A costing framework to support a sustainable approach to end-of-life vehicle recovery

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A Costing Framework to Support a Sustainable Approach to End-of-Life Vehicle Recovery

by

Gareth Coates

A Doctoral Thesis
Submitted in Partial Fulfilment of the Requirements for the Award of Doctor of Philosophy of Loughborough University

Wolfson School of Mechanical and Manufacturing Engineering
July 2007

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ACKNOWLEDGEMENTS

I would like to express my sincere thanks to my supervisor Dr. Shahin Rahimifard, whose tireless support, effort and patience has been a dependable source of insight and advice. I would also like to thank Prof. Steve Newman, Dr. Paul Leaney and Dr. Tracy Bhamra for their early guidance.

I would also like to offer my gratitude to my fellow researchers, especially Chris Edwards, Theodoros Staikos, Muhammad Abu Bakar, Rob Darlington and Srinivas that have offered me friendship and enjoyment throughout my research.

I would like to acknowledge the Wolfson School of Mechanical and Manufacturing Engineering and the Engineering and Physical Science Research Council for their financial support during my research studies, and the technical & administration support staff, particularly Clive Turner, David Walters, Sally Mitton, Bridgett Shipman and Jo Mason.

My thanks also go out to the following industrialist for giving so much of their valuable time; Paul Owen and all the staff of Rozone Ltd., Derek Wilkins, Ray Kirk, Dawn Allan, Andy Langridge, Peter Norgrove, Diana McAfee, Pat Winfield and Paul Farquharson.

Finally, I would like to thank my family for all their unbelievable love and support during my research. To my parents Jennifer Barnard and Howard Coates that have always motivated and encouraged me on whichever path my life has taken. Last but by no means least, I would like to thank the love and support of my future wife, Rebecca Clark. Whose patience and love is equalled only by her enthusiasm and kindness.
SYNOPSIS

This thesis reports on the research undertaken to analyse the factors affecting end-of-life vehicle value, and to investigate a costing framework to assist the vehicle recovery industry in promoting sustainable vehicle recycling. The principal objective of this research is to develop decision support tools for the vehicle recovery sector to adopt more sustainable processing strategies, whilst meeting the requirements of impending and future legislative targets.

The research contributions are divided into three parts. The first part reviews the most relative research in the areas of environmental concerns relating to the automotive sector, end-of-life vehicle recovery and associated costing techniques, to identify the most relevant research directions. The second part consists of a substantial program of data collection, which included; formal interviews, survey of treatment facilities, time-studies and vehicle teardowns, to generate a costing framework for the modelling of indirect and direct costs of both pre and post-fragmentation activities in vehicle recycling. The third part includes the design and implementation of a decision support costing system that enables end-of-life stakeholders to understand the main economics that underpin their operations, and to support future investment in more sustainable vehicle recovery activities.

The applicability of the research concept has been demonstrated via three case studies. The results from the case studies have shown that although most end-of-life vehicle recoverers are currently profitable due to the strong demand for scrap metal, significant improvement in their processes and value recovery is possible through strategic investment. Such strategic investment in process improvement and expansion of recycling activities should be considered in light of future fluctuations in material markets and increasing costs of attaining higher recycling targets.

In summary, this research has concluded that the realisation of environmentally friendly approaches to vehicle recycling and the long-term survival of the ELV recovery sector is very much dependent on the pro-active and direct involvement of automotive manufacturers in end-of-life vehicle recovery.
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<td>ABC</td>
<td>Activity Based Costing</td>
</tr>
<tr>
<td>ATF</td>
<td>Authorised Treatment Facility</td>
</tr>
<tr>
<td>CBR</td>
<td>Case Based Reasoning</td>
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<tr>
<td>CDF</td>
<td>Cumulative Distribution Function</td>
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<tr>
<td>CER</td>
<td>Cost Estimate Relationships</td>
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<tr>
<td>DfR</td>
<td>Design for Recycling</td>
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<td>DfS</td>
<td>Design for Serviceability</td>
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<td>DMS</td>
<td>Dense Media Separators</td>
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<td>ELV</td>
<td>End-of-Life Vehicles</td>
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<td>EOL</td>
<td>End-of-Life</td>
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<td>EPR</td>
<td>Extended Producer Responsibility</td>
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<tr>
<td>IDIS</td>
<td>International Dismantling Information System</td>
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<td>MRR</td>
<td>Material Removal Rate</td>
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<td>OEM</td>
<td>Original Equipment Manufacturer</td>
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<td>PDF</td>
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Chapter 1

Introduction

Sustainability will be one of the core themes of the 21st century, as virgin materials become increasingly scarce and the ecological footprint of products more apparent. Environmental legislation is attempting to address this, and is becoming progressively more prevalent throughout the developed world. The physical interpretation of the "polluter-pays" principal has meant that manufacturers and businesses are becoming ever more accountable for their products environmental effects beyond the traditional boundaries of the product development process. End-of-life disposal and product take-back legislation has taken a proactive stance and has formulated a number of prescriptive European directives encompassing the design, production and end-of-life treatment of a range of products. The automobile, through the End-of-Life Vehicle (ELV) directive has become one of the first consumer products to be actively legislated against under this new wave of Extended Producer Responsibility (EPR) legislation.

Over two million ELVs are produced in the UK each year, containing a range of metallic, ceramic and polymeric materials. The recovery and recycling of these materials at end-of-life has the potential to substantially improve the sustainability of the automobile through resource conservation and waste minimisation. Yet, at present ELV recycling is undertaken by an industry relatively unprepared and unfamiliar with a vehicles manufacturing processes and material composition. The ELV directive has therefore attempted to put right this imbalance and bring vehicle manufacturers closer to the recovery of their products to facilitate a more sustainable closed-loop approach.

The existing vehicle recovery industry is predominantly led by a handful of large metal merchants that are primarily interested in recovering the metallic fraction from ELVs. Geographically distributed scrap-yards (or authorised treatment facilities) serve as collection hubs for these large operators, and are required to make the vehicle environmentally safe via a process of "de-pollution", before passing the vehicles on to
the metal merchants for shredding. Unlike the automotive “supply chain” the vehicle “recovery chain” is a reactive industry, able to function on uncertain return volumes and fluctuating material and part prices. Coupled with the fact that manufacturers have traditionally seen the remit of their responsibilities and influence ending at point of sale, the recovery industry has not seen the technical innovation and drive for efficiency seen within other parts of the vehicle “value chain”. The current transposition of the ELV directive into UK law is set to change this, and forced dramatic reform and investment in an otherwise archaic industry.

The ELV directive requires vehicle manufacturers to provide free take-back and treatment for all its own vehicles post 2006, and meet stringent recycling and recovery targets of 85% and 95% by 2006 and 2015 respectively. Vehicle manufacturers have opted to conform to the legislation by moving away from actively fulfilling the requirements of the directive themselves, in favour of utilising the existing infrastructure and waste reclamation processes within the UK. This has lead to the establishment of “collection contracts”, whereby the existing vehicle recovery industry has agreed to fulfil the requirements laid down by the ELV directive on the vehicle manufacturer’s behalf. The economic support required to fund such an undertaking is estimated to be in the region of £160-£340 million per year from 2006 onwards (DTI 2004). It was widely believed that this would be subsidised by the vehicle manufacturers, yet during the establishment of these manufacturer collection contracts it became apparent that no direct financial support would be paid to the vehicle recovery sector, given the wealth of intrinsic value ELVs possessed at the time of the contract negotiations.

Not only must the recovery sector recoup the financial investment it has made during the directives transposition, but it must also continue to improve its recovery effectiveness in the light of its new legislative commitments. All this points to more elaborate and complex end-of-life processing. Potentially shifting the recovery operators focus away from self-directed environmental improvements, and more towards profit and business survival. Without a subsidised influence from the vehicle manufacturers any end-of-life processing decisions will be based solely on economic merit as opposed to any long-term environmental benefits. This lack of financial
support for the vehicle recovery industry is further compounded by its reactive nature and inexperience in having to adapt and improve its operations. The UK transposition of the ELV directive has therefore done little to strengthen the relationships between the vehicle recovery chain and the vehicle manufacturers, and it can be argued that it has been counter productive to the core themes of long term sustainability.

The research assertion made in this thesis is that the vehicle recovery sector will only promote sustainable vehicle recovery if there is an economic incentive to do so; it is therefore an essential need not only to assist in the increased realisation of end-of-life value, but also to provide economic decision-support to assess any environmentally beneficial alternative processing routes. It is proposed that the first step in achieving this is the development of a costing framework, which is capable of accurately assessing current end-of-life processing economics. The challenge with developing such a framework is that it must not only account for the variations in operating procedure from facility to facility, but also allow for the fluctuations in bespoke end-of-life markets. Hence, this research has considered the creation of an holistic framework, consisting of a number of costing techniques to assess the value added processing of various end-of-life operators.

The overall aim of the thesis is therefore to analyse the fundamental factors affecting end-of-life value, and to investigate a costing framework to assist the current vehicle recovery industry in assessing its ability to promote sustainable vehicle recovery under the constraints of current and future legislative targets.

The research issues addressed in this thesis are:-

- The generation of a framework to cost the economic operations carried out by various end-of-life stakeholders, both pre and post-fragmentation.
- The design and specification of an end-of-life costing model that can highlight the interactions of processing decisions in terms of economic cost.
- The creation of a number of end-of-life vehicle processing strategies that promote sustainability while still maintaining economic viability within the current markets.
Chapter I

The structure of this thesis is broken down into three different sections; research background and overview, theoretical research and model development and research conclusions, as highlighted within Figure 1.1.

The research background and overview section encompasses chapters one to six and provides an introduction to a range of issues regarding vehicle recovery, recycling and costing. Chapter 2 provides a detailed insight into the context of the research and outlines the research aim and objectives. Chapter 3 introduces the reader to the current vehicle recovery sector within the UK, and the main stakeholders, activities and legislation that are currently shaping the industry. Chapter 4 discusses environmental concerns within the automotive sector, and maps the waste hierarchy onto the current activities and technology within the vehicle value chain that support end-of-life reclamation. The final literature review chapter, Chapter 5, reviews previous ELV cost modelling research and provides a review of the current costing techniques available to assist in the development of an end-of-life cost model. Chapter 6 highlights the research methodology and demonstrates the systematic approach taken in determining the context of the research and the development of a novel holistic costing ELV costing framework.

The theoretical research and development section encompasses six chapters, and highlights the thesis’ main contributions to research. Chapter 7 discusses the generation of a holistic costing framework for the end-of-life vehicle recovery sector, based on the application of techniques and approaches highlighted within the research background and overview section. Chapter 8 is the first of the three research chapters which realise the framework and consider the costing of direct and indirect vehicle processing. Chapter 8 specifically looks at indirect reclamation costing, and the inclusion of uncertainty modelling and sampling techniques within the cost modelling process. This is then followed by two chapters focusing uniquely on the direct costs of vehicle processing, and on the problematic issues of pre and post-fragmentation activity costing. Chapter 9 considers the direct pre-fragmentation costs, and selects costing approaches most applicable to the resolution required and data available, while Chapter 10 discusses the development of a direct costing approach to model the
value added processing for automated post-fragmentation separation technologies. Chapter 11 describes the software implementation of these techniques into a ELV cost model, and Chapter 12 highlights suitable case studies to demonstrate the effectiveness and applicability of these techniques.

The final section within the thesis includes two chapters (13 & 14), which discuss the significance of the proposed decision support system in the context of the thesis scope, before drawing the final conclusions from this research and highlighting potential areas of further work.
Figure 1.1, Thesis structure
Chapter 2

Scope of the Research

2.1 Introduction

This chapter discusses the research scope. The preliminary part of the chapter describes the research assertion and aim, which provides the context of the research with respect to the current industrial situation. The later part of the chapter highlights the research objectives for achieving this aim, and the scope of the thesis when considering these objectives.

2.2 Research Assertion

The European Union generates over 9 million tonnes of automotive waste each year. A high percentage of this waste is reused, recycled or recovered in a range of different applications based on the assessment of the materials end-of-life value. Due to the transposition of the ELV directive in the UK, in which a number of automotive recovery consortiums are tasked with fulfilling the manufacturers legislative requirements, this value assessment is undertaken by an industry with little understanding of the products they are recovering. If the ELV directive had made ELV recovery a required part of a vehicle manufacturers core competency, and "vertically integrated" the current vehicle recovery sector within their own organisations, the manufacturers product knowledge would have allowed easier identification of end-of-life value, and in most instances promoted sustainability by allowing a product or assembly to re-enter further up the waste hierarchy (e.g. supplier re-use & supplier reconditioning, as opposed to material recycling or energy recovery). However, the shift away from such vertical integration, and the lack of influence from the vehicle manufacturers during downstream processing, has resulted in a materials reclamiation industry very much guided by "end-of-pipe" economics. Material market prices, fuelled by the growth of developing nations (in particular China and India), ultimately dictates which subset of materials the recovery industry
decides to focus on, and in doing so negates any possibility for the establishment of newer more sustainable recycling and recovery activities.

The retention of economic value and the level of processing required at end-of-life are good indicators as to a product’s sustainability, but it is often the case that the investment required to retain this value can not always be perceivably justified. There is often little understanding as to the exact economics of current processing decisions within the vehicle recovery chain, and even less transparency in terms of the costs of environmental best practice. Hence, the major assertion made in this research is that by providing the mechanism to model current and future vehicle reclamation costs it is possible to improve the recovery sector’s environmental performance by not only demonstrating its ability to accept operational change, but also by providing the means through which to assess the profitability of more sustainable alternatives.

2.3 Research Aims and Objectives

The overall aim of the research is to promote sustainable practices within vehicle reclamation through improved value recovery, whilst meeting the requirements of impending and future legislative targets, by generating:

i) A framework to cost the economic implications of operations carried out by various end-of-life stakeholders, both pre and post-fragmentation.

ii) An end-of-life cost model that highlights the interactions of current and future processing decisions in terms of economic cost.

iii) A number of end-of-life vehicle processing scenarios that model both; the financial robustness of an end-of-life operator to accept operational change under current legislative and market conditions, and the economic viability of an end-of-life operator to adopt more sustainable processing strategies.

To achieve this aim the major research objectives can be defined as:

a.) to review the associated literature on end-of-life vehicle processing, legislative requirements and component/ materials markets.
b.) to investigate economic costing techniques and their applicability to the ELV recovery sector.

c.) to model and analyse the industrial practices within the vehicle recovery sector, and create an information model of typical industrial activities.

d.) to generate a comprehensive framework to effectively cost the economic implications of end-of-life processing activities, and the relationships between them.

e.) to implement an ELV cost model and develop functional viewpoints to effectively support the processing decisions made by pre and post-fragmentation end-of-life operators.

f.) to validate and demonstrate the ELV cost model using appropriate case studies to assess the feasibility of operational change and additional sustainable processing scenarios.

2.4 Research Scope

The recovery and processing of end-of-life passenger vehicles has been identified as the product through which value recovery will be assessed, under the context of legislative reform. Thus, the scope of the end-of-life value recovery research can be outlined as:-

2.4.1 A Review of the Relevant Literature End-of-life Vehicle Processing and its Legislative Requirements

A comprehensive review of literature within the area of vehicle reclamation is required to provide the knowledge with which to direct the industrial focus of the research. A review encompassing not only the traditional work and ideas with regard to product recovery such as reverse logistics and de-manufacture, but also to provide an insight into current end-of-life industrial capabilities. Peripheral to the literature study is the development of an understanding as to the political background and ramifications of the ELV directive, and the potential effects this has on the relationships within product value chain.
2.4.2 Investigation into Costing Techniques and Existing Approaches

The realisation of techniques and systems to not only capture the economics of the end-of-life recovery systems but also to identify potential methodologies to model them, is one of the primary objective of the research. The attribution of overheads and investment costs, the effective modelling of uncertainty and the dynamic modelling of widespread variation within the model need to be investigated and addressed. This requires a review of current costing techniques, together with their suitability to be effective applied to end-of-life recovery costing.

2.4.3 Modelling of Current Activities and Identification of Industrial Requirements

An effective assessment of the current recovery chain within the UK will not only identify the main stakeholders but also determine the effectiveness of the recovery and the levels of value recovered. The generation of a basic information model for each end-of-life stakeholder will facilitate the identification of cost and revenue intensive "hotspots", allowing the subsequent framework to be built around these activities. Generation of these models will also provide an opportunity to assess the stakeholder focus of interest in this research, and can be ascertained through questionnaires and one-to-one interviews.

2.4.4 The Generation of a Costing Framework

This includes the establishment of a methodology to effectively apply the costing tools and techniques within the context of end-of-life product recovery. The framework must consider an holistic approach to end-of-life costing, not only identifying methods of capturing the visible direct costs, but also adopting approaches capable of modelling the more obscure indirect ones. Such an approach which looks at the direct as well as indirect costs will provide stakeholder confidence that the framework will consider all aspects of their business and not just one processing scenario in isolation. The framework is required to encompass the whole vehicle recovery chain, to cost both the pre and post-fragmentation operations.
2.4.5 Realisation of an ELV Cost Model to Support Sustainable ELV Recovery

The realisation of end-of-life vehicle value in terms of a dynamic model that accounts for both direct and indirect costs and revenues. A software based decision support system will be developed, incorporating methods described within the ELV framework. A process of model validation exercises will then be undertaken to demonstrate the ability of the system to replicate real world economics.

2.4.6 Validation of the Research Concept via Case Studies

Suitable case studies will be undertaken to demonstrate applicability to end-of-life operators, and provide analysis as their current situation. Industrial data will be used to populate the model and provide an initial assessment as to the economic robustness of a particular operator to accept operational change. The result of this analysis will then provide a foundation on which to measure an operator's suitability to reform and its ability to adopted more sustainable processing scenarios.
Chapter 3

An Overview of the Vehicle Recovery Sector

3.1 Introduction

This chapter provides an overview of the current vehicle recovery sector within the UK, and its changing status based on the implementation of the ELV directive. The recent inclusion of this sector within the manufacturers core focus has highlighted a distinct lack of technical innovation and healthy competition over the last half century. Prescriptive legislation has brought with it reforms to the sector that have required increased investment, improved operating standards and more rigorous environmental auditing. This chapter provides an overview as to the market scale of ELV recovery, and highlights the main end-of-life actors within the vehicle recovery chain and the legislation currently affecting them.

3.2 Actors within the End-of-life Vehicle Recovery Sector

Since the introduction of the automobile in Europe in the early 1900’s, various recycling industries have emerged to profit from the discarded vehicle waste produced at end-of-life. Traditionally, this profiting by downstream actors bore little influence with the engineers that undertook the original vehicle design. Over the years this situation has changed, with the introduction of recognised end-of-life activities such as reuse, remanufacturing and recycling, and has started to improve the interactions between stakeholders within the vehicle salvage industry, material processors and component manufacturers. The interconnectivity of all these actors as a whole is referred to as the vehicle “value chain” (Roy and Whelan 1992). This is in turn composed of the downstream “recovery chain” (the actors involved in end-of-life reclamation activities), and the upstream “supply chain” (the actors involved in the vehicles manufacture). Figure 3.1 provides an overview of the strength of the main material flows through the automotive value chain (indicated by the thickness of the black lines), with the recovery chain highlighted in blue and the supply chain in pink.
The recent introduction of environmental legislation regarding an automobiles retirement has forced manufacturers to reassess the role the recovery sector plays within the whole value chain. With this reassessment comes an increasing number of conflicting interests and problematic relationships, as described by Deutz (2004). The following sub-sections provide an overview of the main end-of-life actors within the recovery chain, and their changing role within vehicle salvage.

### 3.2.1 Collection Agents

One of the biggest economic hurdles introduced by the ELV directive is the stipulation that last owners should be allowed to return their vehicles at no cost to themselves. Vehicle manufacturers have opted to conform to this and other legislative requirements by moving away from actively getting involved and investing in their own recovery facilities and networks, in favour of utilising the existing end-of-life
actors and waste reclamation processes. This has lead to the establishment of “collection contracts”, whereby third party collection agents (namely, Cartakeback and Autogreen) have agreed to administrate and fulfil the requirements laid down by the ELV directive on the vehicle manufacturer’s behalf (personal communication with Mr. M. Rivers of Ford Motor Company, April 2005). These collection agents are charged with not only making sure each manufacturer’s network has enough capacity, but also has the measures in place to meet the requirements of the directive. Between Cartakeback and Autogreen, 76 of the major vehicle brands that sell within the UK are represented (Eminton 2007). These agents act as contact points for the vehicles last owner, and assist them in locating a local scrap yard and exercise their free retirement right. The rest of the actors within this section can therefore be either contracted or un-contracted to these collection agents.

3.2.2 The Authorised Treatment Facility (ATF)

The traditional idea of returning your vehicle to the local “scrappie” at the end of its life, often paying for the privilege and expecting it to degrade slowly on a piece of waste-land, is not tolerated under the new legislation. Instead, each returning vehicle must be systemically de-polluted to remove the materials and fluids deemed environmentally detrimental or hazardous to the routing of the final waste stream. De-pollution typically takes between 15-30 minutes per vehicle (Coates 2006), before the removal of any key components for resale can begin. The hulks are then compacted for ease of transportation and sold (mainly for their ferrous content) to a local shredder that often services a network of ATFs.

3.2.3 The Shredder Facilities

There are at present 37 shredder sites within the UK, of which 28 are located in England, five in Scotland, three in Wales, and one in Northern Ireland (Kollamthodi et al. 2003). The shredder acts as a central hub at which vehicle hulks from numerous ATFs are mixed with industrial and white goods waste streams (Ambrose et al. 2000). The shredding facilities then use rotatory-hammer mills to fragment the ferrous rich waste stream into particle sizes susceptible to magnetic (over-band magnets), density (cyclone technology) and charge (eddy current technology) separation devices. For a
more detailed review of traditional sector equipment and its current implementation see Rousseau and Melin (1989), and Manouchehri (2006). These are well-established separation technologies that have high levels of throughput and automation. For example, a typical throughput rate of the hammer-mill is 150 tonnes/hour (personal communication with Mr. D. Wilkins of European Metals Recycling, March 2006). Although well distributed in geographical location throughout the UK, many sites are united (either via organisation or opportunity) into shredder groups. Of the 37 shredding sites in the UK half are operated by two main organisations, namely European Metals Recycling and Sims Metal, which have been estimated to process in the region of 70% of all ELV arising (DTI 2004). Surprisingly, despite their capacity these two main operators are not contracted to any of the collection agents. Hence, it is only the remaining shredding sites run by the independent metal merchants that provide this contracted capacity.

3.2.4 The Dense-media Separation Facilities

Once the ferrous content has been removed, the majority of the remaining Shredder Residue (SR) is moved onto the dense-media separation facilities. These operators are mainly focused on recovering the metallic non-ferrous content from the waste, and a combination of density (floatation tanks and cyclone), charge (eddie current), volume (trommel sieving) and recognition (manual sorting, property recognition) technologies purify the waste stream. The un-removed material fraction from these facilities is currently landfilled.

3.3 End-of-life Vehicle Legislation

Legislation continues to play an important and influential role in determining the end-of-life processing treatment of many retired products. The following subsections give a brief background as to the intended direction of European frameworks, before highlighting the main requires of the ELV Directive.
3.3.1 European Union Legislation

EU legislation accounts for an estimated 80% of UK environmental regulations (Lowe and Ward 1998), and has formulated a number of prescriptive directives encompassing the design, production and treatment of a range of industrial and consumer products. All the directives have the philosophy of “extended producer responsibility” (EPR) at their core (Lindhqvist and Lidgren 1990), which aims to promote end-of-life considerations within the product design process, and the reduction of a product’s overall ecological impact. Toffel (2002a), refers to the justification of the manufacturer as being the focus for EPR due to the “critical leverage point” it has in terms of product design. For additional discussion of EPR and its implementation globally, see Sach (2006). Figure 3.2 provides an aggregated

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<td>Energy use in Products (EuP) (2005/32/EC) – 8th July 2005</td>
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### European Union Vehicle Legislation

- **Restriction of Hazardous Substances (2002/95/EC) 27th Jan 2003**
- **Landfill Directive (1999/31/EC) 16th July 1999**
- **Efficiency Labelling (1996/65/EC) 13th Dec 1999**
- **Integrated Pollution Prevention and Control (96/61/EC) 24th Sept 1996**
- **Packaging and Packaging Waste Directive (94/62/EC) 26th Dec 1994**

### Main Transposed UK Vehicle Legislation

- **Pollution Prevention and Control Regulations, 2000 (SI 2000 No 1073) 27th July 2000**
- **Transferred Shipment of Waste Regulations 1994 (SI 1994 No 1123) 22nd April 1994**
- **Batteries and Accumulators (Containing Dangerous Substances) Regulations (SI 1994 No 220) 2nd Feb 1994**
- **Hazardous Waste Regulations 2005 (SI 2005 No 894) 23rd March 2009**

**Figure 3.2**, Summary of the main European Frameworks and Directives that affect the automobile, and the UK regulations generated by their transposition
overview as to some of the main environmental frameworks, directives and transposed statutory instruments relating to the automobile within the UK.

The top of the diagram highlights the main European frameworks, these don't describe one specific policy for one particular sector, but realise the need to have a number of high level themes to ensure that the direction of legislation is consistent. Integrated Product Policy (IPP) is a good example of this in how it attempts to harmonise the efforts of stakeholders through the whole product life-cycle. The actual implementation of sector specific directives into European law is then formulated along side these framework policies, before they are eventually transposed down into UK legislation via the use of statutory instruments. As can be seen within the diagram the majority of the transposed UK legislation has historically only affected the end-of-life waste management sector, with many of the waste handling regulation being originally formulated in the early 90's. The introduction of EPR legislation has only become more prevalent within UK law in the last few years, but is a clear indication as to the changing focus of regulations to encompass vehicle manufacturers and suppliers.

3.3.2 End-of-life Vehicle Directive

The initial concept for the ELV Directive was formulated as far back as 1989, where the automotive sector was identified as one of the priority waste stream to target. The original directive proposal was published in 1997 (CEC 1997/0194 1997), which outlined a number of manufacturer obligations, and made it implicitly clear that the directive was to be mandatory and exclude any sort of voluntary agreements (Zoboli et al. 2000). Then followed a period of intense negotiation and review, and only on the second parliamentary reading in 2000 was the directive approved (European Union - Directive 2000/53/EC 2000). A number of transposition options were available to EU member states, for a more detailed discussion see Perchards (2004) and GHK Consulting Ltd (2006), to implement the requirements of the directive. The UK opted for an “own marquee approach”, which requires each vehicle manufacturer to be individually responsible for its own make of vehicle. The directive was implemented within the UK using two statutory instruments (End-of-life vehicle regulations 2003, and End-of-life vehicle regulations 2005), that introduced
regulations that have effected both upstream design processes and down-stream recovery. The main points of the regulations and the measures that have been implemented are summarised in Table 3.1.

3.3.3 Areas of the Directive Still Outstanding

To-date the implementation of the regulations has moved swiftly into effect, despite the late overrunning of the original directives timetable. By far the most cost intensive part of the directive has been born by the ATFs, which have invested in the region of £210,000 per facility (Coates 2006) in bringing their operations up to scratch with the new standards. The number of vehicle salvage companies that have left the industry in the last 5 years, approximately 2/3 (Environmental Agency 2007), is a testament to the investment commitment required to ensure a legitimate future. Thus far, the pretreatment and de-registration requirements of the ELV Directive have been widely accepted by the vehicle recovery sector, but by far the most controversial and outstanding issue still to be addressed is the achievement of the recycling and recovery targets. Currently set at:

- 85% of an end-of-life vehicle (by weight) to be recycled, reused or recovered from 2006 (80% recycling and reuse, with a 5% allowance for energy recovery)

- 95% of an end-of-life vehicle (by weight) to be recycled, reused or recovered from 2015 (85% recycling and reuse, with a 10% allowance for energy recovery)

UK industry estimates as to the current metallic fraction already recycled, using existing waste stream processing technology, suggests that around 75% of a vehicles weight is already recovered (Weatherhead and Hulse 2005), supported by other European benchmarking exercises (ARN 2005, François 2003, GmbH 2006). Hence, to achieve the levels laid down within the Directive a further 5% is required via the collection of fluids and non-metallic materials (i.e. glass, rubber and plastics). Figure 3.3 provides a rough composition of a typical automobile (ACORD 2001) and the corresponding recycling percentages currently achieved within the UK (Weatherhead and Hulse 2005). Weatherhead concludes that a total recycling and recovery rate of ≈
<table>
<thead>
<tr>
<th>Requirement</th>
<th>Targeted Stakeholder</th>
<th>Enforcement Date</th>
<th>Description</th>
<th>Implemented UK measures</th>
</tr>
</thead>
<tbody>
<tr>
<td>Prohibition of Heavy Metals</td>
<td>Manufacturers</td>
<td>3rd Nov 2003</td>
<td>Materials or components must not contain lead, mercury, cadmium or hexavalent</td>
<td>Introduction of material monitoring systems (e.g. International Materials Data System, IMDS) populated by component suppliers.</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>chromium</td>
<td></td>
</tr>
<tr>
<td>Coding Standards</td>
<td>Manufacturers</td>
<td>3rd Nov 2003</td>
<td>Materials and components must contain identification markings to assist recovery.</td>
<td>ISO 1043-(1&amp;2), ISO 11469 and ISO 1629 adopted as generic standards</td>
</tr>
<tr>
<td>Dismantling Information</td>
<td>Manufacturers</td>
<td>3rd Nov 2003</td>
<td>Dismantling information available to dismantlers within 6 months of a vehicle</td>
<td>Vehicle dismantling database (International Dismantling Information System, IDIS) established.</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>being on the road.</td>
<td></td>
</tr>
<tr>
<td>Reporting and Information</td>
<td>Manufacturers</td>
<td>3rd Nov 2003</td>
<td>Manufacturers will publish information on the design of their vehicles with a</td>
<td>Further legislation is pending within the UK following the European Commission Directive 2005/64/EC.</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>view to their recoverability and recyclability (Type approval testing).</td>
<td></td>
</tr>
<tr>
<td>Certificate of Destruction (CoD) and Delivery</td>
<td>Dismantlers</td>
<td>3rd Nov 2003</td>
<td>Issuing of de-registration certificate and vehicle take-back at no cost to the</td>
<td>Online deregistration system now active: <a href="http://www.dvlaonline.gov.uk/Homepage.asp">www.dvlaonline.gov.uk/Homepage.asp</a></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>last owner</td>
<td></td>
</tr>
<tr>
<td>Recovery Operating Standards</td>
<td>Dismantlers / Shredding sites</td>
<td>3rd Nov 2003</td>
<td>Site licenses issued to vehicle dismantlers that have proven environmental</td>
<td>De-pollution and processing standards currently enforced by the Environmental Agency in accordance with Annex I of the ELV Directive.</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>recovery standards.</td>
<td></td>
</tr>
<tr>
<td>2006 Reuse, Recovery and Recycling Targets</td>
<td>Manufacturers/Dismantlers /</td>
<td>1st Jan 2006 &amp;</td>
<td>85% of a vehicles weight to be reused, recycled or recovered, with 80%</td>
<td>Department of Trade and Industry measured the current recovered fraction at 74.48% to assist with the monitoring of the 2006 target.</td>
</tr>
<tr>
<td></td>
<td>Shredding sites/Non-ferrous</td>
<td>31st Jan 2006</td>
<td>recycled or reused, with a 5% allowance for energy recovery.</td>
<td></td>
</tr>
<tr>
<td></td>
<td>separators</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2015 Reuse, Recovery and Recycling Targets</td>
<td>Manufacturers/Dismantlers/</td>
<td>31st Jan 2014</td>
<td>95% of a vehicles weight to be reused, recycled or recovered, with 85%</td>
<td>Despite a European review the 2015 target remains the same.</td>
</tr>
<tr>
<td></td>
<td>Shredding sites / Non-ferrous</td>
<td></td>
<td>recycled or reused, with a 10% allowance for energy recovery</td>
<td></td>
</tr>
<tr>
<td></td>
<td>separators</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Table 3.1, Summary of UK ELV regulations
Chapter 3

Figure 3.3, Typical composition of a UK passenger vehicle and the corresponding recycling percentage achieved using current UK processes

78% (excluding fuel) is achievable via the current activities, with the remaining 22%, composed mainly of plastics, glass, rubbers and textiles, eventually ending in landfill.

Debate as to the technical ability of the recovery industry to achieve this, and the viability of the economics that under-pin it, has lead many to believe that attainment of the 2006 target will be difficult, and the fulfilment of the higher 95% target in 2015 to be unrealistic. Mark et al. (2004) and Daniels et al. (2004), highlight the increase in vehicle weight and the proliferation of plastics within modern vehicle design, as causal factors as to the achievement of the “challenging” 2015 quota. A stakeholder working group have voiced their concerns during a recent EU consultation period (Stakeholder Working Group on ELVs 2005), only to be subdued by the recommendations of a more comprehensive EU report highlighting the costs and benefits of maintaining the higher 95% target (GHK Consulting Ltd 2006).
3.4 The Effects of the ELV Directive on the Recovery Sector

The ELV Directive has been a catalyst for dramatic reform and investment within the UKs vehicle recovery sector, the following subsections highlight the most prominent of these changes.

3.4.1 The Introduction of “Zero-cost” Collection Contracts

The ELV directive requires vehicle manufacturers to provide free take-back and treatment for all its own vehicles post 2006, and meet stringent recycling and recovery targets of 85% and 95% by 2006 and 2015 respectively. It was widely believed of the three options available, namely “last owner pays”, “exchequer pays” or “producer pays” (Skinner and Fergusson 2003), that the vehicle manufacturers would be the ones fiscally liable, yet during the establishment of these collection contracts it became apparent that no direct financial support would be given to these collection agents or their network members, given the substantial intrinsic value that ELVs possessed at the time of the contract negotiations (Edwards et al. 2006). Hence, the value of ELVs is currently offsetting the costs of legislative conformance, but has ultimately left the vehicle salvage industry in an economically precarious position.

3.4.2 Reform and Investment at the ATFs

The ELV directive has directly influenced these stakeholders in two ways. The first is a massive overhaul in the environmental operating standards of the industry. The recent surge in spending to bring the standards of the ATFs up to scratch has been as a result of an “invest-to-progress” policy by the UK Government. Forcing ATFs to invest heavily in de-pollution equipment on a very stringent time-scale, or risk being put out of business by the Environmental Agency for non-compliance. ATFs have been keen in one sense to invest in this equipment as it has begun to remove the illegal rogue collection element from the industry, while on the other hand there has been a reluctancy to invest given the producer responsibility that the ELV directive advocates. The second is the establishment of contracted and un-contracted ATFs to different collection agents. Varying opinions exist within the industry as to the pro’s and con’s of being a contracted ATF. Many believe that being contracted will restrict
the flow of ELVs through their facilities and incur additional manufacturer branding costs, while others welcome the guaranteed recycling support that being contracted is said to bring. Any vehicle not returned via a vehicle manufacturers contracted network can charge the last-owner, but the ATF accepting it must guarantee that the recycling and recovery targets are achieved in accordance with the End-of-Life Vehicle Regulations (2005).

3.4.3 Improvements in Post-fragmentation Separation Technology

Hulks sold to the shredders provide a substantial source of revenue for the ATFs, as the current high demand for steel from developing countries such as India and China is creating an overriding cost-driver within the vehicle recovery industry (Beck 2005b, Garino 2005). Strong ferrous market prices have traditionally meant that materials such as steel and iron have been well recovered. More recent investment has seen the introduction of a number of dense-media facilities across the UK, focused at recovering the non-ferrous content. The current goal of these facilities is to further refine and purify the shredder residue produced at these sites, namely the glass, rubbers and plastics. For a more detailed discussion of current shredder residue processing technologies see (Ferrão et al. 2006).

