECUs are among the most valuable electronic devices embedded in modern vehicles. They are able to perform the reading of signals coming from sensors embedded in a car, and control the behaviour of many sub-systems, as engine, air conditioning and infotainment systems and safety devices ((Wang and Chen, 2011)). The current amount of electronic components present in cars is impressive, both in numbers and in impact on costs. A modern medium-sized car can embed up to 15 electronic systems on average ((Kripli et al., 2010)) and luxury cars can reach up to 50 among microcomputers and electronic components ((Freiberger et al., 2012)). In terms of production costs, a statistic of the Bayerische Motoren Werke Corporation described that these systems can account for more than 30% of total vehicle cost, reaching more than 50% in luxury cars ((L. Wang and Chen, 2013)). More precise data on the economic value embedded in scrap automotive electronics are also present in literature ((Cucchiella et al., 2016b)). For example, referring to 2015 data, expected annual generated volumes in the only Europe were estimated in 186.5 kilo tons, with an expected NPV going from a minimum of 891 million € up to a maximum of 8,4 billion €. However, current ELV directives (based on weighting principles) seems to do not adequately take into account the management of these types of e-wastes ((Rosa and Terzi, 2016)). Hence, there are no benefits for the actors involved in the automotive reverse logistic chain to invest in dedicated recovery centres ((Cucchiella et al., 2016a)).

5.2.3. SD models and ELV management

The use of SD models to assess different features related to the ELV recovery chain is not a new idea. The scientific literature presents several works going into this direction, under the form of both journal and conference papers. The reason of the adoption of SD for the assessment of this particular research field must be searched within the drivers pushing the selection of any kind of simulation model, or competitive priorities and decision categories (e.g. (Rondini et al., 2017)). The first ones can be seen as key performance indicators, representative of the specific industrial context taken into account, that must be considered during the selection of different simulation methods. Within the context of this thesis, competitive priorities can be classified as follows:

- Need for qualitative and quantitative information (given the scarcity of data in literature);
- Need for a holistic/strategic vision of the context (high level of abstraction);
- Need for gathering long-term effects of decisions (enabling selection of future strategies).

The second ones can be seen as a set of intrinsic benefits related to each simulation method taken into account during selection. Within the context of this thesis, decision categories can be classified as follows:

- Develop long-term politics;
- Manage qualitative and quantitative data together;
- Support strategic decision-making processes;
- Assess non-linear behaviours of complex systems.

Considering all these elements together, it is possible to comprehend why SD was selected by the authors since many years as the reference tool to simulate ELV recovery chains around the world. The first and, maybe, most significant example is given by the work of (Zamudio-Ramirez, 1996). Even if under the form of a thesis (never published), this work assessed the economic performance of the North American automotive recycling chain in 1996, going to evaluate the effects of different policies regarding the materials composition of cars on the final profits of both dismantlers and shredders. From a purely economic view, this is the only example of a SD model focused on the comprehension of the set of internal drivers influencing transactions among all the involved actors and their reciprocal influence on others economic performances. Furthermore, this work was also the first one completely based on the System Dynamics (SD) approach. SD enabled a full comprehension of specific context's dynamics and a direct comparison of predictable future scenarios, especially considering different ELV materials mixes. Subsequently, other SD models were developed in the last decades.

However, their focuses were extremely different. In chronological terms, an example is given by (Amaral et al., 2006). This paper makes use of a system dynamics model, applied to the Portuguese ELV-processing infrastructure, evaluating how current practices under different recycling strategies depend on both the recycled materials market and the car's composition. The following year (Kumar and Yamaoka, 2007) presented another interesting work. Here, different EoL strategies are examined and applied to the Japanese automotive sector. However, the focus is on the effect of national policies (in terms of import and export of cars) on the national ELV recovery chain. Other authors (e.g. (Asif et al., 2010)) assessed through a SD model a new concept – based on remanufacturing principles – to be used in the product realization process for ensuring optimum useable life of products (or parts of them) and enabling multiple lifecycles. Another work (Kibira and Jain, 2011) evaluated the impact of hybrid and electric vehicles on dismantler and shredder profits. The results confirm that a technological development for the recovery of alternative materials than ferrous metals is needed to counteract the continuous evolution of materials embedded into cars. A more detailed assessment of the ELV recovery chain is done by (El Halabi et al., 2012). In this case, the paper presents an approach to extract factors and causal influences from interview data with stakeholders of the Australian automotive recycling system during the SD model conceptualisation stage. Subsequently, they tried also to explore the root causes of the operational challenges facing the automotive recycling business ((El Halabi and Doolan, 2013a); (El Halabi and Doolan, 2013b)). Other authors (Farel et al., 2013) proposed a model investigating the potential cost and benefit of ELV glazing recycling for all of the value-chain stakeholders, and for the network as a whole. The issue related to the involvement of governments in the ELV recovery chain is assessed also by (Wang et al., 2014) with the aim to explore the impact of subsidy policies on the development of the recycling and remanufacturing industry in China using the system dynamics methodology and by simulating the Chinese auto parts industry. The most recent contributions about the application of the SD methodology to the ELV recovery chain are recent. From one side, (Chen et al., 2015) propose a dynamic modelling and cost-benefit analysis investigating how polices, including government subsidies, a value-added tax and a deposit-refund system, may affect the recycling of end-of-life passenger cars in China. From another side, (Soo et al., 2015) present a dynamic hypothesis illustrating the time effect on life cycle analysis of a car for investigating challenges associated to the materials recovery efficiency. Given the interesting results and the strong research affinity with (Zamudio-Ramirez, 1996) work, the same SD model was taken into account as reference point by this paper. The original SD model was updated and upgraded, before applying it to the Italian context. The updating process interested values of constant inputs of the original SD model, substituted with Italian data coming from both field interviews and the scientific literature.

5.3. Research methodology

The SD methodology is adopted in this study for evaluating the economic performance of the Italian ELV recovery chain. Vensim® Professional was the software adopted within this work for the development of the whole SD model, its simulation, validation and optimization. This way, the economic performance of the Italian ELV recovery chain can be fully assessed with a unique decision-support tool. Vensim® Professional was selected as the best solution for doing that because it is the most commonly used software dedicated to SD models design and simulation (e.g. (Sterman, 2000)). Many SD models present in literature – and related to very different contexts (e.g. (Asif et al., 2015); (Dong et al., 2012); (Yuan et al., 2011)) – were developed with this software.

5.3.1. SD mode assumptions

The model taken into account within this paper assumes that the Italian ELV recovery chain is rep-resented by only one automaker, one dismantler and one shredder. This assumption is the same taken into account by (Zamudio-Ramirez, 1996). Another hypothesis is related to the vehicle's materials composition. ELVs are considered to be made only of ferrous metals (steel), nonferrous metals (aluminium, magnesium and copper) and plastics. Other materials are not taken into account because of their low amounts in comparison to the overall mass of a generic car. Automakers are assumed to be able to decide about the materials composition of platforms under development. This way, cars already in the market are not influenced by these changes. Dismantlers are hypothesised to make revenues from two sides, or the selling of spare parts on the secondary market and car hulks to shredders. However, within this paper dismantler's economic performances are considered only in terms of car hulks selling. Finally, shredders are assumed to make money only through the direct selling of ferrous and non-ferrous scrap metals to the secondary raw materials market. External variables are represented by both the Gross Domestic Product (GDP) of the reference nation and materials market prices. GDP is considered to be strongly correlated to car sales and, hence, to the demand of steel. This way, it is indirectly correlated also to the value of car hulks (and related subsystems). Materials market prices heavily influence the expected revenues of shredders. In addition, some cost items are also considered to be exogenous, like the labour and land-filling costs. The whole number of cars on the road is subdivided into seven groups (or cohorts), basing on their age. The considered timeframe goes from 2015 back to 1994. Cars with an age with-in 0 and 9 years are considered as "new cars". Instead, cars from 10 to 21 years are considered as "old cars". New cars (representing approximately 20% of the total) are the preferred source of spare parts for the secondary market. These parts are generally sold at 50% of the new part price. Old cars are processed only for the material value they embed.

5.3.2. Conceptual framework

The conceptual framework of the SD model is represented in **Figure 25**.

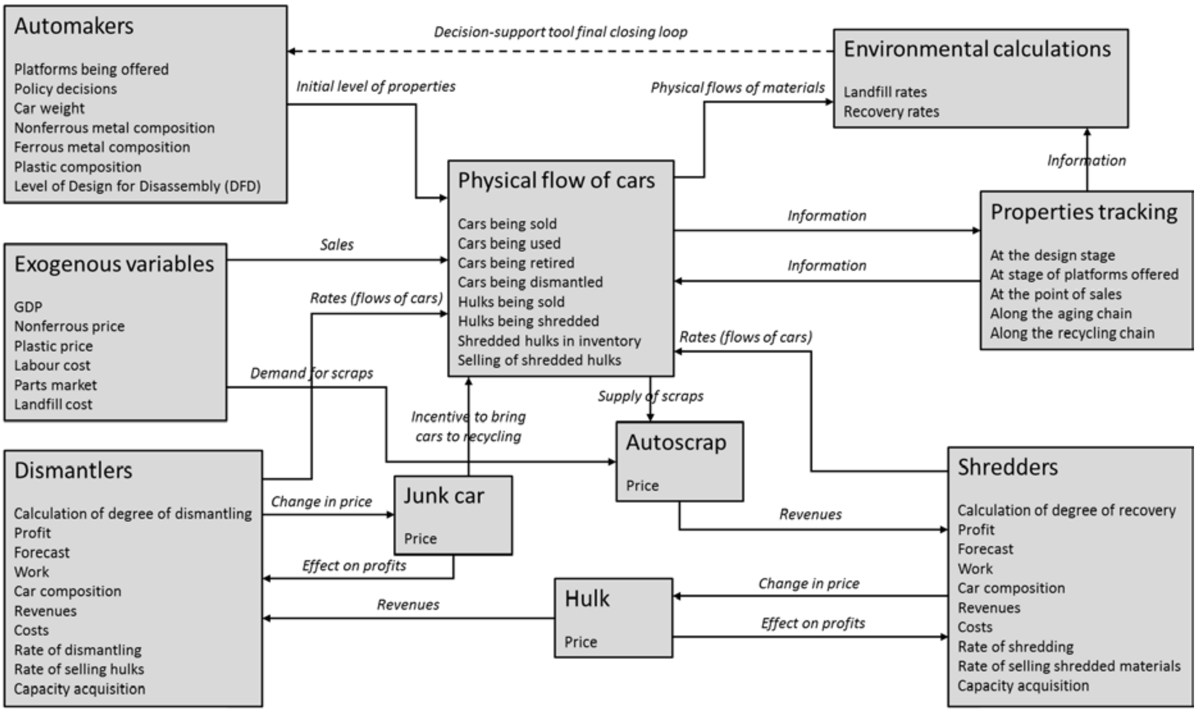

Figure 25. Conceptual framework of the decision-support tool (Adapted from (Zamudio-Ramirez, 1996))

The conceptual framework can be seen as a group of ten elements, three of which relate on the prices of junk cars, car hulks and metal scraps.

The first element (titled "Automakers") represents the automaker's point of view. From its perspective, automakers can influence the ELV recovery chain through several decisions on type, number and composition of new cars launched into the market. Basing on the materials content and the application of design for disassembly (DFD) techniques, ELVs entering the recovery chain can favour (or obstacle) the overall performance of both dismantlers and shredders.

The second element (titled "Exogenous variables") is represented by exogenous variables. These are all the variables that cannot be controlled by any of the actors considered within the context. They can be distinguished among: (i) nation-dependent variables (e.g. GDP), (ii) market-dependent variables (e.g. materials and spare parts prices) and cost items (e.g. labour cost and landfilling fees).

The third element (titled "Physical flow of cars") represents the physical flow of cars. The intent is to map the entire process followed by a car during the recovery chain, trying to continuously check the mass balance between input and output materials.

The fourth element (titled "Environmental calculations") represents the environmental impact of the whole ELV recovery chain. In this case, landfill and recovery rates are the only two dimensions taken into account.

The fifth element (titled "Properties tracking") represent the set of properties related to each car. Its aim is the same as the third element, or guaranteeing the correct balance during the entire process, but at material level.

The sixth and seventh elements (titled “Dismantlers” and “Shredders”) represent the dismantler’s and shredder’s point of view. These are described together because of their similar logic. For both of these actors the model assesses a set of different variables. They can macroscopically be divided between economic and operational ones. Among the economic items there are all the variables related to revenues and costs representative of the activity done by the actors. Instead, operational variables consider elements related to the plant (e.g. capacity, recovery rates and forecasts).
Finally, the eighth, ninth and tenth elements (titled “Junk car”, “Hulk” and “Autoscrap”) represent the mechanisms through which market prices of specific automotive wastes (e.g. junk cars, car hulks and metal scraps, respectively) are defined within the ELV recovery chain.

5.3.3. SD model structure

A representative picture of the SD model structure is reported in **Figure 26** under the form of a simplified causal-loop diagram. Arrows represent a causal relationship between two variables. Positive or negative signs represent the type of relation. This way, a positive (negative) change in the source variable causes a positive (negative) change also in the target variable. The following causal-loop diagram reports the main balancing feedbacks governing the long-run ELV recovery chain.
Given the strong relation between the gross domestic product and the purchasing power of people, the whole structure starts from the GDP value. It influences positively both the production and the sales of new cars. The higher the number of new cars sold, the higher the number of old cars retired from the market. These last ones represent the input material for the ELV recovery chain. An increase in the retirement of cars becomes an increase of dismantled and shredded cars. Then, the recovered material goes to influence both the scrap and raw material prices and, so, the demand of raw materials from the market. The price of scraps influences shredders profit and, so, their investments in additional capacity. Indirectly, the price of scraps influences also the price of hulks, or the trading element between shredders and dismantlers. Subsequently, the price of hulks influences the dismantler’s profit. This way, also the junk car price is influenced and, so, the willingness of private owners to bring their cars to recycling. This last effect can be easily described. The higher the amount of junk cars stored in the backyards of dismantlers, the lower the value they want to pay for other cars. Hence, people are not pushed to buy new cars.
Trying to describe the whole structure, we can assess the effect of a change in GDP on different variables. An increase in GDP can cause a positive change in the amount of dismantled and shredded cars and, so, on the amount of scraps. The augment in scraps lowers their price and the one for virgin materials, by favouring its market demand. However, a reduction of scrap prices negatively influences the shredder’s profit and the price of hulks they want to pay for additional material to treat. This way, dismantlers accumulate hulks in their backyards and the value of junk cars goes down, together with the willingness to pay for additional cars. In contrast, a reduction in GDP decreases the demand for virgin and secondary materials, but enables an increase in scrap prices that favours both shredders and dismantlers. From one side, higher scrap prices support shredders profits. This way, they are willing to pay a higher price to gather additional materials to treat. At the same time, dismantlers can sell their hulks with a higher price, their stocks go down and the value of additional junk cars increases, by enabling the willingness to bring cars to recycle in people.