3.4.4 Relationships between the Recovery Chain and the Vehicle Manufacturers

The de-pollution requirements of the directive combined with the achievement of the recycling targets have meant that the recovery sector has found itself in a unique position. Subject to a massive overhaul in its operating standards by the UK Government, they now retain the tools to carry out the requirements of the legislation (and assist the producers), but are not directly financially liable for the directives successful implementation. This has led many to believe that the only way target attainment can be guaranteed is if manufacturers take direct control of the recovery industry, and make it an integral part of their organisation. Influencing both upstream development and down-stream recovery. Many papers have highlighted the possibility of Original Equipment Manufacturers (OEM) "vertically integrating" into the product recovery chain and the beneficial influence this would have in terms of potential product reuse (Toffel 2002b), available information exchange (Ferguson and Browne
2002, Rahimifard et al. 2003, Thierry 1997) and “Design for” disciplines (dismantling, shredding, serviceability, End-of-life (EOL) value, etc.) in general (Ferrão et al. 2003). Despite some small publicised examples of manufacturer instigating part-resale projects (Toffel 2002b) the reality of vehicle manufacturers moving away from their core competencies into an industry with a well-established experience base and infrastructure is unlikely. As such, any “design for value recovery” considered at the concept design stage is currently being undertaken either due to consumer pressures, legislative requirements or corporate morality, rather than any direct end-of-life economic incentives.

3.5 Implications of the Automotive Sectors Growth Within the UK

The UK accounts for around 3.5% of total global passenger car consumption, in a world market which has an annual world turnover in the region of £48.8 billion (SMMT 2006). Toyota, General Motors and Volkswagon are the global industry leaders (OICA 2006) and accumulatively account for around 1/3 of all passenger cars produced (see Figure 3.4). Whereas in the UK, Ford, General Motors (Vauxhall) and Renault are the manufacturers that currently hold the strongest market positions.

![Figure 3.4, Global manufacturer passenger vehicle sales figures for 2005](image-url)
Chapter 3

There is currently in excess of 31 million vehicles on the UK’s roads (Mintel 2006b), and despite a decline in new registrations from 2003, the number of active vehicles has steadily increased by 10% over the last 6 years (see Figure 3.5). The variations in vehicle purchasing from year to year can be primarily related to the population’s available income (Graham and Glaister 2005) and pre-sale manufacturer registrations (Mintel 2006c). Whereas more long-term sustained growth is more readily attributed to changes in the general population’s attitude towards personnel mobility (indicated by the strength of the low to medium sized vehicle market in the UK), and the need to support new lifestyle choices such as the movement away from centralised city living (European Environmental Agency 2006).

With this persistent market growth comes an increased abundance of end-of-life waste, estimated to be in the region of between 8 and 9 million tonnes per annum for the European Union (European Union - Directive 2000/53/EC 2000). On average 2.1 million passenger cars reach the end of their useful life each year within the UK (ACORD 2001), and the onus of responsibility for their safe and environmental disposal has often been clouded by a number of economic and ownership issues.

![Figure 3.5, UK vehicle car parc and the number of vehicle registrations from 2001-2006](image-url)
Manufacturers have traditionally seen the remit of their responsibility ending at the termination of the vehicles warranty period, with ownership (and hence accountability) passing to the final customer. However, the introduction of the ELV directive aimed at vehicle manufacturers aims to change this, necessitating a rethink of their traditional product lifecycle to encompass more end-of-life considerations, in the hope of promoting more sustainable closed-loop recovery.

3.6 Summary

The literature would suggest that the UK transposition of the ELV directive has done little to strengthen the relationships between the vehicle recovery chain and the vehicle manufacturers, and it can be argued that this lack of synergy has been counter productive to the core themes of sustainability, a sentiment echoed in Forton et al. (2006) and Edwards et al. (2006). Without a subsidised influence from the vehicle manufacturers, any decisions concerning the end-of-life operations carried out on a vehicle will be based solely on process economics as opposed to any long-term environmental benefits. It is therefore vital for the vehicle recovery industry to begin to understand the economics of its own operation, so that future vehicle salvage is based on environmentally sustainable strategies that are also economically feasible.
Chapter 4

An Overview of Research on Environmental Concerns Related to End-of-Life Vehicles

4.1 Introduction

This chapter provides an overview of the main literature and research carried out within the vehicle value chain that support environmental vehicle reclamation practises. The chapter begins by mapping the waste hierarchy onto the current considerations made, both within upstream design and downstream recovery, regarding End-of-life components and materials. The subsequent literature review is catalogued according to this segmentation and provides a detailed discussion of the main research reported within each area.

4.2 Activities that Support Sustainable End-of-Life Recovery

The waste pyramid depicted in Figure 4.1 is widely accepted as the preferred waste management hierarchy for sustainable development. Developed in the mid-seventies as part of the Waste Framework Directive (CEC - 75/442/EEC 1975) and formalised into a hierarchy in 1996 (CEC - CdR-339/96 1996), the pyramid suggests an approach located nearer the summit as opposed to the base will provide a more ecologically friendly solution to product recovery. Numerous decisions taken within the whole life-cycle of a vehicle can impact on its environmental footprint. From upstream material selection and design, through to end-of-life markets and processing technology. Figure 4.1 maps each level onto the activities made by key stakeholders (using colour coding) within the vehicle value chain.

Chapter 3 discussed the stakeholders involved and the relationships created by the implementation of the ELV Directive within the UK. This highlighted some of the economic disparities between the interests of the vehicle manufacturers at end-of-life and those of the vehicle recovery sector. Figure 4.1 shows an interesting contrast to
Figure 4.1, Waste pyramid and its mapping to the end-of-life considerations made throughout the vehicles life-cycle
Chapter 4

this, that despite a third party industry profiting from vehicle salvage, the upstream vehicle developers still retain the greatest amount of influence on how far up the waste pyramid end-of-life materials and components finally end up. Indicated within Figure 4.1 by the assortment of green and yellow disciplines located upstream and the use of more unfavourable amber and red pyramid considerations at end-of-life. The following sections provide a detailed overview as to the main research undertaken on each of the life-cycle considerations included within Figure 4.1.

4.3 Upstream End-of-Life Considerations

The choices the suppliers, manufacturers and owners make when designing, manufacturing and maintaining a vehicle have considerable implications for end-of-life processing and material value. The following subsections provide an overview of how upstream stakeholders can affect downstream vehicle recovery.

4.3.1 Component and Material Suppliers

Suppliers often provide materials or produce components on the manufacturers behalf. As such they are independent of the manufacturer and provide products based on a specification that often defines attributes like quality, aesthetics, reliability, etc. As long as these product requirements are fulfilled, suppliers can potentially re-use and revitalise retired products or reprocess end-of-life materials to be used again. The following sections discuss some of the issues with these two possibilities.

4.3.1.1 Original Equipment Manufacture Remanufacturing and Reconditioning

Remanufacturing is described as a production batch process of disassembly, cleaning, and refurbishment or replacement of parts, from products at end-of-life (Lund 1984), and can be undertaken by the OEM or a third party agent. Remanufactured assemblies require between 20-80% less energy to produce (McCaskey 1994), and can be offered to the customer at a significant reduction of the original cost. The returned product (known as a “core”) can benefit from the existing in-house development knowledge and manufacturing facilities at the OEM. Hence, concessions made within the design process, can more readily assist the various inspection, disassembly, cleaning,
replacement and testing activities. A number of papers have highlighted some of the problematic trade-offs that remanufacturing can have for an OEM; its conflicting focus with other design for disciplines (Shu and Flowers 1999), the indirect assisting of competitors within the same market (Parkinson and Thompson 2003), and the suitability of product leasing versus product collection (Thierry 1997). For a more comprehensive review of the related research regarding “design for remanufacture” see Bras and McIntosh (1999).

4.3.1.2 The Use of Recycled Materials as New Resources

A number of primary materials industries currently incorporate recovered materials with their refined virgin stock. The steel and aluminium industries are prime examples of this, which are capable of recycling up to 95% of materials in some of the more advanced electric arc furnaces (Manouchehri 2006). World crude steel production reached 1.05 billion tonnes in 2004 (Beck 2005a), and uses a 1:5 ratio of additional materials in its production. This highlights the substantial benefits of the inclusion of recycled material as a resource conservation measure, but additional studies have shown the significant energy and emissions savings of using scrap materials (Emi 2005). Table 4.1 is adopted from Chandler (1986) and shows the various environmental savings that a number of material categories can have for material suppliers.

<table>
<thead>
<tr>
<th></th>
<th>Paper</th>
<th>Aluminium</th>
<th>Iron and Steel</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reduction of energy use (BTU)</td>
<td>30-55%</td>
<td>90-95%</td>
<td>60-70%</td>
</tr>
<tr>
<td>Reduction of spoil/solid waste (tonnes)</td>
<td>130%</td>
<td>100%</td>
<td>95%</td>
</tr>
<tr>
<td>Reduction of air pollution (tonnes)</td>
<td>95%</td>
<td>95%</td>
<td>30%</td>
</tr>
</tbody>
</table>

*Table 4.1, Percentage saving per tonne of recycled material, sourced from Chandler (1986)*
Vehicle manufacturers are consistently pressured to incorporate a number of often conflicting design paradigms into their product development process. For example, today's vehicles must be manufactured from lightweight materials promoting reduced emissions and better fuel efficiency, while at the same time maintaining its recyclability and environmental inertness. Often these trade-offs are not as transparent, and a great body of research has been directed at trying to prioritise these competing issues. The following sections specifically focus on design for paradigms that influence end-of-life vehicle processing. Lambert (1999) refers to the important need to balance these considerations with the more traditional "Design for" disciplines (manufacturer, assembly, repair). The following sub-sections discuss the main design paradigms that can affect end-of-life, for a more extensive review of the various "Design for" disciplines and their changing focus see Kuo et al. (2001).

4.3.2.1 Design for Extended Component and Material Life

In terms of sustainability increasing the longevity of a product's life would be the most beneficial situation, but the current emphasis society places on consumerism makes us a very materialistic culture. Technologies and fashions are mutually exclusive to the idea of product longevity, and as such are barriers to sustainability's wider acceptance by the consumer. Industry has a reluctance to produce products that outlive expectation, as this usually involves more costly durable materials and more innovative development practices. Product longevity, although supporting sustainability, also hinders the longer term market growth of a company when the market becomes saturated with a product that simply does not become outdated or obsolete, a sentiment echoed in van Nes and Cramer (2003).

4.3.2.2 Design for Serviceability (DfS)

Manufacturers "design for serviceability" is primarily targeted at supporting their aftermarket services. "After sales service currently account for about 40% of the total cost of owning a car over its lifetime" (PriceWaterHouseCoopers 2003a). Hence, the development of modulised assemblies and the use of widely available machining
techniques have the potential to assist the franchised dealership undertaking the repair on the manufacturers behalf. Indirect end-of-life benefits of manufacturers incorporating such design issues, are the advantageous effects this has on assembly removal, and replacement (easily accessible components), and de-pollution (sump plugs are designed for oil changes but also assist non-hazardous disposal) (Sodhi et al. 2004).

4.3.2.3 Design for Recycling (DfR)

Given the relatively high cost of implementing techniques such as design for dismantling, upstream design initiatives have adapted to incorporate more widely accepted downstream recycling technologies (Leibrecht et al. 2004). Labelled “Design for Recycling” this approach represents the recovery sector’s shift towards shredding technology and large-scale automated separation. A number of more recent pieces of research have been undertaken to understand the links between product attributes and material separation, these include; the effects of design changes on particle size and liberation (Schaik et al. 2004), the assessment of assembly joint type, amount and morphology on liberation (Castro et al. 2005, Castro et al. 2004), and the selective removal of materials to facilitate improved waste stream purity (Ferrão and Amaral 2006b).

4.3.3 The Vehicle Use Stage

In terms of a vehicle’s environmental footprint, the period in which the owner uses the vehicle is by far the most impacting phase. The majority of these effects comes from a vehicle’s emissions and fuel consumption, but the two main factors that effect end-of-life processing; are the vehicle purchasing trends which affect the rate of ELVs produced, and the consumers’ acceptance of second-hand materials and components. The following two sections discuss some of the main trends and barriers relating to these two issues.

4.3.3.1 Sustainable Consumption: Trends in Vehicle Purchasing Habits

From an environmental perspective it would be expected that the increasing reliability of modern vehicles, indicated by the reduction in vehicle failure rates from 37% to
31.7% between 1995 and 2000 (Mintel 2002), would mean that owners would retain their vehicles for longer, reducing the need to manufacture additional units. But more recent figures would suggest that the UK is pushing ever closer to a two-car per household norm (Mintel 2006c), resulting in vehicle retirements in excess of 2 million per year (ACORD 2001). This growth in the number of vehicles on the UK’s roads is fuelled by a change in consumer attitudes towards personal mobility; there is now an expectancy that each person should have the means of travelling when and where they like. Other factors compounding this growth are; the readily available pool of cheap vehicles within the second-hand market, and the fact that the automobile is often used as a status symbol, changed at the whim of fashions. Barber (2004) discusses the intertwined nature of such production and consumption habits, and the knock on effect this has on other resources such as fossil fuels and materials.

4.3.3.2 Consumable parts and Recycled Materials: Consumer Perception

It is generally accepted that sustainability requires the cohesion of three main elements to make it work. The activity needs to prove its environmental performance, its technical and economical viability, and its ability to be widely accepted and adopted within today’s society (otherwise known as the three pillars). Consumer perception of the materials recycled during vehicle reclamation fits into the latter of those three bands, but is as vitally important when trying to move from a “push” to a “pull” recycling market. Examples of end-of-life vehicle materials that have historically failed due to this consumer perception can be seen in the problems the tyre re-treading industry has faced. Retread sales have fallen from 7.5 million units in 1995 to 1.3 million in 2001 (Used Tyre Working Group 2003), due to public fears regarding retread safety. From a sustainable standpoint, a typical retread cycle for a commercial vehicle tyre saves ≈60kg of materials plus an inherent energy saving of ≈37.4 kWh (AEA Technology 2004), making it by far the most environmentally sound processing route. Yet despite a counter market campaign to reassure retread customers, and the introduction of compulsory quality standards (in the UK with The Motor Vehicle Tyre Regulations 2003 and within Europe via adoption of the United Nations 1998 retread specification) the UK’s retread market is still in free fall.
This public perception regarding quality and safety issues is becoming increasingly prevalent within other end-of-life material streams, no more abundantly so than in the plastics recycling sector. The bad publicity regarding material quality that many reprocessed plastics get, despite the evidence to the contrary (Ambrose et al. 2002, Weatherhead 2003, Weatherhead 2005), is another example of the prejudicial perception that sustainable practices must overcome. Strangely, these quality perceptions do not extend to other more robust materials within the vehicle, with 40% of current steel production coming from End-of-Life products (Sullivan 2004).

4.4 Downstream Pre-fragmentation End-of-Life Considerations

Despite the advancement of shredding technology pre-fragmentation vehicle processing is still an essential part of ELV processing. The following section introduces the main pre-fragmentation stakeholder, and discusses the literature surrounding the current activities in relation to the environmental pyramid.

4.4.1 End-of-life Material Recovery at the ATF

The physical returning of ELVs is undertaken by geographically distributed ATFs, of which there are currently 1286 in the UK (Environmental Agency 2007). Vehicles returned to these facilities can be broadly classified into two main vehicle types (Edwards et al. 2006). “Premature ELVs”, which can be either fire, theft or accident damaged, or “Natural ELVs” which are primarily retired due to their age and inconsistent reliability. This distinction has implications on the activities undertaken at the ATFs and the value obtained for each vehicle.

Given the extensive national coverage that each of these stakeholders have, combined with the non-destructive pre-fragmentation activities they undertake already, the potential for them to act as collection hubs to support more sustainable vehicle recovery activities (i.e. re-use, remanufacturing and recycling) should not be overlooked (Fleischmann et al. 1997). The following sub-sections highlight the main research literature regarding these various activities and the effects that both different vehicle types and reverse distribution has on them.
4.4.1.1 Component Salvage Pre-fragmentation

The vehicle recovery sector is still relatively underdeveloped in the UK. The archaic nature of the sector is demonstrated by its relatively slow response to adapt and change to the requirements of the ELV directive, supported in survey findings by Deutz (2001) and KGP (2005). Despite this reluctance to embrace new procedures and recycling technologies, component salvage has always been the backbone revenue stream for many UK scrap-yards. Figure 4.2 is taken from the authorised treatment survey (for further details see Chapter 9) and highlights the most widely removed sub-assemblies that ATFs currently remove.

Literature would also suggest that Premature ELVs present a better opportunity for component re-use than those removed from Natural ELVs, given their typical conditions at time of retirement. This demand variation is shown within Table 4.2 (Ambrose 2001) and highlights the fact that facilities that only process premature ELVs (insurance write-offs), and 0% natural ELVs, on average sell more components.

Car manufacturers typically produce around 20% of the spare-parts themselves, while procuring and selling the rest to their authorised dealers. The spare-parts market is

![Figure 4.2, Most widely removed sub-assemblies taken from ELVs](image-url)
Table 4.2, Table outlining the percentage of parts resold from natural ELVs, sourced from Ambrose (2001)

Table 4.2

<table>
<thead>
<tr>
<th>Dismantler</th>
<th>% Natural ELVs Processed</th>
<th>% Typical Parts Resold</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>0</td>
<td>12</td>
</tr>
<tr>
<td>B</td>
<td>0</td>
<td>12</td>
</tr>
<tr>
<td>C</td>
<td>25</td>
<td>7</td>
</tr>
<tr>
<td>D</td>
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<td>E</td>
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</tbody>
</table>

extremely profitable for vehicle manufacturers, with a typical gross margin of 65% return (PriceWaterHouseCoopers 2003b). Hence, designing as many variations of replacement assemblies is in the vehicle manufacturer's best interest to prevent third party duplication. The knock-on effect of this practice at end-of-life means it is increasingly difficult to catalogue and track each make, model and year variant (personal communication with Mr. C Morgan of ASM Auto's, February 2005), further reducing the effectiveness of second-hand spares. Additional upstream legal issues, such as the recent introduction of new block exemption regulations within the EU allowing spare parts manufacturers to deal directly with end-users (European Commission 2002), have also adversely effected the ATF spare parts sales.

4.4.1.2 Material Recycling at the ATF

The de-pollution process, typically accounting for around 7% of a total vehicle's weight (Weatherhead and Hulse 2005), includes some legislated materials recycling activities (tyres into rubber crumb, oil into recovered fuel oil, etc), but few ATFs regard material collection and recycling as their core focus. Figure 4.3 is taken from the ATF survey (see Chapter 9) and supports this observation.

Vehicle manufacturers are coming under increased pressure (via legislation and green lobbies) to included "closed loop" practices within their new products, via the inclusion of greater quantities of recyclate. Although plastic separation and recycling
Figure 4.3, The number of ATFs that currently undertake different EOL activities

technologies are improving, the wider acceptance of reprocessed materials compared to virgin have a number of issues still unresolved, having a consequential effect on the number of EOL operators that are realistically considering recycling. Technically, certain engineering plastics that are used within structural areas of the vehicle must conform to stringent materials specifications. The inclusion of recyclate within new products has often found its lack of acceptance routed in a misunderstanding of the material’s ability to meet engineering requirements (Bellmann and Khare 2000). Aesthetically, recycled interior trim must attain certain levels of opacity, texturing and colour pigmentation before consideration (Resource Recycling Systems 1998). Although the focus over the past decade for many vehicle manufacturers has been to increase the amount of virgin plastic used within vehicles (so as to increase fuel efficiency while decreasing emissions), conflict has arisen between this approach and the new directive (KGP 2001). Aiming to reduce CO₂ emissions through lightweight design has ultimately made the recovery and recycling of vehicles increasingly difficult, as there is a lack of established technologies at EOL with which to recover these exotic materials. A potential solution to this is to revert back some of the more problematic materials from plastic to metal, so as to increase the overall recoverability of the vehicle. Studies carried out by Corus have shown that reverting only nine
components traditionally manufactured from plastics back into a metallic equivalent, would increase the mass of the vehicle by 25kg, but increases its recyclability by 5% (Sullivan 2004). Figures that counter this argument produced by the Association of Plastic Manufacturers in Europe, have suggested that a 200kg weight saving in material would lead to an estimated 1000 litres of fuel saved over the service life of the vehicle (Jenseit et al. 2003).

One of the most critical issues when regarding material recycling is the availability of markets into which the recovered materials can be sold. Figures produced by the DTI have shown proactive development of the materials recycling market over the last few years, and have placed a figure of 12% of plastic packaging recycled in 2000 compared to 22% in 2003 (DEFRA 2004). Despite the growth of post-consumer packaging recycling within the UK, there is still substantial headway to be made with regards to the recovery of engineering plastics. At present little or no dismantling for recycling is undertaken at the majority of ATFs within the UK. This can be attributed to a number of reasons; problematic high volume/weight ratio of plastics (Recoup 2000), the lack of cheap and accurate analysis equipment (personal communication with Mr. P. Farquharson of Recovered Plastics Ltd, August 2005), and the perceived effort-versus-return of material removal. The use of automotive grade plastics, with their increased mechanical properties when compared to those of packaging plastics, are yet to attract extensive investment and consideration by end-of-life operators. Bellman and Khara (2000) refers to this as the “chicken and the egg” situation, where investment and commitment to the recovery of recyclates will only be undertaken if there’s a market for the re-processed materials, a sentiment echoed in Ambrose et al. (2002) and Mark and Kamprath (2004). These authors all identify the need to establish a “pull” recycling infrastructure, in which supplier demand for cheaper recyclates can establish a market. Surprisingly, this supplier demand may ultimately be strengthened by the very thing that recycled materials are trying to conserve. Plastic and adhesives currently accounting for around £175-£210 in the cost of a typical four-door saloon (Kimberley 2004), and with oil prices hitting record highs in recent years ($78.40/bbl, 13th July 2006), the need for a cheaper alternative may ultimately facilitate the establishment of a strong recyclate market within the UK (Bellmann and Khare 1999).
4.5 Downstream Post-fragmentation End-of-Life Considerations

Once the vehicle has been stripped of its various assemblies and materials at the ATF, the shell (otherwise known as the hulk), is passed on to one of the 37 UK shredding sites. Here the hulk is commutated into fist sized fragments using a rotary hammer mill, before the liberated materials (known as shredder residue) pass through a range of automated separation technologies to extract and purify various materials. Figure 4.4 shows a typical representation of waste flows through the shredding and dense-media facilities, taken from EMR Birmingham and EMR New Market.

Mechanical separation technologies (screens, floatation tanks, over-band magnets, etc.) are currently the preferred method of mass material recycling, and traditionally have been developed for the upstream minerals refinement industry. The ability of mechanical separation technology to segregate the shredder residue is highly dependent on the processes ability to distinguish between the different material properties. Each process uniquely targets a material property within the waste stream that is susceptible to its influence. Wilson et al. (1994) refers to the more traditionally targeted properties such as magnetic susceptibility and density (used within many well established technologies), while others target more unusual differences such as particle resilience and surface friction. Given the important role that mechanical separation is currently playing within the UK’s waste recycling strategy the following section provides an overview of the main literature surrounding many of the main separation technologies, and the selection of waste stream properties that form the basis of their mechanical separation. A brief mention of the literature surrounding some of the more fringe recycling technologies (energy recovery, and feedstock recycling) is also given, before the implications of landfill taxation is highlighted.

4.5.1 End-of-life Material Recovery at the Shredders and Dense Media Plants

Problems that complicate automated mechanical separation are the interparticle interactions that occur between materials in the waste stream (Oberteuffer 1974). These interactions can be as a result of a number of factors; incomplete material liberation at the fragmentation stage, the frictional forces between components, moisture induced bonding or even electro-static attraction forces between materials...
European Metal Recycling
Post-fragmentation Waste Stream Routing (Graphics adapted from Sims Metal Ltd.)

Figure 4.4, A typical shredding and dense-media facility setup
Therefore, the ability of a recycling technology to separate a material is dependent on the composition of the waste stream that is placed through it (Ferrão et al. 2006, Wilson et al. 1994). Different interparticular interactions will occur depending on which materials are concurrently processed together, and only the dissimilarity of the targeted material property will determine whether the materials are ultimately segregated. The following sub-sections give a brief overview of the separation technology and identify the main research regarding the separation modelling and target material properties.

4.5.1.1 Over-band Magnets

Magnetic separation is one of the oldest forms of material separation and has existed in the mineral processing industry since the early 1900’s. High gradient magnetic separators have made the transition from the mineral refining industry to the end-of-life waste management sector, and are now an integral value-added process. The ability of these devices to effectively separate a material is dependent on the superiority of three competing forces; the magnetic forces from the device, the resistive forces when lifting the target substance from the waste stream and the positive and negative inter-particular forces between adjacent materials (Oberteuffer 1974).

For a detailed discussion as to the magnitude of the magnetic force on a waste stream particle see Stradling (1993) and Sheahan (1958). Sheahan states that the force on a

Figure 4.5, A typical over-band magnet found within many post-fragmentation sites
particle within the waste stream affecting its ability to become liberated from the feed, is dependent on the three main forces, magnetic force applied \( (F_m) \), particles own weight due to gravity \( (F_g) \), and resistive force of a particle buried under other material \( (F_p) \). Sheahan's model excluded the inclusion of interparticular forces as it was assumed that their effects were negligible and difficult to quantify. Sheahan therefore expressed these relationships within Equations 4.1 and 4.2 (where; \( \mu_0 = \) permeability of free space, \( m = \) the mass of the particle; \( B = \) is the magnetic induction and \( \chi = \) magnetic susceptibility).

\[
F_z = F_m - F_g - F_p \quad \text{Equation 4.1}
\]

\[
F_z = \frac{m\chi}{\mu_0} B_z \frac{dB_z}{dz} - mg - F_p \quad \text{Equation 4.2}
\]

Of all the parameters within the above equation, magnetic susceptibility \( (\chi) \) is identified as the only waste stream specific property that effects the resultant force. Magnetic susceptibility describes the level of internal magnetisation when subjected to a magnetic field. High magnetic susceptibility values describe the distinctive traits of ferromagnetic materials such as steel and iron that are traditionally separated via magnetic separation devices. Many material databases document an additional parameter to describe the ease with which materials can be magnetised known as the relative permeability \( (K_m) \), Equation 4.3 describes its relationship to that of a material's magnetic susceptibility.

\[
\text{Magnetic susceptibility:} \quad \chi_m = K_m - 1 \quad \text{Equation 4.3}
\]

Paramagnetic and Diamagnetic materials have relative permeability and magnetic susceptibility values close to 1 and zero respectively, while for ferromagnetic materials these values can be considerably larger (Nave and Nave 1985). Table 4.2 describes a range of typical values for \( K_m \).
### Material Relative permeability (Km)

#### Paramagnetic and Diamagnetic materials at 20°C

<table>
<thead>
<tr>
<th>Material</th>
<th>Relative permeability (Km)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Paramagnetic materials</strong></td>
<td></td>
</tr>
<tr>
<td>Iron oxide (FeO)</td>
<td>1.00720</td>
</tr>
<tr>
<td>Platinum</td>
<td>1.00026</td>
</tr>
<tr>
<td>Tungsten</td>
<td>1.000068</td>
</tr>
<tr>
<td>Aluminium</td>
<td>1.000022</td>
</tr>
<tr>
<td>Magnesium</td>
<td>1.000012</td>
</tr>
<tr>
<td><strong>Diamagnetic materials</strong></td>
<td></td>
</tr>
<tr>
<td>Mercury</td>
<td>0.999971</td>
</tr>
<tr>
<td>Carbon (graphite)</td>
<td>0.999984</td>
</tr>
<tr>
<td>Copper</td>
<td>0.99999</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Material</th>
<th>Initial relative permeability</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Ferromagnetic materials</strong></td>
<td></td>
</tr>
<tr>
<td>Iron (99.8% pure)</td>
<td>150</td>
</tr>
<tr>
<td>Iron (99.95% pure)</td>
<td>10,000</td>
</tr>
<tr>
<td>Nickel (99% pure)</td>
<td>110</td>
</tr>
<tr>
<td>Steel (0.9% C)</td>
<td>50</td>
</tr>
<tr>
<td>Cobolt (99% pure)</td>
<td>70</td>
</tr>
</tbody>
</table>

Table 4.3, Parameters describing the differences in magnetic properties of an array of materials commonly found within SR, sourced from Brown (1958)

#### 4.5.1.2 Eddy Current Separation

Eddy current separation technology is primarily focused on the non-ferrous part of the waste stream. The separation is brought about by inducing eddy currents inside the conductive particles of the stream. These currents lend a magnetic moment to the particles which are then propelled by the gradient field of the magnets (Edison 1889). Opposite polarity magnets are laid end-to-end around the circumference of the drum, as the belt moves over the drum the magnetic field produces an electrodynamic force within conductive materials that accelerates them. Conventional separators (mainly utilised within the materials recovery industry) are typically known as horizontal drum separators, an example of which is shown within Figure 4.6.
Chapter 4

Figure 4.6, Separation principles of an Eddy-current separation devices

The selection of an appropriate material properties with which to identify potential separable materials is describe within much of the literature as the separation factor. This value is created by dividing the conductivity of a material ($\sigma$) by its density ($\rho$). Lungu specifically refers to this factor as being an integral parameter in determining the materials resulting trajectory (Lungu and Rem 2002), a sentiment echoed from Schlomann's early theoretical and practical papers in which he stated that a material will be deflected a characteristic distance (proportional to $\sigma/\rho$) substantially independent of the particle's size (Schlomann 1975). Table 4.4 highlights the separation factors for a range of materials found within post-fragmented waste stream (MATWEB 2007). The materials with the highest separation factors within the table correlate well with the materials typically found within the recovered feed of an industrial eddy current separation device.

4.5.1.3 Air Separation Cyclone Technology

Cyclone technology can be broadly classified into two main groups; those that require a suspension liquid (hydro-cyclones), and those that utilise a carrier gas (air-cyclones). The majority of cyclone technology used throughout the UK are air classifiers, and utilise air-flow to separate the light and heavy fractions of the shredded waste. The main separation chamber is a conical design with either a tangential or axial inlet (Boubel et al. 1984). Figure 4.7 has been adapted from the work of Wills (1997), and demonstrates the basic separation principles.
Material Separation factor ($\sigma / \rho$)

<table>
<thead>
<tr>
<th>Material</th>
<th>Separation factor ($\sigma / \rho$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Calcium, Ca</td>
<td>14.1</td>
</tr>
<tr>
<td>Aluminium, Al</td>
<td>13.7</td>
</tr>
<tr>
<td>Magnesium, Mg</td>
<td>12.5</td>
</tr>
<tr>
<td>Copper, Cu</td>
<td>6.6</td>
</tr>
<tr>
<td>Zinc Alloys, General</td>
<td>2.3</td>
</tr>
<tr>
<td>Brass</td>
<td>2.1</td>
</tr>
<tr>
<td>Nickel, Ni</td>
<td>1.8</td>
</tr>
<tr>
<td>Iron, Fe</td>
<td>1.4</td>
</tr>
<tr>
<td>Tin, Sn</td>
<td>1.3</td>
</tr>
<tr>
<td>Tungsten, W</td>
<td>0.9</td>
</tr>
<tr>
<td>Platinum, Pt</td>
<td>0.4</td>
</tr>
<tr>
<td>Lead, Pb</td>
<td>0.4</td>
</tr>
<tr>
<td>Stainless steel</td>
<td>0.2</td>
</tr>
<tr>
<td>Glass</td>
<td>0.0</td>
</tr>
</tbody>
</table>

Polymers and organics have extremely low conductivity values and as such produce extremely small separation.

**Table 4.4,** The separation factors of various materials that is proportional to the trajectory of the particle and its ultimate segregation.

**Figure 4.7,** Typical air movement within a air-cyclone separator.
The input waste is introduced under pressure via the tangential inlet valve and starts swirling within the separation chamber. This creates a vortex, the centre of which generates a low pressure air column to the atmospheric pressure of the overflow pipe. Particles entering the vortex therefore have two competing forces acting on them, the centrifugal force due to its radial rotation around the chamber, and an opposing drag force created by the pressure difference.

The cyclone separation process has been used extensively within the minerals refinement industry, and a great body of literature exists on the predictive modelling of the processes separation efficiencies for both wet and dry cyclones (Avci and Karagoz 2003, Plitt 1976), qualitative testing of process design parameters (Molerus and Glückler 1996), and the development computational optimisation software (Conway 1985). A comprehensive review of collector separation efficiencies, both experimental and theoretical can also be found within Avci et al. (2003).

Specific papers that have directly referred to material properties that effect separation efficiency can be seen in Boubel et al. (1984) in which the particle’s mass is identified via its contribution to centrifugal force.

"For any cyclone regardless of type, the radius of motion (curvature \( R \)), the particle mass \( m \), and the particle velocity \( v \) are the three factors which determine the centrifugal force exerted on the particle. \( F = \frac{mv^2}{R} \)"

Additional papers by Zhang et al. (1988) and Benzer et al. (2001) also suggest that a particle’s physical size (maximum diameter) can be used as a separation metric.

4.5.1.4 Density Media Separation (DMS) Technology

DMS devices can be broadly divided into two distinctive groups, gravity separators and centrifugal separators. Both rely on the differential between specific gravities of the suspension media and the material in question, and are capable of making specific gravity distinctions of 0.1 or less (Wills 1997). Gravity separators tend to be used for
much larger particle sizes (>3mm), where as centrifugal separators segregate materials much smaller (500\(\mu\)m).

Density media separation is typically undertaken at the non-ferrous recovery stage, and utilise heavy liquids such as Magnetite and Ferro-silicate solutions (personal communication with Mr. D Wilkins of European Metals Recycling, March 2006), having specific densities of 1.5 and 3.5, respectively. The media is traditionally agitated to reduce the viscous effect of the separation liquids, and is known as jigging. Given the fundamental operating principal of a density media separator is density, this material property has been widely used within research literature to describe its separation performance. Weiss (1985) and Jong and Dalmijn (1996) proposed the development of a normalized separation metric known as the settling ratio (see Equation 4.4), that identified which materials would be most effectively separated if concurrently processed together.

\[
\frac{v_h}{v_l} = \sqrt{\frac{(\rho_h - \rho_m)}{(\rho_l - \rho_m)}} \quad \text{Equation 4.4}
\]

Where: \(v\) = settling velocity, \(\rho\) = density, \(h, l, m\) = heaviest material, lightest material, liquid separation media. Typical settling ratios can then be generated for a range of standard materials found within the shredder residue mix. Table 4.4 highlights the normalised settling ratios generated for a selection of materials found within shredder residue (MATWEB 2007). Jong concludes that materials with settling ratios greater than or equal to two are likely to be easily separable.

Further research that has considered various performance metrics and waste stream effects include; the influence of particle shape on separation effectiveness (Ferrara et al. 2000), the mathematical modelling of gravity separator performance (Napier-Munn 1991), the size and density of particles on performance (Venkoba et al. 2003) and the mathematical modelling of centrifugal separator performance (Hu et al. 2001).
### Table 4.5, Settling ratios of common materials based on the work of Jong and Dalmijn (1996)

<table>
<thead>
<tr>
<th></th>
<th>Lead [g/cm³]</th>
<th>Copper</th>
<th>Brass</th>
<th>Iron</th>
<th>Alumi</th>
<th>Glass</th>
<th>Rubber</th>
<th>Plastics</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lead</td>
<td>11340</td>
<td>1.0</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Copper</td>
<td>8960</td>
<td>1.1</td>
<td>1.0</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Brass</td>
<td>8660</td>
<td>1.2</td>
<td>1.0</td>
<td>1.0</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Iron</td>
<td>7870</td>
<td>1.2</td>
<td>1.1</td>
<td>1.1</td>
<td>1.0</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Aluminium</td>
<td>2699</td>
<td>2.5</td>
<td>2.2</td>
<td>2.1</td>
<td>2.0</td>
<td>1.0</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Glass</td>
<td>2600</td>
<td>2.5</td>
<td>2.2</td>
<td>2.2</td>
<td>2.1</td>
<td>1.0</td>
<td>1.0</td>
<td></td>
</tr>
<tr>
<td>Rubber</td>
<td>1870</td>
<td>3.4</td>
<td>3.0</td>
<td>3.0</td>
<td>2.8</td>
<td>1.4</td>
<td>1.4</td>
<td>1.0</td>
</tr>
<tr>
<td>Plastics</td>
<td>1210</td>
<td>7.0</td>
<td>6.2</td>
<td>6.0</td>
<td>5.7</td>
<td>2.8</td>
<td>2.8</td>
<td>2.0</td>
</tr>
</tbody>
</table>

4.5.1.5 Screening Technology

The screening of particles acts as a form of size classification to most optimally prepare the waste streams for the further downstream processes. Screening is therefore a way of sorting the waste stream according to particle size once it has been fragmented. It uses consistently defined aperture sizes to filter the waste stream, those particles with sizes smaller than the aperture will be segregated from those larger, bulkier particles. A detailed review of screening technologies can be found in the work of Suttill (1990).

Although a great range of screening equipment exists that vibrate and agitate the waste stream in various directions, one of the most widely adopted techniques is that of the trommel. This uses a number of circularly aligned meshes of varying aperture sizes that slowly rotate as the waste is fed in. Each section of the trommel has progressively larger apertures to further segregate the courser particle sizes. A number of mathematical models have been developed to consider the various design attributes of various screen types on separation effectiveness (Soldinger 2000, Subashinghe et al. 1989a, Subashinghe et al. 1989b). Research aimed at determining a simplified separation metric to describe the separation capability of a screen is presented in Mohanty et al. (2003) and utilises particle diameter and aperture size as a means of determining screening efficiency.
4.5.2 Feedstock and Energy Recovery Technologies

Once fragmented the waste stream undergoes a number of separation processes to remove its more saleable material content, the remaining feed (composed mainly of dirt, light plastics and organic materials) is sent to landfill, or can alternatively be processed by further feedstock or energy recovery technologies. A comprehensive review of various technologies can be found with Ferrao et al. (2006). The majority of the techniques are currently not widely adopted within the UK, and as such are considered beyond the scope of this literature review.

4.5.3 Landfill

Landfill taxing has become a progressively more prevalent method for many Governments to divert waste away from refuge dumps and extract any value that still exists. Landfill taxation is a direct result of the European Landfill Directive (European Union - Directive 1999/31/EC 2000), which requires member states to follow certain standards, targets and administration guidelines regarding the disposal of waste. Landfill taxation is is not solely restricted to Europe, with many other countries that have a high population to landmass ratio (such as Japan) introducing the same kind of economic instruments. Table 4.5 shows landfill costs taken from Kanari et al. (2003)
Chapter 4

<table>
<thead>
<tr>
<th>COUNTRY</th>
<th>COST (£)</th>
<th>POPULATION DENSITY (POPULATION PER KM²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Austria</td>
<td>72</td>
<td>98</td>
</tr>
<tr>
<td>Belgium</td>
<td>28</td>
<td>341</td>
</tr>
<tr>
<td>Denmark</td>
<td>36-72</td>
<td>126</td>
</tr>
<tr>
<td>France</td>
<td>20-31</td>
<td>110</td>
</tr>
<tr>
<td>Germany</td>
<td>31-87</td>
<td>232</td>
</tr>
<tr>
<td>Italy</td>
<td>38-41</td>
<td>193</td>
</tr>
<tr>
<td>Netherlands</td>
<td>36-46</td>
<td>392</td>
</tr>
<tr>
<td>Spain</td>
<td>10-31</td>
<td>85</td>
</tr>
<tr>
<td>Sweden</td>
<td>46-51</td>
<td>20</td>
</tr>
<tr>
<td>UK</td>
<td>15-18</td>
<td>246</td>
</tr>
<tr>
<td>Japan</td>
<td>69-82</td>
<td>339</td>
</tr>
<tr>
<td>US</td>
<td>26-31</td>
<td>31</td>
</tr>
</tbody>
</table>

Table 4.6, A mixture of countries and their associated landfill taxation and population densities and population density figures from United Nations World Populations Prospects Report (UN 2006).

The standard landfill tax rate is currently set at £24 per tonne (Sanderson 2007), the last of a £3 incremental increase (HMCE 2006) before the introduction of an £8 per tonne year-on-year increase from 1st April 2008 (HM Treasury 2007). This tax will become an increasingly influential economic instrument over the coming years, potentially forcing large shredder residue producers to effect change based on their bottom line as opposed to prescriptive ELV legislative targets.

4.6 Summary

As the focus of the research is on supporting the activities and decisions made when a vehicle is retired, the chapter has focused more extensively on end-of-life literature, namely current pre and post fragmentation activities within the UK. At the ATF processing stage this has considered the potential for recycling and re-use in terms of manual material removal and component resale, while at the shredder and dense
media processing stage it considered the current technology available to support recovery and recycling. This chapter has provided an overview of the main environmental considerations of reclamation activities throughout the vehicle value chain. It has mapped the environmental prioritisation criteria specified by the waste pyramid onto the current upstream and downstream ELV recovery activities, and demonstrated the importance of manufacturer and supplier support in promoting sustainable vehicle recovery. It has also provided an understanding as the most favourable sustainable end-of-life processing scenarios currently available, and will assist in selecting a suitable sustainable strategy to be tested for economic viability later in the thesis.

One of the most important issues this literature survey has highlighted is the fundamental role that upstream designers and manufacturers can have on a vehicles recovery regarding end-of-life monetary value and sustainable processing. Combined with the incongruous implementation of the ELV directive within the UK the resulting situation has somewhat isolated the ELV recovery sector. This would therefore suggest that the realisation and transparency of end-of-life value, in terms of supporting downstream pre and post-fragmentation reclamation activities, is the only true way in which the vehicle recovery sector can help itself.
Chapter 5

Review of Costing Techniques and Existing End-of-Life Vehicle Cost Models

5.1 Introduction

This chapter is broken down into two main areas. The first provides an overview of the main quantitative cost modelling techniques available, and their suitability to be applied to vehicle recovery stakeholders and their associated activities. The second is a review of research work carried out in the area of vehicle reclamation costing, and in particular the changing focus and costing resolution that previous work has obtained (from generalised legislative conformance costing through to detailed disassembly and shredding costing). The chapter concludes by highlighting the gaps within the literature that support the research assertion.

5.2 Costing Techniques

Quantitative estimating techniques can be broadly classified into three main groups; parametric models (Parametric and Artificial Neural Networks), analogous models (Case Based Reasoning) and detailed models (Generative-analytical) (Asiedu and Gu 1998). What follows is a brief overview of the various modelling approaches, concluding with an assessment of the various advantages and limitations associated with them.

5.2.1 Parametric Cost Estimating

Parametrics is a term that is used to describe a “top-down” estimating process, and has been accepted by both industry and government organisations for many years as a credible method of cost or price estimating (NASA 1999). Parametrics is usually based on some form of calculation and is reliant on quantitative data. The most widely used and well known form of parametrics is that of regression analysis developed by
the US Department Of Defence in the early 1950's, and has been increasingly expanded over the past half century (Cochran 1976).

The approach identifies relationships between product characteristics and cost dependent variables and uses statistical methods to generate mathematical expressions or formulas. These formulas are known as Cost Estimate Relationships (CER) and are capable of estimating a cost from multiple independent variables. This allows an engineer to specify general or detailed technical values of parameters that they would know, into a plausible estimate. An example of a basic CER could be the time taken to dismantle a component from a vehicle as a function of its mass, see Equation 5.1 and Figure 5.1.

\[
\text{Disassembly Time} = (a \cdot \text{Mass}) + c \quad \text{Equation 5.1}
\]

*Where* "a" *is a derived coefficient and "c" is the intercept offset.*

The correlation used within the CER must be founded on a valid trend. Within the above CER the trend identified is that of a linear correlation between mass and time. The CER can be made more complex by including logarithmic or higher exponent values, and additional parameters that are independent of others within the CER can also be added, see Equation 5.2 and Figure 5.2.

\[
\text{Disassembly Time} = (a \cdot \text{Mass}) + (b \cdot \text{Complexity}) + c \quad \text{Equation 5.2}
\]

*Where* "a,b" *are derived coefficients and "c" is the intercept offset. With all parameters still exhibiting a linear relationship with time*

Although the above examples use CER with very high-level product characteristics that can be determined very early on within the product development process to provide a cost estimate, the CER can also be composed of more accurate design time
Figure 5.1, Linear regression using one dependent variable

\[ y = 294.81x - 15.804 \]

\[ R^2 = 0.9696 \]

Figure 5.2, The surface created by a multi-variable polynomial regression equation
parameters when they become available. For example, rather than basing the parameter of “complexity” on a subjective decision, the knowledge available at design time might allow this parameter to be expanded to include low-level parameters such as “number of fastenings”, a “components’ tolerances”, etc.

Two major decision factors as to the effectiveness of regression analysis is routed in its reliance on quantitative data, and the analysts ability to effectively identify the parameters driving the cost. The first of these problems relates to the fact that CER, when used within a cost model, are only as good as the quantifiable historical or empirical data on which they are based. The problem being not the actual deriving of the formula’s coefficients, but the writer’s ability to collect and collate enough data to make them valid. “It is important to note that no degree of sophistication in the use of advanced mathematical statistics can compensate for a seriously deficient database” (NASA 1999). A sentiment echoed within the Bras and Emblemssvag’s paper (1995) when discussing the modelling of uncertainty within cost models via traditional Gaussian statistics. The second of these issues relates to the fact that a certain amount of expert knowledge may be required to identify the appropriate cost drivers. Cost drivers might not be composed of a single parameter, but instead from a function using a multitude of parameters that are not always immediately apparent (Farineau et al. 2001, Hoult and Meador 1996). An example of a cost driver composed of a function might be the availability of spaces within a dismantling yard. It might be very clear from the regression analysis that there is a clear correlation between number of spaces and the cost of storage, but space might be a function of a number of other parameters. Equation 5.3 gives an example of this.

\[
\text{Cost of storage} = a \cdot \text{NoOf Spaces} + c \quad \text{Equation 5.3}
\]

Where “a” is a derived coefficient and “c” is the intercept offset.

\[
\text{No. of Spaces} = \frac{\text{Length of Yard} \times \text{Width of Yard} - \text{Vehicles In} + \text{Vehicles out}}{\text{Average footprint of vehicle}}
\]

A further detailed review of these points can be found within the work of Matson et al. (1994).
5.2.2 Artificial Neural Networks

Neural Networks have been implemented in a number of various technology sectors, most noticeably in the area of computer signal processing, but only more recently has the approach been used to provide cost estimates. Artificial Neural Networks (ANN) work on the same physiological principles on which the brain operates. Control nodes (known as "neurons") react to the weightings applied to them via the pathways that connect them, otherwise known as "synapses". The level of conduction through the synapses determines a specific strength of connection between the nodes. Knowledge is stored in a "distributed" manner and coded via these weightings (Cavalieri et al. 2004). One of the more salient strengths of an ANN is its ability to learn and adapt. "The network is presented with a set of known cases (the training set) which is used to "train" the network". (Finnie et al. 1997). Initially the network is trained using historical data that calibrates the strength of the synapses connections (see Figure 5.3). Neurons then react accordingly depending on the sum of the weightings of the synapses linked into them. For a more detailed review of the foundation principles of ANN see Haykin (1999).

Unlike regression analysis ANN are a non-parametric estimating tool, in that they do not require specialist knowledge to generate the functions that describe the relationships between the product attributes. Unlike parametrics, neural networks "estimate a function without requiring a mathematical description of how the output functionally depend on the inputs" (Wang et al. 2000). The network automatically calibrates itself by adjusting the weightings between the synapses based on the variation of input and output parameters from the historic data. This means that the approach does not need the user to understand the sometimes complex internal relationships within a cost driver, as the approach will automatically detect any hidden relationships. The down side to this is that the cost engineer might never fully know what the actual cause of an estimate adjustment is. The idea of CER writing themselves, reduces the need for a detailed understanding of the cost
Figure 5.3, An Artificial Neural Network, adapted from the work of Cavalieri et al. (2004) on multi-layer preceptor neural networks

relationship within the recovery process, but an estimate’s lack of traceability can potentially reduce the validity of any model developed. Finnie et al. (1997), concludes that ANN are very effective at capturing the influence of different parameters on the eventual output, but what the approach gains in accuracy it loses in its ability to be traced and comprehended by the individual involved within the estimating process. The technique is not transparent enough for the user to clearly see how an estimate has been formulated. And the strength of any investment decisions based on this method’s output would rely solely on the mysterious “black box” workings of the ANN model.

5.2.3 Case Based Reasoning (CBR)

Analogy-based estimation involves the comparison of a particular instance (target) against a previously defined historic instance (source). One of the more widely recognised analogous techniques is that of Case-Based Reasoning (CBR) and has been adopted extensively within a number of sectors, not least design. The technique
operates on the idea that the relationship between a source cases problem/solution is directly related to a target cases problem/solution (see Figure 5.4).

As with ANN, CBR has been developed from the cognitive psychology of problem solving that humans exhibit, and the idea that problem solving improves with experience. The analogous method works via the use of codification procedures that generate a "case indexation" for each of the instances. This allows a case to be stored according to certain attributes. Once a source case is retrieved and compared against a target case a "similarity measure" is used to determine how far the deviation is, and what the cost correction should be.

Like ANN, CBR allows a system to learn and adapt as new instances are assimilated and become part of the case-base. This would be extremely advantageous when considering application to End-of-life costing, in that the technique would not only provide a means by which the recoverability of a vehicle could be charted over time (which could easily predict future trends), but would adapt any model to include cases where end-of-life issues had already started to have been considered. In brief, this would mean that the model would update itself without the need for a costly and time-consuming re-analysis of data. For a more extensive review of CBR and its implementation and variations see Aamodt and Plaza (1994).

Figure 5.4, Basic representation of the Case Based Reasoning approach, adapted from Duverlie et al. (1999)
5.2.4 Generative-Analytical Models

Detailed models, otherwise known as generative-analytical models, use estimates of labour time and rates and also material quantities and prices to estimate the direct costs of a product or activity (Shields and Young 1991). This is the more traditional approach to developing cost estimates and uses detailed process planning information, as well as product characteristics. Unlike many of the statistical approaches, detailed models are based on a “bottom-up” approach that attempts to capture the detailed cost drivers that go into costing a particular activity or operation. These are very precise allocation rates, which accurately detail how a cost is affected, and require a very detailed knowledge of the product and processes. However, the most accurate cost estimates can be made using this approach (Asiedu and Gu 1998). Equation 5.4 demonstrates the simplistic nature of these equations.

\[
\text{Cost of vehicle de-pollution} = \text{Time (h)} \times \text{Labour rate (£/h)} \quad \text{Equation 5.4}
\]

As the approach focuses on the quantities, processes and rates used at the lowest-level of the estimate, the sheer quantity of these equations and the need to keep them up to date can prove to be problematic if effective data collection techniques are not adopted.

5.3 Discussion of Costing Technique Suitability at End-of-Life

The formulation of a quantitative cost estimate requires an appreciation not only for the suitability of the technique to be implemented at end-of-life, but also for its relative benefits when compared to the others available. The following sub-sections discuss the appropriateness of each technique to be implemented at end-of-life, and the literature considering their comparative merits.

5.3.1 Information Availability from the Product Development Process

Selection of the most appropriate costing technique must also be justified based on its intended use within the product development process. Many of the techniques require specific access to different types of data and knowledge to then function as intended.
An example of this would be if a cost estimate was generated at the early conceptual design stage to consider the costs of recovering the product at end-of-life. At this point in the product development process the designers lack the clarity of vision to answer a lot of the detailed questions (Will any of the components have hazardous materials? What type of fixing mechanisms will be used?). Only limited, high-level product attributes can be specified at this stage (approximate volume, mass and material, etc.). Farineau et al. (2001), attempted to look at these effects of varying degrees of product description on the resolution of an estimate when using regression analysis. He described the processing of a product in three different ways. Highly specific manufacturing processes (such as centreboring, hole drilling, etc), were tested as an example of having detailed design knowledge. Technical grouping of similar manufacturing functions provided the second type of product description, allowing a more generalised definition of the operations that could be defined more readily within the design phase. Finally, the product was described in terms of a high-level complexity function that used general parameters as opposed to processes. Farineau concluded that all three product definitions were satisfactory in the quality of their estimates, but certain descriptions could only be made if the parameters were available to the designers at the time. The work of Seo et al. (2002) further outlines many of these problems. Similarly, estimates generated at end-of-life have the same problem of a lack of available data, although this not due to lack of product definition but rather to the lack of collaboration between manufacturers and recovers.

A general approach for the selection of a cost modelling technique in the context of the product development processes is further discussed within Chauvet’s (1993) paper and shows the suggested application of various techniques at the various development stages (see Figure 5.5).

5.3.2 Data Source Applicability for Allowing Generic Modelling

The availability of upstream design data to facilitate an end-of-life reclamation estimate will be key if adequate accuracy is to be achieved. Ideally, this data should come from multiple sources, as this makes the model more generic and removes any specific manufacturer bias, e.g. the SMART car is highly modularised and easier to
Figure 5.5, The suggested application of the main cost technique groups within the product development process, adapt from the work of Chauvet (1993)

disassemble when compared with other models (van Hoek 2002). As a consequence whether the data is from multiple or an isolated vehicle manufacturer, can potentially affect the costing technique selected. Previous work that has directly assessed the benefits of using multi-organisational versus company specific data using different cost techniques, was carried out by Jeffery et al. (2000). This work provided a comparative study as to which type of data best suited which technique. The techniques under assessment were parametric regression analysis and case-based reasoning. The paper concluded that the regression technique provided considerably more accuracy within its cost estimate than the analogous approach when using multi-organisational data.

5.3.3 Accuracy of Technique and Knowledge Requirements

The relative merits of using different modelling techniques in terms of accuracy are discussed in Briand et al. (2000). Briand concludes that traditional parametric regression analysis outperformed the analogy-based approach when comparing the variances of the two techniques. This is in somewhat contrasted by other published comparisons, Finnie et al. (1997) concluded that both ANN and analogous models perform significantly better at providing estimates than regression models, although it must be noted that this study attempted to estimate effort as opposed to cost. Duverlie et al. (1999) discusses this comparison further and highlights the more precise results obtained via analogous models CBR approach when compared with parametric
regression, but also notes CBR time-consuming setup requirements and regressions rapidity when generating estimates. Bode's (2000) paper on the assessment of ANN in cost estimating concludes that neural networks do provide a better cost estimate than traditional cost estimating methods, but only if sufficient case bases are available to train the network, and that the cost drivers selected are influential. In his conclusions he highlights the criteria for technique selection (see Table 5.1).

5.3.4 Transparency of the Technique

Despite the relative merits of ANN compared to Parametrics in terms of precision, a more decisive trade-off that is not discussed within previous sections is that of the traceability of the estimate. As previously stated Parametrics like ANN operate a "black-box" approach to cost estimating, in that the procedures and systems that are used to generate an estimate are hidden from the user. One of the greatest barriers for the acceptance of a cost estimate is the user's own belief that the model is generating the correct values. Although both techniques lack the transparency to see their internal operations, only Parametrics has the ability to provide some form of traceability via

<table>
<thead>
<tr>
<th>Use neural networks when...</th>
<th>Use parametric cost estimation when...</th>
<th>Use detailed cost estimation when...</th>
</tr>
</thead>
<tbody>
<tr>
<td>...you have quite a few similar cases from the past and...</td>
<td>...you have quite a few similar cases from the past and...</td>
<td>...you know the exact number of work hours and quantities required and...</td>
</tr>
<tr>
<td>...you are quite certain which attributes have a cost effect and...</td>
<td>...you know precisely which attributes have a cost effect and...</td>
<td>...you know precisely which attributes have a cost effect and...</td>
</tr>
<tr>
<td>...cost drivers are few and...</td>
<td>...cost drivers are not too many and...</td>
<td>...you know exactly how drivers influence cost.</td>
</tr>
<tr>
<td>...you do not know how drivers influence cost.</td>
<td>...you are quite certain how drivers influence cost.</td>
<td>...you know exactly how drivers influence cost.</td>
</tr>
</tbody>
</table>

Table 5.1, Criteria selection for cost estimation techniques (Bode 2000)
its CER. A questionable estimate can be related back to a formula. At the basic operator level the user can play with these CER to see how the various parameters ultimately affect the final cost. Neural networks do not provide this insight, and as such lose a lot of user confidence. A sentiment echoed in Cavalieri et al. (2004) comparison of parametrics and neural networks. This traceability and transparency should be a key factor in the technique ultimately selected. Table 5.2 provides a summary of the advantage and disadvantages of the various costing techniques described within the previous sub-sections. Further, more general criteria and shortcomings of the various cost estimating techniques can be found in the work of Layer et al. (2002) and Brinke et al. (2004).

<table>
<thead>
<tr>
<th>▲ – Excellent, AVERAGE Parametrics</th>
<th>Artificial Neural Networks</th>
<th>Cased-based Reasoning</th>
<th>Generative-analytical</th>
</tr>
</thead>
<tbody>
<tr>
<td>Statistical validation (F-test, R², etc.)</td>
<td>▲</td>
<td>▲</td>
<td>▼</td>
</tr>
<tr>
<td>Time taken to produce estimate</td>
<td>▲</td>
<td>▲</td>
<td>▼</td>
</tr>
<tr>
<td>Ease of adapting and learning</td>
<td>▼</td>
<td>▲</td>
<td>▼</td>
</tr>
<tr>
<td>Estimate transparency and traceability</td>
<td>▼</td>
<td>▲</td>
<td>▲</td>
</tr>
<tr>
<td>Can be used with both vague and detailed design knowledge</td>
<td>▲</td>
<td>▲</td>
<td>▼</td>
</tr>
<tr>
<td>Ability to handle variation and innovation</td>
<td>▼</td>
<td>▼</td>
<td>▼</td>
</tr>
</tbody>
</table>

Table 5.2, An overview of the techniques advantages and disadvantages
5.4 Review of End-of-life Vehicle Reclamation Costing

Historically, the majority of end-of-life cost modelling has been focused at upstream manufacturer reclamation costing, in the hope of supporting in-house recovery processing (i.e. de-manufacturing strategies as means of returning value). Only more recently with the advent of EPR has the focus shifted to support the manufacturers in conformance costing their legislative requirements. A great quantity of the research assumes manufacturers would provide both financial and technical support to the vehicle recovery sector in achieving the requirements of the directive, as it would be in their vested interests to do so. Hence, many vehicle reclamation cost estimates have been developed and linked with upstream design attributes or vehicle compositional data. Chapter 3 highlighted the incongruous implementation path of EPR through the ELV directive, and that access to potentially useful data to assist in the formulation of end-of-life cost estimates would not be made available to the current vehicle recovery industry. The following sub-section reviews the main end-of-life costing literature previously undertaken, and attempts to highlight the distinct lack of research focus on the new situation created in the UK's vehicle reclamation sector.

5.4.1 Pre-legislation Vehicle Reclamation Costing

Before the introduction of the ELV legislation in 2000 and the inclusion of producer responsibility, the majority of cost modelling focused on the potential value recovery opportunities using the existing technology and practices. One of the first pieces of costing research that specifically considered the End-of-Life automobile recovery scenarios was that produced by Diffenbach et al. (1993). Even at this early stage shredder residue was highlighted as a problematic waste stream that end-of-life operators should be focusing on, with the paper discussing the development of a "technical cost model" to assess the various recovery possibilities. Hock and Allen (1993) uniquely focused on one of these recovery scenarios (pre-fragmentation material recovery and recycling), considered the economic potential therein, and concluded that there was potential for value recovery if a number of issues could be overcome. The energy requirement of various automobile recovery scenarios and the subsequent economics is also discussed in Das et al. (1995). Further automobile recycling economics within the US was considered within Zamudio-Ramirez's (1996)
thesis, that developed a disassembly optimisation model to facilitate pre-fragmentation value recovery. This value recovery theme is further continued within Gupta and Isaacs (1997) paper that developed “profit function” equations for both the dismantler and shredder to assess the value achievable from each disposal strategy.

5.4.2 Generalised Vehicle Reclamation Costing

Each member state that has transposed the ELV directive has undertaken a number of economic impact assessments. This research tends to be very high-level estimating as to the economic viability of various transposition alternatives. These are not dynamic cost models and tend to provide only a momentary snapshot of the current situation. Examples of this generalised cost modelling can be seen within the François (2003) French example, Straudinger and Keoleianand (2001) US model, and Sakkas and Manios (2003) Greek case-study. Specific modelling that has considered the UK transposition can be found within Skinner and Fergusson (2003) which provides rough costings of each transposition option (i.e. last owner pays, producer pays, exchequer pays), and the Department of Trade and Industries full regulatory impact assessment (DTI 2005).

5.4.3 2006 and 2015 Target Attainment Costing

To-date the majority of research regarding the economics of vehicle reclamation has been born out of a need to conformance cost current and future ELV Directive recycling and recovery targets. One of the first cost models that considered this was produced by Johnson and Wang (2002), and outlined the additional dismantling and energy recovery activities that were required to fulfil the 2006 target. One of the more controversial aspects of this paper was the detrimental economic effects that the 2015 target would have in terms of vehicle dismantling. The paper proposed that a recycling rate of 87.6% could be achieved if all 42 plastic components were removed, but this would incur a cost of $28.16 per ELV. With the economic viability of vehicle dismantling in question, subsequent costing research focused more on the importance of high volume shredding activities. Amaral et al. (2006) modelled the effects of two different plastic separation efficiencies within the shredding process (25% current and
40% optimised), in the hope of fulfilling the 2015 recycling target (85%). The paper concluded that despite the improvements in plastic segregation that an optimised facility would have, additional dismantling activities would be still needed to meet the higher target; incurring plastic dismantling costs of around 43 € per ELV for the lower shredder efficiency (25%), and 3 € per ELV for the higher (40%). Further work produced by Ferrão et al. (2003) have developed more comparative economic analysis of dismantling versus shredding, and highlighted the effects that key factors such as vehicle composition have on dismantler and shredder profitability (Ferrão and Amaral 2006a). Ferrão et al. (2006) continue this economic modelling to incorporate the possibility of energy recovery technologies (Pyrolysis, Gasification, Co-combustion) being used as a means of target attainment, but concluded that traditional mechanical recycling (dismantling & shredding) offer a more realistic possibility to achieve the proposed targets.

5.4.4 Future End-of-life Vehicle Disposal Issues

The likelihood of achieving the 2015 recycling and recovery target is further questioned in Reuter et al. (2006) who investigates the fundamental limits of ELV recycling based on an amalgamation of mechanical, feedstock and energy recovery technologies. Reuter refers to the rigid inflexibility of the current quota driven system, and the problems it will encounter as innovative light-weighting materials will result in more expensive and innovative disposal scenarios, also highlighted in Mark and Kamprath (2004). Boon et al. (2003) further considers the implications of new automobile materials of future end-of-pipe economics, and models the economics of cleaner vehicles such as electric and hybrid electric vehicles, and the resultant effects this will have on dismantler and shredder profitability. Research aimed at adapting existing recovery facilities to deal with these new types of materials in the most effect and profitable way is discussed within Williams et al. (2006). In this paper it is proposed that changes are made to the current shredding processing route based on the altering composition of ELVs. The increasing abundance of aluminium and plastics within a vehicles composition can be modelled to identify the optimum point at which shipment should occur, and how various waste streams can be banded together to improve costs. It should also be noted that this paper is one of the first instances in which environmental improvement is inferred indirectly by improving

65
waste stream value. The optimisation of shredding activities to improve profitability is
the main focus, whereas in the majority of the past literature legislative recovery
targets have forcefully driven recovery activities and costing.

5.5 Summary

This chapter has provided an overview of the different costing approaches, and has
highlighted the main literature considering their comparative merits. This review is
intended to provide a foundation on which the most appropriate techniques can be
selected for use within the ELV cost model framework to be developed.

The chapter has also considered the main literature regarding end-of-life reclamation
costing, and highlighted the changing focus as to why it has been undertaken. The
majority of early research was designed to support manufacturer recovery and de-
manufacturing activities, before moving more towards legislative conformance
costing. The shortcomings of these previous ELV modelling activities are routed in
the assumption that vehicle reclamation costing directly benefits the vehicle
manufacturers, hence many rely on the availability of upstream data (i.e. bills of
materials for disassembly, vehicle compositions for identifying end-of-life value) to
be easily transferred from vehicle designer to end-of-life recover. The reality that the
vehicle recovery sector has been isolated by the ELV directives transposition within
the UK has overridden the assertion behind much of the previous costing research
(e.g. Design for Disassembly, CAD to end-of-life cost).

The future focus for research within this area would suggest a third reason behind
reclamation costing, one not so involved with the automotive manufacturers, but still
born out of legislative necessity as well as sector advancement. Williams et al. (2006)
provides a good example of this, inferring that reclamation costing should be used as
a means of improving operational effectiveness under the backdrop of environmental
regulation, not using environmental regulation as a driver for change in end-of-life
operations. This new costing paradigm within ELV recovery therefore requires an
array of costing approaches (devoid of direct manufacturer assistance) to assess the
economics of vehicle recovery from an end-of-life perspective. This should be
identified as one of the key differentiators between this research and previous ELV cost modelling activities.
Chapter 6

Research Methodology

6.1 Introduction

This chapter outlines the research methodology used within this thesis. The chapter begins by discussing the initial context of the research and the exercises and activities that assisted in formulating the original research assertion. Once the scope and goal of the research has been identified, the approach adopted to develop the costing framework and the industrial data required to realise this has been highlighted. The chapter concludes by describing the proposed industrial case studies and how sustainable processing strategies are to be tested in the context of the ELV cost model.

6.2 Research Methodology

The methodology adopted is in line with that traditionally used within a research program, which consists of four distinct phases; review and background, data collection and framework development, testing and validation, and finally the assessment and formulation of conclusions. Figure 6.1 provides an overview of the research methodology adopted within this thesis.

Development of an industrial viewpoint as to the main drivers and issues within the vehicle recovery sector was supported by a program of industrial visits to the main End-of-life stakeholders, and the deployment of an online survey. These interviews were not limited to just the vehicle recovery sector but also encompassed vehicle manufacturers, component suppliers and material re-processors. This allowed for a more thorough consideration of the relationships between the actors within the vehicle value chain, and provided a more detailed understanding as to the end-of-life data available. For example, the formulation of manufacturer collection contracts during
Figure 6.1, Research methodology used within the thesis
this initial period raised considerable issues with regard to the sharing of upstream manufacturing and assembly data with downstream recoverers. The online survey further supported the initial interviews and was used to collect industrial data and gauge the sectors viewpoint as to the required reform. The results of this investigation into the industrial viewpoint were subsequently used to further refine the research assertion.

Establishment of this industrial viewpoint moved the research into its second phase (data collection, framework development and cost model creation), and ultimately lead to the development of a framework with which ELV processing costs could be analysed. Based on the initial literature review it became apparent that this costing framework would need to encompass both pre and post-fragmentation separation economics, and utilise a varied range of costing techniques, from well-established approaches to cases where innovative new ones were required. It was intended that this framework be realised in a number of smaller spreadsheet cost models, which were then be brought together within an holistic ELV cost model to consider all the reclamation activities.

The third phase of work involved the validation of the various costing approaches within the ELV cost model, via the use of synthesised datasets, before a real case study was then undertaken. This case study was then established as base model (“as-is” model) with which typical processing costs and revenues could be simulated. This was then used to investigate further sustainable processing scenarios, currently not adopted within the UK industry, to test their feasibility and optimisation (“to-be” models). This future modelling was carried out under the context of achieving the 2015 recycling and recovery target.

The final phase of the research methodology was to analyse the research results to develop the concluding discussion within phase three.
Chapter 7

A Framework for Cost Modelling in End-of-Life Vehicle Recovery

7.1 Introduction

This chapter discusses the formulation of a systematic approach to model the end-of-life reclamation activities currently undertaken within the vehicle recovery sector (referred to in this thesis as the “as-is” model). The chapter begins by defining a system boundary for this “as-is” model, and highlights the main stakeholders and activities, before identifying the demographic of vehicle most likely to be processed through this system. The overall aim of this chapter is to provide an understanding as to how a used vehicle affects a recovery process, highlighting the direct and indirect costs/revenues incurred when various activities are undertaken. Identifying these relationships will assist in the generation of a formalised framework, capable of accounting for both the direct and indirect costs of current vehicle reclamation activities.

7.2 Vehicle Recovery Process Model and Associated Cost Types

For the vehicle recovery sector to accurately cost their reclamation activities there is a need to identify and establish costing methods most suitable to the information available and costing resolution required. Chapters 3 and 4 provided an overview of activities and technologies currently influencing the vehicle recovery sector, and highlighted the volatile situation created by the implementation of the ELV Directive. The following sub-sections build on these soft-issues and describes the need to define a typical end-of-life vehicle processing scenario in which costs and revenues can be identified.
7.2.1 A Typical ELV Processing Scenario and Associated Stakeholders

The interface between last owner and authorised treatment facility site is the start of a long and diverse journey for many of the vehicle’s materials, and encompasses a range of operators and industries peripheral to what is traditionally perceived to be the vehicle salvage sector. The more obvious fringe operators such as metal re-processors and the Driver and Vehicle Licensing Agency (DVLA), have become highly visible appendages to the industry in recent years. Yet other stakeholders such as the recovered fuel oil and rubber crumbing sectors, that play important yet subdued roles, are not as transparent. Given the variation in operators and activities that the vehicle recovery sector is subject to, it is necessary to identify a system boundary around the stakeholders to be considered. Figure 7.1 has been developed based on the industrial interviews and literature survey undertaken, and provides a snapshot of the typical operators involved in removing materials in the vehicle retirement process.

Figure 7.1, The main end-of-life stakeholders within the vehicle recovery chain and the typical operators involved in the materials removed from an ELV
The key operators of the ATF, shredder and dense media separator are identified as the main three stakeholders around which the framework will be developed. Figures 7.2-7.5 provides generalised process models (using IDEF0 diagrams) for the main stakeholder within the vehicle recovery process. These models are intended to give a generic overview of the main activities undertaken and the main resources required, which will then underpin the identification of different cost types that need to be considered within the framework. Figure 7.2 is the root IDEF0 diagram highlighting the logistical return of the vehicle and subsequent operators that process the main bulk of the ELV material. Further sub-level activity diagrams, describing the operators activities in more detail are then shown within Figure 7.3 (ATF), 7.4 (Shredder) and 7.5 (Dense-media Separator). These IDEF0 diagrams collectively group the main resources, controls and outputs under generalised headings, and are not intended to be an exhaustive representation of a real world operation. To see such a bespoke IDEF0 model for an industrial application, see section 12.3.1 within the case study chapter.

Figure 7.2, An IDEF0 model representing the main end-of-life stakeholders
Chapter 7

**Figure 7.3,** An IDEF0 model for activities carried out by an ATF

**Figure 7.4,** An IDEF0 model for activities carried out by a shredder
7.2.2 Identification of Different Costs and Revenue Types

To develop an holistic costing approach for the current vehicle reclamation industry an appreciation is needed for the different types of costs incurred. These can be broadly classified into two main groups:

- **direct costs** of ELV processing that are directly relatable to the activity performed, e.g. the cost of vehicle collection is directly dependent on how far the collection truck must travel.

- **indirect costs** of ELV processing that are not directly accountable to a specific process, e.g. the cost of heating and lighting a facility.