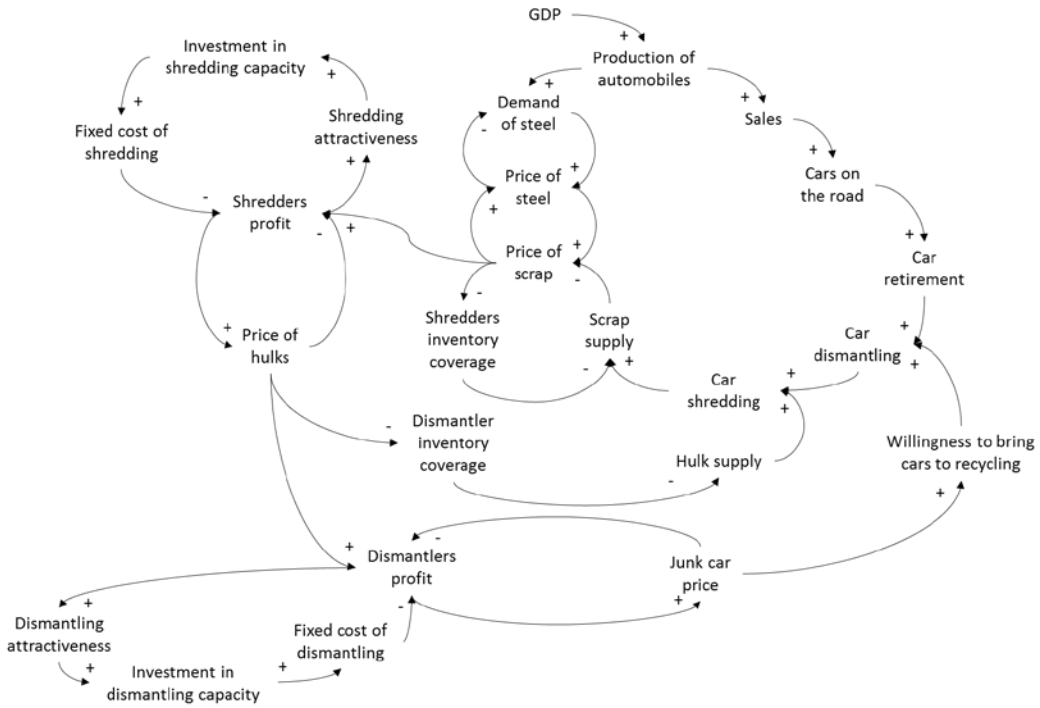

Figure 26. Structure of the SD model (Adapted from (Zamudio-Ramirez, 1996))

5.3.4. ELV recovery chain economic models

The profitability of both a generic dismantler and shredder can be assessed only if their costs and revenues structures are known. For this reason, a dedicated MS Excel economic quantification tool was developed, following what was been already done in the reference work. A simplified version of the economic model developed by the authors can be described below, by distinguishing between dismantlers and shredders equations. From the dismantler's point of view, the economic model can be simplified as follows (see **Table 15** for details):

$$\Pi_{tot} = R_{tot} - C_{tot} \quad (1)$$
$$R_{tot} = R_{recycled\ mat.} + R_{selling\ hulks} + R_{PCB\ recovery} \quad (2)$$
$$R_{recycled\ mat} = p_{recycled\ mat} * Q_{recycled\ mat} \quad (3)$$
$$R_{selling\ hulks} = p_{hulks} * Q_{hulks} \quad (4)$$
$$R_{PCB\ recovery} = p_{PCB} * Q_{PCB} \quad (5)$$
$$C_{tot} = C_{variable} + C_{fixed} \quad (6)$$
$$C_{variable} = C_{junk\ car} + C_{labour} + C_{energy} + C_{PCB\ recovery} \quad (7)$$
$$C_{junk\ car} = p_{junk\ car} * Q_{junk\ car} \quad (8)$$
$$C_{labour} = c_{labour} * Q_{hour/car} \quad (9)$$
$$C_{energy} = c_{energy} * Q_{energy/car} \quad (10)$$
$$C_{PCB\ recovery} = c_{PCB\ recovery} * Q_{PCB} \quad (11)$$
$$C_{fixed} = C_{substitution\ (XX\%)} \quad (12)$$

From the shredder's point of view, the economic model can be simplified as follows:

$\Pi_{tot} = R_{tot} - C_{tot}$ (13)
$R_{tot} = R_{ferrous\ scrap} + R_{nonferrous\ scrap} + R_{PCB\ recovery}$ (14)
$R_{ferrous\ scrap} = p_{ferrous\ scrap} * Q_{ferrous\ scrap}$ (15)
$R_{nonferrous\ scrap} = p_{nonferrous\ scrap} * Q_{nonferrous\ scrap}$ (16)
$R_{PCB\ recovery} = p_{PCB} * Q_{PCB}$ (17)
$C_{tot} = C_{variable} + C_{fixed}$ (18)
$C_{variable} = C_{hulks} + C_{transport} + C_{energy} + C_{var\ maintenance} + C_{landfilling} + C_{PCB\ recovery}$ (19)
$C_{hulks} = p_{hulks} * Q_{hulks}$ (20)
$C_{transport} = p_{transport} * Q_{transport}$ (21)
$C_{energy} = c_{energy} * Q_{energy}$ (22)
$C_{var\ maintenance} = c_{var\ maintenance} * Q_{hulks}$ (23)
$C_{landfilling} = c_{landfilling} * Q_{hulks}$ (24)
$C_{PCB\ recovery} = c_{PCB\ recovery} * Q_{PCB}$ (25)
$C_{fixed} = C_{operator} + C_{administrative} + C_{fix\ maintenance} + C_{insurance} + C_{capacity}$ (26)
$C_{operator} = c_{operator} * Q_{hulks}$ (27)
$C_{administrative} = c_{administrative} * Q_{hulks}$ (28)
$C_{fix\ maintenance} = c_{fix\ maintenance} * Q_{hulks}$ (29)
$C_{insurance} = c_{insurance} * Q_{hulks}$ (30)
$C_{capacity} = c_{capacity} * Q_{hulks}$ (31)

Table 15. Nomenclature of the ELV recovery chain economic model

Nomenclature			
Π_{tot}	Total profit	$C_{substitution\ (XX\%)}$	Plant substitution cost
R_{tot}	Total revenue	$R_{ferrous\ scrap}$	Revenue from ferrous scraps
C_{tot}	Total cost	$R_{nonferrous\ scrap}$	Revenue from nonferrous scraps
$R_{recycled\ mat}$	Revenue from recycling	$p_{ferrous\ scrap}$	Market price of ferrous scraps
$R_{selling\ hulks}$	Revenue from selling hulks	$Q_{ferrous\ scrap}$	Quantity of ferrous scraps per year
$R_{PCB\ recovery}$	Revenue from PCB recovery	$p_{nonferrous\ scrap}$	Market price of nonferrous scraps
$p_{recycled\ mat}$	Market price of recycled materials	$Q_{nonferrous\ scrap}$	Quantity of nonferrous scraps per year
$Q_{recycled\ mat}$	Quantity of recycled materials	C_{hulks}	Hulks cost
p_{hulks}	Market price of hulks	$C_{transport}$	Transportation cost
Q_{hulks}	Quantity of hulks per year	$C_{var\ maintenance}$	Variable maintenance cost
p_{PCB}	Market price of PCBs	$C_{landfilling}$	Landfilling cost
Q_{PCB}	Quantity of PCBs per year	$c_{transport}$	Unit transportation cost (€/ton)
$C_{variable}$	Variable costs	$Q_{transport}$	Quantity of tons transported per year
C_{fixed}	Fixed costs	$c_{var\ maintenance}$	Unit variable maintenance cost (€/ton)
$C_{junk\ car}$	Junk cars cost	Q_{hulks}	Quantity of hulks per year
C_{labour}	Labour cost	$c_{landfilling}$	Unit landfilling cost (€/ton)
C_{energy}	Energy cost	$C_{operator}$	Operator cost

$C_{PCB\ recovery}$	PCB recovery cost	$C_{administrative}$	Administrative cost
$p_{junk\ car}$	Market price of junk cars	$C_{fix\ maintenance}$	Fixed maintenance cost
$Q_{junk\ car}$	Quantity of junk cars per year	$C_{insurance}$	Insurance cost
c_{labour}	Unit labour cost (€/hour)	$C_{capacity}$	Capacity cost
$Q_{hour/car}$	Hours per car	$c_{administrative}$	Unit administrative cost (€/hour)
c_{energy}	Unit energy cost (€/KWh)	$c_{fix\ maintenance}$	Unit fixed maintenance cost (€/ton)
$Q_{energy/car}$	KWh per car	$c_{insurance}$	Unit insurance cost (€/ton)
$c_{PCB\ recovery}$	Unit PCB recovery cost (€/PCB)	$c_{capacity}$	Unit capacity cost (€/ton)

5.3.5. SD model economic and technical input

All the economic data embedded in the MS Excel tool were gathered directly from interviews and the scientific literature. The lacking ones were both autonomously quantified (following the logics of the reference wok) or maintained the same, like in the original SD model (see **Table 16** and **Table 17** for details).

Table 16. Average Italian dismantler economic and technical input – Source: (ACI, 2015), (ADA, 2015), (ANFIA, 2015), (FISE UNIRE, 2015), (Santochi et al., 2002), direct interviews

Sectorial data					
Variable	Value	Unit of measure	Variable	Value	Unit of measure
Total deregistered	200,000	cars/month	Disassembly time	1,22	hour/car
Monthly total capacity	112,000	cars/month	Depollution time	0.78	hour/car
Utilization rate	70%		Total dismantling time	2.00	hour/car
Installed capacity/plant	960	cars/year	Avg. plant lifetime	15	years
Variable costs					
Variable	Value	Unit of measure	Variable	Value	Unit of measure
Junk car price	20.00	€/car	Energy cost	0.63	€/car
Labour cost	40.00	€/car	PCB recovery cost	18.42	€/car
Revenue sources					
Variable	Value	Unit of measure	Variable	Value	Unit of measure
Unit recycling revenues	20.83	€/car	Selling hulks revenues	124.95	€/car
Avg. hulks weight	892.5	kg/car	PCB recovery revenues	46.88	€/car
Avg. hulk price	140.00	€/car			
Fixed costs					
Variable	Value	Unit of measure	Variable	Value	Unit of measure
Substitution cost (100%)	34.72	€/car	Substitution cost (70%)	49.60	€/car
Total costs	128.65 €/car				
Total revenues	192.67 €/car				
Total profits	64.01 €/car				

Table 17. Average Italian shredder economic and technical input – Source: (ACI, 2015), (AIRA, 2015), (ANFIA, 2015), (FISE UNIRE, 2015), (Berzi et al., 2013), direct interviews

Sectorial data					
Variable	Value	Unit of measure	Variable	Value	Unit of measure
Total cars shredded	200,000	cars/month	Installed capacity/plant	7500	tons/month
Total tons shredded	178,500	tons/month	Shredder utilization rate	40%	
Materials distribution data					
Variable	Value	Unit of measure	Variable	Value	Unit of measure
Ferrous metals	70.00%		Avg. car weight	1050	kg/car
Non-ferrous metals	5.00%		Avg. hulks weight	892.5	kg/car
ASR fraction	25.00%		Total cars shredded	3361	cars/month
Variable costs					
Variable	Value	Unit of measure	Variable	Value	Unit of measure
Avg. hulk price	124.95	€/car	Variable maintenance cost	3.51	€/car
Transportation cost	13.39	€/car	ASR landfilling cost	15.17	€/car
Energy cost	1.76	€/car	PCB recovery cost	18.42	€/ton
Revenue sources					
Variable	Value	Unit of measure	Variable	Value	Unit of measure
Scrap ferrous price	240.00	€/ton	Nonferrous revenues	48.11	€/car
Scrap nonferrous price	1100.00	€/ton	PCB recovery revenues	46.88	€/car
Scrap metal revenues	148.44	€/car			
Fixed costs					
Variable	Value	Unit of measure	Variable	Value	Unit of measure
Labour cost	7.24	€/car	Insurance cost	4.10	€/car
Administrative cost	4.00	€/car	Capacity cost	2.98	€/car
Fixed maintenance cost	3.15	€/car			
Total costs	198.66 €/car				
Total revenues	243.43 €/car				
Total profits	44.77 €/car				

5.4. Case study and input data: The Italian ELV recovery chain

5.4.1. The European and Italian ELVs management context

The most updated estimates say that the only Europe generates yearly from 7 to 14 million tons of ELVs, with an annual growth rate from 2% to 8%. A general perspective of EU ELVs volumes can be seen in the following **Table 18**.

Table 18. European ELV volumes – Source: (Eurostat, 2015b)

	2007	2008	2009	2010	2011	2012	2013
EU-28 (¹)	**:**	**:**	**:**	**:**	**:**	**6,290,000**	**6,250,000**
EU-27 (¹)	**6,500,000**	**6,270,000**	**9,000,000**	**7,350,000**	**6,750,000**	**6,250,000**	**6,220,000**
Belgium	127,949	141,521	140,993	170,562	165,016	160,615	134,506
Bulgaria	23,433	38,600	55,330	69,287	62,937	57,532	61,673
Czech Republic	72,941	147,259	155,425	145,447	132,452	125,587	121,838
Denmark	99,391	101,042	96,830	100,480	93,487	106,504	125,650
Germany	456,436	417,534	1,778,593	500,193	466,160	476,601	500,322
Estonia	12,664	13,843	7,528	7,268	11,413	12,835	14,712
Ireland	112,243	127,612	152,455	158,237	134,960	102,073	92,467
Greece	47,414	55,201	115,670	95,162	112,454	84,456	86,205
Spain	881,164	748,071	952,367	839,637	671,927	687,824	734,776
France	946,497	1,109,876	1,570,593	1,583,283	1,515,432	1,209,477	1,115,280
Croatia	:	:	:	:	:	35,213	32,135
Italy	**1,692,136**	**1,203,184**	**1,610,137**	**1,246,546**	**952,461**	**902,611**	**876,052**
Cyprus	2,136	14,273	17,303	13,219	17,145	17,547	13,212
Latvia	11,882	10,968	10,590	10,640	9,387	10,228	9,003
Lithuania	15,906	19,534	19,656	23,351	26,619	22,885	26,482
Luxembourg	3,536	2,865	6,908	6,303	2,341	2,834	2,290
Hungary	43,433	37,196	26,020	15,907	13,043	15,357	14,897
Malta	:	:	:	330	2,526	2,530	1,198
Netherlands	166,004	152,175	191,980	232,448	195,052	187,143	183,451
Austria	62,042	63,975	87,364	82,144	80,004	64,809	73,993
Poland	171,258	189,871	210,218	259,576	295,152	344,809	402,416
Portugal	90,509	107,746	107,946	107,419	77,929	92,008	92,112
Romania	36,363	51,577	55,875	190,790	128,839	57,950	:
Slovenia	8,409	6,780	7,043	6,807	6,598	5,447	:
Slovakia	28,487	39,769	67,795	35,174	39,717	33,469	36,858
Finland	15,792	103,000	96,270	119,000	136,000	119,000	99,300
Sweden	228,646	150,197	133,589	170,658	184,105	185,616	189,748
United Kingdom	1,138,496	1,210,294	1,327,517	1,157,438	1,220,873	1,163,123	1,149,459
Iceland	:	9,386	5,109	4,195	4,075	5,824	4,463
Liechtenstein	82	91	72	107	94	114	326
Norway	95,128	130,018	95,000	112,537	124,563	119,905	141,452

(¹) Eurostat estimates for 2007–09 and 2013. For reasons of comparison, EU-27 data are also shown for 2012 and 2013, although EU-28 data are available.

Considering the previous 2013 data reported in **Table 18**, United Kingdom is the first source of ELVs in Europe, followed by France, Italy, Spain and Germany. Italy alone generated in 2013 an annual amount of about 876k tons of ELVs.

In a broad perspective, data about Italian ELVs management practices can be gathered from several information sources. They can be distinguished in: (i) official database (e.g. Eurostat), (ii) official institutions (e.g. the Italian Ministry of Infrastructures and Transports (MIT), the

Italian Automobile Club (ACI), the Italian Automotive Public Register (UMC), the Italian Institute for the Environmental Protection and Research (ISPRA)), (iii) industrial entities (e.g. the Italian Confederation of Industries (CONFINDUSTRIA)) and (iv) national sectorial associations (e.g. the Italian Association of the Automotive supply chain (ANFIA), the Italian Associations of Automotive Dismantlers (CAR and ADA), the Italian Association of Automotive Recyclers (AIRA)). All of these sources were assessed by the authors. Some of them were reached only through their official website, others were directly interviewed (see **Table 19 - Table 23** for details).