Direct costs and revenues are often more visible and easier to catalogue, as their links with throughput are more readily seen and understood. Component removal is an example of a direct cost as this is directly dependent on the number sub-assemblies to be removed and the labour effort required; consequently revenue is only generated...
once this activity has been performed. Other more obscure direct costs and revenues are those incurred during the post-fragmentation separation of materials. Automated separation technologies can be considered as a direct revenue due to the variation in value achieved when considering different compositions of waste. The process capabilities of the equipment used within the separation activity are always static, but their effectiveness in extracting value is not. For example, consider the value-added processing of a magnetic separation device on a highly ferrous rich waste stream when compared to one composed mainly of plastics. The overall value improvement of the ferrous stream is significantly greater than that of the plastic, yet the cost of processing one tonne of waste is the same in each instance. The direct revenue from the waste stream is ultimately a function of its compositional contamination and pure material market value. Hence, the mixture of the waste that is placed into an automated process bears a direct relationship on its resulting value.

The other type of costs are indirect costs. These costs make a substantial contribution to the cost of processing a particular vehicle, but often trying to attribute them to a particular product or process is not straightforward. Therefore, indirect costs rely heavily on a cost estimator's ability to understand the processes and resources required to achieve a specific activity. An example within the vehicle recovery sector might be the cost incurred due to a new piece of equipment, say a vehicle de-pollution rig. The rig incurs more obvious indirect costs such as depreciation, power, maintenance, etc., which must all be absorbed and recouped by the vehicles it assists in de-polluting. Not as obvious are those other indirect costs required to have a de-pollution rig in the first place. A facility to house the rig is required (which provides heating and lighting), the rig must sit on a sealed concrete foundation (which incurs Uniform Business Rates, additional employee insurance, environmental agency site licences), the waste produced by the rig must also be managed and accounted for - the list is extensive. All these indirect costs need to be factored into the cost of processing an ELV but traditionally do not lend themselves well to straightforward product attribution.

Based on the literature review, semi-structured interviews and facility site visits, the generalised IDEF0 model discussed within the previous section was used to identify
Figure 7.6, Direct and indirect costs, revenues and the main externalities influencing ELV recovery

the main direct and indirect costs associated with the main end-of-life operators. Figure 7.6 provides an aggregated summary of these main direct and indirect costs experienced by each stakeholder, which highlight the various areas for consideration in the EOL costing framework.

7.3 Demographic of Vehicles to Support Data Collection

The EOL costing framework provides an understanding as to the main cost intensive areas to be considered within the vehicle reclamation process. This framework provides an appropriate starting point from which subsequent data collection activities can be undertaken to identify the most appropriate costing techniques. Given the large variety of vehicle make and model variations that exist within the current market, it is necessary to make some assertions as to the typical profile of an ELV to be processed so that these data collection activities are most effectively targeted. Each vehicle type has specific materials and components bespoke to its own design, hence when trying to catalogue direct costs such as vehicle dismantling costs (either for reuse or recycling) a more condensed sample of 'typical' vehicles are required.
The average age of a natural ELV (mechanical failure or retired) within the UK is roughly 12.8 years, and a premature ELV (insurance write-offs) is approximately 6.7 years (Kollamthodi et al. 2003). Therefore, to gain a representative sample of the most popular passenger cars coming to the end of their useful lives around the current time period, it is necessary to highlight the most prominent vehicles sold within the years identified by the average vehicle ages (13 and 7 years, respectively). Figure 7.7 was compiled with the assistance of the Society of Motor Manufacturers and Traders (SMMT), and highlights the top three selling vehicles that correspond to the typical demographic of Natural and Premature ELVs around the key target attainment dates.

Selection of these natural and premature ELVs provides a starting point from which further data collection activities can be directed, to assist in the formulation of both the "as-is" and "to-be" models. These makes and models will primarily be used to understand the cost and revenues associated with pre-fragmentation material removal operations and post-fragmentation waste stream compositions, and will be further utilised in Chapters 9 and 10.

Figure 7.7, The top three selling vehicles due for retirement around key target attainment dates and their corresponding market share in their year of manufacture
7.4 ELV Costing Framework

Given the drastic reform and investment the recovery industry is currently undergoing, combined with the dependency of the sector on the long-term stability of only a few key market drivers, the future financial profitability of the sector is highly uncertain. To expect operators to make any commitments in the face of this uncertainty and adopt more resource intensive environmental practices, without first giving them the ability to assess the economic feasibility of the various options, would not promote pro-active investment.

This research has therefore highlighted the necessity to demonstrate a “win-win” situation, highlighting the economic benefits that an end-of-life operator would gain if they adopted a more sustainable approach to vehicle recovery. This increase in value recovery could possibly require the diversifying of an end-of-life stakeholders main core competency, or the focused improvement of existing operations. In both these instances the starting point for this economic understanding is the generation of a formulised set of costing methods most suitable to the information available and costing resolution required at end-of-life. Selection and development of these techniques is referred to as the ELV costing framework, and will bring together a range of costing approaches most suitable to the end-of-life perspective of ELV recovery costing. One of the novel aspects of this framework is that it is very much based on the technical capabilities and information available to current end-of-life operators, and does not rely on substantial upstream manufacturer data and assistance.

The ELV costing framework is outlined within Figure 7.8, and provides an overview of the costing approaches selected to cost ELV reclamation activities within the UK. The ELV framework brings together a whole range of different costing modelling techniques, from those readily adopted within industry (i.e. Activity Based Costing), to the application of fringe techniques within this sector (i.e. parametric regression analysis), to situations in which a radically new costing approach was required all together (post-fragmentation cost modelling). The following three research chapters provide justification and development insight for each of the main costing techniques adopted within the ELV framework.
**INDIRECT COSTS**

- Facilities & Buildings
- Business rates
- Power consumption
- Equipment investment
- Maintenance
- Management
- Training & Insurance

**ACTIVITY BASED COSTING AND UNCERTAINTY MODELLING**

(Chapter 8)

---

**DIRECT COSTS**

Collection and Distribution Costs (Apply to all stakeholders)

- De-pollution costing
- Parts resale costing
- Material recycling costing
- Post-fragmentation materials separation costing
- Non-ferrous materials separation costing

**GENERATIVE-ANALYTICAL**

(Chapter 9)

**PARAMETRICS**

**THEORETICAL SEPARATION MODELLING**

(Chapter 10)

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**Figure 7.8**, The ELV Costing Framework highlighting the costing approaches to modelling end-of-life reclamation costs
Chapter 8

Indirect Vehicle Recovery Costing

8.1 Introduction

The ELV costing framework discussed within the previous chapter highlighted a requirement to model both the apparent direct costs and the more unintuitive indirect ones. This chapter discusses the need to consider indirect vehicle processing costs and their pivotal role in producing realistic cost estimates. Adoption of a cost accounting technique known as Activity Based Costing (ABC), capable of attributing indirect costs, will be demonstrated. Adoption of this relatively new cost accounting method also introduces additional issues peripheral to indirect cost attribution, namely the inclusion of estimate uncertainty. The latter part of the chapter discusses the need to move away from using just one value in isolation (single-point estimating), and the requirement to incorporate uncertainty modelling into the final indirect costing approach.

8.2 Indirect Pre & Post-fragmentation Cost Areas

Indirect vehicle processing costs are not unique to a particular stakeholder within the vehicle recovery process. Whether considering a small scale ATF or large dense-media separation plant each facility incurs indirect processing costs. These indirect cost areas include:

- Operating equipment, transport and building depreciation
- Business rates (U.B.R. - Uniform Business Rates)
- Power consumption costs (heating, light, equipment)
- Fuel costs (recovery transportation trucks, forklifts)
- Maintenance and consumables
- Taxation, licences, training and insurance
- Employee fringe benefits
Although diverse in range these areas can be significant in cost, and contribute heavily to a facilities overall expenditure. In addition, these cost areas impinge on a whole myriad of different end-of-life activities throughout the vehicle reclaimation process, and involve the costs of operating resources that are typically used within an activity. There are exceptions to this rule, in which the above cost areas are not linked with a resource, facility operating licenses being a good example of this. Here the cost of obtaining a facility license (be it for waste processing, ISO accreditation) cannot be directly related to a specific group of resources. In these situations each activity must absorb these un-attributed indirect costs to maintain the integrity of the model. Figure 8.1 provides a visual representation of these activity attribution issues, which are also further discussed within the next section under the limitations of the indirect cost modelling approach.

8.3 Modelling Indirect Costs via Activity Based Costing

Traditional cost accounting has always attributed indirect costs using direct cost-drivers (such as labour). The inadequacies of such approaches are well documented and have led to the development of Activity Based Costing (ABC) (Bruns and Kaplan 1987). ABC assumes that activities consume resources, and as such indirect costs such as overheads and equipment depreciation can be directly linked to a machines.

**Figure 8.1, Issues with attribution of indirect cost to activities**
utilisation and throughput. Attributing a resource to a typical activity can be often difficult if that resource is shared among a number of operations. Hence, effectively capturing these links and the sharing of resources (otherwise known as "cost-drivers") allows the attribution of the total operating cost of an activity to unit, batch or line level quantities. ABC is a cost traceability tool and provides a more detailed picture to management as to the cost intensive areas of their business.

8.3.1 The Activity Based Costing Approach

Figure 8.2 highlights the processing stages within the ABC methodology, which is outlined by No and Kleiner (1997), and is used by this research to model indirect costs. The first stage is to gain a detailed understanding of the processing activities the returning vehicle goes through. A structured modelling tool such as IDEF0 or Value Stream Mapping can be adopted to facilitate this process. This modelling can be undertaken at different levels of resolution depending on the detail required for the cost estimate. For example, the activity of vehicle collection can be further broken down into the sub-activities of outbound journey, vehicle loading and inbound journey, if the resolution required it. It is up to the cost estimator to aggregate the activities down to a level that gives the most reasonable amount of accuracy for the least amount of resolution. Figure 8.3 provides an example of this activity aggregation for a typical vehicle reclamation process. Stages 3 and 4 within the ABC methodology identify, quantify and attribute the cost areas previously highlighted within section 8.2 to each activity. It is important that this attribution of costs not only identifies the correct activities that consume the resources, but also weighs their influence accordingly. For example, if a forklift is used by four different activities (the unloading of the vehicle from recovery truck, the movement into and out of the de-pollution bay, and the placement of the vehicle into the bailer) then the total cost of running and maintaining the forklift must be proportioned to each activity based on some utilisation metric (allocated activity times been the most widely used). Once this has been achieved each activity has an allotted pool of costs, composed from numerous resources, each with different allocation weightings.
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Figure 8.2, The Activity Based Costing approach

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<tr>
<th>Level 1</th>
<th>Level 2</th>
<th>Level 3</th>
<th>Level 4</th>
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<tbody>
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<td>Vehicle return</td>
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<td>Journey to collection</td>
<td>Recovery vehicle loading</td>
<td>Journey to ATF</td>
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<td>Cost of buy-back</td>
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<td>Documentation processing</td>
<td>VM located on vehicle</td>
<td>Details and CoD issued</td>
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<td>Vehicle moved for processing</td>
<td>Vehicle moved</td>
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<td>Vehicle assessment and prep</td>
<td>Vehicle storage</td>
<td>Operation identification</td>
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<td>Top accessible fluids removed</td>
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<td>Filler caps opened</td>
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<td>De-pollution</td>
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<td>ATF processing</td>
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<td>Catalytic converter removed</td>
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<td>Remove part</td>
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<td>Removal of hazardous substances</td>
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<td>Catalogue part</td>
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<td>Assessment of dismantling</td>
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<td>Information gathering</td>
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<td>Parts/Material dismantling</td>
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<td>Destructive dismantling</td>
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<td>Removal of resale parts</td>
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<td>Sorting/storing of material</td>
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<td>Removal of material</td>
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<td>Remove part</td>
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<td>Removal of re-conditioning parts</td>
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<td>Catalogue part</td>
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<td>Crushing and compacting</td>
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<td>Vehicle loaded</td>
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<td>Compaction</td>
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<td>Hulk unloaded</td>
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<td>Transportation to shredders</td>
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<td>Journey in</td>
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Figure 8.3, Typical activity aggregation within an ATF
Stage 5 requires the identification of cost-drivers that directly apply the costs to the activity in question. These cost-drivers are quantifiable values that describe the operational parameters that facilitate the activity to be undertaken. Cost drivers can be chosen at the unit, batch, product or facility level, and are selected based on their influence over activity consumption. For example, the costs associated with vehicle de-pollution could possibly be related to the number of setup changes required, the number of units processed, or the time to complete per ELV. Once a suitable cost-driver has been selected and quantities for these drivers have been determined, the total indirect cost for the activity can be divided by the cost driver quantity to generate the consumption intensities (e.g. cost per unit). These consumption intensities (rates) demonstrated in Equation 8.1 are then used as multipliers when user-defined cost-driver quantities are specified.

\[
\text{Consumption intensities (£/unit) = \frac{\text{Total allocated activity cost}}{\text{Cost driver quantity}}} \quad \text{Equation 8.1}
\]

### 8.3.2 Limitations of Activity Based Costing

The attribution problems associated with indirect costing, highlighted within Figure 8.1, is no more apparent when considering the limitations of ABC. The example of attributing management salaries to an activity is often given within many accounting textbook. The manager of a facility incurs substantial labour costs which need to be allocated to an activity, but as their position within the company is strategic, there are no tangible links to a particular activity. Hence, in these instances costs tend to get accumulatively added to every operation, somewhat negating the principle behind ABC due the lack of cost traceability.

### 8.4 Uncertainty Modelling within Indirect Costing of ELV Recovery

A modelling consideration peripheral to the ABC methodology, but still an integral part in terms of generating a plausible model, is that of uncertainty modelling. The inclusion of uncertainty within the ELV cost model and the assessment of the probability of achieving a calculated estimate is important to maintain the model credibility. By moving away from straight-forward *single point* estimating,
uncertainty modelling can account for a serious lack of available information within the input parameters and create an estimate "confidence level" within the models output parameters. These two advantages of uncertainty modelling are particular relevant given the deficiency in manufacturer support created by the transposition of the ELV directive within the UK. The remainder of this chapter introduces the uncertainty modelling approach adopted when considering indirect ELV recovery costs, in particular; the selection of a suitable distribution to easily capture ELV costing knowledge, and the adoption of an appropriate random sampling technique to effective incorporate uncertainty.

8.4.1 Selection of an Input Distribution to Capture Indirect ELV Recovery Costs

When determining values for the indirect cost areas (as described in section 8.2) it is advantageous to move away from using one single value in isolation to formulate a cost estimate. Using single-point estimating can produce a highly detailed and bespoke cost model, but requires widespread data collection activities and costing expertise (which are not extensively undertaken or widely available within the current recovery sector). Hence, to overcome this problem the research has specified a range of possible values for the indirect cost areas considered. These ranges, otherwise known as input distributions, assist in describing the variations in end-of-life costs that a particular resource might incur.

As well as describing the range between which an ELV recovery cost might lie, these input distributions can also provide an indication as to the probability of a particular value occurring within that range. When the likelihood of a cost occurring is also included within the input distribution it becomes known as a Probability Density Function (PDF). Within these PDFs the area underneath the distribution is always equal to one, the x-axis refers to the parameter in question (in this case cost) and the y-axis is represents the probability of occurrence.

The selection of the most appropriate PDF is very much dependent on the availability of historic facility purchasing data, and the ability of an end-of-life operator to match a distribution to a trend seen within it. Due to the archaic nature of the reclamation sector, combined with a lack of abundant historic investment prior to the ELV


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directive, it has been extremely difficult to select bespoke distribution types for each of the cost areas considered. The research has therefore proposed that one single distribution type be adopted for all the indirect ELV costing areas, which allows for uncertainty modelling to be included within the ELV cost model without the need for lengthy data collection exercises. The distribution adopted is that of the BetaPERT distribution, which has been widely used within many commercial cost estimating packages to overcome the data availability issues. The main PDF equation for the BetaPERT distribution is given Equation 8.2 and shown in Figure 8.4.

**Figure 8.4** The probability distribution functions for three BetaPERT distributions

**Equation 8.2** The Equation for calculating the PDF of a BetaPERT distribution,

where \( B(\alpha, \alpha_2) \) is the Beta function
8.4.2 Random Input Distribution Sampling to Include Uncertainty Modelling

Once the research had selected a suitable PDF with which the variation in cost ranges could be described, the question remained as to which of the values within the distribution do you adopt for the ABC model. Determining the mean or 50th percentile for a BetaPERT distribution will always be consistent, and will only alter if one of the three input parameters (maximum, minimum and most likely) are changed. The BetaPERT PDF created by Equation 8.2 is an idealised case, and although it is advantageous to use idealised curves to generate consistent values, this is still a roundabout way of generating a single point cost estimate. There is therefore a need to regenerate the BetaPERT PDF with uncertainty included, so as to account for the lack of ELV costing data available.

Monte Carlo sampling is one the most widely adopted technique for incorporating uncertainty into a PDF distribution. It works by taking the idealised PDF and mapping it with its Cumulative Distribution Function (CDF). The CDF is another graphical representation of the PDF, with the x-axis still representing the parameter in question (e.g. catalytic converter price), but the y-axis highlighting the cumulative area under the PDF curve. As previously stated, the area under a PDF always equates to one hence when a PDF is converted to its CDF equivalent the maximum value represented on a CDF graph is also one (see Figure 8.5). Once the idealised BetaPERT PDF has been transposed into its CDF equivalent it is in a suitable form to be randomly sampled. This is implemented within the indirect costing approach by utilising a random number generator that selects a number between zero and one. This selects a random point on the y-axis of the CDF, and uses an inversed form of the CDF equation to determine the x-axis value (e.g. if the random number generated was 0.5 within the CDF described in Figure 8.5 the inverse form of the CDF would return a parameter value of £15). Each time a random number is generated a parameter value is identified (on the x-axis) based on the CDF, and can be directly re-transposed onto the original PDF. It is convenient to segment the CDF and PDF into intervals with which parameters values can fall, otherwise a great quantity of random numbers are required to be generated to make sure a parameter value stands a chance of being sampled and appearing on the re-generated PDF. Figure 8.6 shows the idealised PDF
Figure 8.5, The BetaPERT probability density function \( f(x) \) mapped onto its associated cumulative distribution function \( F(x) \)

Figure 8.6, The regenerated BetaPERT PDF histogram via Monte Carlo sampling
and its transposed CDF equivalent, along with the predefined interval sizes and regenerated PDF histogram.

As can be seen within the regenerated BetaPERT PDF within Figure 8.6, the histogram assumes the approximate shape of the idealised original, but incorporates uncertainty into the indirect vehicle recovery costing approach. These basic uncertainty modelling techniques are to be used in conjunction with the indirect ABC methodology previous described (see figure 8.7).

**Figure 8.7**, The application of uncertainty modelling into the ABC costing methodology adopted within research
Chapter 9

Direct Pre-fragmentation Costing

9.1 Introduction

This chapter discusses some of the main issues regarding pre-fragmentation material removal, and the costing techniques adopted by this research in light of the various pre-fragmentation activities currently possible. Chapter 7 identified a particular vehicle demographic which was representative of the makes and models of vehicles expected to be returned around key target attainment dates (i.e. 2006 and 2015). In this chapter these “typical” vehicle models are used to facilitate further data collection activities within the three areas identified as being the most common pre-fragmentation processes, namely, de-pollution, parts resale and material recycling. This data collection activities enables the main factors affecting costs for each process to be identified, so that the most suitable costing technique can be selected. This chapter specifically highlights, the application of generative-analytical costing to vehicle de-pollution and component removal, and the use of parametric regression analysis to develop make and model specific costing equations for manual material removal and recycling.

9.2 Data Collection Activities Relating to Direct Pre-Fragmentation Costing

When developing the ELV costing framework it became apparent as to the distinct lack of available end-of-life data relating to pre-fragmentation reclamation activities. Hence, as part of the research it was decided to undertake a program of industrial visits to key stakeholders. The results of these interviews directed the research to consider further data collection activities in the form of a web-based survey.
9.2.1 Industrial Interviews

An extensive program of industrial interviews were undertaken that encompassed stakeholders throughout the vehicle value chain. These stakeholders included:- (the number in brackets refers to the number of different organisations interviewed)

- UK policy makers (1)
- Tier 1 supplier (1)
- Vehicle manufacturers (3)
- Authorised Treatment Facilities (5)
- Shredding sites (2)
- Dense-media plants (2)
- Plastics re-processors (2)

The majority of the output from these interviews assisted to refining the industrial focus of the research, but also highlighted the distinct lack of collaboration between various value chain stakeholders. Many were keen to share their opinions as to the current implementation ELV directive, but none were willing to share extensive vehicle manufacturing and reclamation data.

9.2.2 Web-based Data Collection Survey

To overcome the issues identified with missing data and to also provide a more comprehensive understanding as to the wider opinions of the pre-fragmentation sector, a set of questions were developed and distributed to over 270 ATF sites within the UK. The majority were contacted via email and directed to an online web survey (see figure 9.1).

Twenty-four sites completed the survey creating a response rate of around 8.9%, which represented 2% of the UK’s pre-fragmentation capacity. The data collated and opinions presented through this exercise are subsequently utilised within the following chapter to assess the factors affecting pre-fragmentation costing.
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Figure 9.1, The web-based pre-fragmentation data collection survey

9.3 Current Pre-Fragmentation Practises

Material recovery pre-fragmentation can be broadly classified into two main groups; those materials removed for environmental considerations (carried out via the de-pollution process) and those materials removed for monetary value (parts resale and low-end recycling). As a legal requirement vehicle de-pollution is a perquisite, whereas component removal and plastics recovery are often carried out on a case by case basis. The direct labour costs incurred and revenues generated can therefore be extremely varied, and at times, quite substantial when compared to other ELV processing costs. There is therefore a requirement to determine the underlying cost factors that determine these vehicle specific costs and revenues during pre-fragmentation processing, and to attribute the most suitable costing approach to reflect the metrics and data available. Figure 9.2 provides a visualised overview of the main vehicle processing activities carried out pre-fragmentation and their associated removal classifications.
Environmental considerations:
De-pollution activities
Monetary value:
Parts resale activities
Recycling activities

PU foam removed from seat cushions
Airbag deployed
Wing-mirrors
Engine removed for resale, remanufacture, or component recycling
Battery removed
Headlamp lens
Coolant, screen wash, AC fluid, Power steering fluid
Catalyst removed for precious metals
Tyres & lead weights

Underside fuel and oil removed, shock absorbers drained

Figure 9.2, An overview of main pre-fragmentation activities and their associated removal classification.
9.3.1 Vehicle De-pollution

De-pollution is the process by which end-of-life vehicles are made environmentally inert, and has become an integral and cost intensive part of the ELV recovery process. This involves the removal of a number of key materials and fluids during the pre-fragmentation processing stage, to try and avoid further hazardous leaching of materials to the surrounding environment during downstream reclamation activities. The process typically involves a de-pollution rig that contains suction hoses and collection tanks for a range of end-of-life fluids. Depending on the price of the rig it normally allows a vehicle to be elevated, which assists operators in reaching underside fuel tanks and engine sumps.

9.3.1.1 Cost Factors Affecting Vehicle De-pollution

An appropriate starting point for the determination of vehicle de-pollution costing is to gain an understanding as to cost factors that drive the economics of the process. These can then be related to a suitable costing methodology most applicable to the situation. For costing the de-pollution process the initial approach involved a series of stakeholder interviews and survey questions, to determine the operators own opinions as to the factors affecting vehicle de-pollution time (see Figure 9.3).

Interestingly, the most prominent response in the survey had no relationship to vehicle de-pollution technology or design, but instead cited the thoroughness of the Environmental Agency (EA) in policing the current regulations as being the most influential factor. A response typical of many ATFs during the interview period, highlighting an underlying feeling that they have been unduly targeted despite their recent investment and that regulatory enforcement of the directive still varied from region to region. Additional influential factors cited included; the use of recommended Department of Trade and Industry (DTI) equipment, design variations between different makes and models and the external condition of the vehicle. Issues regarding the use of the correct industrial equipment are very much explained via the demographic of respondents surveyed, with many keen to demonstrate and justify their own facilities forward thinking investment during a time of increased industry...
Factors affecting the depollution of vehicles

Figure 9.3, Typical factors identified by facility managers as to the main causes of vehicle de-pollution processing uncertainty. This somewhat discounted the use of either the EA strictness or the sophistication of the de-pollution technology as a measurable cost factor, as these are very much facility specific factors, and do not lend themselves well to identifying time-related dependencies. This therefore leaves make and model design variations and external vehicle condition as the two areas highlighted by the survey as the most quantifiable de-pollution cost factors.

To investigate the results of the survey data further a series of time-studies were undertaken to gain a more accurate representation as to the main factors affecting vehicle de-pollution time. This data collection was undertaken at two different ATF sites (Albert Looms & ASM Autos – see figure 9.4) and involved observing the de-pollution activities for a range of natural and premature ELVs. Activity removal times were also catalogued for each de-pollution process to identify the most resource intensive activities (see Appendix A1.3 for complete listings). Potential inaccuracies within this data collection process were identified as:

- The use of non-standardised equipment and techniques at different facilities.
The varying skill levels of employees carrying out the work.

The variable degrees of effort in carrying out every de-pollution activity.

It became clear early within the observations that despite the initial expectation that each make and model of vehicle would have unique de-pollution activity times, that this hypothesis was incorrect. Instead, the time study highlighted that factors such as; the corrosive condition of the fixings, the amount of each of the fluids to be removed, contributed more to the overall de-polluting time than specific vehicle design attributes. This would therefore suggest that de-pollution cost factors are based on relatively indeterminable end-of-life metrics (i.e. the amount of fuel at time of retirement, the corrosive condition of tyre bolts, the ability of the operator to gain access to the vehicle to deploy the airbag, etc.).

9.3.1.2 Selection of Costing Technique for the De-pollution Process

Due to the indeterminable end-of-life factors that affect vehicle de-pollution time it is not possible to make a selection from a wide range of costing techniques. With no easily determinable input parameters and no extensive historic quantity of data to analyse it is proposed that a simplified version of the generative-analytical costing approach be adopted to cost vehicle de-pollution activities. In this approach standard de-pollution activity times will be generated, which in turn can be multiplied by the facility specific labour rates to create a standard activity cost. A vehicle specific cost

Figure 9.4, The observation of vehicle de-pollution at Albert Looms and ASM Autos
estimate is then generated by accumulatively appending the cost of other activities, if
the vehicle in question requires that operation to be undertaken. For example, if a
catalyst is still present then the standard processing time for catalyst removal is
appended, if the vehicle has been in a frontal impact the time taken to drain the top
level fluids is excluded from the estimate, etc.

An example sub-set of the de-pollution data catalogued from the time study at Albert
Looms is formally represented using the Program Evaluation and Review Technique
(PERT) in Figure 9.5 (Malcolm et al. 1959). The critical path within the PERT
diagram has been highlighted in red. However, it should be noted that this assumes
that a number of operators are available to process the vehicle in parallel, whereas as
in practise typically only one employee works on the vehicle at any one time.

9.3.2 Component Removal and Part-resale

Parts-resale has traditionally been the core market for many pre-fragmentation
treatment facilities. During recent times facilities have diversified their operations to
focus more on metal reclamation activities, given the reduction of the components
resale market and the increase in value of a vehicles metallic fraction. The sustained
low interest rates and 24% growth in consumer expenditure in the last 5 years (Mintel
2006a), are key factors as to the changing mentality of automobile owners.
Consumers no longer spend time fixing or upgrading their vehicles themselves,
instead they use attractive financing agreements to buy new ones exemplified by the
fact that in 2005, 46% of all finance agreements were taken through car dealers
(Motor Trader Magazine 2006). Also, the increased use of integrated electronics
within components, and the increasingly sophisticated manufacturing techniques used
by many component suppliers (high-end laser and spot welding equipment) has
successfully dissuaded many would-be amateur mechanics from repairing their own
vehicles. As a result the parts-resale business has suffered, supported by 2/3 of
respondents with the authorised treatment survey suggesting that in their opinion the
longer-term sales of components for reuse within the UK will further decline.
<table>
<thead>
<tr>
<th>Number</th>
<th>Description</th>
<th>Duration</th>
<th>Type</th>
<th>Dependency</th>
<th>Number</th>
<th>Description</th>
<th>Duration</th>
<th>Type</th>
<th>Dependency</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Airbag deployment</td>
<td>30</td>
<td>Sequential</td>
<td>-</td>
<td>10</td>
<td>Shock absorbers clipped</td>
<td>59</td>
<td>Sequential</td>
<td>9</td>
</tr>
<tr>
<td>2</td>
<td>Pick and place</td>
<td>60</td>
<td>Sequential</td>
<td>1</td>
<td>11</td>
<td>Shock absorbers drip drained</td>
<td>165</td>
<td>Parallel</td>
<td>10</td>
</tr>
<tr>
<td>3</td>
<td>Bonnet popped, battery removed</td>
<td>60</td>
<td>Sequential</td>
<td>2</td>
<td>12</td>
<td>Sump drilled / crack / opened</td>
<td>10</td>
<td>Sequential</td>
<td>9</td>
</tr>
<tr>
<td>4</td>
<td>Fuel caps opened</td>
<td>20</td>
<td>Parallel</td>
<td>3</td>
<td>13</td>
<td>Sump drip drained</td>
<td>240</td>
<td>Parallel</td>
<td>12</td>
</tr>
<tr>
<td>5</td>
<td>Coolant setup and drain</td>
<td>75</td>
<td>Parallel</td>
<td>4</td>
<td>14</td>
<td>Fuel tank drilled</td>
<td>10</td>
<td>Sequential</td>
<td>9</td>
</tr>
<tr>
<td>6</td>
<td>Screen-wash setup and drain</td>
<td>75</td>
<td>Parallel</td>
<td>4</td>
<td>15</td>
<td>Fuel tank suction drain</td>
<td>395</td>
<td>Parallel</td>
<td>14</td>
</tr>
<tr>
<td>7</td>
<td>Brake fluid setup and drain</td>
<td>75</td>
<td>Parallel</td>
<td>4</td>
<td>16</td>
<td>Catalytic converter removed</td>
<td>55</td>
<td>Sequential</td>
<td>9</td>
</tr>
<tr>
<td>8</td>
<td>Tyres unlocked, wheels removed</td>
<td>65</td>
<td>Parallel</td>
<td>2</td>
<td>17</td>
<td>Placed back in yard</td>
<td>60</td>
<td>Sequential</td>
<td>11,13,15,16</td>
</tr>
<tr>
<td>9</td>
<td>Pick and placed on rig</td>
<td>70</td>
<td>Sequential</td>
<td>5,6,7,8</td>
<td>-</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

**Figure 9.5**, The use of Program Evaluation and Review Technique (PERT) on the vehicle de-pollution process to highlight the parallel and sequential processing of each activity.
9.3.2.1 Modelling the Revenue from Component Removal

Varying degrees of investment have been made with regard to parts resale facilities, from open compounds where customers remove their own components, to internet based salvage operators allowing online parts request and payment. The distinction between these types of facilities is important as different factors need to be considered when determining the revenue generated from a components resale (i.e. if mechanic labour costs need to absorbed, or postal charges are included, etc.). An additional factor to also consider is the inconstancy in component pricing methods at each facility. The vehicle salvage industry sees huge variations in the types of vehicle it processes, and as a result component pricing is often generated on an adhoc basis when a certain vehicle is made available and a customer parts request is received. The yard foreman often estimates a price based on his own knowledge as to the components market availability, typical service life, virgin price and labour required to remove the component. Hence, potential revenue is not solely a function of a facilities level of technical capability but also its operators ability to provide a realistic price estimate.

Throughout the research, it has become apparent that for the purposes of including component revenue generation within the costing of the vehicle reclamation process it is necessary to determine a generalised facility setup, and one which is “typical” of the majority of vehicle dismantlers within the UK. As no statistical data exists with regard to the level of technical capabilities between different ATFs in terms of component removal, the authors own experience as to the most common working practises were be assumed. The majority of facilities visited during the initial interview phase of the research were sites that offered basic online breaking lists (vehicles currently being processed), and component prices based on onsite collection. Removal was typically undertaken by facility employees and not the general public (dismantlers were keen to promote this method of working as it allows them to include a component removal charge within the price, and also stops amateur dismantlers potentially damaging other salvageable components). Table 9.1 has been collated from ATF sites that exhibit the work practises identified above, and describes the average prices charged for the most commonly removed components, for those typical ELVs identified within Chapter 7 due for retirement in 2006 and 2015.


<table>
<thead>
<tr>
<th>Component</th>
<th>Average Natural ELV 2006</th>
<th>Average Premature ELV 2006</th>
<th>Average Natural ELV 2015</th>
</tr>
</thead>
<tbody>
<tr>
<td>Engines</td>
<td>£132.24</td>
<td>£418.00</td>
<td>£471.67</td>
</tr>
<tr>
<td>Gearboxes</td>
<td>£112.51</td>
<td>£205.92</td>
<td>£188.83</td>
</tr>
<tr>
<td>Carburettors</td>
<td>£30.32</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Alternators</td>
<td>£24.73</td>
<td>£41.16</td>
<td>£55.67</td>
</tr>
<tr>
<td>Starter motors</td>
<td>£30.41</td>
<td>£38.45</td>
<td>£35.67</td>
</tr>
<tr>
<td>Distributors</td>
<td>£22.80</td>
<td>£37.01</td>
<td>-</td>
</tr>
<tr>
<td>Head lamps</td>
<td>£13.11</td>
<td>£25.43</td>
<td>£35.00</td>
</tr>
<tr>
<td>Quarter glass</td>
<td>£22.40</td>
<td>£25.30</td>
<td>£22.83</td>
</tr>
<tr>
<td>Brake discs</td>
<td>-</td>
<td>£6.15</td>
<td>-</td>
</tr>
<tr>
<td>Brake callipers</td>
<td>£27.65</td>
<td>£31.98</td>
<td>£38.25</td>
</tr>
<tr>
<td>Steel wheels</td>
<td>-</td>
<td>£13.50</td>
<td>£37.50</td>
</tr>
<tr>
<td>Alloy wheels</td>
<td>-</td>
<td>£43.18</td>
<td>£37.17</td>
</tr>
<tr>
<td>Radiators</td>
<td>£20.53</td>
<td>£37.23</td>
<td>£41.50</td>
</tr>
<tr>
<td>Average (£)</td>
<td>£43.67</td>
<td>£76.94</td>
<td>£96.41</td>
</tr>
</tbody>
</table>

Table 9.1, Average component prices based on demographically selected vehicles from www.carparts-uk.com as of Nov 2006

9.3.2.2 Cost Factors Affecting Component Removal

Unlike the vehicle de-pollution process that has a range of set procedures that can be carried out on all generic vehicle types, component removal and plastics recovery is very much a vehicle specific cost issue. The commonalities between vehicles is that they all share the same standard functional assemblies (e.g. starter motors, radiators, wing mirrors, etc.), which can always be assumed within each returning ELV. However, their placement and accessibility is not as consistently assured, with specific vehicles often requiring unique dismantler knowledge as to how best to remove their components (e.g. prerequisite assembly removal to gain access to the required component, specialised tooling considerations, etc.). The fact that component removal is often non-destructive and requires disassembly knowledge adds an extra level of complexity to the activity, making it distinctly different from that of material removal for end-of-life recycling. In the case of parts re-sale the functionality of the component must be retained, and requires a certain level of mechanical understanding to prevent damage, whereas in the mining of materials for recycling these considerations are not as vital. Hence, the cost implications of component removal again points to cost factors that are not easily measurable and quantifiable (i.e.
employee experience in understanding the best practises, specific vehicle accessibility attributes). Figure 9.6 highlights the main cost factors identified by ATF sites as contributing to the time taken to remove component assemblies, which include: ease of access, fastener attributes and condition, facility specific tooling, and vehicle design attributes.