Table 19. Some Italian cars 2015 data – Source: (ACI, 2015), (ANFIA, 2015)

Cars age	Circulating 2015	Deregistered 2015	Demolished 2015
0-3 years	4.374.400	31.051	20.751
3-6 years	5.018.196	54.363	36.331
6-9 years	6.506.229	86.007	57.478
9-12 years	6.092.160	176.949	118.254
12-15 years	5.100.336	312.093	208.571
15-18 years	3.583.165	327.445	218.830
18+ years	6.676.747	361.729	241.742
Total 2015	37.351.233	1.349.637	901.957

Table 19 reports some interesting data about the Italian automotive market. Since this table it is possible to comprehend that there is a wide distinction between deregistered and demolished cars. Italy, like most of the other EU countries having the widest number of circulating cars (e.g. Germany, France and Spain) suffer of a typical issue of illegal exports of ELVs. Those ELVs that should be recycled in Italy, instead, are sold by official Italian dealers to foreign traders as "second-hand cars". However, those "cars" will not be re-registered in the final market of the trader, but recycled there. This way, an illegal transfer of automotive wastes (only in Italy represents 30% of generated ELVs yearly) continues to expand, especially in eastern EU countries. Given this effect, ELV recovery chains in western EU countries are obliged to either buy volumes abroad or reduce their capacity to survive.

Table 20. Some Italian cars 2015 monthly data – Source: (ACI, 2015), (ANFIA, 2015)

Cars age	Deregistered/month 2015	Demolished/month 2015	Monthly/Total deregistered (%)	Cumulated amount
0-3 years	2.588	1.729	0,02301	0,02301
3-6 years	4.530	3.028	0,04028	0,06329
6-9 years	7.167	4.790	0,06373	0,12701
9-12 years	14.746	9.855	0,13111	0,25812
12-15 years	26.008	17.381	0,23124	0,48936
15-18 years	27.287	18.236	0,24262	0,73198
18+ years	30.144	20.145	0,26802	1,00000

Table 20 shows the distribution of 2015 ELV monthly volumes basing on the car ages. This amount was, then, used to define the monthly/total deregistered ratio and, consequently, the

cumulated amount of ELVs by age. Those data will be exploited by the next calculation steps to define the historical and equilibrium amounts and percentages of ELVs to be embedded into the SD model.

Table 21. Italian 2015 new registered cars – Source: (Eurostat, 2015b), (MIT, 2015), (ACI, 2015)

Year	New registrations	Year	New registrations	Year	New registrations	Year	New registrations
1996	1.843.366	**2001**	2.379.980	**2006**	2.543.157	**2011**	1.764.592
1997	2.389.892	**2002**	2.235.948	**2007**	2.514.905	**2012**	1.403.043
1998	2.437.718	**2003**	2.516.972	**2008**	2.193.822	**2013**	1.311.334
1999	2.312.309	**2004**	2.743.769	**2009**	2.176.940	**2014**	1.376.185
2000	2.359.674	**2005**	2.441.978	**2010**	1.971.830	**2015**	1.593.857
Annual average sales 1996-2015				2.125.564			
Monthly average sales 1996-2015				177.130			
Annual average sales 1996-2007 (pre-crisis)				2.393.306			
Monthly average sales 1996-2007 (pre-crisis)				199.442			

Considering **Table 21** data, it is possible to compare the automotive sector health in pre- and post- economic crisis time periods. Annual and Monthly sales were higher in the 1996-2007 period than in 2015 (-1.1 million cars sold than in 2004). To this aim, the 1996-2007 average data (e.g. 2.4 million cars sold/year and 200k cars sold/month) were considered as more representative of the real Italian market potential and adopted within the SD model like the equilibrium level. Instead, the annual and monthly 2015 data (e.g. 1.6 million cars sold/year and 133k cars sold/month) were considered in the SD model as the historical level.

Table 22. Calculation of Italian 2015 new and old deregistered cars – Source: (MIT, 2015), (ACI, 2015), (ANFIA, 2015)

Groups	Circulating cars				Monthly deregistered cars					
	Hist.	**%**	**Equil.**	**%**	**Hist.**	**%**	**Equil.**	**%**	**Hist. new**	**Equil. new**
0-3 years	4.374.400	11,7%	7.200.000	21,4%	2.588	2,3%	4.601	2,3%	14.285	40.215
3-6 years	5.018.196	13,4%	7.034.352	20,9%	4.530	4,0%	12.366	6,2%		
6-9 years	6.506.229	17,4%	6.589.170	19,6%	7.167	6,4%	23.247	11,6%	12,70%	20,11%
9-12 years	6.092.160	16,3%	5.752.262	17,1%	14.746	13,1%	41.244	20,6%	**Hist. old**	**Equil. old**
12-15 years	5.100.336	13,7%	4.267.483	12,7%	26.008	23,1%	58.010	29,0%	98.185	159.785
15-18 years	3.583.165	9,6%	2.179.132	6,5%	27.287	24,3%	44.308	22,2%		
18+ years	6.676.747	17,9%	584.050	1,7%	30.144	26,8%	16.224	8,1%	87,30%	79,89%
Total 2015	37.351.233		33.606.448		112.470		200.000			

Table 22 reports the overall data used to define the Italian new and old deregistered cars at historical and equilibrium levels. Starting from historical 2015 data about aging groups of circulating cars and a hypothesised equilibrium amount (average value) of new registered cars in a period of three years, it was possible to define the overall amount of monthly historical and equilibrium deregistered cars, dividing them between new (age from 0 to 9 years) and old (age from 9 to 18+) ones. Like described above, this calculation was possible only by considering monthly deregistered cars percentages and cumulated amounts. These last amounts (both percentages and amounts) represent input data of the SD model.

5.4.2. Italian ELVs management inputs

The current Italian ELV recovery chain is constituted by several actors that, acting independently, are able to recover materials and parts from cars. Generally, an ELV recovery chain can be described as a group of three actors, like automakers, dismantlers and shredders. In addition, a series of auxiliary actors can be present, like foundries, raw material brokers, secondary spare parts and metal scrap traders.

Table 23. Italian official ELVs dismantlers and shredders – Source: (FISE UNIRE, 2015)

	2003	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**
Dismantlers	1562	1418	1489	1388	1407	1313	:	:	:	1348	1500	1510
Shredders	:	:	27	:	:	36	:	:	33	36	33	35

Considering data reported in **Table 23**, there is a relevant presence of dismantlers (estimated in about 1510 companies in 2014) – uniformly distributed in the country – and about 35 shredders mainly distributed in the northern part of Italy. From the automaker's perspective, only one Italian company is taken into account within this study. During this work, many of these actors were directly interviewed, asking for quantitative data to be embedded within the SD model under development. However, average data will be presented and discussed within the paper, for confidential reasons. Considering an initial list of 51 companies (among sectorial associations, automakers, dismantlers, shredders and spare parts traders), 30 of them accepted to be interviewed, by sharing valuable information about their business. The interview was always implemented through a set of open questions regarding all of the constant inputs of the SD model.

Trying to define some realistic hypotheses about Italian dismantlers' and shredders' plants installed capacity, some initial distinctions were done. First of all, the overall installed capacity is currently unsaturated, given illegal flows previously described. This way, annual demolished amounts represent only a 70% of the potential capacity available. Secondly, not all ELVs are cars, but also trucks, buses, motorcycles and any other kind of vehicle considered by the ELV Directive. This way, cars represent the 90% of yearly generated ELVs. By taking into account all of these information, a set of important data to be embedded within the SD model were found. From the dismantlers' point of view, the authors considered an average of 1400 plants, with an overall installed capacity of about 1.3 million cars/year (about 112k cars/month) and a utilization rate of 70%. This way, a generic Italian dismantling centre's capacity can be quantified in about 960 cars/year (80 cars/month). From the shredders' point of view, the authors considered an average of 35 plants, with an overall installed capacity of about 3.1 million tons/year (about 262k tons/month) and a utilization rate of 70%. %. This way, a generic Italian shredding centre's capacity can be quantified in about 90,000 tons/year (7500 tons/month).

5.5. Results and discussion

5.5.1. SD model validation

Once identified all the equations reported in Section 3.4 and all the input data reported in Section 3.5, the SD model was tested intensively, by varying values of constant inputs at the base of the model and assessing different scenarios. Results were, then, discussed with a selected group of experts for a final refinement of specific parameters. **Figure 27** and **Figure 28** illustrate the response of the model to the introduction of the automotive electronics recovery process within the current ELV recovery chain. The first picture considers that the automotive electronics recovery process will be managed by shredders. Contrarily, the second picture considers that dismantlers will manage those process. Taking into account the graphs reported in **Figure 27**, there is a clear distinction in terms of effects between dismantlers and shredders. In this scenario, dismantlers will be only partially responsible for the recovery of automotive electronics. Their involvement will consider the only disassembly of automotive electronic components from ELVs and their direct selling to shredders. Instead, shredders will be responsible for the final recovery of materials from scrap automotive components. This means that a dedicated recovery plant will be owned by shredders, in addition to current shredding and separation equipments. **Figure 27a** and **Figure 27b** show the impact of the adoption of automotive electronics recovery processes on the economic performance of dismantlers. Here, like in all of the following pictures, economic performances of both dismantlers and shredders are mapped into two ways, or cumulative and semester profits. In this first scenario, even if dismantlers will participate only partially to the recovery process, it is evident that the recovery of electronic components could offer a relevant improvement also for them, even immediately. The cumulative profit of dismantlers could strongly increase. Considering the hypothesized trend, in 50 years the overall business of dismantlers could go from 5.8 billion Euros to 7.5 billion Euros. The same behaviour can be found also in terms of semester profits of the overall sector, stabilizing in 50 years to about 80 million Euros (instead of about 37.5 million Euros without electronics). Obviously, the best improvement in terms of economic performances can be expected for shredders. Even if the recovery of automotive electronics will require a considerable investment in additional plants, these expenses will be completely recovered in few years. Cumulative profits related to the overall business of shredders could strongly increase, reaching in 50 years a potential level of about 12.4 billion Euros (instead of about 4.8 billion Euros without electronics). The same effect can be described in terms of semester profits, stabilizing in 142.5 million Euros in 50 years instead of 56.3 million Euros without electronics.

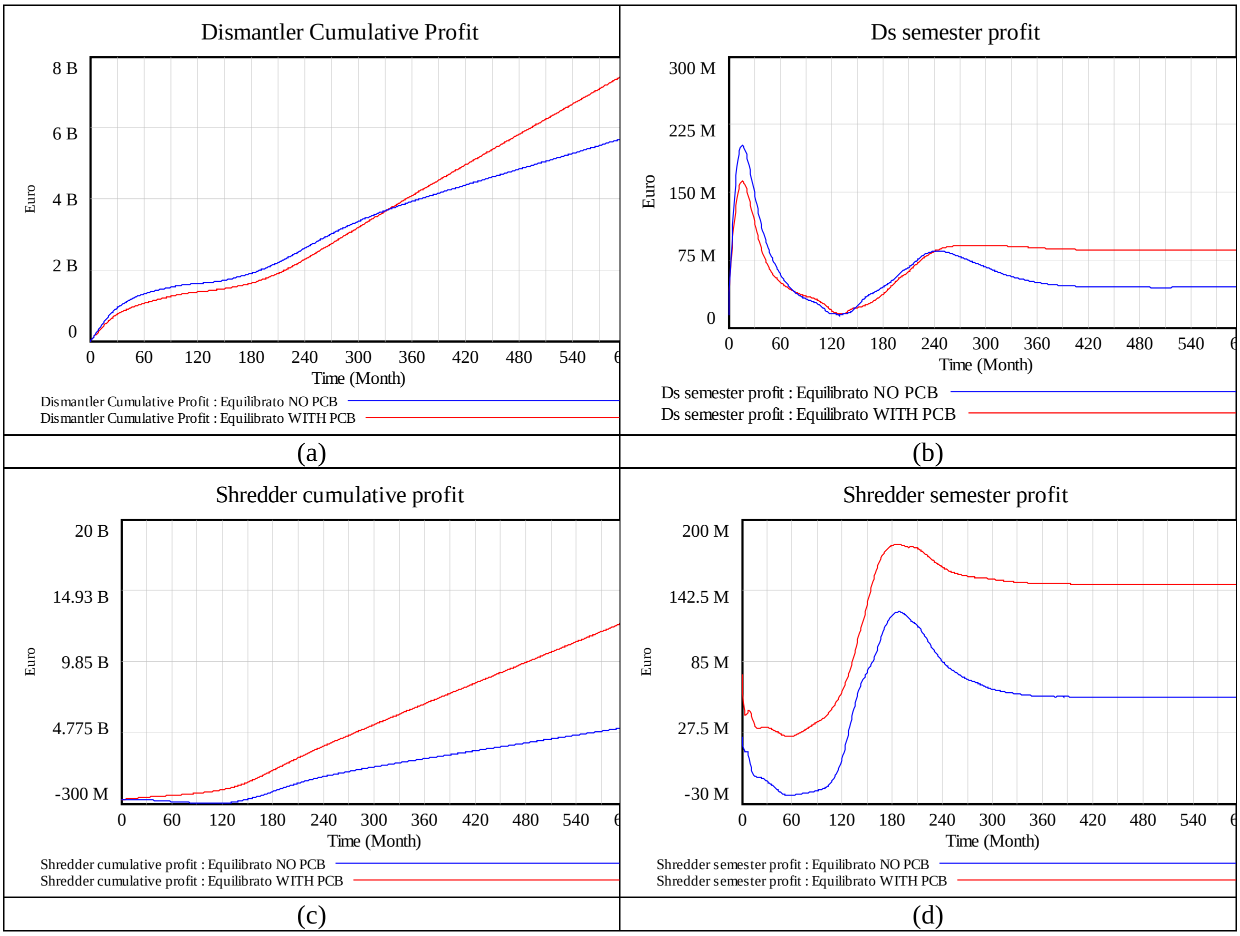

Figure 27. Effects of automotive PCB recovery on Sh and Ds profits – Shredders lead

A similar result can be obtained by taking into account the graphs reported in **Figure 28**. Also in this case, there is a clear distinction in terms of effects between dismantlers and shredders. However, in this scenario shredders will not be involved in the recovery of automotive electronics, a process completely managed by dismantlers. Dismantlers, in addition to usual disassembly and collection activities, will be responsible for the final recovery of materials from scrap automotive components. This means that a dedicated recovery plant will be owned by dismantlers. **Figure 28a** and **Figure 28b** show the impact of the adoption of automotive electronics recovery processes on the economic performance of dismantlers. Also here, economic performances of both dismantlers and shredders are mapped into two ways, or cumulative and semester profits. In this second scenario, it is evident that the recovery of electronic components could offer a relevant improvement for dismantlers, even immediately. The cumulative profit of dismantlers could strongly increase. Considering the hypothesized trend, in 50 years the overall business of dismantlers could go from 5.8 billion Euros to 15 billion Euros. The same behaviour can be found also in terms of semester profits of the overall sector, stabilizing in 50 years to about 150 million Euros (instead of about 37.5 million Euros without electronics). From the shredders side, they would continue in doing their business as usual, without any impact coming from this new kind of activity done by dismantlers.

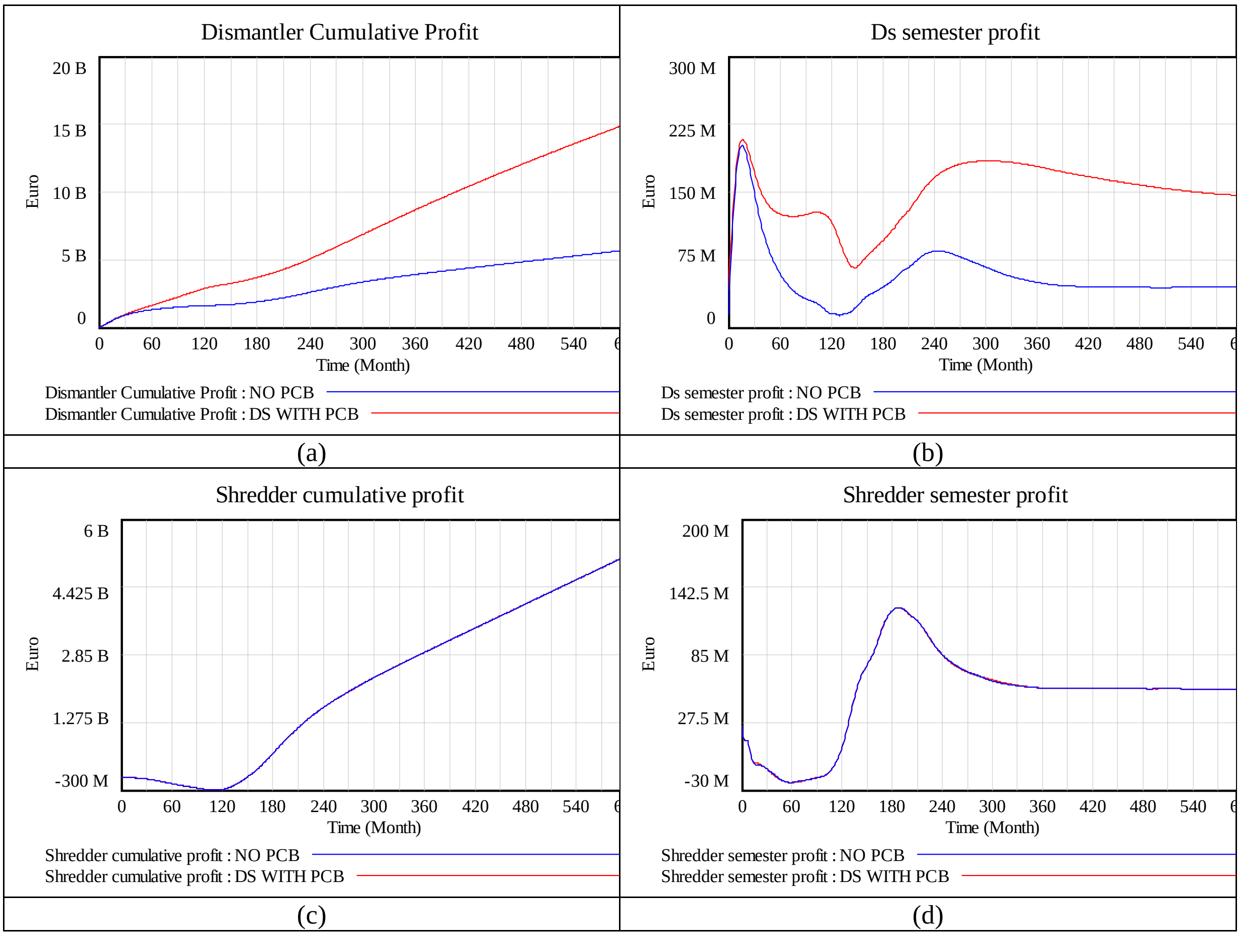

Figure 28. Effects of automotive PCB recovery on Sh and Ds profits – Dismantlers lead

5.5.2. Analysis of factors influencing the economic performance

A direct involvement of managers from several companies involved in the Italian ELV recovery chain allowed defining what are the main variables embedded in the current SD model influencing their economic performances once automotive electronics will start to be recovered correctly. Those variables were identified in:

- Constant properties (e.g. nonferrous metals composition) of ELVs entering the recovery process
- Price of nonferrous metals leaving the recovery process. A detail on the impact of a change in those two variables is described in the next sub sections.

Constant properties (nonferrous metals composition) of ELVs

One of the first variables identified by industrial actors as potentially influenced by the increasing presence of electronics in cars is the overall materials composition of vehicles. Even if limited (estimates speak of about 1% on the total average mass of a car), the presence of electronics in cars could reduce the overall amount of iron. Given that iron is, currently, the main source of revenue for both dismantlers and shredders, its reduction could mean a direct reduction of their revenues. However, a good answer to this issue can be found in the following **Figure 29**. Also in this case, cumulative and semester profits of both dismantlers and shredders have been compared. The main hypothesis done during this simulation was that the percentage in nonferrous metals could rise from current 9% to about 10%). From a dismantler's view, the reduction in iron could negatively affects their business, especially in the first assessed period. However, wit time this change in materials composition of cars could offer a better economic perspective than now (this is visible especially in terms of semester profits). From a shredder's perspective, the effect is less evident. Both in terms of cumulative and semester profits their economic performances will be very similar.

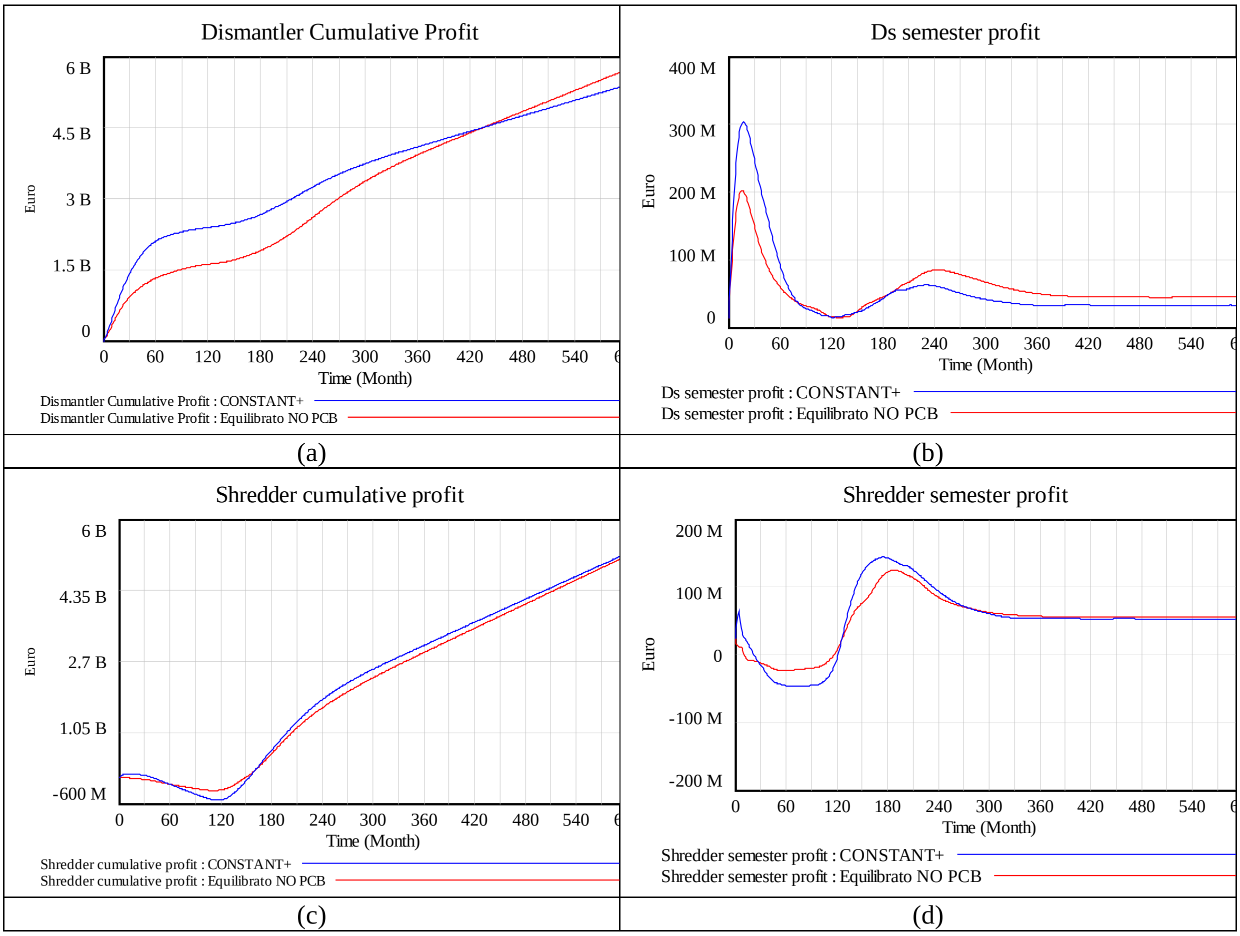

Figure 29. Effect of an increase in constant properties on Sh and Ds profits

Nonferrous prices

One of the other variables potentially influenced by the introduction of automotive electronics recovery in current ELV recovery chains is represented by the nonferrous market price. In this case, with this term the authors refer to all the nonferrous materials present in a car. This way, several metals can be considered within this category, being them basic, critical or precious ones. The intent of the simulation process this time was the assessment of an improvement in the nonferrous market price (going from to current 1.1 Euro/kg to 4 Euro/kg) on the overall economic performance of both dismantlers and shredders. **Figure 30** reports the resulting effects. Also in this case, economic performances of the actors were assessed in terms of cumulative and semester profits. From a dismantler's perspective, an improvement in nonferrous market prices has a relevant effect on the cumulative profit of the entire sector, measurable in 7.5 billion Euros on a period of 50 years (5.8 without automotive electronics). Instead, the effect of this improvement seems to be limited to a certain period (about 10 years) if considering the semester profit. Contrarily to authors' expectations, an improvement in nonferrous marker prices would have an irrelevant effect on shredders economic performances. The cumulative profit of the entire sector will follow, more or less, the same trend. Some more visible effects can be found in terms of semester profits. After some evident under- and overshooting in the first period of time (about 20 years), the system will stabilize at the same level that could be reached without the introduction of automotive electronics recovery processes. However, this equilibrium seems to be not a stable one.

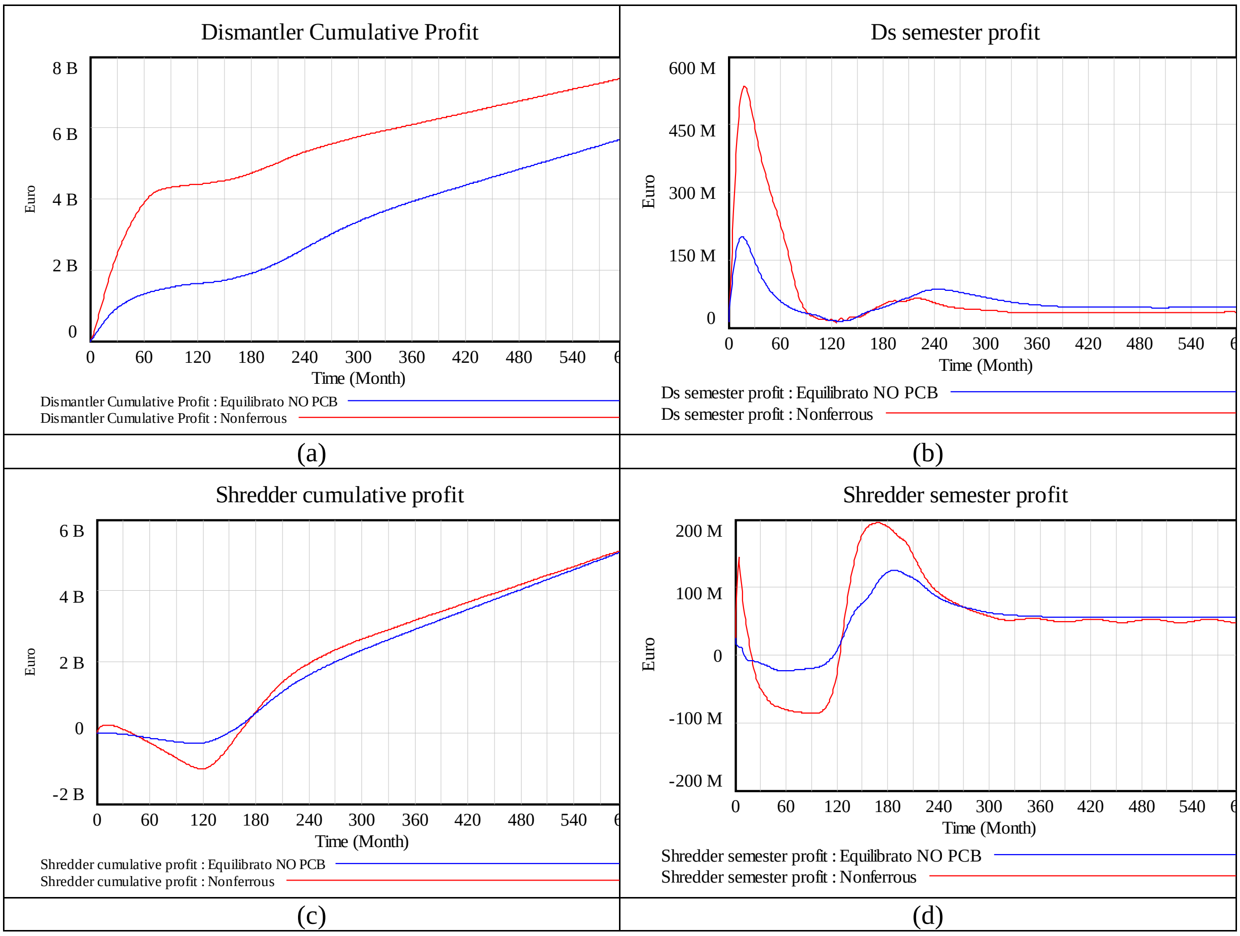

Figure 30. Effect of an increase in nonferrous price on Sh and Ds profits

5.5.3. Main findings

Considering the previous **Figure 27** and **Figure 28**, important benefits are expected from the introduction of automotive electronics recovery processes in current ELV recovery chains. Whoever will be the owner of the recovery plant (either shredders or dismantlers), the economic performance is clearly better than the one reachable through current procedures. If we would like to select the best solution considering the direct point of view of industrial actors, it seems that the cheapest and logical solution (considering the strong technological affinity between ELV and WEEE recovery processes) should be the one seeing shredders as direct recyclers of scrap automotive PCBs. However, several limitations have to be considered before selecting a solution.

Issues related to illegal transfer of cars among nations and lack of official information about automotive electronics recovery trends influence negatively any kind of study on this topic. This way, several hypotheses were considered during the work. Firstly, input volumes of cars entering the ELV recovery chain were represented only by treated volumes (not generated ones). Secondly, only fixed plants performances were considered. Third, costs and revenue structures of both Italian dismantlers and shredders were extrapolated from interviews. Sensitive data were translated in average values and they were adopted in all calculations presented within the paper. The same logic was followed also in defining costs and revenue from automotive electronics recovery. Fourth, all the economic performances mapped through simulation are referring to scenarios where no national subsidies exist (like in reality). Wherever there could be any sort of national subsidy, economic performances could be better than those reported in this paper. It is possible to say the same thing also in terms of both estimated automotive electronic volumes, material market prices, materials purity levels, costs and revenue structures of the actors. All these elements were assessed both through field interviews or literature reviews. This way, they could represent only partially the real state of the industrial context. Just as an example, automotive electronics volumes could be higher than expected. Especially in the last decade, common cars saw a strong augment in electronic components. In addition, hybrid and electric cars are embedding more electronics than common vehicles, without considering auto-guided ones. This consideration allows the reader to comprehend the level of uncertainty influencing this research field.