9.3.2.3 Selection of Costing Technique for Component Removal Process

The diverse range of factors identified within the survey, combined the inability of respondents to identify a uniquely distinctive metric with which to relate costs again highlights the need to use a generative-analytical approach to model the costs of component removal. In this approach a generalised standard sub-assembly removal time is created from a range of interviewed vehicle dismantlers for the components considered. These estimates are then multiplied by the facility specific labour rates, and appended to the cost estimate if the activity is undertaken. Figure 9.7 highlights the average sub-assembly removal times for each component, and the range variation

![Figure 9.6, Typical factors identified by facility managers as to the main causes of component removal times](image-url)
Figure 9.7, Average sub-assembly removal times and removal estimate variation

given with the facilities interviewed. These times will ultimately be utilised to cost component re-use, and recycling target achievement within the cost model.

9.3.2.4 Concerns with Part Resale Fulfilling ELV Recycling Targets

Despite the economic and sustainable advantages of parts resale (re-use), survey data would suggest that component removal cannot make substantial headway into improving the recycling and reuse targets laid down by the ELV directive, as the majority of removed sub-assemblies are metallic and are currently counted within the assumed recycled fraction processed during post-fragmentation. Only components composed of plastics, rubbers or glass can count towards target attainment, and only the headlamps, door mirrors and tyres were listed within the top 10 of most commonly removed assemblies that fulfil this criterion (see ATF survey Appendix A1.2). Hence, this research has argued that future ELV directive target attainment must come from either manual plastics dismantling at the ATF, or automated plastics recovery post-fragmentation. The consideration for the improvement in automated plastics recovery in post-fragmentation is beyond the scope of this research. However, the costing of manual removing of plastics via dismantling for target attainment as further described in the remaining sections of this chapter.
9.4 Material Removal for Value and Target Attainment

Pre-fragmentation material recycling is currently not a widespread practice within UK ATFs, as it is widely perceived within that the economics of manual material removal is not viable based on UK labour rates. Hence, the only realistic situation in which further vehicle dismantling will be undertaken is if the 2015 target remains the same and post-fragmentation technology is unable to meet the higher recycling target (85%), or if the value received for recycled materials increases enough to make dismantling economically viable. In either of these instances the inclusion of pre-fragmentation costing is a necessity not only to determine when and if vehicle dismantling becomes economically feasible, but also to assist in supporting component selection decisions when targeting the most removable and valuable assemblies.

9.4.1 Data Requirements for Pre-fragmentation Material Recovery

The selection of previous costing techniques for component removal and vehicle de-pollution is facilitated by the fact that these activities are currently undertaken at a number of ATFs across the UK. Conversely, the destructive dismantling of vehicles for material removal is not, and as a result there is a distinct lack of data with which to determine cost related factors. Disassembly information generated for upstream design sources such as manufacturer and third party tear-down databases is therefore the only abundant source of reference. Vehicle manufacturers such as the Ford Motor Company have a designated vehicle disassembly facility (in Cologne, Germany) in which new models are systematically dismantled into their individual components. This assists with the generation of repair and maintenance manuals, and provides insight into many ‘design for X’ disciplines. Additionally, third party companies, such as AutoBench (www.autobench.com), provide a similar service to the automotive sector by undertaking vehicle teardown as part of its competitor analysis reporting. Unfortunately, the nature of this information is both highly propriety and expensive to obtain, but is really the only realistic approximation to the activities carried out during manual vehicle dismantling. The limitations of these data sources are that they only consider non-destructive vehicle dismantling times (e.g. the removal of each assembly systematically), and are not representative of the suggested material yield rates.
obtainable via destructive methods. This lack of data availability, combined with the limitation of having no comprehensive destructive dismantling times, would suggest that further data collection is needed with which to infer an appropriate costing approach, and is outlined in the section below.

9.4.2 Vehicle Teardown Study

As part of data requirements needed to select a suitable costing approach a dismantling study was conducted by the author at a local ATF to generate a range of component dismantling times for a number of natural ELVs. These vehicles were selected based on the aforementioned vehicle demographic of a natural ELV in 2006 (1993 - Astra, Escort and Fiesta), and involved the use of standard facility tools (electric screwdrivers, chisels, hammers, crowbars, etc.) to obtain the highest material yield rates possible. It should be noted that the study was based on a worst case scenario for the salvaging of ELVs. The author had no prior design knowledge of the vehicles used within the study, and the condition of fasteners had been fatigued by the longest possible service life. Hence, it is expected that the recorded removal times will be slightly higher than those achievable by an ATF.

Despite the difficult relationships between the vehicle manufacturers and end-of-life recovery sector previous discussed, one end-of-life software tool that has been made
available to assist vehicle dismantling is that of the International Dismantling Information System (IDIS). IDIS is a requirement of the ELV directive that forces manufacturer to make limited vehicle design information available to end-of-life operators. This software tool was used to identify and assist the removal of approximately 117 individual components, while additional material separation and stripping times were also catalogued. An extensive listing of the dismantling data obtained can be found within Appendix A1.4.

During the study some of the main factors affecting the material removal rates became apparent. Each vehicle had unique design attributes that made it significantly different to dismantle than the others. Figure 9.9 gives an indication of this variation by highlighting the average component removal times for each zonal location within IDIS of each model of vehicle dismantled. This highlights the lack of consistency in component removal times between different model variations, and would suggest that a costing approach is required that is capable of making this distinction. Given the variation that exists between a materials location, quantity and type between different makes and model, it is advantageous to select a costing technique capable of generating dismantling times for the specific vehicle considered.

### 9.4.3 Selection of Costing Technique for Material Removal for Recycling

Despite the lack of upstream manufacturer data with regard to vehicle dismantling, the aforementioned study provides an accurate and consistent data pool with which to consider a more diverse range of costing approaches. Based on the cost modelling requirements identified within Chapter 5, it was decided to adopt a Parametric Regression approach, based on the statistical data collated during the vehicle dismantling study. The most beneficial attributes of this cost modelling approach is its ability to generate Cost Estimate Relationships (CER) that are very straightforward to use, and produce a statistically measurable output (providing a good assessment of estimate confidence). CER can be based on any number of relevant parameters, and can potentially be linked to both upstream design and downstream recovery data sources. The IDIS database previously used to assist the vehicle dismantling study, is
Figure 9.9, Average component removal times for each zonal location and type of vehicle considered within the dismantling study

one such data source widely made available to end-of-life operators. This software system catalogues not only the potentially recoverable materials from each make and model of vehicle, but also provides basic component parameter data for each instance. Therefore, relating IDIS component parameters to the data obtained from the study can be analysed using parametric regression analysis to see if reliable CER can be determined. CER developed based on parameter data within IDIS would provide the vehicle specific costing identified during the dismantling study. Therefore, this investigation highlighted that if a component’s attributes can be statistically linked to its removal time, and those attributes can be determined for any other make or model of vehicle (i.e. catalogued within the database), then a dismantling time and labour cost can be generated without physically having to perform the work (see Figure 9.10). With over 1069 vehicle variants and 59,000 removable assemblies this costing approach is highly advantageous.

9.4.4 Implementing Parametric Regression Analysis

Once the appropriate data pool was established through the aforementioned study, an
iterative process of testing various component parameters was adopted to investigate if there was a statistical relationship between disassembly time. Given the complex nature of establishing CER via parametric regression analysis, a systematic approach was adopted, shown within Figure 9.11. The approach uses an equation development process adapted from Levine et al. (2005), and uses various statistical performance measures to refine the CER.

The starting point for these relationships requires the estimator to hypothesize as to the standard parameters affecting dismantling time (accessibility, fixturing, etc.), and the availability of these parameters within the obtainable data source (i.e. IDIS). Parameters (explanatory variables) must appear statistically independent of one another to be included within the analysis, and must contribute to improving the correlation between the predicted and actual disassembly times. The equation performance metrics (Variance Inflationary Factor (VIF), the $C_p$ statistic, coefficient of determination, T-stat, P-value, F-stat) utilised at different stages of the equation
Chapter 9

Figure 9.11, The equation development process adapted from (Levine et al. 2005) for generating the CER for direct ELV costing

development process, assisted in selecting the most appropriate parametric equation based on the available data. For a more detailed discussion of the performance metrics, search algorithms and analyse types identified within Figure. 9.11 see Levine et al. (2005).

It was decided to consider each of the IDIS zonal areas independently when developing the dismantling equations, as it became apparent during the study that there was a clear distinction between the effort required and the region of the vehicle worked upon. Table 9.2 provides details of the dismantling equations created and the statistical significance of their coefficients. For a further detailed understanding as to these statistical measures the reader is referred to the parametric estimating handbook (NASA 1999). A further critical review as to the reliability of the IDIS data source when compared to the dismantling study data can be found within Appendix A1.4.
<table>
<thead>
<tr>
<th>IDIS Zonal Area</th>
<th>Dismantling time equation</th>
<th>$R^2$</th>
<th>$R^2_{adj}$</th>
<th>SE</th>
<th>F-Stat</th>
<th>Parameters</th>
<th>Coeff.</th>
<th>SE</th>
<th>$t$ Stat</th>
<th>$P$-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dashboard</td>
<td>$Y = 26.76X_1 + 123.47\sqrt{X_2} - 132.27$</td>
<td>0.59</td>
<td>0.54</td>
<td>52.89</td>
<td>11.36</td>
<td>*No. attachments $(X_1)$</td>
<td>26.76</td>
<td>7.18</td>
<td>3.73</td>
<td>1.84E-03</td>
</tr>
<tr>
<td>(20 observations)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>*Cleaning effort $(X_2)$</td>
<td>123.47</td>
<td>32.53</td>
<td>3.80</td>
<td>1.59E-03</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Intercept</td>
<td>-132.27</td>
<td>54.04</td>
<td>-2.45</td>
<td>2.63E-02</td>
</tr>
<tr>
<td>Door and Glaze</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Limited datasets available</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(4 observations)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Seats</td>
<td>$Y = -14.12X_1^2 + 158.26X_1 + 8.81$</td>
<td>0.94</td>
<td>0.92</td>
<td>40.54</td>
<td>50.65</td>
<td>Mass $(X_1)$</td>
<td>158.26</td>
<td>33.52</td>
<td>4.72</td>
<td>3.25E-03</td>
</tr>
<tr>
<td>(9 observations)</td>
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<td></td>
<td></td>
<td></td>
<td></td>
<td>Mass*</td>
<td>-14.12</td>
<td>5.66</td>
<td>-2.49</td>
<td>4.70E-02</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Intercept</td>
<td>8.81</td>
<td>20.44</td>
<td>0.43</td>
<td>6.81E-01</td>
</tr>
<tr>
<td>Exterior</td>
<td>$Y = 118.88X_1^2 + 52.08$</td>
<td>0.78</td>
<td>0.76</td>
<td>63.76</td>
<td>38.75</td>
<td>Mass $(X_1)$</td>
<td>118.88</td>
<td>19.10</td>
<td>6.22</td>
<td>6.49E-05</td>
</tr>
<tr>
<td>(12 observations)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Mass*</td>
<td>-14.12</td>
<td>5.66</td>
<td>-2.49</td>
<td>4.70E-02</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Intercept</td>
<td>8.81</td>
<td>20.44</td>
<td>0.43</td>
<td>6.81E-01</td>
</tr>
<tr>
<td>Interior</td>
<td>$Y = 459.94\sqrt{X_1} - 129.73$</td>
<td>0.79</td>
<td>0.77</td>
<td>39.63</td>
<td>42.16</td>
<td>Mass $(X_1)$</td>
<td>459.94</td>
<td>70.83</td>
<td>6.49</td>
<td>4.47E-05</td>
</tr>
<tr>
<td>(12 observations)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Mass*</td>
<td>-129.73</td>
<td>37.46</td>
<td>-3.46</td>
<td>5.30E-03</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Intercept</td>
<td>65.26</td>
<td>25.66</td>
<td>2.53</td>
<td>6.74E-02</td>
</tr>
<tr>
<td>Engine compartment</td>
<td>$Y = 879.62\sqrt{X_1} - 448.07$</td>
<td>0.65</td>
<td>0.57</td>
<td>92.77</td>
<td>9.10</td>
<td>Mass $(X_1)$</td>
<td>879.62</td>
<td>291.62</td>
<td>3.02</td>
<td>2.95E-02</td>
</tr>
<tr>
<td>(7 observations)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Mass*</td>
<td>-448.07</td>
<td>206.77</td>
<td>-2.17</td>
<td>8.25E-02</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Intercept</td>
<td>20.74</td>
<td>108.01</td>
<td>-0.22</td>
<td>1.13E-01</td>
</tr>
<tr>
<td>Load space</td>
<td>$Y = -6.37X_1^2 + 91.72X_1 - 239.94\sqrt{X_2} + 53.35$</td>
<td>0.85</td>
<td>0.69</td>
<td>24.83</td>
<td>5.49</td>
<td>*No. attachments $(X_1)$</td>
<td>91.72</td>
<td>27.40</td>
<td>3.35</td>
<td>4.41E-02</td>
</tr>
<tr>
<td>(8 observations)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td><em>No. attachments</em> $(X_2)$</td>
<td>-6.37</td>
<td>1.95</td>
<td>-3.28</td>
<td>4.72E-02</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Mass $(X_2)$</td>
<td>-239.94</td>
<td>108.01</td>
<td>-2.22</td>
<td>1.13E-01</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Intercept</td>
<td>53.35</td>
<td>40.66</td>
<td>1.31</td>
<td>2.81E-01</td>
</tr>
</tbody>
</table>

* Cleaning effort was a quantitative measure developed during the study that categorises the level of additional post-removal cleaning required

* No. attachments refers to the number of mechanically removable fastenings (e.g. clips, screws, bolts)

**Table 9.2**, Statistical significance testing of the generated dismantling equations
Using the parametric equations developed, and the additional parameter data located within IDIS, it is possible to predict the expected direct labour costs incurred to disassemble a range of typical passenger vehicles. A unique dismantling time can be generated for an individual component based on the explanatory variables identified within the regression analysis. As stated, the author is of the opinion that the removal of material during pre-fragmentation processing is one of the most environmental/sustainable approaches to ELV directive target attainment. Hence, in assessing the viability of this approach the dismantling metrics of Mass Removal Rate (MRR - Equation 9.1) and the Value Removal Rate (VRR - Equation 9.2), developed by Coutler et al. (1996) will be used to determine the economic feasibility.

\[
\text{Material Removal Rate (kg/sec)} = \frac{\text{Material(kg)}}{\text{Time(sec)}} \\
\text{Value Removal Rate (£/s)} = \frac{(\text{Material(kg)}) \times (\text{Value(£/kg)})}{\text{Time(sec)}}
\]

The use of these metrics to select plastic components should be used based on the goal of the dismantler. If target attainment is required the Material Removal Rate should be used, as this identifies the heaviest and easiest components to remove first, and gives a better mass-versus-effort return. A recent governmental report estimated the deficit to the 2015 recycling target to be approximately 5.18% of a vehicle's weight (Weatherhead and Hulse 2005), hence component removal can be optimised to make up this shortfall. Alternatively, if a dismantler is interested in knowing if there are any components on a vehicle that can return a profit (when compared to a workers labour rate (£/s)), then the Value Removal Rate should be used, as this considers the value of the component removed as well as its weight.

9.4.6 Typical Material Yield Rates via Dismantling

A further consideration as to the feasibility of manual material removal is that of
achievable material yield rates. The aforementioned value removal rate utilises material value estimates based on minimum recycled quantities. Hence, to realistically consider manual material removal a consideration must be made as to the vehicle throughput required to achieve minimum re-processor specifications. During the study 22 different material types were removed, with the nine most abundant materials (> 0.25 kg) producing 28 kilos per vehicle. These quantities can then be factored up based on the typical number of vehicles processed at an ATF per day (see ATF survey within Appendix A1.2), and are listed within Table 9.3.

The data catalogued within Table 9.3 is representative of those materials found within natural ELVs, but it is envisaged that the material types and quantities will be reduced over time as more “design for recycling” issues are considered by vehicle manufacturers. The current quantities obtained via the study would suggest that only a few key material types (PP, PU & ABS) would produce enough material to satisfy the typical minimum quantities required by plastic re-processors, and justify their removal.

<table>
<thead>
<tr>
<th>Material</th>
<th>Abbreviation</th>
<th>Mass per vehicle (kg)</th>
<th>Total possible per month (tonnes)</th>
<th>Total possible per year (tonnes)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Polypropylene</td>
<td>PP</td>
<td>11.45</td>
<td>5.8</td>
<td>69.1</td>
</tr>
<tr>
<td>Polyurethane</td>
<td>PUR</td>
<td>8.12</td>
<td>4.1</td>
<td>49.0</td>
</tr>
<tr>
<td>Acrylonitrile-butadiene-styrene</td>
<td>ABS</td>
<td>3.78</td>
<td>1.9</td>
<td>22.8</td>
</tr>
<tr>
<td>Polypropylene-Talcum 20%</td>
<td>PP-T20</td>
<td>1.57</td>
<td>0.8</td>
<td>9.5</td>
</tr>
<tr>
<td>Polyamide</td>
<td>PA</td>
<td>0.83</td>
<td>0.4</td>
<td>5.0</td>
</tr>
<tr>
<td>Polypropylene-Ethylene-PropyleneDiene terpolymer</td>
<td>PP-EPDM</td>
<td>0.82</td>
<td>0.4</td>
<td>4.9</td>
</tr>
<tr>
<td>Poly(ethylene terephthalate)</td>
<td>PET</td>
<td>0.65</td>
<td>0.3</td>
<td>3.9</td>
</tr>
<tr>
<td>Poly(vinyl chloride)</td>
<td>PVC</td>
<td>0.50</td>
<td>0.3</td>
<td>3.0</td>
</tr>
<tr>
<td>Polycarbonate Acrylonitrile-butadiene-styrene Blend</td>
<td>PC+ABS</td>
<td>0.25</td>
<td>0.1</td>
<td>1.5</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>Total</strong></td>
<td><strong>27.97</strong></td>
<td><strong>14.1</strong></td>
<td><strong>168.7</strong></td>
</tr>
</tbody>
</table>

Table 9.3, The main types and quantities of material removed during the dismantling study
9.5 Summary

This chapter outlines the approach adopted by the research to identify and model the direct costs involved in pre-fragmentation ELV recovery. This related to the activities of de-pollution, component removal and material recycling. In the case of de-pollution and component removal the lack of easily determinable end-of-life metrics would suggest that a basic form of the Generative-Analytical costing approach is used. The author has then argued that material removal for target attainment should be considered, and has utilised Regression Analysis and material selection metrics to develop a model for assessing the viability of pre-fragmentation material recycling.


Chapter 10

Direct Post-Fragmentation Costing

10.1 Introduction

This chapter describes the development of a post-fragmentation separation model, capable of modelling the value-added processing that a piece of automated separation equipment can have on a fragmented waste stream. The model takes the input composition of the vehicle and determines the most likely route of each material through separation processes outlined in Chapter 4, and based on its material attributes and component interactions generates a material value as a function of its post-separation contamination. The research has identified a range of challenges in modelling the costs of a post-fragmentation process. These included the modelling the inefficiencies of the technology, the affects of material entanglement on separation, determination of typical material sizing, and an appreciation for compositional value. A number of mathematical/statistical techniques, including process partition curves, Monte Carlo analysis, material interaction matrices and material value curves have been used to aid the modelling of the direct costs of post-fragmentation processing. These challenges together with the techniques used to model these costs are described in the later sections of this chapter.

10.2 Modelling of Post-fragmentation Separation Processes

The current investment in post-fragmentation technologies by many major UK end-of-life material processors is a testament to the ever increasing demand for closed-loop resources. Currently, only high value metals are being extracted from the waste, before the remaining residue is either sold as aggregate or placed into landfill. It is envisaged that one day this residue will also be recovered, either due to legislative targets, increased landfill taxation or economic value. The review and survey conducted by this research would suggest that large scale automated separation will be the preferred waste management route for many different product groups in the
future. It is therefore necessary to develop ways of ascertaining the value-added these technologies achieve. The following sub-sections provide an overview as to a method of modelling the post-fragmentation process and some of the considerations that need to be made, before a more detailed discussion of the various elements of the model is presented.

10.2.1 Linking Waste Stream Material Parameters to Separation Technologies

Chapter 4 investigated the current automated separation technologies used within many of the post-fragmentation facilities in the UK. For each technology discussed within the literature review a parameter was identified, used by the process to distinguish a target material from the rest of the waste stream (e.g. density for media separation tanks, shredded particle size for screening, etc.). These parameters target either the physical characteristics (morphology) of a shredded waste stream (shape, size, etc.), or its material characteristics (density, conductivity, etc.). Table 10.1 provides a summary of these targeted parameters for a range of the most common separation technologies. The parameters identified within this table will be used as a means of benchmarking the separation capabilities of a particular technology, and is described further within the following section.

10.2.2 Using Targeted Parameters for Material Separation

Figure 10.1 describes the overall post-fragmentation modelling approach adopted by this research, and how the targeted parameters are used to benchmark the inefficiencies of the various separation technologies, so that its value-added processing can be ascertained. The model works by considering each material from the input composition individually in turn. A fragmented material has both physical and material characteristics, some of which are used by the various separation technologies to make distinctions between other materials within the waste stream. By utilising these targeted parameters, combined with an appreciation for the separation inefficiencies due to the equipment and imperfect liberation, it is possible to predict in which processes a particular material is most likely to be separated.
<table>
<thead>
<tr>
<th>Process Name</th>
<th>Description</th>
<th>Targeted parameter(s)</th>
<th>Physical or Material</th>
</tr>
</thead>
<tbody>
<tr>
<td>Screens</td>
<td>Used within trommels and vibrating tables. Screen aperture size determines separation results.</td>
<td>Particle size</td>
<td>Physical</td>
</tr>
<tr>
<td>Over-band magnets</td>
<td>Used to segregate the ferrous fraction from the waste stream. Power and orientation vary depending on the position in the process.</td>
<td>Magnetic susceptibility</td>
<td>Material</td>
</tr>
<tr>
<td>Eddy-current devices</td>
<td>A rotating magnet induces eddy currents within a conductive material and propels materials further.</td>
<td>Separation factor (Conductivity + Density)</td>
<td>Material</td>
</tr>
<tr>
<td>Cyclone separators</td>
<td>A rotating air or liquid column creates a vortex and displaces materials according to their weight.</td>
<td>Mass (Density × Volume)</td>
<td>Physical / Material</td>
</tr>
<tr>
<td>Density media separators</td>
<td>The liquid separation media's density floats and sinks waste stream materials according to their densities.</td>
<td>Density</td>
<td>Material</td>
</tr>
</tbody>
</table>

Table 10.1, Summary of targeted material parameters for a range of common separation technologies

The initial stage is to gain an understanding of the percentage composition of the product being processed. This not only allows an understanding of the various quantities of each material to be considered, but also provides an indication as to the different contamination ratios, which are important in determining the input waste streams value. The benefits of using compositional data as an input into the post-fragmentation model is that this data can be readily obtained from upstream design sources, but can be equally sourced using onsite shredder sampling. Certain European density media sites already undertake this type of sampling, for example RNS in the Netherlands provides a post-fragmentation service to end-of-life waste producers and
Modelling post-fragmentation process inefficiency and value-added waste stream processing

**INPUT VALUE**

- Vehicle input composition (%)
  - Materials considered one at a time

**PROCESS INEFFICIENCY**

- Materials attribute database (Generic material properties)
- Physical characteristics
- Material characteristics

**10.2.3**

**10.2.4**

- Size distribution Monte-Carlo analysis (Accounts for the randomness of the shredding process)

**10.2.5**

- Materials interaction matrix (Describes typical material interactions for a given waste stream)

**OUTPUT VALUE**

- Process partition curves (Equipment specific process inefficiency curves)

**10.2.6**

- Output waste stream A (%)

**10.2.7**

- Output waste stream B (%)

Value determined by the purity of the input waste stream.

A materials attribute database provides generic material property data. Material size distributions determined by uncertainty analysis, un-liberated material interactions by an interaction matrix.

Value determined by the purity of each output waste stream.

**Figure 10.1,** Post-fragmentation separation modelling approach (the sub-sections number in this chapter of each stage is also listed)
riously tests the composition of the input waste entering their facility so the provider can be suitably recompensed. Although this is currently not the case within the UK it is envisaged that such compositional sampling will become common place in the future.

10.2.3 Physical and Material Characteristics within an Attribute Database

All data pertaining to the physical or material characteristics are stored within a materials attribute database (see Figure 10.1), which is continually interrogated for each new material. Within this database there are a range of typical materials found within end-of-life products (plastics, metals, ceramics) and approximate values for each of the targeted parameters identified within Table 10.1 (density, size, conductivity, etc.). The different material types have every parameter listed, as this would allow for the possibility of each material to pass through every separation process, regardless of the process having any substantial affect or not.

10.2.4 Determining Physical Waste Stream Characteristics for Different Material Types

Processes that base their separation on physical characteristics (e.g. screening) require each material to have some consideration as to the range of typical particle sizes created during shredding. This means that aside from not only knowing the material characteristics of a particular material within a waste stream, the model also requires an understanding of a materials typical geometry (i.e. steel particles fragment into size ranges of between 0-100mm, rubber particles range between 0-50mm, etc.). Material characteristics can be readily sourced from a materials database, whereas the fragmentation effects of the shredding process are not consistent, producing particles of all sizes depending on the random effects of the liberation process. One way of overcoming this is to try and link the material characteristics of a waste stream constituent (e.g. density, izod impact strength) to a resultant particle size, but this requires a detailed understanding of the complex science of impact phenomena and is considered beyond the scope of this thesis. The other approach is to try and build particle size uncertainty into the model, capturing the random sizing effects of the shredding process based on the analysis of the output. Using uncertainty modelling
approaches (Monte-Carlo simulation), and size distribution data taken from existing fragmentation studies for a range of different materials (Harder 2002), it is possible to account for the randomness of the shredding process. The generation of material size distribution profiles (see Figure 10.1) is discussed in more detail in section 10.4.

### 10.2.5 Modelling Waste Stream Interactions on Separation Technology Efficiency

An important area which must be considered, before looking at the specific inefficiencies of the separation technology, is an appreciation for the ability of the shredding process to isolate each material from the rest of its waste stream. One of the biggest separation influences within any post-fragmentation model is the extent to which the hammer-mill can liberated each material, and the level of material interaction that occurs (entanglement, electro-static adhesion, moisture induce bonding, etc.). In an ideal situation each material would perfectly segregate from the rest, but industrial data would suggest that a great deal of interactions occur between certain material types. This has been accounted for within the model via the use of a materials interaction matrix (see Figure 10.1), which describes typical material interactions for a particular type of waste stream (automotive, WEEE, etc.). Section 10.5 within the chapter describes the need for the inclusion of this additional modelling, and demonstrates how the materials interaction matrix works.

### 10.2.6 Recognised Methods of Modelling Technology Inefficiency

When all the size distribution and materials interaction data has been generated it is then a case of describing the separation inefficiencies of the technology relative to the targeted process parameter identified. This is done via the use of process partition curves (see Figure 10.1), typically found within the minerals refinement industry, that describe the percentage of a material that end up within each of the output waste streams. Section 10.3 of this chapter highlights the inclusion of these curves.

### 10.2.7 Determining Waste Stream Value

Finally, once the partition curve has distributed each material between the two output waste streams a new waste stream value can be calculated based on the new waste
stream composition. The main problems in identifying this compositional value are the lack of industry material specifications regarding the contamination criteria of recovered materials. Only well established material groups, such as Steels and Aluminiums, have market prices and material specifications readily available (www.lme.com) that are linked to their contamination. Hence, an approach is required for those materials that don't have material specifications, but can return value if purified enough. Section 10.6 describes the approach adopted in assessing the value of the separated waste streams and the development of material value curves to estimate the possible end-of-life revenue.

10.3 Technology Inefficiencies: Process Partition Curves

Tromp/Partition curves have long been used within the minerals refinement industry to describe the effectiveness of various separation processes, but have never been applied to end-of-life waste stream reclamation. The tromp curve is made up of three main parameters; the cut-point ($x^*$), the probable error of separation ($E_p$), and the cut-point off-set (present in some curves but not others). The tromp curve models the partition co-efficient which is the percentage of the feed material that ends up within the incorrect fraction at the specific material parameter value identified (see Figure 10.2). Hence the cut-point ($x^*$) represents value at which half the feed reports to the

![Diagram of typical process partition curves using an inverse exponential function](image)

**Figure 10.2**, Typical process partition curves using an inverse exponential function
wrong output fraction. The $E_p$ is described as the density at which 75% of the waste will go to the wrong place ($A$), minus the density at which 25% will go to the wrong place ($B$), divided by 2 (See Equation 10.1).

$$E_p = (A - B) / 2$$

Equation 10.1, Equation for calculating the probable error of separation

Using these three parameters and standard measurement procedures, it is possible to describe the separation efficiency for a particular process setup. The majority of separation processes exhibit a tromp curve similar to that described by an inverse exponential function (also see Equation 10.2). These curves can be generated to describe process inefficiency relative to a selected separation parameter, as identified in Chapter 4.

$$y = \frac{1}{1 + \exp(1.099 \times (x_{s0} - x) / E_p)}$$

Equation 10.2, Inverse exponential function taken from the work of Napier-Munn (1991) that describes the separation effectiveness of a density media device

These curves are designed to model the inefficiencies of a particular process and vary depending on the machines design parameters (e.g. a smaller air-gap between the feed and magnet within an magnetic separator may pick up more ferrous metal, or a stronger vortex within a cyclone separator may lift heavier materials). Therefore, depending on how a process is step-up and the machine parameters for that process, the shape of these curves can vary quite substantially from facility to facility. By specifying three variables within the model ($x_{s0}$, $E_p$ and offset) the variations between equipment and operating parameters can be accounted for; it is simply a case of benchmarking the equipment at the facility in question. These process partition curves can then be used in the direct costing of ELV recovery to model process inefficiencies.
10.4 Determining Particle Size

The majority of the separation technologies only require an understanding of the material characteristics of the waste stream to model the separation effects, but other technologies (such as screening and air-separation) require an appreciation of the physical characteristics as well (average particle sizes). The mathematical modelling of particle impacts indicative of those found within a shredder-mill is an extensive and complex research field, and is considered beyond the scope of this research. There is nevertheless a requirement within the model to describe the effects of the fragmentation process, and the basic particle sizes produced by it. This has been incorporated within the modelling approach by considering the fragmentation process as a number of random liberation events. Data exists as to the typical size ranges of various material types once they have been through the shredding process (Harder 2002). For example, foams tend to produce particle size between 4mm-100mm wide, but have a typical average value of a round 10mm. These kinds of ranges lend themselves very well to the Beta-PERT probability distribution curves described in Chapter 8, initially used to facilitate indirect cost uncertainty modelling. These distribution curves (based on published data) can then be used to describe the likelihood of different particle sizes occurring, and can also be catalogued as additional information within the material attribute database. These typical size ranges generate a size distribution profile for each material. It is then simply a case of undertaking a random sampling analysis on each material distribution which simulates the random liberation affects of the shredding process. In summary, the published data on typical shredded particle sizes, described using BetaPERT distributions and sampled using Monte Carlo analysis, has been used within this research to determine average particle sizes.

10.5 Entanglement In-efficiencies: Materials Interaction Matrix

If these process partition curves are used in isolation to describe the separation effectiveness of typical end-of-life processes a huge assumption is needed to make the model valid. This assumption is that every waste stream constituent is perfectly isolated from the rest of the feed during the fragmentation process, with zero inter-particular interactions between materials. An approximate analogy would be to
consider a shredded waste stream as thousands of perfectly spherical balls, each an individual material and unable to interact and entangle with any other part of the waste. Each product would have been perfectly liberated into its individual materials, with geometry that would not tangle or bind with any other waste stream constituent. Each of these individual balls can then have their targeted material parameter tested by the technologies process partition curve to determine their predicted separation. In reality we know this is not the case. Unlike the minerals refining process the end-of-life shredding process produces materials that are rarely of singular composition, and are often attached to other materials either due to incomplete liberation or post-fragmentation entanglement. Therefore, determining a value for the targeted material parameter to be used within a processes partition curve is not as straight-forward, due to the need to consider the material properties of a combination of materials as opposed to one in isolation. The following example describes the limitations of developing a post-fragmentation separation model based solely on process partition curves, and the inadequacies of not including the inefficiencies due to inter-particular interactions within the model.

10.5.1 The Inadequacies of Using Process Partition Curves in Isolation

The inadequacies of generating a post-fragmentation costing approach using only process partition curves are no more apparent when considering the separation of a non-magnetic shredded fraction using a water elutriation tank (the input composition is given within Table 10.2). Within this process materials are separated using water as the separation media, dense materials will sink and lighter materials float. If a separation model only utilises process partition curves to model the separation, the situation within Figure 10.3 is created, in which any material with a density suitably far from that of the separation media’s density (1000 kgm$^{-3}$) generates a partition coefficient approximately equal to 100%. This means that each of these waste materials will sink, and be perfectly separated from other waste stream constituents (such as glass, rubbers and plastics). In practise this is not the case. Table 10.2 shows a typical waste composition before the water elutriation stage, the predicted separation
Figure 10.3, The effect of developing a model which only considers the inefficiencies of the processes

<table>
<thead>
<tr>
<th>Material name</th>
<th>Input composition (%)</th>
<th>Predicted recovery (%) (using only process efficiency curves)</th>
<th>Actual recovery (%) (taken from the USBM study)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Copper (Cu)</td>
<td>6</td>
<td>100</td>
<td>49</td>
</tr>
<tr>
<td>Zinc (Zn)</td>
<td>33</td>
<td>100</td>
<td>97</td>
</tr>
<tr>
<td>Aluminium (Al)</td>
<td>8</td>
<td>100</td>
<td>52</td>
</tr>
<tr>
<td>Lead (Pb)</td>
<td>3</td>
<td>100</td>
<td>100</td>
</tr>
<tr>
<td>Iron (Fe)</td>
<td>32</td>
<td>100</td>
<td>96</td>
</tr>
<tr>
<td>Other (Plastics/Glass Rubber, etc.)</td>
<td>18</td>
<td>n/a</td>
<td>n/a</td>
</tr>
</tbody>
</table>

- Predicted and Actual recovery % refers to the % of the input composition that ends up in the correct fraction

Table 10.2, Predicted data and actual data (Froisland et al. 1975) showing the affect of a water elutriation process on the main metals targeted during post-fragmentation ELV waste separation ($x_w = 1000 \text{ kgm}^{-3}$ and $E_r = 100 \text{ kgm}^{-3}$)
using only process partition curves, and real data taken from a study conducted by U.S. Bureau of Mines (Froisland et al. 1975). As can be seen from Table 10.2, the process partition curves predict that perfect separation occurs. Although some materials (Zinc, Lead and Iron) correlate well with the predicted recovery values, materials such as copper and aluminium deviate wildly from those predicted, and exhibit no correlation. Possible reasons for this are the typical components these materials are found in. The majority of copper is usually used within electrical wiring. Although the shredding processes is capable of comminuting products to small fractions, components such as wiring are unable to be completely separated. As a result the copper is sheathed within a Polyvinyl chloride (PVC) jacket that has a different density to that of copper (PVC 1420 kg\textpercm\(^3\) and Copper 8960 kg\textpercm\(^3\)). The material that is processed by the water elutriation tank has the material properties of both copper and PVC, the apportionment of which depends on the contamination quantities. Electrical wiring is just one example, but the interaction of all materials with each others within the waste stream has the potential to seriously affect the predicted recovery values.

The predicted recovery values for other separation technologies, based on different targeted material parameters also exhibit a similar lack of correlation to real world values. This would suggest that the effects of inter-particular interactions and cross-material contamination greatly affect the efficiency of post-fragmentation separation technologies, and cannot be ignored when modelling the post-fragmentation process. Therefore, this research has generated a novel method of describing material interactions as will be further described in the following sections of this chapter.

10.5.2 Modelling Material Interactions

To account for the inefficiencies in separation due to material interactions the research has developed a method that is capable of describing the compositional contamination between various material types. Given that materials interact differently depending on the composition of the waste stream they are processed within (e.g. shredded electronics will interact differently than a shredded automobile) this contamination description must be changeable depending on the products that the shredder is processing.
The most logical place to catalogue these material interactions is at the shredding stage, before any separation technology has altered the waste stream. At this point it is possible to sample the shredder output to determine which materials have been un-liberated and cross-contaminated, and in what percentages. This would be a rather laborious task, but once achieved it would provide an accurate snapshot of the typical waste stream entanglements for a range of waste stream types (electronics, automotive, industrial scrap, etc.). The following sub-sections describe how these entanglement descriptions can be incorporated within the post-fragmentation model. To begin with this will be described using a basic two material interaction, before discussing the need to further increase this to consider multi-component interactions.

10.5.2.1 Modelling the Interaction of Two Materials

When sampling the shredder output it is possible to determine a percentage fraction of a particular material that is un-liberated during the shredding process, or that easily binds with other materials when in close contact. The un-liberated material must be given a new material property based on the types and quantities of material it is most likely to interact with. Consider the interaction of polypropylene (PP) and copper within a water elutriation tank, as illustrated Figure 10.4. In the example PP can be considered the target material, and accounts for 20% of the input streams mass and typically has a 5% un-liberated fraction (which is 1% of the total input mass), while copper accounts for 80% of the input streams mass and typically has a 10% un-liberated fraction (which is 8% of the total input mass). If PP has zero contamination with copper the material will have the property of PP (937 $kgm^{-3}$), conversely if copper accounts for 100% of the mix the material will have the property of copper (8960 $kgm^{-3}$). If it is assumed a linear relationship between the two (highlighting this as the main assumption of the model) it is possible to use the percentage quantities of the input feed into the shredder (20% and 80% respectively) as a weighting factor. This will then generate a new material property (based on the mixture of PP and Copper) for the percentage of un-liberated PP identified. Figure 10.5 demonstrates the compositional quantities, and Equation 10.3 determines the resultant density. The
combination of 8% Cu (10% un-liberated from the 80% input mass) is processed with 1% PP (5% un-liberated from 20% input mass).

\[
\delta_{\text{Combined}} = \frac{(\text{Density}_1 \times \%\text{of Input}_1) + (\text{Density}_2 \times \%\text{of Input}_2)}{\%\text{of Input}_1 + \%\text{of Input}_2}
\]

\[
\delta_{\text{Combined}} = \frac{(937 \times 1\%) + (8960 \times 8\%)}{9\%}
\]

\[
\delta_{\text{Combined}} = 8068 \text{ kgm}^{-3}
\]

**Equation 10.3**, A typical material characteristic weighting calculation

![Diagram showing partition coefficient (%) vs. density (kgm^-3) with two curves for Copper and Polypropylene. The density for Copper is 937 kgm^-3, and for Polypropylene is 8960 kgm^-3.]

**Figure 10.4**, Highlighting the assumed linear relationship between the two components interacting. The process tromp curve would process 72% of copper at a density of 8960 kgm^-3, 19% of PP at a density of 937 kgm^-3, and 9% combination of copper and PP at 8068 kgm^-3
10.5.2.2 Modelling Multiple Material Interactions

The previous section demonstrated a very simplistic example of a waste stream composed of two materials. This approach can be further extended to consider the interactions between multi material waste streams (i.e. typical shredder output). In this instance an additional parameter is required to describe the attribution of a material to each and every other within the waste stream. Within the previous example as there were only two materials within the waste stream, their un-liberated fractions could only interact with one another. Hence, generating the weighting factors was straightforward, but within a multi-material system each material can potentially interact with any other, in varying quantities or simply not at all. Therefore, another level of apportionment is needed. To model the interaction between multiple materials not only does the % un-liberated quantity have to be specified, but of that percentage how much interacts with each material).

The most effective way of describing this multiple materials interaction within the post-fragmentation model is to use a matrix, as depicted in Figure 10.6. By locating the materials within the waste stream, along the top and down the side a materials
interaction matrix can be developed that describes the percentage entanglement of an un-liberated material to the rest of the waste stream. The vertical columns identify the targeted material considered; entries within this column highlight the materials that typically interact with it. In this approach the diagonal within the matrix is always set to zero, as materials can not interact with themselves. Every column must equate to 100% to fully attribute the un-liberated fraction, and every entry must have a transposed entry about the diagonal (e.g. if Copper interact with PP, PP must interact with Copper). Figure 10.6 provides an example matrix of a waste stream composed of six different materials. Within this example, the un-liberated percentage of Copper interacts with Iron and PVC, in the quantities 40% and 60% respectively. Conversely, 100% of the un-liberated Iron interacts with Copper, and 50% of the un-liberated PVC mixes with Copper, and finally, the other 50% of the un-liberated PVC interacts with all the un-liberated rubber. Apportioned weighting factors can now be generated for each combination of interactions if the input stream composition is known.

Using the matrix in Figure 10.6 and assuming that Iron, PVC and Copper account for 20% each of the shredder output, and each one has a 10% un-liberated fraction, then Copper’s interactions can be calculated as follows:

<table>
<thead>
<tr>
<th></th>
<th>Copper</th>
<th>Iron</th>
<th>Rubber</th>
<th>Glass</th>
<th>PVC</th>
<th>PU</th>
</tr>
</thead>
<tbody>
<tr>
<td>Copper</td>
<td>100%</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Iron</td>
<td>40%</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Rubber</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Glass</td>
<td></td>
<td></td>
<td></td>
<td>50%</td>
<td></td>
<td></td>
</tr>
<tr>
<td>PVC</td>
<td>60%</td>
<td>100%</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>PU</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>100%</td>
</tr>
<tr>
<td>Total</td>
<td>100%</td>
<td>100%</td>
<td>100%</td>
<td>100%</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Figure 10.6, An example materials interaction matrix
Chapter 10

The process tromp curve would therefore process:

18% of Copper calculated at a density of 8960 kgm\(^{-3}\).
18% of Iron calculated at a density of 7870 kgm\(^{-3}\).
18% of PVC calculated at a density of 1420 kgm\(^{-3}\).
2.8% Copper / Iron mix at a density of 8181 kgm\(^{-3}\).
2.2% Copper / PVC mix at a density of 5533 kgm\(^{-3}\).

Figure 10.7 provides a visual representation of coppers apportionment and the percentage of each un-liberated fraction which will pass through the process partition curve at the newly calculated densities.

Figure 10.7, Coppers interactions using the interaction matrix defined within Figure 10.6 and assuming that PVC, Fe and Cu make up 60% of the waste stream (20% each), and have a 10% un-liberated fraction.
10.5.3 Limitations of the Material Interaction Matrix Modelling Approach

The materials interaction matrix has been developed by this research in response to the realisation that process partition curves are not capable of modelling the separation capabilities of UK separation facilities in isolation. The inclusion of the materials interaction approach therefore provides an additional means by which process ineffectiveness can be further described. Within the above examples this is limited to a 2D matrix, and provides only a basic demonstration as to the description of multiple two material entanglements. To incorporate additional complexity into this modelling approach, beyond that outlined above, cannot be justified due to the lack of data available in existing shredding facilities. It is therefore considered beyond the scope of this thesis to account for complex multi-material interactions, but the author recognises the need to expand this approach in the future when the acquisition of more detailed processing data becomes a more widely adopted practice.

10.6 Determining Waste Stream Value via Material Value Curves

The composition of a fragmented waste stream varies depending on the type of product being shredded. Shredding facilities within the UK process a number of white goods and industrial waste streams as well as end-of-life vehicles; this results in a large variety of different material types and compositional grades. Due to the lack of established specifications for each liberated material, the secondary markets for these materials are limited. Materials are often collectively grouped based on the requirements of the material re-processor, e.g. all grades of aluminium are generically classified as one, all grades of steel classified as another. Within these material groupings market value is clearly a function of a waste's percentage contamination (grade). Therefore, to work out the economic value of a waste stream, a clear understanding is required as to the purification capabilities of the post-fragmentation separation processes, and their effects on a material grade.

Currently there is not any formalised specifications available for the majority of the shredded materials produced during post-fragmentation processing. The research has therefore developed a method to estimate the level of achievable value based on its final purity. This method is based on an exponentially decaying value curve, as shown
within Figure 10.8. This type of curve has been adopted due to its close correlation with the typical material recycling scenario found within industry. It is often the case that large purification increases can only raise the market value so far, it is the reduction of the contaminants to zero over the last few percent that significant boast its market value. In terms of available market prices that can be used to generate these material value curves certain assumption need to be made. These are shown in Figure 10.8 and include:-

- At zero percent contamination the material market value for virgin material can be adopted ($V_{\text{Max}}$).
- Inert wastes streams that cannot be successfully purified to the levels required for recycling can be sold for a positive value as industrial aggregate ($V_{\text{Min}}$).
- Certain waste streams will require minimum contamination or they will have to be disposed of via a negative landfill charge (see dashed line in Figure 10.8).
- The rate of end-of-life value improvement can be controlled via the curve decay rate ($k$).

![Market value vs % contamination](image)

**Figure 10.8,** The generated exponentially decaying value curve used to determine end-of-life market value for a range of materials.
A suitable equation that incorporates these three control parameters ($V_{\text{Max}}$, $V_{\text{Min}}$ and $k$) that generates the £/tonne value of a material with $x$ contamination, is given in Equation 10.6 and is plotted in Figure 10.8. This equation was created based on adoption of the exponential decay function and requirement to incorporate the market prices available. Initially, the exponential decay curve with an input parameter $x$ and a decay constant of $k$, produces a graph that has a range between 0-1.

$$y = e^{-kx} \quad \text{Equation 10.4}$$

Requiring zero percentage contamination to cross the axis at the virgin material market price ($V_{\text{Max}}$) requires the exponential decay function to be multiplied by this value.

$$y = V_{\text{Max}} e^{-kx} \quad \text{Equation 10.5}$$

If the fragmented waste streams are not refined enough to a suitable level of purity for traditional recycling many will be utilised as industrial aggregate. Hence, within the above value curve equation an increase material contamination should converge towards this value. The final version of the equation can there be written as seen in Equation 10.6, which incorporates $V_{\text{Min}}$.

$$\text{Value(£/tonne)} = \left(V_{\text{Max}} e^{-kx} + V_{\text{Min}} \right) - \left(V_{\text{Min}} e^{-kx} \right) \quad \text{Equation 10.6}, \text{Developed value curve equation that incorporates the industrially obtainable parameters of } V_{\text{Max}}, V_{\text{Min}} \text{ and } k$$

Once the value curves have been developed for a range of end-of-life materials they can then be applied as a conversion tool. Translating the predict output grades (%) produced by the post-fragmentation model into a realistic assessment of recycled market value.
10.7 The Calibration of the Post-fragmentation Model

This chapter has so far described various challenges involved in modelling the direct costs of post-fragmentation activities. These challenges together with the associated modelling techniques are summarised in Figure 10.9. The following section discusses the calibration of this post-fragmentation cost model via the use of industrial data, and considers its ability to be adjusted to real world data.

10.7.1 Facility Layout Adopted for Calibration of Post-fragmentation Model

The archaic nature of the ELV recovery sector is exemplified by the lack of an abundant amount of end-of-life processing data. The detailed analysis and data collection that is undertaken in many vehicle suppliers is currently not reciprocated by end-of-life operators. Only recently has the first comprehensive ELV shredder trials been held within the UK, to determine which materials are recovered and in what quantities (Weatherhead and Hulse 2005). This lack of industrial data presents a particularly difficult problem when trying to validate the post-fragmentation model generated by this research. It was therefore decided to calibrate and test the approach using the shredder trial data intended to benchmark the UK’s current recovery

![Figure 10.9, The post-fragmentation model and associated techniques](image)
Chapter 10

capabilities. This study systematically de-polluted and shredded over 400 ELVs, and provided a reasonable indication as to the segregation of the main material groups, from the shredding site through to the density media separator. The facilities used to undertake this study were the Sims shredding site in Newport, and their density media plant in Long Marston. For the purposes of applying the post-fragmentation modelling approach the layout for the Newport shredding site was adopted, as depicted in Figure 10.10.

10.7.2 The ELV vehicle Composition Utilised as the Input Feed

The shredder trial data, which will be used to calibrate the post-fragmentation model, was original collated using natural ELVs that had a registration date around 1990. Ideally, the input composition into the model should be from a vehicle produced around the same year. Numerous sources of literature exist regarding the changing composition of ELVs over the last two decades (ACORD 2001, Hooper et al. 2001, Montedison 1992, Staudinger and Keoleian 2001). For the purposes of model calibration, compositional data from the Association of Plastic Manufacturers in Europe (1999) describing a 1995 vehicle will be adopted (See Figure 10.11).

All the vehicles used within the shredding trial were systematically de-polluted to remove the rubber types and fluids; hence for consistency the composition listed in Figure 10.11 was adjusted to simulate the removal of these materials.

![Diagram](image)

**Figure 10.10**, An overview of the Newport shredder site setup and the four main output waste streams
10.7.3 The Post-fragmentation Cost Model Replicating the Effects Seen in Industry

Based on the processing scenario described in Figure 10.10 and the input composition adopted, the models process partition curves, materials interaction matrix and size distribution profiles were adjusted to replicate the output seen within the shredder trial data. Table 10.3 shows the main material groups used within the input composition and the predicted distribution of the materials amongst the four main output streams at the Newport facility. This effectively demonstrates the post-fragmentation models ability to replicate the separation capabilities seen within industry. The calibration of this model would have preferably been undertaken using meticulous facility data from one of the many UK post-fragmentation facilities, but despite prolonged industry contact and numerous site visits, the requested data collection exercise was considered beyond the scope of their current operating capabilities. This lack of industrial data with which to validate the modelling approach can be identified as a modelling limitation, and therefore highlighted as a possible area for further investigation and work within the future.
### Adjusted 1995 Vehicle Composition, Main Material Groups

<table>
<thead>
<tr>
<th>Material Groups</th>
<th>% grade input</th>
<th>% to scrap metal waste stream</th>
<th>% to density media separator waste stream</th>
<th>% to scrap metal waste stream from light fraction</th>
<th>% to landfill waste stream</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ferrous metals</td>
<td>70.84</td>
<td>68.98</td>
<td>1.27</td>
<td>0.51</td>
<td>0.04</td>
</tr>
<tr>
<td>Non-ferrous metals</td>
<td>10.82</td>
<td>0.31</td>
<td>9.59</td>
<td>0.05</td>
<td>0.83</td>
</tr>
<tr>
<td>Rubber</td>
<td>1.98</td>
<td>-</td>
<td>0.92</td>
<td>-</td>
<td>1.06</td>
</tr>
<tr>
<td>Glass</td>
<td>3.14</td>
<td>-</td>
<td>1.02</td>
<td>-</td>
<td>2.12</td>
</tr>
<tr>
<td>Other</td>
<td>3.15</td>
<td>-</td>
<td>1.24</td>
<td>-</td>
<td>1.90</td>
</tr>
<tr>
<td>Plastics</td>
<td>10.07</td>
<td>-</td>
<td>7.20</td>
<td>-</td>
<td>2.86</td>
</tr>
<tr>
<td><strong>Predicted</strong></td>
<td><strong>100.00</strong></td>
<td><strong>69.30</strong></td>
<td><strong>21.25</strong></td>
<td><strong>0.56</strong></td>
<td><strong>8.80</strong></td>
</tr>
<tr>
<td><strong>Actual</strong></td>
<td><strong>69.72</strong></td>
<td><strong>20.88</strong></td>
<td><strong>0.43</strong></td>
<td><strong>8.97</strong></td>
<td></td>
</tr>
</tbody>
</table>

**Table 10.3.** The predicted material distribution of various waste streams when the Newport shredding site is modelling using the post-fragmentation modelling approach.
Chapter 11

Implementation of the ELV Decision-Support Costing System

11.1 Introduction

The previous three chapters have discussed the various aspects of the ELV costing framework outlined in Chapter 7, to facilitate the establishment of an "as-is" economic model for the vehicle reclamation sector. This chapter describes the integration of all the direct and indirect costing approaches into one homogenised system, to assist in the modelling of an end-of-life vehicle facility. The implementation of the prototype software costing system is intended to be a portal for the creation of bespoke, quick and accurate cost estimates, the application of which will be further demonstrated within the Chapter 12 through a number of industrial case studies.

11.2 Software Implementation Overview

The research assertion within this thesis proposed that the end-of-life vehicle sector in the UK would not adopt a more diverse range of ELV processing activities (that are potentially more sustainable and resource effective) unless the economic feasibility of these scenarios could be realistically demonstrated. Hence, the literature survey and outlined costing framework thus far have aimed at identifying the most appropriate information, technologies and costing techniques to facilitate the development of an ELV costing system. The main aim of this system is to provide decision support to one of the three main ELV stakeholders (ATF, Shredder, Dense-media separator). To achieve this, certain modelling requirements must be fulfilled:

a) Any model developed must be dynamic and allow for fluctuations in the main market drivers which under-pin its economic analysis (material prices, labour rates, waste management costs).
b) The model must be enough to allow customisation for a specific end-of-life operator (i.e. one generalised operator model is not acceptable). For example, the ATF survey demonstrated the wide variation in processes, vehicle throughput and equipment investment from facility to facility, and therefore there is a need to provide the modelling resolution necessary to make this distinction.

Figure 11.1 provides an overview of the architecture of the ELV costing system. The ELV costing system is composed of three main subsystems; a dynamic database link that catalogues end-of-life processing data, a user-interface assisting in the creation of bespoke stakeholder estimates, and the core costing modules that implement the cost modelling approaches discussed within the previous three chapters. Furthermore, the core costing modules are classified into three main areas; the modelling of indirect...
processing costs (Chapter 8), the costing of direct pre-fragmentation material recovery (Chapter 9) and the costing of direct post-fragmentation material recovery (Chapter 10). These three areas have formed the core engine for the ELV costing system (as shown in Figure 11.1), and will be focused upon within the following sections of this chapter. The system was developed using a combination of Visual Basic.NET, Microsoft Excel, Microsoft Word and Microsoft Access.

11.3 Software Implementation of Indirect Cost Modelling

Chapter 8 discussed the implementation of a recognised costing methodology known as Activity Based Costing, and highlighted its strengths in attributing indirect processing costs. Given the ridged and systematic stages of ABCs methodology it was implemented within the ELV costing system as a linear five step modelling approach, which are summarised within Figure 11.2.

Figure 11.2, The introductory screen highlighting the five step software implementation of the ABC methodology
In step 1 the ABC process begins by identifying the end-of-life costing database to be used as the basis for the formulation of an EOL stakeholder estimate. The database was developed in Microsoft Access and catalogues the typical equipment costs, market prices and waste management costs, which were accumulated during the stakeholder interview period of the research. The database is intended to function as a data pool from which typical prices and rates can be extracted (using typical and non-specific prices reduces the need for highly detailed model development each time a new model is required, and avoids additional confidentially issues). In step 2 basic “ABC - consumption intensities” are then defined for the main stakeholder types. These consumption intensities are the fundamental values that affect the magnitude of the estimate. For example vehicle throughput per day, facility operating hours, percentage of different vehicle types, etc. and must be defined at the start in order to be applied throughout the model. Resource identification is then undertaken in step 3, allowing labour, vehicles, equipment and building costs to be fully described. Figure 11.3, 11.4 and 11.5 shows the initial three steps described above.

![Figure 11.3, Step1: database opening within the ABC methodology](image-url)
Figure 11.4, Step 2: identification of basic consumption intensities within the ABC approach

Figure 11.5, Step 3: identification and description of facility resources
Once a facility’s resources have been fully defined they must be individually attributed to a specific activity which utilises them. Therefore, this requires an activity definition stage (step 4), and a system whereby the cost of resources can be shared (step 5). Figure 11.6 shows the activity definition screen used to create and edit activities, while figure 11.7 shows the specific activity description process and allocation of the previously defined resources.

Throughout the five step ABC methodology the end-of-life costing database is periodically interrogated to access the prices and rates. To avoid single point estimating three values are returned from the database for a particular piece of equipment or rate; the minimum, maximum and most likely values. These are then used in conjunction with the Beta-PERT function, and Monte Carlo sample techniques to represent uncertainty in the cost model (see section 8.4).

Figure 11.6, Step 4: activity definition modelling screen showing a range of activities already defined
Chapter II

11.4 Software Implementation of Direct Pre-fragmentation Cost Modelling

The direct costs of de-pollution, part removal and plastics recovery (activities within pre-fragmentation vehicle recovery) have been implemented within the costing system using the approaches identified within Chapter 9, namely, the use of generative-analytical for de-pollution and part removal, and parametric regression analysis for plastics removal. The following subsections demonstrate the software implementation within the ELV costing system of these two approaches.

Figure 11.7, Step 5: activity description and resource attribution process

On completion of the Activity Based Costing methodology a binary file is created whichcatalogues each individual activity and its allocated indirect costs and revenues. Given the high number of activities and resources a specific stakeholder can have, the binary file acts as a record which can subsequently accessed and edited at a later date. This is important when moving from “as-is” to “to-be” modelling (see section 12.2).
11.4.1 Implementation of Generative-analytical Costing Approaches for De-pollution and Part Removal

The user interfaces for costing vehicle de-pollution and parts removal activities are very similar. Both interfaces establish a link with the end-of-life cost database and undertake an analysis according to the options selected. Figure 11.8 shows the assembly selection screen for parts removal. For example, if the premature ELV radio button is selected it pulls the average weight of a premature vehicle from the database. For the de-pollution interface this recalculates the percentage affect on the ELV Directive targets based on this new weight, and for the parts removal interface this alters the achievable revenues and weights for each sub-assembly type. A vehicle specific cost estimate can then be formulated based on these point and click options.

11.4.2 Implementation Parametric Cost Estimating for Plastics Recovery

The complexity of parametric regression analysis necessitates the inclusion of more selection options when compared to those required for generative-analytical. Figure 11.9 provides a screenshot from the pre-fragmentation module within the ELV costing system.
Figure 11.9, Pre-fragmentation material recovery screenshot that implement the parametric cost estimating approach
system that implements the Cost Estimate Relationships developed within Chapter 9, and gives an indication of sub-assembly removal effort and value in terms of the metrics also identified within this same chapter (namely, Material Removal Rate [MRR] and Value Recovery Rate [VRR]).

The end-of-life database catalogues the make and model parameters required by the CER to calculate the disassembly time. The system then uses this disassembly time combined with material market prices and component weights to calculate direct labour cost, recycling revenue and contribution to recycling target attainment. The aforementioned removal metrics (MRR and VRR) are then used to assist in sub-assembly selection (shown within the bar-graph in Figure 11.9), identifying the most efficient components in terms of mass and value. The user then has the option of manually appending specific components to the removal list, or allowing the ELV cost model to select assemblies automatically according to the removal metrics. A report building function has been incorporated into the model so a dismantling report can be quickly generated to assess the variation in effort and value between different makes and model of vehicle held within the End-of-life database (see Figure 11.10).

![Focus - 1999+ dismantle report](image)

**Figure 11.10, VRR dismantling report generated within Microsoft Word**
11.5 Software Implementation of Direct Post-fragmentation Cost Modelling

Chapter 10 discussed the considerations required to model the inefficiencies of the post-fragmentation process, and highlighted a method of determining waste stream segregation via the use of physical and material characteristics. This approach utilised equipment efficiency curves (known as process partition curves, see section 10.3), the random sizing effects of the shredding process (using size distribution profiles, see section 10.4) and a method of describing typical waste stream interactions (using an interaction matrix, see section 10.5), to model the recovery capabilities of each separation technology. These methods have been developed in Microsoft Excel, and utilise many embedded functions and graphing tools.

The post-fragmentation model begins with the user entering the typical composition of the waste stream processed (e.g. generalised automotive, industrial scrap, etc.). Material types and their relevant parameters (name, density, conductivity, typical size distribution once shredded, etc.) are extracted from the end-of-life database and placed in a separate worksheet within the Excel model (referred to as the materials attribute worksheet). The user then describes the material composition of the input waste stream with drop-down menus linked to this materials attribute worksheet. Once identified the user then selects the separation technologies the waste stream will pass through (see Figure 11.11). Each separation technology has a designated worksheet describing its separation efficiency based on the parameters required to generate the process partition curve (i.e. the cut-point \([x_{20}]\), probable error of separation \([E_p]\) and the cut-point offset, see section 10.3). By changing these parameters on each worksheet, the separation efficiencies of each of the technologies can be altered in line with the capabilities of the facility in question. Figure 11.12 shows the worksheet for altering the air-separation process partition curve, with the input waste composition entering the separation process listed below the efficiency performance parameters.
Figure 11.11, User description of waste stream process route and separation technologies

Figure 11.12, Air separation worksheet showing the process efficiency metrics of cut-point, probable error of separation and cut-point offset
In addition to utilising material characteristics to determine waste stream segregation, Chapter 10 also highlighted the need to determine physical particle sizes, to assist with the description of partition curves for trammels and cyclones. A method of using size distribution curves randomly sampled to replicate the effects of the shredding process was described. Figure 11.13 shows the randomly generated size distribution profile for the stainless steel fraction of the waste stream and the Monte-carlo sampling parameters available to alter the analysis. The final part of the post-fragmentation cost modelling is an allowance made for the imperfect contamination of end-of-life waste streams, and the fact that the shredding process rarely liberates the mix into perfect isolation. Section 10.5 discussed the need to incorporate material interactions within the post-fragmentation model, given the required idealised operating conditions of many process partition curves. These typical material interactions are described within the model using a 2D interactions matrix (see Figure 11.14) and should be completed for a typical waste stream type (i.e. automotive,

**Figure 11.13**, Randomly generating particle size distributions
Figure 11.14, Material interaction matrix describing un-liberated material mixes WEEE, industrial waste). This matrix identifies those combinations of materials that typical do not separate well, and can be rarely considered to be in isolation when passing through subsequent separation technologies.

Figure 11.15 summarises all of the aforementioned modelling approaches and how they map onto the original post-fragmentation modelling approach (see Figure 10.1). This demonstrates the realisation of the original post-fragmentation modelling approach with the ELV costing system.

11.6 Costing System Analysis Features

The primary output from the ELV costing system is a number of cost and revenue break-down statistics, an automatically generated report that identifies the cost intensive hotspots in an organisation, a facility specific measure of their current
Modelling post-fragmentation process inefficiency and value-added waste stream processing

**INPUT VALUE**

Vehicle input composition

**OUTPUT VALUE**

Output waste stream A (%)

Output waste stream B (%)

**Materials attribute worksheet**

**PROCESS INEFFICIENCY**

Process partition curves

**Materials interaction matrix**

Value determined by the purity of the input waste stream.

A materials attribute database provides generic material property data. Material size distributions determined by uncertainty analysis, un-liberated material interactions by an interaction matrix.

Value determined by the purity of each output waste stream.

**Figure 11.15**, The mapping of the developed Excel methods to the original post-fragmentation framework in Chapter 10
recycling and recovery levels, and a range of dismantling vehicle reports. The main aim of this output is to identify a facilities strengths and weaknesses, in terms of revenue generation and incurred costs. By identifying these activities, it allows an end-of-life stakeholder to building subsequent processing scenarios around these assets and pitfalls. Figure 11.16 illustrates the main analysis screen, in which all direct and indirect costs are brought together. This provides a facility specific overview as to

![Image of the analysis screen](https://example.com/image)

Figure 11.16, The ELV costing system cost and revenue analysis screen, and facility specific target attainment
the costs incurred and the revenues generated, along with an assessment of a facilities current recycling and recovery levels.

Depending on the type of end-of-life stakeholder considered, combined with the availability of data for that stakeholder, one or all of the costing modules and analysis features can be used. Figure 11.17 provides montage of the main costing system analysis screens. These modules and features within the ELV costing system can be used to develop an “as-is” snapshot of an end-of-life stakeholder, and can use a mixture of direct and indirect costing modules depending on the goal of the analysis, before being used in a more predictive way (“to-be”) to consider the economic feasibility processing alternatives. The “as-is” and “to-be” functionality of the ELV costing system is further tested and demonstrated in the case studies in Chapter 12.

Figure 11.17, Main software menu and analysis features of the ELV costing system
Chapter 12

Case Study

12.1 Introduction

This chapter utilises case studies to demonstrate the various stages of the ELV costing framework discussed within this thesis. The chapter begins by providing an overview of how the costing techniques can be used in both a practical and predictive way. Three case-studies are then presented, each demonstrating a fundamental element of the ELV costing framework, these being: a) the use of the indirect costing approach to assess the current activities of an end-of-life operator, b) the economic feasibility of them adopting more sustainable processing scenarios (dismantling vehicles for material recycling), c) the use of the post-fragmentation costing approach to predict the changes in output compositions at a UK shredding site.

12.2 Case Study Overview

The aim of the research reported in this thesis is to provide costing and revenue transparency to the vehicle recovery sector so that various vehicle processing scenarios, be they imposed by legislative requirements, improved value recovery or proactive approaches reduce environmental concerns, can be easily assessed. The foundation for this assessment is the establishment of a base cost model, referred to as the “as-is” model, which accurately reflects a current facility’s costs and revenues. This understanding in itself can provide a valuable insight for a sector that has not historically understood the economics of its own operation very well. The development of this “as-is” model will be demonstrated for a specific UK recovery facility, and will be used to gain an insight into its operating profitability. This modelling is vitally important as it provides a gauge as to the economic stability of a particular operator, which will ultimately measure their suitability to adopting new more sustainable ways of working. Operators that turn a reasonable profit, despite
their ELV directive commitments, are more likely to adopt new vehicle processing scenarios than those that are simply surviving.

In the next stage, once a facility’s economic strengths and weaknesses have been identified, the costing approaches within the framework will be utilised to hypothesise as to the viability of the “to-be” situation. This “to-be” modelling allows a facility to consider the pros and cons of processing strategies yet to be adopted, and allows a theoretical glimpse of various investment decisions. The “to-be” case study will be demonstrated using two instances, both highlighting processing strategies that are not currently widely adopted within the UK recovery industry. The first will be the adoption of material removal for recycling pre-fragmentation (i.e. vehicle dismantling for material recycling), and the second will be the alteration of post-fragmentation waste streams based on changes in its input composition (i.e. moving towards batch processing of different waste stream types).

Given that the remit of the framework introduces direct and indirect costing approaches that influence both pre-fragmentation (ATFs) and post-fragmentation (shredders and dense media separators) stakeholders, the following case-studies are aimed at demonstrating the techniques discussed within the previous chapters and how they map onto the “as-is” and “to-be” case study models (see Figure 12.1). The case study will be broken down into three sections; the establishment of a pre-fragmentation “as-is” model that will make use of the indirect costing approaches discussed in Chapter 8, the development of a “to-be” model looking at the profitability of pre-fragmentation material recycling that utilises the parametric techniques highlighted in Chapter 9, and the development of another “to-be” model looking at the routing of post-fragmentation waste based on the costing method discussed within Chapter 10. For both pre-fragmentation case study models (“as-is” and “to-be”) the ATF facility of Albert Looms located on the outskirts of Derby (www.albertlooms.co.uk) will be used. For the post-fragmentation case study the shredder site layout adopted within the UK shredder trials in Newport will be used. The following sub-sections provide a brief overview of these two organisations.
Figure 12.1, Overview of case studies, the organisations involved and main costing approaches adopted

12.2.1 Pre-fragmentation Company Background: Albert Looms

Albert Looms (known simply as “Looms”) is a family run business located on the outskirts of Derby-UK, in the small town of Spondon. Established in the 1930’s the business has always been a breakers yard, and has provided vehicle spares to generations of hobbyists and mechanics in the surrounding areas. The site occupies a region of approximately 17,000 m², and employees 19 staff of varying skill levels and status. The business operates a 58 hour working week (approximately 13 hours of which are at the weekend) for 52 weeks of the year.

Unlike many modern breakers yards, Looms still allows its customers to remove vehicle components themselves. They do not stack their vehicles but place them end to end within the main compound. Over the years many breaker sites have changed this policy given the health and safety concerns of customers climbing over aging automobiles, and the fact that untrained dismantlers can potentially devalue vehicles by damaging other saleable assemblies. Looms receives approximately 20 new ELVs...
per day, of which 60% are delivered and 40% are collected. For onsite deliveries
Looms operates a vehicle buyback policy, the value of which is dependent on the
price of scrap hulks and the number of saleable components. Vehicle collection
operates within the Derby area up to a radius of around 15 miles and impacts on the
buyback price given to the last owner. Looms typically deals in natural ELVs, with a
ratio of around 2:1 of natural against premature. Once on site, certain category B
(breakable salvage) vehicles are placed in a holding compound that are still under
dispute with the insurance companies. All other vehicles are taken directly to the
vehicle de-pollution compound, before being placed into the main dismantling yard.
Looms crushes and ships hulks to the shredder each day, and replaces the stock by
rotating anti-clockwise around the yard. Clearing 20-30 of the oldest vehicles and
replacing them with the new arrivals. A typical vehicle is held on site for around 4
weeks. For a more detailed insight into the activities and processing stages undertaken
at Looms see section 12.3.1.

Figure 12.2, Photos of the Albert Looms staff and site
12.2.2 Post-fragmentation Company Background: Sims Group Ltd.

Sims Group Ltd is one of the main players in the UK waste metal market and sells roughly 2 million tonnes of ferrous and non-ferrous material each year. The company currently has 120 locations across 4 continents. Sims operates 5 shredder sites within the UK, with the one located in Newport being used as the example facility within the "to-be" post-fragmentation case-study.

The crushed or bailed hulks arrive on site and are lifted by two material handlers onto a conveyor belt that feeds the rotary hammer mill. The hammer mill is a high speed rotating drum composed of rows of high strength hammers that continually fragment the waste until it can pass through meshed openings within the chamber walls. The mill is by far the most vital piece of equipment at the facility as downtime on this affects the whole system. The Newport site was redeveloped in 2004 and has a 9,000 horsepower shredder onsite. Hammers are typically replaced every 8,000 tonnes. Once fragmented the comminuted waste passes through a series of automated separation technologies to extract the directly saleable ferrous metal, the non-ferrous fraction (which requires further processing), and the material destined for landfill. Section 10.7 shows the facilities separation activities.

12.3 Case Study 1: The "as-is" model for Albert Looms

The "as-is" case study is intended to provide a foundation to the economic health of an ELV operator, and to identify their main value recovery "hotspots" around which a more robust business model can be built. Although this case study generates the "as-is" snapshot for an ATF, the process can equally be applied to other operators further downstream in the vehicle recovery chain (i.e. shredders and non-ferrous separators). The following sub-sections apply the indirect costing techniques discussed within Chapter 8 using real life data to demonstrate the benefits of ELV reclamation costing.

12.3.1 Information Models for Albert Looms Current Activities

An IDEF0 modelling approach was adopted to provide a comprehensive snapshot of activities undertaken, and was completed with the assistance of the Looms' operations
manager in August 2006. Figures 12.3-12.11 provide a detailed account of the resources required for an end-of-life vehicle to be processed at Albert Looms. The root node is given in Figure 12.3, with subsequent child nodes listed on the pages following. This thorough information modelling exercise provides a detailed starting point for the activity based costing approach required to develop the Albert Looms “as-is” model.