5.6. Conclusions

End of life vehicles are re-known by the experts as one of the main sources of secondary raw materials. Even if the ELV recovery chain is an already established industry in Europe, its management strategies remained the same since many decades, completely based on raw materials market prices. However, in the last decades this way of doing exposed national ELV recovery chains to several market risks, like unpredictable fluctuations of raw materials market prices, uncontrolled illegal transfer of vehicles among nations and the unavoidable evolution of ELVs' materials mix. Trying to control and partially limit this exposition to materials market risks, the experts proposed several interesting works about new methods for predicting ELV composition and materials flows, comparing different waste management strategies, mapping ELV reverse logistic chains and assessing the effect of new environmental policies. This way, many simulation approaches were adopted. However, all of them were focused on either a specific part of the ELV recovery chain or a specific issue affecting the industrial context. Only in one case a whole national ELV recovery chain was assessed completely, trying to comprehend the set of internal drivers influencing transactions among all the involved actors through System Dynamics. The same SD model was taken as reference work by this paper. The original SD model was updated and upgraded, before applying it to the Italian context. The updating process interested values of constant inputs of the original SD model, substituted with Italian data coming from both field interviews and the scientific literature. The upgrading process, instead, interested the materials mix taken into account by the original SD model, by considering also the ones coming from the recovery of automotive electronic components (e.g. hazardous, rare and precious metals, rare earths and epoxy resins). Current economic performances of the Italian ELV recovery chain were assessed and compared with the ones potentially reachable through the management of automotive electronics. Results – validated by relevant industrial experts – demonstrate how the new approach could potentially improve the economic potential of both dismantlers and shredders, depending on the scenario taken into account. Considering comments from the experts, it seems that the scenario seeing shredders as main responsible of the automotive electronics recovery could be the most realistic and logical, because of the high technological affinity between ELV and PCB recovery processes. Contrarily, considering what reported by the German Ministry of the Environment within its 2016-2019 work program (BMUB, 2016), the best solution should see dismantlers as the main responsible for the recovery of automotive electronics from ELVs. However, whatever the scenario considered, the benefits coming from the management of scrap automotive electronics are visible and measurable in any configuration of the industrial context.

Post-publication potential improvements

Once again, the limited space offered by the journal format did not allowed to describe with more detail what is the background knowledge about System Dynamics (e.g. main logics and steps linking qualitative explanations and simulation results). This way, the added value of the modelling tool is only partially emphasized and results seems to be weakly related with the overall model. An interesting improvement could be given by describing other works adopting SD to map circular economy (e.g. see (Diez et al., 2017)).

6. General conclusions

6.1. Introduction

This last section of the thesis elaborates the main conclusions on the topics exposed in the previous sections. Like evidenced since the introduction, the thesis is constituted by two main visions on the same topic. These two points of view derives directly from the fact that the entire work is based on two main research questions, better contextualized through the literature review presented in Sections 2 and 3. The first part of the thesis presented in Section 4 quantifies the potential profits coming from the recovery of automotive electronic wastes, while the second part (presented in Section 5) identifies a set of innovative operation strategies related to the recovery of wasted automotive electronics, by measuring their effects on the overall economic performance of the Italian ELV recovery chain.

6.2. Contribution to knowledge: Answering to research questions

This thesis is focused particularly on the recovery of electronic wastes from ELVs, both from a theoretical and practical perspective. The main intent is to make the actors involved in national ELV recovery chains aware about the current state of knowledge and what kind of perspectives could be exploited in the future for the survival of the overall industrial context. This issue represents the "fil rouge" of the entire work and it was assessed within the thesis into two ways. These views were identified thorough the following two research questions:

RQ1: How much is the potential profit coming from the recovery of electronic wastes from ELVs?

One important message offered by the first part of this thesis (see Sections 2, 3 and 4 for details) refers about the strategic importance of a combined management of e-wastes, whatever their origin. Electronics is becoming a relevant element not only in mass electronic products, but also in other industrial fields like the automotive sector. Given that, current recovery processes must adequate themselves to both new market conditions and new products features, trying to recover as many materials as possible, particularly the most critical and precious ones. Even if several inequalities exist between WEEE and wasted automotive electronics management, the fact that some authors demonstrated the similarity between PCBs embedded in these two kinds of e-wastes enabled new hypotheses towards a more sustainable recovery of these components. These potential improvements were also quantified in terms of both volumes and profits. To this aim, an economic model evaluating potential revenues and costs coming from the recycling of scrap automotive electronics was implemented and described into detail. The obtained indexes (e.g. NPV and DPBT) demonstrated the validity of investments in two different types of plants (mobile and field ones) and for all the four types of WPCBs considered. Economic values obtained from the model are so high, and different from common values available in literature, because of the relevant presence of Au in automotive WPCBs. A sensitivity analysis done on critical variables (e.g. plant saturation level, Au content, Au market price, Au final purity level, WPCBs purchasing cost and opportunity cost) allowed to test the robustness of theoretical evaluations. Finally, the matching of predicted ELVs volumes in the next fifteen years and expected NPV allowed to define the potential dimension of a market dedicated to the treatment of these wastes.

RQ2: How the recovery of electronic wastes from ELVs could improve the economic performance of the whole recovery chain?
Another important message coming from the second part of this thesis (see Section 5 for details) is about the effect that an adoption of innovative operation strategies focused on the recovery of wasted automotive electronics could have on the overall economic performances of a typical ELV recovery chain. Trying to better comprehend these effects, the quantification of potential profits coming from this kind of activities must be linked to the current state of things, by considering together all of the key elements driving the decisional process within a generic ELV recovery chain. The SD model taken into account, once updated with Italian data and upgraded to consider automotive electronics, allowed to evidence current economic performances of the Italian ELV recovery chain and comparing them to the ones potentially reachable through the management of automotive electronics. Results demonstrate that more sustainable practices could enable a positive improvement of economic performances of both dismantlers and shredders, depending on the scenario taken into account. Considering comments from the experts, it seems that the scenario seeing shredders as main responsible of the automotive electronics recovery could be the most realistic and logical, because of the high technological affinity between ELV and PCB recovery processes. Contrarily, considering what reported by the German Ministry of the Environment within its 2016-2019 work program (BMUB, 2016), the best solution should see dismantlers as the main responsible for the recovery of automotive electronics from ELVs. However, whatever the scenario considered, the benefits coming from the management of scrap automotive electronics are visible and measurable in any configuration of the industrial context.

6.3. Contribution to practice: Managerial implications

In general terms, the obtained results could be of interest for several actors involved in any ELV recovery chain, being them industrial or political ones. From the industrial side, the main intent of his work is the enhancement in awareness towards the embedded value of wasted automotive electronics, currently not recovered in the correct way. Following this logic, results coming from the two models could be useful for both car manufacturers, car dismantlers and car shredders. Manufacturers could assess in real time the effect of a change in materials composition of their cars on the entire ELV recovery chain, by identifying most feasible alternatives. Dismantlers and shredders, instead, could do the same, but from a more restricted perspective, basing on their role within the ELV recovery chain. From the political side, the main intent of this work is the enhancement of current EU ELV Directive gaps and the identification of potential corrective actions. Following this logic, results could be of interest for both national/international environmental associations, national ministries of environment and supra-national entities. Like what happened in Germany and Switzerland, inspiring from the results coming from this thesis environmental associations could push national governments in modifying their regulations according to new kinds of electronic wastes to be taken into account in future revisions of e-waste management laws. This change, if spread in many nations, could also influence a change in supra-national directives, like the European WEEE and ELV Directives in force. Once directives will oblige companies in behaving in a more sustainable way (i.e. recovering electronic products from wasted cars), a real improvement in the sustainability level of the entire European economy could happen and new business opportunities for companies could rise in the next future.

6.4. Limits and advantages of the proposed models

The two models presented within this thesis, even if interesting and innovative in comparison with what available from the literature, present a series of limitations. Form the mathematical model side, the quantification of potential profits is completely based on several assumptions done by the authors because of the lack of information already described in the previous sections. Hypotheses about the cost and revenue structures of mobile and field plants, ELV volumes and average weight of automotive PCBs are only some of the most important estimates adopted within the work. Also from the SD model side many assumptions were adopted, especially for those variables of the SD model not quantified by the industrial actors. In this case, hypotheses were done, for example, on percentages of cars yearly dismantled by the ELV recovery chain, the average amount of junk cars and car hulks stored in dismantler's and shredder's backyards, all the durations of decisional processes or the equilibrium transfer price of both junk cars and car hulks. However, results were validated by industrial experts that evidenced their realistic value. The two models presented within this thesis present also many advantages. From the mathematical model side, the use of the DCF method allows industrial actors to better comprehend and have a direct idea about the quantification of the investment required when someone wants to adopt this new kind of technology. The use of official materials market prices together with the exploitation of cost structures defined by the literature basing on real cases guarantee about the goodness of numbers. In addition, the consideration of Eurostat estimates allows to comprehend the current and future evolution of the wasted automotive electronics market. From the SD model side, several advantages can be described. First of all, the same model can be exploited to assess historical and future expected performances. Again, within the same simulation environment, it is possible to compare in real time different scenarios and continuous variations of the key variables, by making a direct identification of the best operation strategies to implement. In addition, within the same SD model it is possible to simulate the behaviour of different actors involved in the same ELV recovery chain, by assessing the effects of independent decisions on others economic performances.

6.5. Future developments

The product end-of-life management is a new research area established in the last decades. Before the Kyoto Protocol of 1997, no one gave importance to this side of the product lifecycle. This aspect opens the door to many innovative contributions, given the lack of information in many industrial fields. Because of the amazing volumes, e-wastes from electrical and electronic equipments were the focus of thousands of papers in the last decades. However, like hugely evidenced within this thesis, there are also other sources of e-wastes – currently not considered by the literature – that could become a relevant source of materials in the near future. Given the even more importance of electronics within several kinds of products and industrial sectors, the chances to propose something of innovative to the research community are enormous. For example, always considering the management e-wastes from cars, a good research stream could be the assessment of recycling issues coming from the treatment of hybrid and full-electric cars or auto-guided vehicles. The expected result is an increase in volumes of e-wastes from cars, given the high presence of electronic components in these kinds of new vehicles. Again, considering current evolutions in German ELV regulation and Swiss WEEE regulation, the next revisions of both ELV and WEEE European Directives will include certainly the management of automotive electronic wastes. this way, their management will become mandatory for all the actors involved in both ELV and WEEE recovery chains. Following this regulatory evolution, interesting researches could be done also in terms of comparison of alternative subsidy policies and impact of new policies on current way of doing of industrial actors. Together with technical issues, also economic, environmental and political ones could be a good ground for future researches and projects. All these considerations could be of utmost importance for governments and supranational organizations in defining new amendments of current international regulations, trying to make aware industrial companies about the importance of circular economy principles. However, the adoption of circularity principles depends by some relevant variables (dimensions) that are intrinsically related to each enterprise, like its product complexity and useful life and the value chain complexity. Unfortunately, a clear management of these variables is still lacking in the scientific literature. A good manner to enable to introduction of circular economy – and sustainability in general – within companies should be to make the research more exploitable by them. This way, more research efforts should be spent in the area of product end of life management, always considering the "cost impact" of the proposed solutions. Enterprises (in particular SMEs) are interested in having a short Return-On-Investment (ROI) for their projects and in a way for measuring it. Then, an interesting aspect to be analysed in the future is the cost and performance measurement for circular economy projects. A practical demonstration (e.g. through pilot plants) of the real potential of new recovery technologies – especially in economic terms – could be a good strategy going in this direction. In addition, best practices in circular economy should be tracked and good practices standardized and shared with other industrial sectors. The research community has a strong influence and role in sharing knowledge about circular economy and end-of-life management practices. Like evidenced also in this thesis, the absence of information and data in a particular research field implicitly transfer to companies a negative message, by making them (incorrectly) aware about the goodness of their current practices and abandoning any sort of innovative idea. This is enormously dangerous when these companies are playing with the environmental resources, making at risk the survival of our world. So, it is of utmost importance to deliver methodologies and technologies improving the sustainability level of enterprises involved in any kind of reverse logistic chain. Again, methodologies and technologies should be focused on a better integration of reverse logistic chains, even more made up of few medium-big companies and lots of SMEs. Their collaboration could take advantage from available competencies and flexibility, for properly answering to an increasing demand complexity. To this aim, the development of common reference database in which companies involved in any kind of reverse logistic chain is mandatory. Only this way they could gather more information on materials embedded into products, so enabling a more engineered and structured approach for the evaluation of different operational strategies. The existence of networked infrastructures could uniform also the decisional process.

6.6. General conclusions

As usual, it is not so easy to conclude such a kind of work. Lot of activities, experiences, meetings and persons are arising at the mind. The author hopes to have been clear, even if English is not his primary language. He hopes also to have contributed to the research community. Only the time will decide it.

7. References

Achillas, C., Moussiopoulos, N., Karagiannidis, A., Banias, G., Perkoulidis, G., 2013. The use of multi-criteria decision analysis to tackle waste management problems: a literature review. Waste Manag. Res. 31, 115–29. doi:10.1177/0734242X12470203

ACI, 2015. Italian circulating cars: detailed data [WWW Document].

ADA, 2015. Italian dismantlers data [WWW Document].

AIRA, 2015. Italian shredders data [WWW Document].

Amaral, J., Ferrão, P., Rosas, C., 2006. Is recycling technology innovation a major driver for technology shift in the automobile industry under an EU context? Int. J. Technol. Policy Manag. 6, 385–398. doi:10.1504/IJTPM.2006.011723

Andersen, F.M., Larsen, H. V, Skovgaard, M., Isoard, S., 2008. Projection of end-of-life vehicles - Development of a projection model and estimates of ELVs for 2005-2030. Copenhagen, Denmark.

ANFIA, 2015. Italian circulating cars: detailed data [WWW Document].

Ardente, F., Mathieux, F., Recchioni, M., 2014. Recycling of electronic displays: Analysis of pre-processing and potential ecodesign improvements. Resour. Conserv. Recycl. 92, 158–171. doi:10.1016/j.resconrec.2014.09.005

Asif, F.M.A., Rashid, A., Bianchi, C., Nicolescu, C.M., 2015. System dynamics models for decision making in product multiple lifecycles. Resour. Conserv. Recycl. 101, 20–33. doi:10.1016/j.resconrec.2015.05.002

Asif, F.M.A., Semere, D.., Nicolescu, C.., Haumann, M., 2010. Mehods analysis of remanufacturing options for repeated lifecycle of starters and alternators, in: DAAAM 2010 - 7th International Baltic Conference on Industrial Engineering. Tallinn.

Baldé, C.., Wang, F., Kuehr, R., Huisman, J., 2014. The Global E-Waste Monitor - 2014. United Nations University, IAS-SCYCLE, Bonn, Germany.

Behnamfard, A., Salarirad, M.M., Vegliò, F., 2013. Process development for recovery of copper and precious metals from waste printed circuit boards with emphasize on palladium and gold leaching and precipitation. Waste Manag. 33, 2354–2363. doi:10.1016/j.wasman.2013.07.017

Berzi, L., Delogu, M., Giorgetti, A., Pierini, M., 2013. On-field investigation and process modelling of End-of-Life Vehicles treatment in the context of Italian craft-type Authorized Treatment Facilities. Waste Manag. 33, 892–906. doi:10.1016/j.wasman.2012.12.004

Birloaga, I., Coman, V., Kopacek, B., Vegliò, F., 2014. An advanced study on the hydrometallurgical processing of waste computer printed circuit boards to extract their valuable content of metals. Waste Manag. 34, 2581–2586. doi:10.1016/j.wasman.2014.08.028

Birloaga, I., De Michelis, I., Ferella, F., Buzatu, M., Vegliò, F., 2013. Study on the influence of various factors in the hydrometallurgical processing of waste printed circuit boards for copper and gold recovery. Waste Manag. 33, 935–941. doi:10.1016/j.wasman.2013.01.003

Blume, T., Walther, M., 2013. The End-of-life Vehicle Ordinance in the German automotive industry - Corporate sense making illustrated. J. Clean. Prod. 56, 29–38. doi:10.1016/j.jclepro.2012.05.020

BMUB, 2016. German Resource Efficiency Programme II (ProgRess II), Programme for the sustainable use and conservation of natural resources.

Brealey, R.A., Myers, S.C., Allen, F., 2011. Principles of Corporate Finance, 5th editio. ed. McGraw Hill/Irwin, New York (USA).

Buekens, A., Zhou, X., 2014. Recycling plastics from automotive shredder residues: A review. J. Mater. Cycles Waste Manag. 16, 398–414. doi:10.1007/s10163-014-0244-z

Castro, M.B.G., Remmerswaal, J.A.M., Reuter, M.A., Boin, U.J.M., 2004. A thermodynamic approach to the compatibility of materials combinations for recycling. Resour. Conserv. Recycl. 43, 1–19. doi:10.1016/j.resconrec.2004.04.011

Chancerel, P., Rotter, V.S., 2009. Recycling-oriented characterization of small waste electrical and electronic equipment. Waste Manag. 29, 2336–2352. doi:10.1016/j.wasman.2009.04.003

Chatterjee, S., 2012. Sustainable Electronic Waste Management and Recycling Process. Am. J. Environ. Eng. 2, 23–33. doi:10.5923/j.ajee.20120201.05

Chau, K., Muttil, N., 2007. Data mining and multivariate statistical analysis for ecological system in coastal

waters. J. Hydroinformatics 9, 305. doi:10.2166/hydro.2007.003
Che, J., Yu, J. soo, Kevin, R.S., 2011. End-of-life vehicle recycling and international cooperation between Japan, China and Korea: Present and future scenario analysis. J. Environ. Sci. 23, S162–S166. doi:10.1016/S1001-0742(11)61103-0
Chen, Z., Chen, D., Wang, T., Hu, S., 2015. Policies on end-of-life passenger cars in China: Dynamic modeling and cost-benefit analysis. J. Clean. Prod. 108, 1140–1148. doi:10.1016/j.jclepro.2015.07.093
Cheng, Y.W., Cheng, J.H., Wu, C.L., Lin, C.H., 2012. Operational characteristics and performance evaluation of the ELV recycling industry in Taiwan. Resour. Conserv. Recycl. 65, 29–35. doi:10.1016/j.resconrec.2012.05.001
Chiaroni, D., Chiesa, V., Colasanti, L., Cucchiella, F., D'Adamo, I., Frattini, F., 2014. Evaluating solar energy profitability: A focus on the role of self-consumption. Energy Convers. Manag. 88, 317–331. doi:10.1016/j.enconman.2014.08.044
Clappier, M., Helbig, T., Königsberger, J., Schütten-Breitenbach, J., Taheri, K., 2014. A decision concept for the economic evaluation of different recycling paths in the dismantling of end-of-life vehicles. Front. Artif. Intell. Appl. 262, 38–47. doi:10.3233/978-1-61499-405-3-38
Coates, G., Rahimifard, S., 2008. A cost estimation framework to support increased value recovery from end-of-life vehicles. Int. J. Comput. Integr. Manuf. 21, 895–910. doi:10.1080/09511920801915218
Copani, G., Rosa, P., 2014. DEMAT: Sustainability assessment of new flexibility-oriented business models in the machine tools industry. Int. J. Comput. Integr. Manuf. 28. doi:10.1080/0951192X.2014.924160
Courtney, H., Kirkland, J., Viguerie, P., 1997. Strategy Under Uncertainty : Scenario Planning. Harv. Bus. Rev. November-D, 1–51.
Cucchiella, F., D'Adamo, I., 2013. Issue on supply chain of renewable energy. Energy Convers. Manag. 76, 774–780. doi:10.1016/j.enconman.2013.07.081
Cucchiella, F., D'Adamo, I., Gastaldi, M., 2014a. Sustainable management of waste-to-energy facilities. Renew. Sustain. Energy Rev. 33, 719–728. doi:10.1016/j.rser.2014.02.015
Cucchiella, F., D'Adamo, I., Gastaldi, M., 2014b. Strategic municipal solid waste management: A quantitative model for Italian regions. Energy Convers. Manag. 77, 709–720. doi:10.1016/j.enconman.2013.10.024
Cucchiella, F., D'Adamo, I., Gastaldi, M., D'Adamo, I., Koh, S.C.L., 2014c. Implementation of a real option in a sustainable supply chain: an empirical study of alkaline battery recycling. Int. J. Syst. Sci. 45, 1268–1282. doi:10.1080/00207721.2012.761458
Cucchiella, F., D'Adamo, I., Koh, S.C.L., 2015a. Environmental and economic analysis of building integrated photovoltaic systems in Italian regions. J. Clean. Prod. 98, 241–252. doi:10.1016/j.jclepro.2013.10.043
Cucchiella, F., D'Adamo, I., Koh, S.C.L., Rosa, P., 2015b. Recycling of WEEEs: An economic assessment of present and future e-waste streams. Renew. Sustain. Energy Rev. 51, 263–272. doi:10.1016/j.rser.2015.06.010
Cucchiella, F., D'Adamo, I., Rosa, P., 2015c. End-of-Life of used photovoltaic modules: A financial analysis. Renew. Sustain. Energy Rev. 47. doi:10.1016/j.rser.2015.03.076
Cucchiella, F., D'Adamo, I., Rosa, P., Terzi, S., 2016a. Scrap automotive electronics : A mini-review of current management practices. Waste Manag. Res. 34, 3–10. doi:10.1177/0734242X15607429
Cucchiella, F., D'Adamo, I., Rosa, P., Terzi, S., 2016b. Automotive Printed Circuit Boards Recycling: an Economic Analysis. J. Clean. Prod. 121, 130–141. doi:10.1016/j.jclepro.2015.09.122
D'Adamo, I., Rosa, P., 2016. Remanufacturing in industry: advices from the field. Int. J. Adv. Manuf. Technol. doi:10.1007/s00170-016-8346-5
Dalrymple, I., Wright, N., Kellner, R., Bains, N., Geraghty, K., Goosey, M., Lightfoot, L., 2007. An integrated approach to electronic waste (WEEE) recycling, C-Tech Innovation Ltd. doi:10.1108/03056120710750256
Demirel, E., Demirel, N., Gökçen, H., 2016. A mixed integer linear programming model to optimize reverse logistics activities of end-of-life vehicles in Turkey. J. Clean. Prod. 112, 2101–2113. doi:10.1016/j.jclepro.2014.10.079
Despeisse, M., Kishita, Y., Nakano, M., Barwood, M., 2015. Towards a circular economy for end-of-life vehicles: A comparative study UK - Japan, in: Procedia CIRP. Elsevier B.V., pp. 668–673. doi:10.1016/j.procir.2015.02.122
Diez, L., Marangé, P., Levrat, É., 2017. Regeneration Management Tool for Industrial Ecosystem. IFAC-

PapersOnLine 50, 12950–12955. doi:10.1016/j.ifacol.2017.08.1797
Dimitrakakis, E., Janz, A., Bilitewski, B., Gidarakos, E., 2009. Small WEEE: Determining recyclables and hazardous substances in plastics. J. Hazard. Mater. 161, 913–919. doi:10.1016/j.jhazmat.2008.04.054
Dong, X., Li, C., Li, J., Huang, W., Wang, J., Liao, R., 2012. Application of a system dynamics approach for assessment of the impact of regulations on cleaner production in the electroplating industry in China. J. Clean. Prod. 20, 72–81. doi:10.1016/j.jclepro.2011.08.014
Duan, C., Wen, X., Shi, C., Zhao, Y., Wen, B., He, Y., 2009. Recovery of metals from waste printed circuit boards by a mechanical method using a water medium. J. Hazard. Mater. 166, 478–482. doi:10.1016/j.jhazmat.2008.11.060
Duan, H., Hou, K., Li, J., Zhu, X., 2011. Examining the technology acceptance for dismantling of waste printed circuit boards in light of recycling and environmental concerns. J. Environ. Manage. 92, 392–399. doi:10.1016/j.jenvman.2010.10.057
El Halabi, E., Doolan, M., 2013a. Causal loops in automotive recycling, in: 56th Annual Meeting of the ISSS-2012. pp. 1–16. doi:10.1073/pnas.0703993104
El Halabi, E., Doolan, M., 2013b. Operational Challenges in the Automotive Recycling Business: A System Dynamics Perspective, in: 20th CIRP International Conference on Life Cycle Engineering. p. 5.
El Halabi, E., Doolan, M., Cardew-Hall, M., 2012. Extracting Variables and Causal Links from Interview Data. 30th Int. Conf. Syst. Dyn. Soc.
Ellen MacArthur, 2013. Towards the Circular Economy: Economic and business rationale for an accelerated transition. doi:10.1007/b116400
European Union, 2013. EU Directive 2013/28/EU of 17 May 2013 amending Annex II to Directive 2000/53/EC of the European Parliament and of the Council on end-of-life vehicles. Off. J. Eur. Union.
European Union, 2012. EU Directive 2012/19/EU of the European Parliament and of the Council of 4 July 2012 on waste electrical and electronic equipment (WEEE) (recast). Off. J. Eur. Union 1–34.
European Union, 2011. EU Directive 2011/65/EU of the European Parliament and of the Council of 8 June 2011 on the restriction of the use of certain hazardous substances in electrical and electronic equipment (recast). Off. J. Eur. Union. doi:10.1017/CBO9781107415324.004
European Union, 2005. EU Directive 2005/64/EC of the European Parliament and of the Council of 26 October 2005 on the type-approval of motor vehicles with regard to their reusability, recyclability and recoverability and amending Council. Off. J. Eur. Union L310.
European Union, 2000. EU Directive 2000/53/EC of the European Parliament and the Council of 18 September 2000 on end-of-life vehicles. Off. J. Eur. Union 34–43.
European Union regional policy, 2008. Guide to Cost Benefit Analysis of Investment Projects.
Eurostat, 2015a. Waste Electrical and Electronics Equipments: Detailed data [WWW Document].
Eurostat, 2015b. End-of-life vehicles: Detailed data [WWW Document].
Fan, S.K.S., Fan, C., Yang, J.H., Liu, K.F.R., 2013. Disassembly and recycling cost analysis of waste notebook and the efficiency improvement by re-design process. J. Clean. Prod. 39, 209–219. doi:10.1016/j.jclepro.2012.08.014
Farel, R., Yannou, B., Ghaffari, A., Leroy, Y., 2013. A cost and benefit analysis of future end-of-life vehicle glazing recycling in France: A systematic approach. Resour. Conserv. Recycl. 74, 54–65. doi:10.1016/j.resconrec.2013.02.013
Ferrão, P., Amaral, J., 2006. Assessing the economics of auto recycling activities in relation to European Union Directive on end of life vehicles. Technol. Forecast. Soc. Change 73, 277–289. doi:10.1016/j.techfore.2004.03.010
Ferrão, P., Nazareth, P., Amaral, J., 2006. Strategies for Meeting EU End-of-Life Vehicle Reuse / Recovery Targets. J. Ind. Ecol. 10, 77–93. doi:10.1162/jiec.2006.10.4.77
Fiore, S., Ruffino, B., Zanetti, M.C., 2012. Automobile Shredder Residues in Italy: Characterization and valorization opportunities. Waste Manag. 32, 1548–1559. doi:10.1016/j.wasman.2012.03.026
Fischer, C., Junker, H., Mazzanti, M., Paleari, S., Wuttke, J., Zoboli, R., 2012. Transboundary shipments of waste in the European Union Reflections on data, environmental impacts and drivers, European Environment Agency (EEA) report.
FISE UNIRE, 2015. L'Italia del riciclo 2015.
Freiberger, S., Albrecht, M., Käufl, J., 2011. New technologies for remanufacturing of automotive systems

communicating via CAN bus. J. Remanufacturing 1, 1–14. doi:10.1007/978-3-642-19692-8-16
Freiberger, S., Nagel, A., Steinhilper, R., 2012. WebCAN for Remanufacturers: A New Automotive CAN-Bus Tool Analyzing and File Sharing Application, in: Seliger, G. (Ed.), Sustainable Manufacturing. Springer-Verlag Berlin Heidelberg, pp. 143–148. doi:10.1007/978-3-642-27290-5
Garcia, J., Millet, D., Tonnelier, P., 2015. A tool to evaluate the impacts of an innovation on a product's recyclability rate by adopting a modular approach: automotive sector application. Int. J. Veh. Des. 67, 1–18. doi:10.1504/IJVD.2015.068143
Garetti, M., Rosa, P., Terzi, S., 2012. Life Cycle Simulation for the design of Product-Service Systems. Comput. Ind. 63. doi:10.1016/j.compind.2012.02.007
Gaustad, G., Olivetti, E., Kirchain, R.E., 2012. Improving aluminum recycling: A survey of sorting and impurity removal technologies. Resour. Conserv. Recycl. 58, 79–87. doi:10.1016/j.resconrec.2011.10.010
Gerrard, J., Kandlikar, M., 2007. Is European end-of-life vehicle legislation living up to expectations? Assessing the impact of the ELV Directive on "green" innovation and vehicle recovery. J. Clean. Prod. 15, 17–27. doi:10.1016/j.jclepro.2005.06.004
Ghosh, B., Ghosh, M.K., Parhi, P., Mukherjee, P.S., Mishra, B.K., 2015. Waste Printed Circuit Boards recycling: an extensive assessment of current status. J. Clean. Prod. 94, 5–19. doi:10.1016/j.jclepro.2015.02.024
Go, T.F., Wahab, D.A., Rahman, M.N.A., Ramli, R., Azhari, C.H., 2011. Disassemblability of end-of-life vehicle: A critical review of evaluation methods. J. Clean. Prod. 19, 1536–1546. doi:10.1016/j.jclepro.2011.05.003
Golinska, P., Kosacka, M., Relich, M., Peiró-Signes, A., Payá-Martínez, A., Segarra-Oña, M.-V., De-Miguel-Molina, M., Prendi, L., Che Wen Tseng, S., Tam, E.K.L., Rychlik, A., Jasiulewicz-Kaczmarek, M., Bocewicz, G., Sitek, P., Wikarek, J., Suzuki, K., Aiura, N., Pehlken, A., Kaerger, W., Chen, M., Mueller, D.H., Sawyer-Beaulieu, S.S., Stagner, J.A., 2014. Environmental Issues in Automotive Industry, EcoProduct. ed. Springer, Poznan, Poland. doi:10.1007/978-3-642-23837-6
Graedel, T.E., Allwood, J., Birat, J.-P., Buchert, M., Hagelüken, C., Reck, B.K., Sibley, S.F., Sonnemann, G., 2011. What Do We Know About Metal Recycling Rates? J. Ind. Ecol. 15, 355–366. doi:10.1111/j.1530-9290.2011.00342.x
Gu, Q., Gao, T., 2011. System dynamics analysis of RFID-EPC's impact on reverse supply chain, in: The 18th International Conference on Management Science and Engineering. IEEE Explore, Rome, Italy, pp. 250–255. doi:10.1109/ICMSE.2011.6069972
Guo, C., Wang, H., Liang, W., Fu, J., Yi, X., 2011. Liberation characteristic and physical separation of printed circuit board (PCB). Waste Manag. 31, 2161–2166. doi:10.1016/j.wasman.2011.05.011
Guo, J., Tang, Y., Xu, Z., 2010. Wood Plastic Composite Produced by Nonmetals from Pulverized Waste Printed Circuit Boards. Environ. Sci. Technol. 44, 463–468.
Hadi, P., Barford, J.P., McKay, G., 2013a. Toxic heavy metal capture using a novel electronic waste-based material - Mechanism, modeling and comparison. Environ. Sci. Technol. 47, 8248–8255. doi:10.1021/es4001664
Hadi, P., Gao, P., Barford, J.P., McKay, G., 2013b. Novel application of the nonmetallic fraction of the recycled printed circuit boards as a toxic heavy metal adsorbent. J. Hazard. Mater. 252–253, 166–170. doi:10.1016/j.jhazmat.2013.02.037
He, Y., Xu, Z., 2014. The status and development of treatment techniques of typical waste electrical and electronic equipment in China: a review. Waste Manag. Res. 32, 254–269. doi:10.1177/0734242X14525824
Hiratsuka, J., Sato, N., Yoshida, H., 2014. Current status and future perspectives in end-of-life vehicle recycling in Japan. J. Mater. Cycles Waste Manag. 16, 21–30. doi:10.1007/s10163-013-0168-z
Hu, S., Wen, Z., 2015. Why does the informal sector of end-of-life vehicle treatment thrive? A case study of China and lessons for developing countries in motorization process. Resour. Conserv. Recycl. 95, 91–99. doi:10.1016/j.resconrec.2014.12.003
Huang, K., Guo, J., Xu, Z., 2009. Recycling of waste printed circuit boards: A review of current technologies and treatment status in China. J. Hazard. Mater. 164, 399–408. doi:10.1016/j.jhazmat.2008.08.051
Huang, L., Wang, Q., 2007. The bullwhip effect in the closed loop supply chain, in: The 2007 International Conference of the Systems Dynamics Society. Boston, Massachusetts (USA), pp. 1–24.