12.3.2 Activity Based Costing Implementation: Activity Aggregation

Activity aggregation is important as it provides a level of resolution that is most applicable to the requirements of a model. This enables the redundant complexity that does not significantly contribute to the understanding of the overall cost to be omitted. Therefore, the activities identified within the previous IDEF0 modelling stage were further refined to remove those activities which demonstrated general procedure as opposed to value-added operations.
Figure 12.3, A0: Root diagram for Albert Looms
Figure 12.4, A1: The reverse logistic of returning vehicle

Figure 12.5, A2: The vehicle de-registration process
Figure 12.6, A3: The vehicle de-pollution process

Figure 12.7, A33: Detailed removal activities of top-level fluids
Figure 12.8, A334: The tyre removal process

Figure 12.9, A34: Detailed removal activities of bottom-level fluids
Figure 12.10, A4: The removal of parts for resale

Figure 12.11, A5: The compaction and transportation of hulks to the shredder
12.3.3 Activity Based Costing Implementation: Determining Applicable Indirect Costs

The next, and perhaps most important stage of the ABC approach, is to identify all the resources that Albert Looms uses, and the indirect costs that apply to using those resources. Within Chapter 8 the following indirect costing areas were identified as those incurred by using various facility resources:

i.) Operating equipment, transport and building depreciation
ii.) Business rates (U.B.R. - Uniform Business Rates)
iii.) Power consumption costs (heating, light, equipment)
iv.) Fuel costs (recovery transportation trucks, forklifts)
v.) Maintenance and consumables
vi.) Taxation, licences, training and insurance
vii.) Employee fringe benefits

Obviously all these cost areas aren’t incurred by every resource. For example, a building would require a depreciation value, a business rate, a power consumption figure, a maintenance charge and possibly an insurance cost. Whereas a resource such as a collection truck might require taxation, insurance, fuel costs and a depreciation value. To combat this issue, four resource classifications were developed that incorporated the most appropriate costing areas, these were; Buildings & Land, Vehicles, Equipment and Labour. It should be noted that the model allows labour to be defined as an indirect as well as a direct cost, due to the requirement for representing holistic employee wages and attributing management salaries.

The majority of the data that assisted with describing the indirect costs of the resources was obtained from Albert Looms records. Where exact cost could not be obtained the operation manager would provide minimum, maximum and most likely estimates, which could be used within the uncertainty modelling techniques adopted within the ELV cost model. The other costing data, such as the calculation of the Uniform Business Rates (UBR) were allocated to resources using online tax records (www.ratinglists.voa.gov.uk). Table 12.1 shows equipment cost data released by Looms, and Figure 12.12 shows its definition within the ELV-cost model. Figure 12.13 provides an aerial shot of the facility which highlights the incurred UBR costs.
Table 12.1, Albert Looms equipment investment records over the last 8 years

<table>
<thead>
<tr>
<th>Equipment</th>
<th>Year of purchase</th>
<th>Exact cost</th>
<th>Minimum</th>
<th>Average</th>
<th>Maximum</th>
<th>Depreciation period (years)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Priestman (Magnet)</td>
<td>1999</td>
<td>£2,585</td>
<td></td>
<td></td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>Generator</td>
<td>1999</td>
<td>£250</td>
<td></td>
<td></td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>Magnet gear</td>
<td>2000</td>
<td>£525</td>
<td></td>
<td></td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>Priestman</td>
<td>2001</td>
<td>£3,575</td>
<td></td>
<td></td>
<td></td>
<td>8</td>
</tr>
<tr>
<td>Tyre machine x2</td>
<td>2001</td>
<td>£2,400</td>
<td></td>
<td></td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>Hatachi EY700-1991</td>
<td>2002</td>
<td>£22,500</td>
<td></td>
<td></td>
<td></td>
<td>8</td>
</tr>
<tr>
<td>Beach Breaker</td>
<td>2002</td>
<td>£240</td>
<td></td>
<td></td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>Compressor</td>
<td>2002</td>
<td>£1,950</td>
<td></td>
<td></td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>Priestman 216</td>
<td>2002</td>
<td>£7,175</td>
<td></td>
<td></td>
<td></td>
<td>8</td>
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<tr>
<td>Tyre balancer etc.</td>
<td>2002</td>
<td>£918</td>
<td></td>
<td></td>
<td></td>
<td>10</td>
</tr>
<tr>
<td>Scrap Handling Jig</td>
<td>2003</td>
<td>£750</td>
<td></td>
<td></td>
<td></td>
<td>10</td>
</tr>
<tr>
<td>Crusher and installation</td>
<td>2003</td>
<td>£14,441</td>
<td></td>
<td></td>
<td></td>
<td>20</td>
</tr>
<tr>
<td>Wheel crusher</td>
<td>2004</td>
<td>£5,200</td>
<td></td>
<td></td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>Steam cleaner</td>
<td>2004</td>
<td>£1,550</td>
<td></td>
<td></td>
<td></td>
<td>10</td>
</tr>
<tr>
<td>Oil storage tank</td>
<td>2004</td>
<td>£1,220</td>
<td></td>
<td></td>
<td></td>
<td>20</td>
</tr>
<tr>
<td>Concrete float</td>
<td>2004</td>
<td>£281</td>
<td></td>
<td></td>
<td></td>
<td>20</td>
</tr>
<tr>
<td>Engine bay</td>
<td>2004</td>
<td>£7,235</td>
<td></td>
<td></td>
<td></td>
<td>20</td>
</tr>
<tr>
<td>Interceptor</td>
<td>2004</td>
<td>£12,093</td>
<td></td>
<td></td>
<td></td>
<td>25</td>
</tr>
<tr>
<td>Engine Bay</td>
<td>2005</td>
<td>£2,642</td>
<td></td>
<td></td>
<td></td>
<td>20</td>
</tr>
<tr>
<td>Canopy alarm/wiring</td>
<td>2005</td>
<td>£434</td>
<td></td>
<td></td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>Depollution equipment</td>
<td>2005</td>
<td>£23,959</td>
<td></td>
<td></td>
<td></td>
<td>20</td>
</tr>
<tr>
<td>Petrol bund/wiring/alarm</td>
<td>2005</td>
<td>£1,474</td>
<td></td>
<td></td>
<td></td>
<td>10</td>
</tr>
<tr>
<td>Air con equipment</td>
<td>2005</td>
<td>£1,166</td>
<td></td>
<td></td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>Shocking equipment</td>
<td>2005</td>
<td>£481</td>
<td></td>
<td></td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>Cutting equipment</td>
<td>2005</td>
<td>£3,425</td>
<td></td>
<td></td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>Industrial cutters</td>
<td>2006</td>
<td>£1,950</td>
<td></td>
<td></td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>Air bag deployers</td>
<td>2006</td>
<td>£675</td>
<td></td>
<td></td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>Shocking depollution</td>
<td>2006</td>
<td>£3,000</td>
<td></td>
<td></td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>Noise meter</td>
<td>2006</td>
<td>£20</td>
<td></td>
<td></td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>Pat tester</td>
<td>2006</td>
<td>£384</td>
<td></td>
<td></td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>Diesel tank</td>
<td>2006</td>
<td>£999</td>
<td></td>
<td></td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>Part removal tool boxes</td>
<td>unknown</td>
<td>£2,000</td>
<td>£2,000</td>
<td>£2,000</td>
<td></td>
<td>10</td>
</tr>
<tr>
<td>Collection vehicle - 7.5t</td>
<td>unknown</td>
<td>£8,000</td>
<td>£9,000</td>
<td>£10,000</td>
<td></td>
<td>8</td>
</tr>
<tr>
<td>Internet enabling equipment</td>
<td>unknown</td>
<td>£550</td>
<td>£550</td>
<td>£550</td>
<td></td>
<td>1</td>
</tr>
<tr>
<td>Standard PC</td>
<td>unknown</td>
<td>£1,000</td>
<td>£1,000</td>
<td>£1,000</td>
<td></td>
<td>5</td>
</tr>
<tr>
<td>Forklift 1 (Diesel) hired</td>
<td>unknown</td>
<td>£800</td>
<td>£800</td>
<td>£800</td>
<td></td>
<td>1</td>
</tr>
<tr>
<td>Forklift 2 (Diesel) hired</td>
<td>unknown</td>
<td>£800</td>
<td>£800</td>
<td>£800</td>
<td></td>
<td>1</td>
</tr>
<tr>
<td>PC 2</td>
<td>unknown</td>
<td>£1,000</td>
<td>£1,000</td>
<td>£1,000</td>
<td></td>
<td>5</td>
</tr>
</tbody>
</table>

Figure 12.12, Application of indirect costs areas to equipment resources
Figure 12.13, Overlay of Uniform Business Rates onto the Albert Loom site
12.3.4 Activity Based Costing Implementation: Attribution of Resources to Activities

Once the indirect costing data has been allocated to each resource, the resources are then attributed to the activities to which they relate. The information required to effectively complete this stage has already been encapsulated within the IDEF0 model. This is simply a case of taking the aggregated activities defined within section 12.3.2 and attributing the various resources defined within 12.3.3. The only additional consideration that must be made during this process is if activity attribution is to be weighted, i.e. a resource is clearly used more extensively by one activity than another, where equally dividing the cost of the resources would not accurately reflect this utilisation. For the purposes of the Albert Looms model, all resources have been equally shared between the activities that use them, as the inclusion of weightings would have required additional time on site to model not just the process but also the resource utilisation levels. Appendix A1.5 provides a detailed attribution map of which resources are applied to which aggregated activities, and Figure 12.14 highlights the representation of this map within the ELV-cost model.

Figure 12.14, Attribution of Albert Looms resources to aggregated activities
12.3.5 Analysis of Albert Looms “As-is” Modelling Results

The ELV cost model was run for Albert Looms using the procedures and data described within the previous sub-sections.

During the execution of the model it became apparent that a number of assumptions and allowances had to be made. For example, it was decided to consider the activities of vehicle de-pollution and parts-removal not using all of the direct costing elements of the ELV-cost model. For the purposes of the “as-is” model it was decided to use the indirect ABC approach to model the costs (inclusive of labour), while using the vehicle specific direct costing approaches to determine the achievable revenue. This was done for two reasons. Firstly, indirect attribution of cost via ABC gives a more complete picture of resources (particularly labour) that can be extremely influential on the final cost. Secondly, by still retaining revenue determination using a direct costing technique (which is highly vehicle specific); the two parts of the analysis can be kept separate and the results are more easily understood in isolation.

Figure 12.15 shows the ELV-costing systems output. The facility incurred £121.87 per ELV in processing costs, which includes all resource areas previously mentioned (labour, depreciation, rates, etc.). Meanwhile, they achieved a £142.30 per ELV in revenue (excluding part-resale revenue), based on the material market prices obtained in August 2006. This gives a net profit of £20.43 per vehicle, and this figure can be significantly boosted with the addition of any revenue from parts resale, the costs of which have already been included into the indirect ABC model (mechanics, removal equipment, etc.). Considering part resale revenue in this way allows us to understand Looms basic operating profit via its secondary activities (i.e. de-pollution, waste segregation, hulk sales), and then superimpose the extremely variable revenue effects of its core focus, i.e. a typical vehicle passing through the facility will make the business around £20.43 in profit, any component that a customer decides to strip and buy can then be appended directly to this profit margin.
Figure 12.15, Albert Looms' costs and revenues by aggregated activities (excludes the revenues from parts-resale)

The results would suggest that at the time of the model development Albert Looms was in a financially robust situation, adequately able to cover the additional investment and waste management costs required by the ELV Directive. The results show that hulk sales to the shredder provide an important revenue stream for the facility, but equally the segregation of other valuable materials (catalysts and engines) are beginning to mitigate some of the financial risk. Any revenue generated from parts resale can significantly bolster this financial robustness further, and should be highlighted as a significant contributor to Albert Looms current business model.

It must be noted that the quoted profit figure of £20.43 per ELV is an instantaneous snapshot of Looms’ operating margins based on the material market prices available in August 2006. These market prices are continually fluctuating, and it cannot be assumed that this margin will be consistently maintained. A drop in any one of the main material markets that determines the price of hulk sales could potentially eat into this margin very quickly. The vehicle buyback price that Looms pays to the final customer is a good cushion for these variable market effects (e.g. if hulk prices fall by £10 then the buyback price given to the last owner could also be dropped by £10).
This currently offers a certain level of control over the achievable profit margin, but this control will only last as long as the hulk price allows Looms to pay the last owner. If the situation occurs that Looms is unable to pay the last owner then the profit margin will be more susceptible to effects of material market price variations.

All of the above results and analysis would suggest that Albert Looms is in healthy financial shape, making it a prime example of an ATF that would be more receptive to operating procedural changes (i.e. re-ordering of activities), or the addition of new core competencies (e.g. onsite rubber crumbing or windscreen PBT recovery). What this “as-is” analysis has shown is the need to consider each end-of-life stakeholder as an individual entity, and that by assessing the economic health of that stakeholder it is possible to make a determination as to their ability to move towards an improved, more sustainable, “to-be” scenario.

12.4 Case Study 2: The “to-be” model for Albert Looms

Numerous “to-be” modelling scenarios can be considered for Albert Looms in light of the economic robustness demonstrated within the first case study. This speculative modelling could potentially encompass anything, from re-evaluating the facilities current activities, through to considering if radical new ones could be easily integrated. In line with the research aim of this thesis, the “to-be” model will therefore be focused on trying to promote sustainable vehicle reclamation strategies. By far the most widely contentious reclamation strategy the current vehicle recovery chain is still struggling with, is “at what point does automobile dismantling for material recycling become economically viable”. This question is becoming increasingly important as the UK moves closer to the ELV Directive 2015 recycling and recovery target of 95%, and the technological and investment leap required to meet this target becomes more apparent. Within the ELV costing framework discussed in Chapter 7, an allowance was made to consider the economics of pre-fragmentation material removal. A parametric costing approach was further discussed within Chapter 9 based on a plastics dismantling study conducted at Albert Looms (see section 9.3.2). The costing techniques developed during this process will be further utilised within this case study to take a projected look at how much it will cost Albert Looms to conform to the higher recycling and recovery target of 95% in 2015 (solely via dismantling),
and whether plastics dismantling can potentially ever return Looms a reasonable profit.

12.4.1 Parametric Implementation: Vehicle Demographic and Component Selection

Within Chapter 9, parametric estimating techniques were used to generate vehicle dismantling equations with the intention of utilising data held within the IDIS to cost other makes and model of vehicle. To gain an insight into the “to-be” situation in 2015, it is possible to select vehicles from the IDIS database that that are most likely to be natural ELVs in 2015, based on the assumption that an average vehicle has a life of around 13 years. Therefore, the top selling vehicles in 2002 will provide a good approximation as to the demographic of vehicles processed in 2015.

The vehicle dismantling equations developed use various component parameter data held within the IDIS database to produce a bespoke removal time for a particular assembly, which can subsequently be used to determine the direct labour costs incurred. These dismantling times can also form the basis of component selection metrics, two of which were highlighted within section 9.3.6 as the Mass Removal Rate (MRR), and the Value Removal Rate (VRR). The use of these metrics to select plastic components should be based on the goal of Albert Looms. If target attainment is required, the Material Removal Rate should be used, as this identifies the heaviest and easiest components to remove first, and gives a better mass-versus-labour return. Alternatively, if Looms is interested in knowing if there are any components on a vehicle that can return a profit (when compared to a workers wage rate [£/s]), then the Value Removal Rate should be used, as this considers the value of the component removed as well as its weight.

Table 12.2 demonstrates both of these scenarios, which include the use of the Material Removal Rate to select components to fulfil the reported 5.18% deficit to the 2015 recycling target (Weatherhead and Hulse, 2005), and the use of the Value Removal Rate on the same vehicles to select components capable of returning a profit. The vehicles selected are the UK’s top selling vehicles in 2002, representative of typical natural ELVs in 2015.
Chapter 12

<table>
<thead>
<tr>
<th>2002 sales</th>
<th>Vehicle</th>
<th>Weight (kg)</th>
<th>No. of components</th>
<th>Dismantling Time</th>
<th>Labour cost £</th>
<th>Revenue £</th>
</tr>
</thead>
<tbody>
<tr>
<td>1st</td>
<td>Ford focus 1998+</td>
<td>50.70 kg</td>
<td>33</td>
<td>1h 40 mins</td>
<td>£11.67</td>
<td>£6.46</td>
</tr>
<tr>
<td>2nd</td>
<td>Vauxhall Corsa</td>
<td>50.45 kg</td>
<td>20</td>
<td>1h 38 mins</td>
<td>£11.43</td>
<td>£7.68</td>
</tr>
<tr>
<td>3rd</td>
<td>Vauxhall Astra</td>
<td>50.38 kg</td>
<td>21</td>
<td>1h 50 mins</td>
<td>£12.83</td>
<td>£6.24</td>
</tr>
<tr>
<td>4th</td>
<td>Peugeot 206</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Data unavailable in DIS</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>5th</td>
<td>Ford Fiesta</td>
<td>53.49 kg</td>
<td>20</td>
<td>1h 52 mins</td>
<td>£13.07</td>
<td>£6.53</td>
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<tr>
<td>Averages</td>
<td>51.26 kg</td>
<td>24</td>
<td></td>
<td>1h 45 mins</td>
<td>£12.25</td>
<td>£6.72</td>
</tr>
</tbody>
</table>

Material Removal Rate used to identify components up to 50.30 kg (5.18%) of a vehicle's weight

<table>
<thead>
<tr>
<th>Value Removal Rate used to identify components that return a profit when compared to an hourly rate</th>
</tr>
</thead>
<tbody>
<tr>
<td>1st</td>
</tr>
<tr>
<td>2nd</td>
</tr>
<tr>
<td>3rd</td>
</tr>
<tr>
<td>4th</td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td>5th</td>
</tr>
<tr>
<td>Averages</td>
</tr>
</tbody>
</table>

# Table 12.2, Estimating the achievable costs and revenues from pre-fragmentation dismantling based on MRR and VRR

The worker’s wage rate adopted in the above table is taken from ATF interviews undertaken in 2006 and uses a rate of £7.00 per hour (£5.82 per hour in wages and £1.18 per hour in fringe benefits). Figure 12.16 provides a sensitivity analysis around this adopted rate to demonstrate its impact on the resulting net revenue (recycling revenue minus removal cost).

An additional cost which must also be considered when assessing the dismantling of components is the logistical cost of transporting the removed materials to a recycling facility (if Looms decided not to manually sort and grind onsite). Previous dismantling studies have assumed a flat transport fee of around £50 per tonne (Weatherhead 2005), which based on the average mass removed per ELV within both methods (i.e. 51.26 kg and 7.24 kg respectively, see Table 12.2) equating to £2.56 per vehicle for the MRR scenario, and £0.36 per vehicle for the VRR scenario.
12.4.2 Analysis of Albert Looms "to-be" Modelling Results

The purpose of this "to-be" modelling is to offer analysis as to the cost of Albert Looms meeting the 2015 recycling target, and the potential for them to use pre-fragmentation dismantling as a means of obtaining value. Table 12.3 provides an aggregated summary of the costs and revenues associated with the dismantling of vehicles for target attainment (mass removal scenario), and the dismantling of vehicles for value recovery (value removal scenario).

If Albert Looms were to meet the 2015 recycling target today through further manual dismantling, it would result in an estimated net cost per ELV of around £8.09. The additional investment costs of new equipment and storage facilities would also need to be factored into this if they decided to adopt this practice. What this estimate indicates is the substantial cost burden that will be required to meet the future target if the recycling levels remain the same and the investment and technology in UK post-fragmentation facilities is not significantly improved. It must be stated that the likelihood of this eventuality (achievement of the 2015 target via material
Table 12.3, Summary of costs & revenues incurred via different processing scenarios

dismantling) based on the strategic direction of other more proactive EU member states, would suggest that this will not be the case in the UK. Given the achievable throughput rates of automated waste stream technologies, it is envisaged that post-fragmentation processing will provide the most economically viable point at which to recover the additional rubbers, glass and plastics which are currently sent to landfill. As to whom within the vehicle value chain will ultimately be financially responsible for either of these scenarios is yet to be made clear. Previous investment and de-pollution costs have been offset by extremely strong scrap metal prices, but questions as to this markets long term stability may negate the possibility of scrap revenue supporting the vehicle recovery sector in the future.

The previous analysis has also offered an opportunity to assess the feasibility of Albert Looms disregarding the 2015 target (assuming post-fragmentation conformance), and only dismantling components that can offset the incurred labour costs of their removal. These would be large, heavy sub-assemblies (such as bumpers and internal trim) that are relatively easy to remove. In these instances the recycling revenue generated (£1.96, see Table 12.3) is capable of offsetting the direct labour costs incurred (£1.16, see Table 12.3), producing a net revenue per ELV of around £0.80. This revenue is reduced when the material recovered is transported to a recycling facility for shredding and granulation, resulting in an overall net profit of £0.43 per ELV. Although the small profit margin per ELV does not support
significant investment in expansion of processes and facility, in future as plastic markets become more established and prices increase, this may be considered a viable option. This analysis was undertaken using a wage rate of £7 per hour, Figure 12.17 presents the variation in net profit when considering wage rates either side of this value when the logistical costs of material transport is also included.

Unlike the smooth trend seen within Figure 12.16 the resulting net profit seen in Figure 12.17 fluctuates substantially. This is due to the number of competing parameters that affect the final cost. A lower wage rate reduces the threshold at which the value removal rate selects components, as more components are capable of offsetting the direct labour incurred. Material type determines the obtainable revenue, while the quantity of material removed increases the logistical costs of transportation. The variation between these two figures highlights the significant economic impact of transportation costs, in addition to commonly reported negative environmental impacts associated with reverse logistics. This would perhaps suggest a strong case for a more geographically concentrated approach to vehicle recovery and material recycling. If Albert Looms was to diversify its core competencies to incorporate plastics recycling this could potentially allow them to sell reprocessed granulate directly back to the product suppliers and attain high revenues. These recycling

![Graph of net profit variation](image)

**Figure 12.17**, The impact on average overall profit per ELV for different wage rates when the logistical costs of material transport is included
activities need not be exclusively focused on recovering just automotive polymers, but could also encompass additional product waste streams (consumer packaging, industrial scrap, plastic from WEEE) which will become increasingly more abundant from surrounding businesses as end-of-life legislation becomes more established within the UK.

12.5 Case Study 3: The "to-be" Situation for Sims Group Ltd

The Newport shredding facility was selected to form the basis of the post-fragmentation cost modelling techniques due to the availability of published data. This site was selected by the DTI in 2005 to benchmark the technical capabilities of UK post-fragmentation technology, and to determine the average metallic fraction currently recovered (using 400 ELVs which were approximately 15 years of age). The data included within this report gave a comprehensive look at the separation effectiveness of current technologies, and provides a good overview of the typical waste stream routing of a fragmented ELV. Based on these figures a representation of this facility was generated using the post-fragmentation costing techniques established within Chapter 10. The following sub-sections take this modelling a step further and use the Newport shredder model to predict the likely waste stream segregation, based on the changing composition of ELVs in the next 8 years and due for retirement in 2015.

12.5.1 Selecting a New Vehicle Composition

The post-fragmentation costing techniques discussed within Chapter 10 presented an approach to determine the value-added processing of automated separation technologies. This value determination was based on trying to establish the separation effectiveness of current technologies and the percentage segregation of waste at various points in the processing chain. It made an implicit link between the material characteristics of a waste stream and the separation capabilities of the processes it passed through. Hence, once the effects of the technologies had been calibrated, only changes in the input composition could affect what went where. The original calibration of the model was undertaken using a 1995 vehicle composition taken from
the Association of Plastic Manufacturers in Europe (APME 1999). These calibration exercises along with its limitations are discussed within section 10.7.

When considering the future "to-be" situation for the Newport shredder site it is assumed that they will keep the same processing route that is currently in operation. The main four outputs from each of the nodes are; ferrous metals, non-ferrous to density media separators, ferrous metal from light fraction and landfill (see Figure 12.18). The aim of the "to-be" model in this instance is to determine the expected changes in output composition that the four main output waste streams will see when a more up to date ELV material composition is passed through the Newport model. As with the "to-be" model for Albert Looms the most ideal vehicle compositions would be obtained from those top-selling vehicles manufactured in 2002, as these would give an indication as to the typical vehicle types processed around the key target attainment date of 2015. Unfortunately, as the majority of these vehicles still have models currently in production, detailed compositional data is difficult to obtain due to the confidentiality issues. It is therefore proposed to formulate a generalised composition of a 2002 vehicle from literature discussing general automotive material trends. Figure 12.19 is taken from Giannouli et al. (2007), and highlights the main compositional changes the automobile has undergone. The reduction of a vehicles' ferrous content is balanced by an increased use of more light-weight and fuel efficient material types. The increased abundance of engineering plastics and aluminium to replace the traditional heavy-duty ferrous components will have significant ramifications on the separation capabilities of current end-of-life technologies.

Figure 12.18, An overview of the Newport shredder site setup and the four main output waste streams
The composition adopted within the “to-be” model was made up of: 61% ferrous, 11% non-ferrous, 16% plastics and 12% Other. As with the 1995 composition the fluids (2.1% of weight) and tyres (3.5% of weight) were removed from the “other” material category to simulate de-pollution, and the compositional percentages were then recalculated to allow for this removal. The original model was run using ten material types, so the recalculated 4 main groups were then redistributed based on these ten material types and 1995 weightings (see Table 12.4).

12.5.2 Analysis of Newport Shredder Site “to-be” Modelling Results

The modelling of the Newport shredder site has provided an opportunity to demonstrate the application of the post-fragmentation costing approach within the ELV costing framework. The original Newport model was run using the estimated 2002 vehicle composition listed in Table 12.4, with the resulting waste stream outputs and their comparison with the 1995 vehicle composition listed within Table 12.5.
The predicted changes in waste stream compositions produces some interesting results, not least in terms of the shift in output quantities that each end node experiences. These main changes can be summarised thus:

- The main ferrous output stream is significantly reduced from the 1995 levels (-6.15%). This drop is due to the reduction of the ferrous input composition, and is not due to it being diverted to other output waste streams.

- The waste stream that is currently sent to the density media separators substantially increases under the new composition (+5.00%). This is due to the increased abundance of plastics which provides the most significant increase, and is also supported by a slight increase in the non-ferrous content diverted to this stream.
<table>
<thead>
<tr>
<th>Vehicle Composition</th>
<th>% grade input</th>
<th>% to scrap metal waste stream</th>
<th>% to density media separator waste stream</th>
<th>% to scrap metal waste stream from light fraction</th>
<th>% to landfill waste stream</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ferrous metals</td>
<td>70.84</td>
<td>64.62</td>
<td>68.98</td>
<td>62.83</td>
<td>1.27</td>
</tr>
<tr>
<td>Non-ferrous metals</td>
<td>10.82</td>
<td>11.65</td>
<td>0.31</td>
<td>0.34</td>
<td>9.59</td>
</tr>
<tr>
<td>Rubber</td>
<td>1.98</td>
<td>1.62</td>
<td>-</td>
<td>-</td>
<td>0.92</td>
</tr>
<tr>
<td>Glass</td>
<td>3.14</td>
<td>2.57</td>
<td>-</td>
<td>-</td>
<td>1.02</td>
</tr>
<tr>
<td>Other</td>
<td>3.15</td>
<td>2.58</td>
<td>-</td>
<td>-</td>
<td>1.24</td>
</tr>
<tr>
<td>Plastics</td>
<td>10.07</td>
<td>16.95</td>
<td>-</td>
<td>-</td>
<td>7.20</td>
</tr>
<tr>
<td><strong>Totals</strong></td>
<td><strong>100.00</strong></td>
<td><strong>69.30</strong></td>
<td><strong>63.16</strong></td>
<td></td>
<td><strong>21.25</strong></td>
</tr>
</tbody>
</table>

*Model rounding errors means 0.09% and 0.06% of the input waste are respectively unaccounted for in the 1995 and 2002 models.

**Table 12.5**, Predicted material segregation between 1995 and 2002 vehicles

**Figure 12.20**, Predicted changes in waste stream outputs for the Newport shredder between 1995 and 2002 vehicle compositions

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• Landfill waste increases (+1.11%) when processing the 2002 vehicle composition which sets an alarming precedent.

In summary this “to-be” modelling of the Newport shredder site has demonstrated the challenging shift in material segregation which will become more apparent in the next decade of vehicle processing. A reduction in the core ferrous content of ELVs and an increase in the use of plastics will further impede the achievement of the 2015 recycling and recovery target. The Newport study has presented a strong case for the introduction of additional automated separation technologies to further channel highly contaminated waste away from landfill and towards the additional processing provided by the density media separators. The post-fragmentation costing approaches implemented within this case-study could potentially be used to model these facility changes, but additional data collection and validation should be undertaken first.

12.6 Summary of Case Study Results

The case study chapter has successfully shown the implementation of the ELV costing framework in terms of pre and post-fragmentation activity costing, and has been used to analyse the “as-is” and “to-be” situation. The results of this analysis can be summarised thus:-

• Case study 1 has demonstrated the need to establish an awareness of the current reclamation costs that an end-of-life stakeholder incurs, to gauge their susceptibility to adopting sustainable change in light of their current legislative commitments. Albert Looms showed a healthy economic robustness and setup, making it an ideal candidate to accept operational reform.

• With this in mind case study 2 then considered a possible “to-be” scenario, namely pre-fragmentation vehicle dismantling for material recycling. Demonstrating that this course of action could potentially return Albert Looms a small operating profit if selective assemblies were targeted, but equally showed
the substantial cost burden that Looms would incur if the 2015 recycling target was to be achieved via manual dismantling.

- Case study 3 has demonstrated another “to-be” scenario, this time considering the application of the post-fragmentation costing techniques to predict the changes in waste stream routing expected around the 2015 recycling target. Altering the material composition entering a UK shredding site to reflect these changes has shown a considerable shift in expected material segregation, and has provided a strong case for additional investment to improve the Newport sites separation capabilities in the coming decade.
Chapter 13

Concluding Discussion

13.1 Introduction

This chapter brings together the main research issues highlighted within this thesis and discusses the conclusions that can be drawn from these issues. The first part of the chapter highlights the research contributions, while the latter part presents a discussion of the major conclusions that can be formulated under the broad headings of the research scope within Chapter 2.

13.2 Research Contributions

The research within this thesis has investigated the economics of the operations of the vehicle recovery sector within the UK. The major research contributions of these activities can be summarised as:-

i) Generation of methods and tools to assess the impact of the ELV directive legislation within the UK, and to support a more sustainable approach to the recovery of vehicles.

ii) A substantial program of data collection, which included; formal interviews, survey of ATFs, time-studies and vehicle teardown studies, to generate a significant amount of costing data not currently available but of paramount importance in understanding the economics of the vehicle recovery sector.

iii) Design and implementation of a costing framework to model the diverse range of ELV reclamation costs, which include both indirect and direct costs of pre and post-fragmentation activities.

iv) A mathematical approach for modelling the theoretical separation capabilities of post-fragmentation technologies to improve efficiency of waste stream routing.
Development of a decision support costing system that enables ELV stakeholders to understand the main economics that underpin their operations, and to support future investment in more sustainable vehicle recovery activities.

13.3 Concluding Discussions

The following sub-sections draw together and discuss the results of the main research activities outlined as part of the thesis scope.

13.3.1 A Review of Previous ELV Costing Work and Environmental Concerns of Vehicle Recovery

A comprehensive literature review has highlighted the significant amount of past research considering the economic impact of the ELV Directive on the current recovery sector. This previous research not only considered the direct investment required by end-of-life operators, but also the legislative costs of target attainment. A substantial quantity of this literature has investigated the idealistic view of direct manufacturer involvement in ELV reclamation activities, and the availability of upstream data to support downstream recovery. However, through this research it was identified that the implementation of the ELV directive within the UK had followed a different path than had been originally intended, with the economic responsibility of the directive having shifted from the vehicle manufacturers to the end-of-life recoverers (via contractual agreements). The research also identified that the implementation of the legislation had isolated the ELV recovery sector, and in doing so had stifled the potential and predicted end-of-life investment by manufacturers. This lack of synergy between manufacturers and recoverers has created a situation in which business survival and end-of-pipe economics, as opposed to long-term environmental benefits, are guiding vehicle processing decisions. This highlights the need to produce an economic decision support tool, such as the costing system investigated by this research, for end-of-life operators to support their value recovery activities independently of manufacturers assistance. Such tools enable the vehicle recovery sector to assess the economic feasibility of their own operations and provide
a level of transparency required to achieve a more sustainable approach to vehicle recovery.

In addition a comprehensive review of end-of-life environmental literature assisted in mapping the environmental prioritisation criteria specified by the waste management hierarchy onto the current upstream and downstream ELV recovery activities. This demonstrated the importance of manufacturer and supplier influence on end-of-life vehicle recovery, and also helped to identify suitable sustainable processing scenarios tested later within the thesis.

13.3.2 Development of an ELV Costing Framework

In the initial part of the research, it became apparent that there was not one all encompassing costing technique that was capable of modelling the various cost types identified within the ELV recovery sector. Furthermore, there were common instances of incomplete, missing and unavailable (confidential) data that significantly hindered the development of an holistic cost model for vehicle recovery. This necessitated both a comprehensive program of data collection, and the integration of a range of diverse costing techniques, which included the use of well established approaches, as well as the development of innovative new ones.

This novel framework for the first time gives the ELV recovery sector the ability to model their costs and to understand the economics of their operations, providing support for investment decisions from an end-of-life perspective. Furthermore, the economic insight that the framework provides can be uniquely tailored to the recovery operator under consideration.

13.3.3 The Cost Modelling of Pre-fragmentation Activities in Automotive Recovery

The research identified two main activity groups at the pre-fragmentation processing stage; those activities carried out for environmental considerations (vehicle de-pollution), and those carried out for monetary value (parts resale and material recycling). The mandatory requirements for de-pollution processing as part of the ELV directive has forced ATFs to typically invest in the region of £210,000 on
updating their equipment and resources. During the interviews and survey undertaken as part of this research it became apparent that ATFs felt there were unfairly targeted by the Environmental Agency in the UK to absorb such high investment without the financial assistance of the manufacturers. It was also clearly evident as to their reluctance to further invest in more environmentally friendly processes without an understanding of their economic implications.

During the investigation into the cost drivers that affect pre-fragmentation activities, it became clear that although the overall demand for second-hand parts is diminishing in the UK, a profitable part resale strategy still plays a critical role in the long-term viability of many ATFs. Furthermore, the results of the ATF survey clearly highlighted the inability to meet the 2015 ELV directive recycling targets through part resale and de-pollution activities. This directed the research to give a more thorough consideration for the dismantling of vehicles for material recycling. Pre-fragmentation vehicle dismantling for recycling is not a widely adopted practise within the UK, and as a result data with which to select a suitable costing approach was not available. Therefore, the research has undertaken one of the first comprehensive UK teardown studies, to generate destructive dismantling data and to assess the feasibility of UK facilities to adopt such material recycling practises. Through this research it was also observed that there is a strong industry resistance to adopting pre-fragmentation material removal based on current market prices and UK labour rates, but the author highlights its ability in fulfilling the 2015 recycling targets if post-fragmentation conformance can not be achieved.

13.3.4 The Cost Modelling of Post-fragmentation Activities in Automotive Recovery

The automated separation processes adopted within post-fragmentation facilities will be an integral part of the UK’s waste management strategy in achieving ELV recycling targets, and reducing the volume of waste that ends up in landfill. The volumes-of-scale combined with its historic effectiveness in mechanically separating valuable materials will ensure that post-fragmentation activities will have continued importance within ELV reclamation for years to come. Post-fragmentation was therefore highlighted in the research as a fundamentally important area in which an appreciation for end-of-life recovery costing was essential. Alarmingly, during the
industrial visits to post-fragmentation facilities it became apparent that there was a distinct lack of understanding as to the scale of the value-added processing that these front-line recycling industries were achieving. The research identified that a commonly adopted practise was to establish a rigid facility layout based on only a handful of core material types (e.g. steels and light irons), irrelevant of whether a particular waste stream contained an abundance or scarcity of these core materials. This research therefore developed a novel post-fragmentation simulation model, based on the theoretical separation capabilities of post-fragmentation technologies, to assess the value-added processing that can be achieved. This direct post-fragmentation costing approach has given an indication of end-of-life value for a range of waste streams currently not targeted for recovery (e.g. plastics and rubbers), and has generated a post-fragmentation modelling approach that could account for the variation in the separation capabilities from facility to facility. The implementation of this post-fragmentation costing approach with the ELV costing system has provided the foundation on which the financial viability of current facilities can be modelled and assessed. In addition the post-fragmentation model provides the ability to examine various “what-if” processing scenarios to maximise efficiency and value recovery. The author argues that by identifying end-of-life value and making the revenues associated with waste stream purification more transparent, it is possible to change the current post-fragmentation mentality to consider alternative processing routes that are sympathetic to the composition of the input waste stream, that are not just based on core material types.