Ijomah, W.L., Bennett, J.P., Pearce, J., 1999. Remanufacturing: evidence of environmentally conscious business practice in the UK, in: First International Symposium on Environmentally Conscious Design and Inverse Manufacturing - EcoDesign '99. pp. 192–196. doi:10.1109/ecodim.1999.747607

IMDS, 2015. Database.

Infomine, 2014. Website [WWW Document]. URL http://www.infomine.com

Jain, S., Lamba, T., Verma, V., Gupta, A., Agarwal, S., 2015. Simulated Annealing for Green Vehicle Routing Problem. Int. Conf. Adv. Res. Innov. 103–111.

Jalkanen, H., 2006. On the recycling of automotive shredder residue and electronic scrap in metallurgical industry. Acta Metall. Slovaca 12, 160–166.

Johansson, J., Luttropp, C., 2009. Material hygiene: improving recycling of WEEE demonstrated on dishwashers. J. Clean. Prod. 17, 26–35. doi:10.1016/j.jclepro.2008.02.010

Kamberović, Ž., Korać, M., Ranitović, M., 2011. Hydrometallurgical Process for Extraction of Metals from Eectronic Waste-Part II: Development of the Processes for the Recovery of Copper from Printed Circuit Boards (PCB). Metalurgija-MJoM 17, 139–149. doi:628.477.6

Kanari, N., Pineau, J.-L., Shallari, S., 2003. End-of-life vehicle recycling in the european union. JOM - J. Miner. Met. Mater. Soc. 55, 15–19. doi:10.1007/s11837-003-0098-7

Kasper, A.C., Berselli, G.B.T., Freitas, B.D., Tenório, J.A.S., Bernardes, A.M., Veit, H.M., 2011. Printed wiring boards for mobile phones: Characterization and recycling of copper. Waste Manag. 31, 2536–2545. doi:10.1016/j.wasman.2011.08.013

Keivanpour, S., Ait-Kadi, D., Mascle, C., 2017. Automobile manufacturers' strategic choice in applying green practices: joint application of evolutionary game theory and fuzzy rule-based approach. Int. J. Prod. Res. 55, 1312–1335. doi:10.1080/00207543.2016.1203077

Keivanpour, S., Kadi, D.A., Mascle, C., 2013a. Economic sustainability of end-of-life vehicle recycling infrastructure under uncertainty a fuzzy logic approach, in: 2013 International Conference on Industrial Engineering and Systems Management, IEEE - IESM 2013.

Keivanpour, S., Mascle, C., Ait-Kadi, D., 2013b. Modeling Robustness of Business Ecosystem of End of Life Vehicles Players. J. Sustain. Dev. Stud. 4, 1–35.

Kibira, D., Jain, S., 2011. Impact of hybrid and electric vehicles on automobile recycling infrastructure, in: 2011 Winter Simulation Conference (WSC). IEEE, Phoenix, AZ, pp. 1072–1083. doi:10.1017/CBO9781107415324.004

Kiddee, P., Naidu, R., Wong, M.H., 2013. Electronic waste management approaches: An overview. Waste Manag. 33, 1237–1250. doi:10.1016/j.wasman.2013.01.006

Kim, J.Y., Jeong, S.J., Cho, Y.J., Kim, K.S., 2014. Eco-friendly manufacturing strategies for simultaneous consideration between productivity and environmental performances: A case study on a printed circuit board manufacturing. J. Clean. Prod. 67, 249–257. doi:10.1016/j.jclepro.2013.12.043

Kim, K.H., Joung, H.T., Nam, H., Seo, Y.C., Hong, J.H., Yoo, T.W., Lim, B.S., Park, J.H., 2004. Management status of end-of-life vehicles and characteristics of automobile shredder residues in Korea. Waste Manag. 24, 533–540. doi:10.1016/j.wasman.2004.02.012

Kohlmeyer, R., 2012. End-of-life vehicles recycling in Germany and recent developments, in: International Workshop on 3R Strategy and ELV Recycling 2012. Nagoya - Japan, 19–21 September 2012.

Kripli, J., Vandenberg, R., Steinhilper, R., Freiberger, S., Weiland, F., 2010. Remanufacturing Automotive Mechatronics & Electronics.

Kumar, S., Yamaoka, T., 2007. System dynamics study of the Japanese automotive industry closed loop supply chain. J. Manuf. Technol. 18, 115–138. doi:http://dx.doi.org/10.1108/09564230910978511

Kumar, V., Sutherland, J.W., 2009. Development and assessment of strategies to ensure economic sustainability of the U.S. automotive recovery infrastructure. Resour. Conserv. Recycl. 53, 470–477. doi:10.1016/j.resconrec.2009.03.012

Kunze, P., 2012. Raw materials – challenges and solutions for the future?, in: US-EU Workshop on Mineral Raw Materials Data. Brussels - Belgium, 12 September 2012.

Larsson, M., Jansson, M., Grönkvist, S., Alvfors, P., 2015. Techno-economic assessment of anaerobic digestion in a typical Kraft pulp mill to produce biomethane for the road transport sector. J. Clean. Prod. 104, 460–467. doi:10.1016/j.jclepro.2015.05.054

Lecler, M.-T., Zimmermann, F., Silvente, E., Clerc, F., Chollot, A., Grosjean, J., 2015. Exposure to hazardous

substances in Cathode Ray Tube (CRT) recycling sites in France. Waste Manag. 39, 226–235. doi:10.1016/j.wasman.2015.02.027
Lee, C.-H., Truc, N.T.T., Lee, B.-K., Mitoma, Y., Mallampati, S.R., 2015. Evaluation of heavy metals in hazardous automobile shredder residue thermal residue and immobilization with novel nano-size calcium dispersed reagent. J. Hazard. Mater. 296, 239–247. doi:10.1016/j.jhazmat.2015.04.039
Lehr, C.B., Thun, J.-H., Milling, P.M., 2013. From waste to value – a system dynamics model for strategic decision-making in closed-loop supply chains. Int. J. Prod. Res. 51, 4105–4116. doi:10.1080/00207543.2013.774488
Li, J., Lopez N., B.N., Liu, L., Zhao, N., Yu, K., Zheng, L., 2013. Regional or global WEEE recycling. Where to go? Waste Manag. 33, 923–934. doi:10.1016/j.wasman.2012.11.011
Li, J., Shrivastava, P., Gao, Z., Zhang, H.-C., 2004. Printed Circuit Board Recycling: A State-of-the-Art Survey, in: IEEE Transactions on Electronics Packaging Manufacturing. pp. 33–42.
Li, J., Xu, Z., 2010. Environmental Friendly Automatic Line for Recovering Metal from Waste Printed Circuit Boards. Environ. Sci. Technol. 44, 1418–1423.
Li, J., Yu, K., Gao, P., 2014a. Recycling and pollution control of the End of Life Vehicles in China. J. Mater. Cycles Waste Manag. 16, 31–38. doi:10.1007/s10163-013-0226-6
Li, J., Zhao, N., Liu, X., Wu, X., 2014b. Promoting environmentally sound management of polybrominated diphenyl ethers in Asia. Waste Manag. Res. 32, 527–535. doi:10.1177/0734242X14534517
Li, W., Zhi, Y., Dong, Q., Liu, L., Li, J., Liu, S., Xie, H., 2012. Research Progress on the Recycling Technology for Nonmetallic Materials from Wasted Printed Circuit Board. Procedia Environ. Sci. 16, 569–575. doi:http://dx.doi.org/10.1016/j.proenv.2012.10.078
Liang, G., Mo, Y., Zhou, Q., 2010. Novel strategies of bioleaching metals from printed circuit boards (PCBs) in mixed cultivation of two acidophiles. Enzyme Microb. Technol. 47, 322–326. doi:10.1016/j.enzmictec.2010.08.002
LME, 2014. Website [WWW Document]. URL https://www.lme.com
Lu, Y., Broughton, J., Winfield, P., 2014. A review of innovations in disbonding techniques for repair and recycling of automotive vehicles. Int. J. Adhes. Adhes. 50, 119–127. doi:10.1016/j.ijadhadh.2014.01.021
Mancini, G., Viotti, P., Luciano, A., Fino, D., 2014. On the ASR and ASR thermal residues characterization of full scale treatment plant. Waste Manag. 34, 448–457. doi:10.1016/j.wasman.2013.11.002
Mankhand, T.R., K. Singh, K., Kumar Gupta, S., Das, S., 2013. Pyrolysis of Printed Circuit Boards. Int. J. Metall. Eng. 1, 102–107. doi:10.5923/j.ijmee.20120106.01
Mazzanti, M., Zoboli, R., 2006. Economic instruments and induced innovation: The European policies on end-of-life vehicles. Ecol. Econ. 58, 318–337. doi:10.1016/j.ecolecon.2005.06.008
McKenna, R., Reith, S., Cail, S., Kessler, A., Fichtner, W., 2013. Energy savings through direct secondary reuse: An exemplary analysis of the German automotive sector. J. Clean. Prod. 52, 103–112. doi:10.1016/j.jclepro.2013.02.032
Metalprices, 2014. Website [WWW Document]. URL http://metalprices.com
MIT, 2015. Italian circulating cars: detailed data [WWW Document].
Morselli, L., Santini, A., Passarini, F., Vassura, I., 2010. Automotive shredder residue (ASR) characterization for a valuable management. Waste Manag. 30, 2228–2234. doi:10.1016/j.wasman.2010.05.017
Muttil, N., Chau, K.-W., 2007. Machine-learning paradigms for selecting ecologically significant input variables. Eng. Appl. Artif. Intell. 20, 735–744. doi:10.1016/j.engappai.2006.11.016
Muttil, N., Chau, K.-W., 2006. Neural Network and Genetic Programming for Modelling Coastal Algal Blooms. Int. J. Environ. Pollut. 28, 223–238.
National Instruments, 2009. ECU Designing and Testing Using National Instruments Products.
Ni, F., Chen, M., 2014. Research on ASR in China and its energy recycling with pyrolysis method. J. Mater. Cycles Waste Manag. 17, 107–117. doi:10.1007/s10163-014-0232-3
Niu, Q., Liu, X., Shi, C., Xiang, D., Duan, G., 2007. The recycle model of printed circuit board and its economy evaluation, in: IEEE International Symposium on Electronics and the Environment. pp. 106–111. doi:10.1109/ISEE.2007.369376
Ongondo, F.O., Williams, I.D., Cherrett, T.J., 2011. How are WEEE doing? A global review of the management of electrical and electronic wastes. Waste Manag. 31, 714–730. doi:10.1016/j.wasman.2010.10.023
Pan, S.-Y., Du, M.A., Huang, I.-T., Liu, I.-H., Chang, E.-E., Chiang, P.C., 2015. Strategies on implementation

of waste-to-energy (WTE) supply chain for circular economy system: A review. J. Clean. Prod. 108, 409–421. doi:10.1016/j.jclepro.2015.06.124

Park, J.W., Yi, H.C., Park, M.W., Sohn, Y.T., 2014. A monitoring system architecture and calculation of practical recycling rate for end-of-life vehicle recycling in Korea. Int. J. Precis. Eng. Manuf. - Green Technol. 1, 49–57. doi:10.1007/s40684-014-0008-1

Peters, S., Lanza, G., Ni, J., Xiaoning, J., Pei-Yun, Y., Colledani, M., 2014. Automotive manufacturing technologies – an international viewpoint. Manuf. Rev. 1, 10. doi:10.1051/mfreview/2014010

Raboni, M., Torretta, V., Urbini, G., Viotti, P., 2015. Automotive shredder residue: A survey of the hazardous organic micro-pollutants spectrum in landfill biogas. Waste Manag. Res. 33, 48–54. doi:10.1177/0734242X14559300

Raugei, M., El Fakir, O., Wang, L., Lin, J., Morrey, D., 2014. Life cycle assessment of the potential environmental benefits of a novel hot forming process in automotive manufacturing. J. Clean. Prod. 83, 80–86. doi:10.1016/j.jclepro.2014.07.037

Reichel, A., Saki, Ö., Fischer, C., Milios, L., Ryberg, M., 2012. Movements of waste across the EU's internal and external borders, EEA (European Environment Agency) report. doi:10.2800/62637

Reuter, M.A., Hudson, C., van Schaik, A., Heiskanen, K., Meskers, C., Hagelüken, C., 2013. Metal Recycling: Opportunities, Limits, Infrastructure, A report of the Working Group on the Global Metal Flows to the International Resource Panel. UNEP.

Reuter, M.A., Van Schaik, A., Ignatenko, O., De Haan, G.J., 2006. Fundamental limits for the recycling of end-of-life vehicles. Miner. Eng. 19, 433–449. doi:10.1016/j.mineng.2005.08.014

Richa, K., Babbitt, C.W., Gaustad, G., Wang, X., 2014. A future perspective on lithium-ion battery waste flows from electric vehicles. Resour. Conserv. Recycl. 83, 63–76. doi:10.1016/j.resconrec.2013.11.008

Rocchetti, L., Vegliò, F., Kopacek, B., Beolchini, F., 2013. Environmental impact assessment of hydrometallurgical processes for metal recovery from WEEE residues using a portable prototype plant. Environ. Sci. Technol. 47, 1581–1588. doi:10.1021/es302192t

Rochat, D., Hagelüken, C., Keller, M., Widmer, R., 2007. Optimal recycling for printed wiring boards (PWBs) in India, in: R'07 Conference on Recovery of Materials and Energy for Resource Efficiency.

Rondini, A., Tornese, F., Gnoni, M.G., Pezzotta, G., Pinto, R., 2017. Hybrid simulation modelling as a supporting tool for sustainable product service systems: a critical analysis. Int. J. Prod. Res. 7543, 1–14. doi:10.1080/00207543.2017.1330569

Rosa, P., Terzi, S., 2016. Comparison of current practices for a combined management of printed circuit boards from different waste streams. J. Clean. Prod. 137, 300–312. doi:10.1016/j.jclepro.2016.07.089

Rosa, P., Terzi, S., 2014. Proposal of a global product development strategy assessment tool and its application in the Italian industrial context. Int. J. Prod. Lifecycle Manag. 7. doi:10.1504/IJPLM.2014.066810

Sakai, S. ichi, Yoshida, H., Hiratsuka, J., Vandecasteele, C., Kohlmeyer, R., Rotter, V.S., Passarini, F., Santini, A., Peeler, M., Li, J., Oh, G.J., Chi, N.K., Bastian, L., Moore, S., Kajiwara, N., Takigami, H., Itai, T., Takahashi, S., Tanabe, S., Tomoda, K., Hirakawa, T., Hirai, Y., Asari, M., Yano, J., 2014. An international comparative study of end-of-life vehicle (ELV) recycling systems. J. Mater. Cycles Waste Manag. 16, 1–20. doi:10.1007/s10163-013-0173-2

Sakai, S.I., Noma, Y., Kida, A., 2007. End-of-life vehicle recycling and automobile shredder residue management in Japan. J. Mater. Cycles Waste Manag. 9, 151–158. doi:10.1007/s10163-007-0180-2

Santochi, M., Dini, G., Failli, F., 2002. Computer Aided Disassembly Planning: State of the Art and Perspectives. CIRP Ann. - Manuf. Technol. 51, 507–529. doi:10.1016/S0007-8506(07)61698-9

Schmid, A., Naquin, P., Gourdon, R., 2013. Incidence of the level of deconstruction on material reuse, recycling and recovery from end-of life vehicles: An industrial-scale experimental study. Resour. Conserv. Recycl. 72, 118–126. doi:10.1016/j.resconrec.2013.01.007

Simic, V., 2015. Fuzzy risk explicit interval linear programming model for end-of-life vehicle recycling planning in the EU. Waste Manag. 35, 265–282. doi:10.1016/j.wasman.2014.09.013

Simic, V., Dimitrijevic, B., 2012. Modelling production processes in a vehicle recycling plant. Waste Manag. Res. 30, 940–948. doi:10.1177/0734242X12454695

Sohaili, J., Muniyandi, S.K., Mohamad, S.S., 2012. A Review on Printed Circuit Board Recycling Technology. J. Emerg. Trends Eng. Appl. Sci. 3, 12–18.

Song, Q., Wang, Z., Li, J., Zeng, X., 2013. The life cycle assessment of an e-waste treatment enterprise in

China. J. Mater. Cycles Waste Manag. 15, 469–475. doi:10.1007/s10163-013-0152-7

Soo, V.K., Compston, P., Doolan, M., 2015. Interaction between New Car Design and Recycling Impact on Life Cycle Assessment, in: The 22nd CIRP Conference on Life Cycle Engineering. Elsevier B.V., pp. 426–431. doi:10.1016/j.procir.2015.02.055

Steinhilper, R., Freiberger, S., Nagel, A., Staarman, L.K., Kohler, D., Tom, E.R., de Winter, S., Weiland, F., 2012. European Automotive Remanufacturing. FJW Consulting.

Steinhilper, R., Rosemann, B., Freiberger, S., 2006. Product and Process Assessment for Remanufacturing of Computer Controlled Automotive Components, in: 13th CIRP International Conference on Life Cycle Engineering. pp. 441–446.

Sterman, J.D., 2000. Business Dynamics: Systems Thinking and Modeling for a Complex World, Management. Irwin McGraw-Hill, Boston, Massachusetts (USA). doi:10.1108/13673270210417646

Sthiannopkao, S., Wong, M.H., 2013. Handling e-waste in developed and developing countries: Initiatives, practices, and consequences. Sci. Total Environ. 463–464, 1147–1153. doi:10.1016/j.scitotenv.2012.06.088

Subramoniam, R., Huisingh, D., Chinnam, R.B., Subramoniam, S., 2013. Remanufacturing Decision-Making Framework (RDMF): Research validation using the analytical hierarchical process. J. Clean. Prod. 40, 212–220. doi:10.1016/j.jclepro.2011.09.004

Taylor, R., Ray, R., Chapman, C., 2013. Advanced thermal treatment of auto shredder residue and refuse derived fuel. Fuel 106, 401–409. doi:10.1016/j.fuel.2012.11.071

Tian, J., Chen, M., 2014. Sustainable design for automotive products: Dismantling and recycling of end-of-life vehicles. Waste Manag. 34, 458–467. doi:10.1016/j.wasman.2013.11.005

Tian, J., Ni, F., Chen, M., 2015. Application of pyrolysis in dealing with end-of-life vehicular products: A case study on car bumpers. J. Clean. Prod. 108, 1177–1183. doi:10.1016/j.jclepro.2015.05.096

Tytgat, J., 2013. The recycling efficiency of Li-ion EV batteries according to the European Commission Regulation, and the relation with the end-of-life vehicles directive recycling rate. World Electr. Veh. J. 6, 1039–1047. doi:10.1109/EVS.2013.6914885

Uan, J.Y., Cho, C.Y., Liu, K.T., 2007. Generation of hydrogen from magnesium alloy scraps catalyzed by platinum-coated titanium net in NaCl aqueous solution. Int. J. Hydrogen Energy 32, 2337–2343. doi:10.1016/j.ijhydene.2007.03.014

UNEP-DTIE-IETC, 2012. E-waste Volume III: WEEE/E-waste “Take-back system.”

van Schaik, A., Reuter, M.A., 2004. The time-varying factors influencing the recycling rate of products. Resour. Conserv. Recycl. 40, 301–328. doi:10.1016/S0921-3449(03)00074-0

Vermeulen, I., Van Caneghem, J., Block, C., Baeyens, J., Vandecasteele, C., 2011. Automotive shredder residue (ASR): Reviewing its production from end-of-life vehicles (ELVs) and its recycling, energy or chemicals’ valorisation. J. Hazard. Mater. 190, 8–27. doi:10.1016/j.jhazmat.2011.02.088

Viganò, F., Consonni, S., Grosso, M., Rigamonti, L., 2010. Material and energy recovery from Automotive Shredded Residues (ASR) via sequential gasification and combustion. Waste Manag. 30, 145–153. doi:10.1016/j.wasman.2009.06.009

Vlachos, D., Georgiadis, P., Iakovou, E., 2007. A system dynamics model for dynamic capacity planning of remanufacturing in closed-loop supply chains. Comput. Oper. Res. 34, 367–394. doi:10.1016/j.cor.2005.03.005

Wang, J., Chen, M., 2013a. Technology innovation of used automotive electronic control components recycling in China. Adv. Mater. Res. 610–613, 2346–2349.

Wang, J., Chen, M., 2013b. Remanufacturing process for used automotive electronic control components in China. J. Remanufacturing 3, 1–17. doi:10.1186/2210-4690-3-9

Wang, J., Chen, M., 2012. Management status of end-of-life vehicles and development strategies of used automotive electronic control components recycling industry in China. Waste Manag. Res. 30, 1198–207. doi:10.1177/0734242X12453976

Wang, J., Chen, M., 2011. Recycling of electronic control units from end-of-life vehicles in China. JOM - J. Miner. Met. Mater. Soc. 63, 42–47. doi:10.1007/s11837-011-0136-9

Wang, J., Li, Y., Song, J., He, M., Song, J., Xia, K., 2015. Recycling of acrylonitrile–butadiene–styrene (ABS) copolymers from waste electrical and electronic equipment (WEEE), through using an epoxy-based chain extender. Polym. Degrad. Stab. 112, 167–174. doi:10.1016/j.polymdegradstab.2014.12.025

Wang, J., Wang, L., Chen, M., 2012. Automotive Electronic Control Components Energy Consumption and Environmental Emissions Analysis in China Based on Economic Input-Output Life-Cycle Assessment Model. Adv. Mater. Res. 479, 2177–2181.

Wang, J., Xu, Z., 2015. Disposing and Recycling Waste Printed Circuit Boards: Disconnecting, Resource Recovery, and Pollution Control. Environ. Sci. Technol. 49, 721–733. doi:10.1021/es504833y

Wang, L., Chen, M., 2013. Policies and perspective on end-of-life vehicles in China. J. Clean. Prod. 44, 168–176. doi:10.1016/j.jclepro.2012.11.036

Wang, X., Gaustad, G., 2012. Prioritizing material recovery for end-of-life printed circuit boards. Waste Manag. 32, 1903–1913. doi:10.1016/j.wasman.2012.05.005

Wang, Y., Chang, X., Chen, Z., Zhong, Y., Fan, T., 2014. Impact of subsidy policies on recycling and remanufacturing using system dynamics methodology: a case of auto parts in China. J. Clean. Prod. 74, 161–171. doi:10.1016/j.jclepro.2014.03.023

Weigel, M., Fischedick, M., Marzinkowski, J., Winzer, P., 2016. Multicriteria analysis of primary steelmaking technologies. J. Clean. Prod. 112, 1064–1076. doi:10.1016/j.jclepro.2015.07.132

Williams, J.A.S., Wongweragiat, S., Qu, X., McGlinch, J.B., Bonawi-tan, W., Choi, J.K., Schiff, J., 2006. An automotive bulk recycling planning model. Eur. J. Oper. Res. 177, 969–981. doi:10.1016/j.ejor.2006.01.031

Wu, C.L., Chau, K.-W., 2006. Mathematical model of water quality rehabilitation with rainwater utilization - a case study at Haigang. Int. J. Environ. Pollut. 28, 534–545.

Xie, J.-X., Cheng, C.-T., Chau, K.-W., Pei, Y.-Z., 2006. A hybrid adaptive time-delay neural network model for multi-step-ahead prediction of sunspot activity. Int. J. Environ. Pollut. 28, 364–381. doi:10.1504/IJEP.2006.011217

Xie, Y.H., Yu, H.J., Li, C.D., 2014. Present situation and prospect of lithium-ion traction batteries for electric vehicles domestic and overseas standards, in: IEEE Transportation Electrification Conference and Expo, ITEC Asia-Pacific 2014 - Conference Proceedings. pp. 1–4. doi:10.1109/ITEC-AP.2014.6940614

Xu, Y., Sanchez, J.F., Njuguna, J., 2014. Cost modelling to support optimised selection of End-of-Life options for automotive components. Int. J. Adv. Manuf. Technol. 73, 399–407. doi:10.1007/s00170-014-5804-9

Xue, M., Li, J., Xu, Z., 2013. Management strategies on the industrialization road of state-of- the-art technologies for e-waste recycling: the case study of electrostatic separation--a review. Waste Manag. Res. 31, 130–140. doi:10.1177/0734242X12465464

Xue, M., Li, J., Xu, Z., 2012a. Environmental friendly crush-magnetic separation technology for recycling metal-plated plastics from end-of-life vehicles. Environ. Sci. Technol. 46, 2661–2667. doi:10.1021/es202886a

Xue, M., Yang, Y., Ruan, J., Xu, Z., 2012b. Assessment of noise and heavy metals (Cr, Cu, Cd, Pb) in the ambience of the production line for recycling waste printed circuit boards. Environ. Sci. Technol. 46, 494–499. doi:10.1021/es202513b

Yahaya, N.R., 2012. Environmental Impact of Electricity Consumption in Crushing and Grinding Processes of Traditional and Urban Gold Mining by Using Life Cycle Assessment (LCA). Iran. J. Energy Environ. 3, 66–73. doi:10.5829/idosi.ijee.2012.03.05.11

Yamane, L.H., de Moraes, V.T., Espinosa, D.C.R., Tenório, J.A.S., 2011. Recycling of WEEE: Characterization of spent printed circuit boards from mobile phones and computers. Waste Manag. 31, 2553–2558. doi:10.1016/j.wasman.2011.07.006

Yang, H., Liu, J., Yang, J., 2011. Leaching copper from shredded particles of waste printed circuit boards. J. Hazard. Mater. 187, 393–400. doi:10.1016/j.jhazmat.2011.01.051

Yano, J., Muroi, T., Sakai, S. ichi, 2015. Rare earth element recovery potentials from end-of-life hybrid electric vehicle components in 2010-2030. J. Mater. Cycles Waste Manag. 18, 655–664. doi:10.1007/s10163-015-0360-4

Yu, J., Williams, E., Ju, M., 2009. Review and prospects of recycling methods for waste printed circuit boards, in: IEEE International Symposium on Sustainable Systems and Technology, ISSST ’09 in Cooperation with 2009 IEEE International Symposium on Technology and Society, ISTAS. doi:10.1109/ISSST.2009.5156727

Yuan, H.P., Shen, L.Y., Hao, J.J.L., Lu, W.S., 2011. A model for cost-benefit analysis of construction and demolition waste management throughout the waste chain. Resour. Conserv. Recycl. 55, 604–612.

doi:10.1016/j.resconrec.2010.06.004
Zamudio-Ramirez, P., 1996. Economics of Automobile Recycling. Massachusetts Institute of Technology; “Unpublished results.”
Zeng, X., Li, J., Xie, H., Liu, L., 2013. A novel dismantling process of waste printed circuit boards using water-soluble ionic liquid. Chemosphere 93, 1288–1294. doi:10.1016/j.chemosphere.2013.06.063
Zeng, X., Song, Q., Li, J., Yuan, W., Duan, H., Liu, L., 2015. Solving e-waste problem using an integrated mobile recycling plant. J. Clean. Prod. 90, 55–59. doi:10.1016/j.jclepro.2014.10.026
Zhang, H., Chen, M., 2014. Current recycling regulations and technologies for the typical plastic components of end-of-life passenger vehicles: A meaningful lesson for China. J. Mater. Cycles Waste Manag. 16, 187–200. doi:10.1007/s10163-013-0180-3
Zhao, M.-Y., Cheng, C.-T., Chau, K.-W., Li, G., 2006. Multiple criteria data envelopment analysis for full ranking units associated to environment impact assessment. Int. J. Environ. Pollut. 28, 448–464.
Zhao, Q., Chen, M., 2011. A comparison of ELV recycling system in China and Japan and China’s strategies. Resour. Conserv. Recycl. 57, 15–21. doi:10.1016/j.resconrec.2011.09.010
Zhou, X., Guo, J., Lin, K., Huang, K., Deng, J., 2013. Leaching characteristics of heavy metals and brominated flame retardants from waste printed circuit boards. J. Hazard. Mater. 246–247, 96–102. doi:10.1016/j.jhazmat.2012.11.065
Zhou, Z., Dai, G., 2012. Research of Flexible Dismantling Cell for End-of-Life Vehicle Recycling. ICEWE 2012 - Int. Conf. Ecol. Waste Recycl. Environ. 7, 73–79.
Zhu, N., Xiang, Y., Zhang, T., Wu, P., Dang, Z., Li, P., Wu, J., 2011. Bioleaching of metal concentrates of waste printed circuit boards by mixed culture of acidophilic bacteria. J. Hazard. Mater. 192, 614–619. doi:10.1016/j.jhazmat.2011.05.062
Zorpas, A.A., Inglezakis, V.J., 2012. Automotive industry challenges in meeting EU 2015 environmental standard. Technol. Soc. 34, 55–83. doi:10.1016/j.techsoc.2011.12.006

List of acronyms

ANN	Artificial Neural Network	ICT	Information and Communication Technologies
ASR	Automotive Shredder Residue	IRR	Internal Rate of Return
ATF	Authorized Treatment Facility	NPV	Net Present Value
CRM	Critical Raw Material	OEM	Original Equipment Manufacturer
DCF	Discounted Cash Flow	PBT	Pay Back Time
Df3R	Design for Reuse, Remanufacturing and Recycling	PCB	Printed Circuit Board
DFD	Design For Disassembly	PELM	Product End-of-Life Management
DPBT	Discounted Pay Back Time	PGM	Platinum Group Metals
ECU	Electronic Control Unit	REE	Rare Earth Elements
ELV	End-of-Life Vehicle	SCE	Standard Coal Equivalent
EoL	End of Life	SD	System Dynamics
EU	European Union	SME	Small and Medium Enterprises
GDP	Gross Domestic Product	WEEE	Waste Electrical and Electronic Equipments
GHG	Green House Gas	WPCB	Wasted Printed Circuit Boards