13.3.5 The Realisation of ELV Cost Models to Support Sustainable ELV Recovery

The various cost modelling approaches highlighted within the ELV costing framework were brought together within a software ELV costing system. Given the complex nature and interrelationships identified between various costing approaches a software system had to be developed to automate the task. This established a fast and effective means of realising the ELV framework for a bespoke stakeholder, that can assess various “as-is” and “to-be” modelling scenarios that would otherwise require intensive data collection and laborious analysis if done manually.
One of the novel attributes of the ELV costing system is the ability to model indirect costs through a systematic ABC methodology. It is argued that the inclusion of such indirect costing considerations is of paramount importance to providing an accurate insight into the financial “hotspot” within an ELV operator. Furthermore, the diversity and the range of facilities, combined with the availability of data, provided significant challenges in the design and implementation of the ELV costing system. This necessitated the integration of a number of well established costing techniques with a range of innovative approaches developed by this research. The applicability of this ELV costing system has been briefly demonstrated through the case studies, as will be further discussed below.

13.3.6 Demonstration of Research Applicability Through Case Studies

To validate and demonstrate the results of the research two industrial end-of-life operators have been used, namely, Albert Looms-Derby (an ATF) and Sims-Newport (a shredding site). The applicability of the ELV costing system has be demonstrated both as a short-term operational tool (via an “as-is” model), and a long-term strategic decision support tool (via “to-be” model). The first case study provided a health check as to the economic viability of a typical ATF adopting operational change (via assessing its profitability), also highlighting its main economic strengths and weakness. This modelling assessed an end-of-life operator’s suitability to change and established a case to support the “to-be” model. The ATF case-study demonstrated good financial health, having recently invested in equipment to fulfil the ELV directive requirements, and made them an ideal candidate to consider the inclusion of more environmentally sound practises. The author argues that despite the profitability demonstrated through the Albert Looms case study this exemplifies the typical situation within many UK ATFs, in which extremely strong market prices have successfully absorbed the cost burdens of the ELV directive and hidden the necessity for significant process improvements.

The second case study took a distinctly different approach, and considered pre-fragmentation dismantling for material recycling. This is one of the scenarios currently considered by industry as a method of achieving 2015 targets, and argued by the author as a sustainable addition to the ATFs core competency. The results have
demonstrated the significant economic burden that pre-fragmentation material removal would place on UK ATFs if undertaken purely as a means of recycling target attainment. It has also considered the marginal profit that could be achieved if the ATF adopts pre-fragmentation material recycling as part of its current activities. Though these marginal profits currently do not make the case for extensive investment in pre-fragmentation material recycling, this is not to say that such activities will not become more favourable as plastic recycling markets become more established.

The third case study based on the activities of a typical post-fragmentation site, explored the implication of changes to a vehicle's material composition expected to be processed around the key target attainment of 2015. The “to-be” model of the post-fragmentation highlighted the challenges relating to material segregation which will become apparent in the next decade of vehicle processing, due to the continued shift away from the inclusion of ferrous metal towards plastics and non-ferrous. The author argues that based on the results of this case study, changes within the fundamental waste streams produced at the Newport facility provides a strong justification for the investment in further processing technologies to counter act these additional input compositional affects.

13.3.7 Constraints, Limitations and Assumptions

The research reported in this thesis has investigated a sector which has historically been undeveloped and outmoded in its attitudes towards operational improvement. This is further compounded by the distinct lack of data produced by, and available to this sector. The range of end-of-life costing techniques adopted by this research was therefore selected under these constraints. The limitations and assumptions of these techniques have been discussed in Chapters 7-10, and are summarised in the following sub-sections.

13.3.7.1 Constraints and Modelling Assumptions in the ELV Costing Framework

The intended aim of the ELV costing framework was to provide a set of costing tools which could support the main economic activities and value-added processes within
vehicle recovery. This framework was generated based on industrial visits, review of literature and the authors own experience of the core activities carried out by the recovery sector, these included; collection, de-pollution, dismantling, shredding and separation. Therefore, it should be acknowledged that deviation away from these core activities (e.g. end-of-life recondition of components at the ATF, incineration of waste at the shredder), that existed in the UK as isolated cases, were assumed to be beyond the scope of the research, and the lack of ability to model such ad hoc processes can be considered a limitation of the current framework. As the vehicle reclamation sector potentially diversifies its core competencies in the future, either due to business survival, risk mitigation or recycling target attainment, there will ultimately be a necessity to incorporate the additional direct processing costs of these activities into the existing framework. When this occurs it is assumed that an additional assessment would be needed to determine the main cost drivers affecting each of these new activities, before a suitable costing approach can be selected and integrated into the existing framework.

13.3.7.2 Limitations of Activity Based Costing and Uncertainty Modelling

The inclusion of ABC within the ELV costing framework highlighted the attribution problems associated with indirect costing, specifically the attribution of cost areas that were not associated with a particular resource. For example, the attribution of the labour costs of facility site managers whose contributions could not be clearly attributed to a specific activity highlights this attribution problem. Additional limitations associated with the modelling of indirect costs within the ELV framework were concerned with the limited availability of historic purchasing data on which to select suitable uncertainty modelling curves.

13.3.7.3 Limitations of Pre-fragmentation Costing

The limitations associated with the use of the generative-analytical approach for both de-pollution and parts removal costing are related to the availability of a large pool of end-of-life processing data. The response rate of the survey (completed by approximately 2% of the UK’s recovery capacity), used partly as a means of data collection to provide component removal times, and to generate costing data that was not as comprehensive as within other areas of the ELV costing framework.
Parametric regression analysis was utilised to develop vehicle dismantling equations within the same pre-fragmentation chapter. The statistical significance of the dismantling equations developed is documented within the thesis, along with the reliability of the catalogued parameter variables used within the analysis. Limitations of this cost modelling approach were highlighted as the distinct lack of upstream design parameters available within the IDIS database with which to relate the removal data obtained from the vehicle teardown study, and the need to have a more comprehensive statistical data pool available.

13.3.7.4 Limitations of Post-fragmentation Costing

The consideration of the value added processing that post-fragmentation facilities have on a commuted waste stream and the development of a modelling procedure to account for separation inefficiencies of various automated technologies, also have a number of modelling limitations associated with them. The post-fragmentation material interactions are described using a two dimensional interaction matrix. The research has highlighted the limitations of using this type of matrix in describing only basic material contaminations, and recognises the need to further expand on this modelling approach when post-fragmentation sampling becomes more wide-spread.

The final limitation of the post-fragmentation modelling approach is routed in the industries inability to provide detailed and comprehensive facility benchmarking data. Within the research the model is calibrated using data from an industrial study conducted by the Department of Trade and Industry, and simulates the separation effects seen within industry. Ideally, the modelling approach is required to be validated using an extensive facility specific data collection exercise, but this is currently restricted due to the willingness and readiness of post-fragmentation facilities to catalogue this data.

13.3.7.5 Assumptions Influencing the Generality of the Modelling

The case studies have provided an opportunity to demonstrate the intended application of the ELV costing framework within two UK recovery operators (Albert Looms and Sims Metal). An issue peripheral to this modelling is a consideration as to the
comparability of these stakeholders to others within their own sector, and the
generality of the results discussed. Given that the investment and reform undertaken
by different recovery sector operators has been uniquely different from facility to
facility, the recommendation to adopt additional sustainable processing activities is
very much dependent on a facility’s ability to demonstrate financial robustness to
operational change.

The detailed recommendations made within the case-studies for both Albert Looms
and Sims metal, are very much operator and time frame specific. Therefore, factors
such as the variability in labour rates both within the UK and EU and the variation in
material market prices will significantly impact the results and recommendations for
further investment. Hence, it is important to make the distinction between the
framework being used as a set of costing approaches and methods that can be
generically applied to the vehicle recovery sector, and the highly bespoke results,
analysis and recommendations generated as part of each case study.

13.3.8 The Vision for The Future of ELV Recovery in the UK

It is argued that the “free-market” solution and “zero-cost” contracts which have been
established by the UK’s vehicle manufacturers to deal with their requirements of the
ELV directive has been a short-term “knee jerk” reaction to stop the financial liability
of ELV recovery costs appearing on their balance sheets. The disjointed relationships
this situation has created between vehicle manufacturers and recovers may seem
workable under current market conditions, but a moderate reduction in the scrap
material markets will expose an industry that is ill-equipped to deal with the pressures
of European legislation on its own. If the recovery sector is unable to mitigate this risk
by either improving its operational effectiveness or diversifying their activities and
markets, it may result in a number of operators going under. Resulting in the alarming
situation of further losses in UK recovery capacity, at a time when the number of ELV
retirements is increasing.

The research has also highlighted the difficulties in achieving the 2015 target, and
therefore it is argued that this will lead to a situation in which either; the target must
be rectified to reflect predicted recycling capabilities in 2015, the investment in UK
post-fragmentation technologies needs to be significantly increased, or finally pre-fragmentation removal for recycling needs to be further considered.

By far one of the greatest challenges the ELV recovery sector will face in the coming years, is the recycling difficulties generated by the material compositional changes introduced by manufacturers to decrease vehicle emissions and improve performance. The trade-off between light-weighting versus recyclability will be a key determinant in the achievement of the ELV directives targets, and will become a key debate in the next decade of vehicle manufacturing.
Chapter 14

Conclusions and Further Work

14.1 Introduction

This chapter draws together the conclusions of the research presented within this thesis, and proposes possible directions for further extension of the work.

14.2 Conclusions from the Research

The conclusions that can be formulated from this research are as follows:-

i) The increasing number of passenger vehicles on the UK’s roads, combined with their relatively short lifespan, clearly indicates a significant rise in the expected/predicted number of future ELVs. This escalation in ELV retirements is being undertaken during a time of increasing environmental regulation, which clearly highlights the need for further investment by both the manufacturers and recovery to improve the design of the product and the effectiveness of end-of-life recovery processes.

ii) The ELV directive was aimed at extending the vehicle manufacturers responsibility to encompass end-of-life considerations and make them directly involved in the recovery of their products. However, the “free-market” solutions which has been instigated by the automotive manufacturers, in which the responsibility for the recovery of vehicles has been shifted onto the ELV recovery sector, has widened the gap between manufacturers and recoverers.

iii) The “free-market” solution and “zero-cost contracts” has resulted in the vehicle manufacturers having no direct economic interest or influence on ELV recovery. Hence, the current cost burdens and increased environmental expectation required on behalf of the ELV operators clearly points to the need
to obtain better value recovery to mitigate some of the financial risk. This research has shown that this situation will not facilitate end-of-life investment in additional technologies nor assist in the consideration of more sustainable processing scenarios, unless recoverers can gain an insight into the economical implications of their activities.

iv) The "free-market" solution has not only shifted the financial burden of directive conformance onto the vehicle recovery sector, but in doing so has also restricted the flow of information from the manufacturer to the recoverer. This research has therefore highlighted that such information is of paramount importance to provide such economic transparency for ELV recoverers. This research has therefore adopted a novel approach to integrate a number of existing and new costing techniques within an ELV costing framework, which provides a powerful approach to develop detailed cost models within a sector that has historically not understood its costs or catalogued its data.

v) The diverse range of cost and revenues within the ELV recovery, combined with the range of operational and strategic decisions involved in maintaining sustained profitability, highlights the need to design and implement a flexible decision support costing tool that enables ELV operators to identify value recovery "hot-spots" in both pre and post-fragmentation activities. The ELV costing system developed as part of this research through its ability to model both "as-is" and "to-be" scenarios provides powerful support for these operational and strategic decisions.

vi) The results from the case studies undertaken as part of this research has indicated that the predicted cost of ELV directive implementation within the UK has been offset by the strength of ever-growing demand and consequently usual high value of scrap metal. Furthermore, these case studies have shown that although most ELV recoverers are currently profitable due this strong demand for scrap metal, significant improvement in their processes and value recovery is possible through strategic investment. Such strategic investment in process improvement and expansion of recycling activities should be
considered in light of future fluctuations in material markets and increasing costs of attaining higher recycling and recovery targets by 2015.

vii) The results from the case studies has also highlighted the significant cost per vehicle to achieve the 2015 recycling target based on current vehicle recovery activities. If these targets remain unchanged, either the efficiency of current processes need to be significantly improved, or the currently less economically viable approach of pre-fragmentation material recycling investigated by this research should be considered.

viii) It should be noted that although the current transposition of the ELV directive in the UK has placed the responsibility of fulfilling the legislative requirements on the ELV recoverers, ultimately the EU producer responsibility dictates that the vehicle manufacturers are financially responsible for vehicle take-back and recycling. Therefore, the author is of the opinion that automotive manufacturers must take a more pro-active and direct involvement in ELV recovery to promote an environmentally friendly approach to vehicle recycling that ensures the long-term sustainability and survival of the ELV recovery sector.

14.3 Further Work

The author recognises the additional areas of further work arising from the research, a number of which are described below:

14.3.1 Environmental Assessment of Vehicle Recovery Activities

This research has focused on the costing implications of ELV recovery. A more holistic approach to considering sustainable recovery would be to also consider the environmental impact of these activities in addition to the cost. Various tools and techniques have been developed for such environmental appraisals, which include life-cycle assessment (LCA), eco-indicators, material input per service unit and accumulative energy demand. The author proposes a future research direction to be
the extension of the ELV decision support tool to assess the detailed environmental implications of recycling activities.

14.3.2 Industrial Validation of Post-fragmentation Modelling

The research has developed a theoretical simulation of material separation processes during post-fragmentation activities. Given the lack of extensive post-fragmentation data currently collected from UK facilities, this was calibrated using data generated through a governmental benchmarking study. The author is of the opinion that this work has significant potential to be further exploited through a comprehensive program of industrial validation exercises. Once validated for a particular facility it would generate valuable knowledge as to the best method of utilising separation technologies to most effectively mine the waste streams processed.

14.3.3 Point of Disassembly Decision Support

Currently due to the level of customisation within vehicles there is significant variation between different makes and models in terms of sub-assemblies for resale and plastics for recycling. One of the greatest barriers this research has identified is the distinct lack of operational support tools and available information to assist with the micro-level decisions that must be made between vehicles. Hence, a further research direction would be to investigate the application of the costing approaches discussed within this thesis to most effectively integrate them in a facility's current operations.
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Appendix I

DATA COLLECTION

Introduction

This appendix contains the data collected from the ATF survey, de-pollution time studies, vehicle tear-down and Albert Looms resource attribution map.

A1.1 ATF Survey (postal version)
A1.2 ATF Survey Results: Issues and Trends within the Vehicle Recovery Sector
A1.3 De-pollution Time Study Data
A1.4 Vehicle Dismantling Study Data
A1.5 Albert Looms Resource Attribution Map
This survey has been created as part of a 2 year nationally funded research project undertaken by the sustainable manufacturing research group at Loughborough University. The aim of the project is to gain an understanding as to the economics of the vehicle recovery chain within the UK.

By better understanding how ATF’s operate the research group hopes to develop tools that maximise the available revenues from returning end-of-life vehicles to greater benefit UK ATFs. For further details of the project please see www.lboro.ac.uk/smart

To complete this work and give the project a stronger industrial basis we would be very grateful if you could spend 5 minutes giving us the benefit of your experience. The survey has been divided into 5 sections, all of which might not be undertaken at your facility, but please try and answer as much as you can.

Thank you in advance for your cooperation and help.

Disclaimer: The responses of this survey are confidential and are intended for research purposes only. All responses will be held on a private secure server in accordance with data protection act, and publication of any obtained results will be confined to academic publication. All opinions held there within are assumed to be that of the respondent(s), and may not be representative of the organisations you are associated with.

Your Background

Name:

Position:

Company (ATF name):

Contact email:

or Contact telephone number:

Company background

1.) What activities are included within your companies operation? (Please tick)

- Vehicle collection: □
- De-pollution: □
- Spare parts resale: □
- Car bailing: □
- Part reconditioning: □
- Plastic recycling: □

Other: ........................................................................................................................................
2.) How many workers are employed at your facility? (Please tick)

- 1-5 employees: □
- 16-20 employees: □
- 6-10 employees: □
- 21-25 employees: □
- 11-15 employees: □
- 26+ employees: □

3.) What is the average number of End-of-life vehicles received per day? ........................................

4.) What is the approximate ratio of premature to natural ELV’s that you deal with?

- Natural: □ %
- Premature: □ %

De-pollution

5.) How many de-pollution rigs do you operate at your facility? ............................................................

6.) What is the typical time for you to completely de-pollute an ELV? .............................................. minutes

7.) How important would you consider the following factors to be on the time taken to de-pollute an end-of-life vehicle? (Please circle a number that best describes the statement)

<table>
<thead>
<tr>
<th>Importance has been rated from 5-1, 5 being “Of very great importance” and 1 being “Hard to say”)</th>
</tr>
</thead>
<tbody>
<tr>
<td>The make &amp; model of vehicle returned:</td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td>Whether the vehicle is a natural or premature ELV:</td>
</tr>
<tr>
<td>Having the DTI recommended de-pollution equipment:</td>
</tr>
<tr>
<td>Strictness of the Environmental Agency:</td>
</tr>
<tr>
<td>The internal condition of the vehicle:</td>
</tr>
<tr>
<td>The external condition of the vehicle:</td>
</tr>
</tbody>
</table>
Appendix I

Parts removal and resale

8.) How important would you consider the following factors to be on the time taken to remove a part? (Please circle a number that best describes the statement)

*Importance has been rated from 5-1, 5 being “Of very great importance” and 1 being “Hard to say”*

<table>
<thead>
<tr>
<th>Factor</th>
<th>Of very great importance</th>
<th>Of great importance</th>
<th>Of some importance</th>
<th>Makes no difference</th>
<th>Hard to say</th>
</tr>
</thead>
<tbody>
<tr>
<td>The ease of access to the part:</td>
<td>5</td>
<td>4</td>
<td>3</td>
<td>2</td>
<td>1</td>
</tr>
<tr>
<td>The need to have a specific tools:</td>
<td>5</td>
<td>4</td>
<td>3</td>
<td>2</td>
<td>1</td>
</tr>
<tr>
<td>The need to have dismantling information at hand:</td>
<td>5</td>
<td>4</td>
<td>3</td>
<td>2</td>
<td>1</td>
</tr>
<tr>
<td>The number of attachment points for the part:</td>
<td>5</td>
<td>4</td>
<td>3</td>
<td>2</td>
<td>1</td>
</tr>
<tr>
<td>The type of mechanical fastenings used:</td>
<td>5</td>
<td>4</td>
<td>3</td>
<td>2</td>
<td>1</td>
</tr>
<tr>
<td>The force required to unfasten bolts:</td>
<td>5</td>
<td>4</td>
<td>3</td>
<td>2</td>
<td>1</td>
</tr>
<tr>
<td>Whether adhesives have been used:</td>
<td>5</td>
<td>4</td>
<td>3</td>
<td>2</td>
<td>1</td>
</tr>
<tr>
<td>The condition of the mechanical fasteners holding the part:</td>
<td>5</td>
<td>4</td>
<td>3</td>
<td>2</td>
<td>1</td>
</tr>
<tr>
<td>The type of material the part is made from:</td>
<td>5</td>
<td>4</td>
<td>3</td>
<td>2</td>
<td>1</td>
</tr>
</tbody>
</table>

9.) Listed below are 10 components typically removed for part-resale. Can you provide an approximate estimate of the time taken to remove each of the items:

<table>
<thead>
<tr>
<th>Component</th>
<th>Time (Hours:Minutes)</th>
<th>Never removed (Tick)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 Engine</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2 Gearbox</td>
<td></td>
<td></td>
</tr>
<tr>
<td>3 Carburettor</td>
<td></td>
<td></td>
</tr>
<tr>
<td>4 Alternator</td>
<td></td>
<td></td>
</tr>
<tr>
<td>5 Starter motor</td>
<td></td>
<td></td>
</tr>
<tr>
<td>6 Distributor</td>
<td></td>
<td></td>
</tr>
<tr>
<td>7 Head lamp assemblies</td>
<td></td>
<td></td>
</tr>
<tr>
<td>8 Quarter glass</td>
<td></td>
<td></td>
</tr>
<tr>
<td>9 Radiator</td>
<td></td>
<td></td>
</tr>
<tr>
<td>10 Wing mirror</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
10.) What would you consider to be the top 3 most widely removed components?

1. .................................................................
2. .................................................................
3. .................................................................

11.) How do your mechanics decide which parts to remove from an end-of-life vehicle? (Please tick as many as required)

- Through experience of what sells: 
- They receive specific requests from customers: 
- Via a computerised inventory system: 
- Adhoc (they remove what they feel like): 
- Other: .................................................................

12.) Do you export removed parts to foreign countries? ........................................... Yes.../... No.....

12b.) If “yes”.... is this more profitable than the sale of the components within the UK? ....Yes.../... No.....

13.) Do you use any of the following information sources to work out how to remove parts?

- Computer based Technical Information Systems (e.g. Ford’s ITS): 
- Vehicle manufacturer’s reference manuals: 
- Third party manuals (e.g. Haines): 
- Web-based discussion forums: 
- Other: .................................................................
14.) In your opinion what do you think of the longer term future holds for the vehicle spares market within the UK? (Please tick one)

- It will grow
- It will remain the same
- It will decline
- Don't know

Comments:.................................................................................................................................

Markets and future revenues

15.) Which of the following statements best describes your organisations dependency on hulk sales to the shredders? (Please tick one)

- "They provide a substantial part of our revenues" .........................
- "They provide only part of our revenues, but are still important"
- "They contribute very little to our revenues, and we have other more substantial revenue streams"
- "We don't view hulk sales as important"
- Don't know

16.) What are your opinions on the long term (next 10 years) stability of the scrap steel prices within the UK?

17.) Have you ever considered recycling or taking plastics out of the End-of-life vehicles?  
...Yes.../...No......
17b.) Would you consider plastics removal to be particularly profitable? .......Yes.../...No......

18.) How much do you think your facility has spent to date on conforming to the ELV directive?

Any other comments: (relating to the survey, or problems your organisation is facing with the ELV directive)

Thank you for your help

Please return to ....
G. Coates
Wolfson School of Mechanical and Manufacturing Engineering
Loughborough University
Loughborough,
Leicestershire,
LE11 3TU.

Any further questions or comments regarding our research work please contact:
Gareth Coates
Tel: 01509-227683
Email: G.Coates@lboro.ac.uk
Appendix I

An Authorised Treatment Facility Survey: Issues and trends within the Vehicle Recovery Sector

A report produced by the centre for Sustainable Manufacturing and Reuse/recycling Technologies (SMART),

Wolfson school of Mechanical and Manufacturing Engineering,

Loughborough University.

July 2006

Gareth Coates
SMART RESEARCH GROUP
1.0 Background to ATF survey

The following survey was created and distributed as part of the IMCRC project "Cost Oriented Approaches to the Design and Recovery of End-of-Life Vehicles", to facilitate an understanding of the current issues for the ELV recovery sector within the United Kingdom. Given the reform the industry is currently undergoing and the establishment of Manufacturer collection contracts, the context of any research must be positioned to take into account the industries willingness and ability to change. The survey is therefore a combination of qualitative questions and specific data collection tables.

The main aim of this survey is to generate industrial data to facilitate an economic understanding of ELV recovery, while gaining an appreciation for some of the problems and issues the industry is facing.

2.0 Survey Distribution

An online question was created and hosted on the SMART website during the period of December 2004 and March 2005. The survey was distributed to 220 Authorised Treatment Facilities (ATFs) by email, and fifty surveys by post. Additional distribution was provided by the British Vehicle Salvage Federation (BVSF) who forwarded the website link on to an undisclosed number of members.

3.0 Demographic of respondents

Of the 270 facilities contacted, twenty-four replied with a complete questionnaire, creating a response rate of around 8.9%. There are currently (May 06) 1175 facilities within the UK that are listed with the Environmental Agency as being legally allowed to collect and de-pollute end-of-life vehicles, highlighting the fact that the respondents account for approximately 2% of the total industry.

Given the variation that exists between the processing capabilities of each ATF, the initial part of the survey considered questions regarding their general operation. This was done, not only to provide some way of classifying each facility, but also served as a means of establishing a "typical" (or average) ATF.

Of the facilities responding to the ATF survey all provided vehicle collection and de-pollution operations, slightly less provided parts resale and vehicle compaction facilities, but only two considered any type of part reconditioning or plastics recycling (see figure 1).

In terms of the number of employees working at the facilities surveyed the distribution tended to be at the extremes. The majority of ATFs either had only a small handful of staff (1-10 employees), or went completely the other way and had in excess of 26 (see figure 2).

The average number of ELVs processed at each facility each day was twenty-one, of which the average ratio of natural to premature was 58:42.
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Fig. 1: Respondents core activities

Fig. 2: Employee distribution
Comments: Statistics relating to the typical activities carried out by ATFs correspond well with the author’s experience of facilities visited during the initial stages of the project. With de-pollution now been a legal requirement, and high scrap steel prices absorbing the costs of vehicle collection, it is no surprise these were offered by all the facilities surveyed. Part resale and vehicle bailing are again well established activities, based on historic industry drivers. Not so well established is the recovery and recycling of plastics. This is supported by the current industrial view that there is little infrastructure in place to distinguish and make use of recovered engineering plastics (pre-fragmentation), and even less economic incentive given the high dismantling costs. Part reconditioning is also another area not very well established within the vehicle recovery sector; the lack of relationships between the vehicle supply chain with the main end-of-life operators would support this conclusion.
4.0 Vehicle De-pollution

The de-pollution process has now become an integral part of an ATFs operations, given the legal requirements now placed on the industry. This part of the survey was to gain feedback as to how successful the implementation of this legislation has been, and how much capacity and time the processes take up.

The average de-pollution time for all the facilities surveyed is 23 minutes and 32 seconds, with 2.2 de-pollution rigs per facility (see figure 3). In terms of assessing the factors that effect the time taken to de-pollute an end-of-life vehicle the most influential were identified as the strictness of the Environmental Agency (EA) in policing the new legislation, and the use of Department Trade and Industry (DTI) recommended equipment to de-pollute the vehicle (see figure 4).

**Comments:** As with the employee distribution graph, the number of rigs per facility follows the same trend. With the majority of smaller operators only requiring 1-2 rigs to satisfy their requirements, while the bigger operators with 26+ employees and greater throughput tend to utilise 5+ rigs.
Factors effecting the time taken to de-pollute vehicles were mainly identified as the EA’s ability to enforce the de-pollution requirements, and a facilities use of the recommended equipment to carry out the work. Additional comments that supported this industry view are listed below:

- "Needs to be a level playing field for ATF sites"

- "Legislation is one of my concerns as you setup for new laws and they come back with something else to do, but without help financially to cope with the extra work load."
5.0 Component removal

The parts resale business has historically been the main core competency that many ATFs have focused on to sustain their businesses during turbulent times within scrap materials markets. This part of the survey has been developed to gain a snapshot of the currently industrial situation for the parts resale market, and to generate certain sub-assembly removal times for the most widely removed components.

The survey data would suggest that the most widely removed parts are headlamps, engines, gearboxes and door-mirrors (see figure 5). In terms of removal times for these sub-assemblies Table 1 highlights the costs, revenues and target attainment achievable.

![Most widely removed components](image)

Fig. 5: Most common components removed from ELVs

<table>
<thead>
<tr>
<th>Component</th>
<th>Average removal time</th>
<th>Labour cost (£)*</th>
<th>Resale price for a Premature ELV (£)**</th>
<th>Resale price for a Natural ELV (£)**</th>
<th>Improvement in recycling and reuse target***</th>
</tr>
</thead>
<tbody>
<tr>
<td>Engine</td>
<td>1 h 11 min</td>
<td>12.35</td>
<td>607</td>
<td>192</td>
<td>0.00%</td>
</tr>
<tr>
<td>Gearbox</td>
<td>52 min</td>
<td>9.04</td>
<td>299</td>
<td>163</td>
<td>0.00%</td>
</tr>
<tr>
<td>Alternator</td>
<td>15 min</td>
<td>2.61</td>
<td>60</td>
<td>36</td>
<td>0.00%</td>
</tr>
<tr>
<td>Starter motor</td>
<td>17 min</td>
<td>2.96</td>
<td>56</td>
<td>44</td>
<td>0.00%</td>
</tr>
<tr>
<td>Distributor</td>
<td>10 min</td>
<td>1.74</td>
<td>56</td>
<td>33</td>
<td>0.00%</td>
</tr>
<tr>
<td>Head-lamp assembly</td>
<td>12 min</td>
<td>2.09</td>
<td>37</td>
<td>19</td>
<td>0.17%</td>
</tr>
<tr>
<td>Quarter glass</td>
<td>14 min</td>
<td>2.43</td>
<td>37</td>
<td>33</td>
<td>0.64%</td>
</tr>
<tr>
<td>Radiator</td>
<td>16 min</td>
<td>2.78</td>
<td>54</td>
<td>30</td>
<td>0.00%</td>
</tr>
<tr>
<td>Wing mirror</td>
<td>9 min</td>
<td>1.56</td>
<td>43</td>
<td>27</td>
<td>0.13%</td>
</tr>
<tr>
<td><strong>Totals</strong></td>
<td><strong>3 hours 37 minutes</strong></td>
<td><strong>£37.56</strong></td>
<td><strong>£1249</strong></td>
<td><strong>£577</strong></td>
<td><strong>0.94%</strong></td>
</tr>
</tbody>
</table>

Table 1: Average sub-assembly removal times and cost/revenues generated from survey data.

* Based on £ 21,700 per annum mechanic / technicians wage working a 40 hour week.
Factors that contribute to the time taken to remove a component from a vehicle were highlighted as the ability of mechanics to gain access to the component, the condition of the fasteners holding the component in place and the use of adhesives (see figure 6).

The final questions posed within the component removal part of the survey attempted to quiz the participants as to their opinions of the longer term stability of the parts resale market within the UK. Interestingly two thirds of respondents said that in their opinion they believed the industry would decline in the foreseeable future (see figure 7). This is supported by the statistic that 50% of the ATFs surveyed are now selling their components abroad, despite the lack of profitability of foreign markets compared to those in the UK (see figure 8).
Appendix I

What is your opinion as to long-term future of the component removal market?

- It will grow: 8%
- Don't know: 21%
- Remain the same: 4%
- It will decline: 67%

Fig. 7: Respondent's opinion as the long-term future of the parts resale market

Components for export

- Do you export vehicle components? 50%
- Yes: 50%
- No: 50%

- Would you consider this to be more profitable than UK sales? 36%
- Yes: 14%
- No: 50%

Fig. 8: Percentage of respondents that export components and those that find it more profitable than UK sales

Comments: The survey results highlight the good value versus effort returns that component removal and resale can have for an ATF. The resale values of “premature” ELVs components are significantly higher than that achievable from a “natural” ELVs components, due to an abundance of middle-aged vehicles still on the road compared to those due for retirement.

The survey data would suggest that component removal cannot make substantial headway into improving the recycling and reuse target laid down by the ELV directive, as the majority of removed sub-assemblies are metallic and currently
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counted within the assumed recycling fraction processed during post-fragmentation (74.48%). Only components composed of plastics, rubbers or glass can count towards the target, and only the headlamps, door mirrors and tyres were listed within the top 10 of most commonly removed assemblies that fulfil this criterion.

Results highlighting the factors effecting the time taken to remove components from a vehicle correspond well to the research group’s own experiences of sub-assembly removal. A recent dismantling study run at a local ATF assessing the removal of plastic components also highlighted fastener degradation and fixture accessibility as being critical factors relating to removal time.

The industries scepticism as to the continued growth of the parts resale industry stems from year-on-year decreases in the achievable revenue from component resale. Possible reasons for this is the low-cost of borrowing to finance newer vehicles, an increase in component complexity due to integrated electronics (dissuading amateur hobbyists), and an increased number of component variants between different makes and models. Also, the knock on effects of the Block Exception Regulations (BER) implemented by the European Commission in 2003 are having an adverse effect on second-hand part resale. (The BER allows spare parts manufacturers to deal directly with the end-user, opening their markets dramatically and avoiding the costly mark-up prices associated with “own-brand” spares.)
6.0 Hulk sales and investment costs

The high price for scrap steel that many ATFs are receiving for their hulks, have gone someway to offsetting the investment and labour costs required in meeting the depollution requirements of the ELV directive, and have formed the basis for negotiations during the establishment of many vehicle collection contracts. The ability of the scrap steel market to remain high and to continue to support the recovery industry will be an important factor in whether both the 2006 and 2015 recycling and recovery targets are achieved. It is therefore essential to gauge the industries dependence on this market driver and their opinion as to its long-term stability.

![ATF dependency on hulk sales](image)

88% of the facilities survey would describe hulk sales as important to their operating revenue (see figure 9). Opinion as to the ability of the scrap steel prices to remain high, generally paints an optimistic picture. The comments below are taken from a cross-section of respondents:

- "Very volatile but would expect it to remain good."
- "Stable for the next 10 years"
- "Because of the expansion of new vehicle sales and the emphasis on recycling I think there will always be a market for scrap steel."
- "Scrap prices have allowed fluctuation; I would like to think that there could be some stability, but history does seem to repeat itself."
- "They will fall."

Those comments that make specific reference to the potential problem of a scrap steel market price crash are given below:
- "Ask the men in China. It is a demand market. If there is a demand the price is easily controlled. If there is no demand there will be a big problem!"

- "At some point in that period [10 years] prices will decline, just as they have before and some vehicle manufacturers monies will be needed to fund recycling of ELV's. Of course it can just as easily go back up!"

- "[Hulk prices received will] depending on the costs to recycle the vehicles and who will pay I.E. Customer/Manufacturer/Metal industry."

Given the potential impact of a scrap steel market crash on the vehicle recovery sector, suggestions have arisen that ATFs could potentially diversify their operations to focus on other core competencies. One such suggestion is the recovery of plastics by ATFs pre-fragmentation. Advantages of this approach is that recovery of these components can count towards the recycling target, and if a suitable price can be obtained for the engineering polymers some of the ATFs risk can be mitigated. One of the strongest hurdles in achieving this is a widely held perception within the industry that the cost of plastics recovery is uneconomical (see figure 10). Only 64% of respondents had ever considered removing plastics, but 85% of that fraction believes plastic recovery not to be economically viable.

The final question in the ATF survey was in regard to the estimated costs each facility believes they have invested to date in bringing their operation up to scratch for the ELV directive. The distribution of these costs is quite varied, with some spending as little as a few thousand pounds to others that have invested in excess of 1 million. The average cost of all the facilities surveyed was £217,000.

**Fig 10:** The percentage of respondents that have considered removing plastics, and those that believe it is currently profitable
Comments: This section has highlighted some interesting perceptions within the vehicle recovery sector regarding key market drivers. The revenue generated from vehicle hulk sale is very much an integral part of an ATFs income. Some ATFs have mentioned the potential problems the industry could face if this situation changes, but many are upbeat as the longer-term stability of the scrap steel market. The proliferation of ATFs signing up to Manufacturer collection networks, to ensure a "reasonable level" of capacity for the future would support this industry optimism. The only real test will come if scrap metal prices do fall, and the recovery sector is required to absorb the cost of de-pollution and hope a floundering parts resale market is enough to support them through difficult times. This is when the meaning of producer responsibility will really hit home for the vehicle manufacturers and Government legislators, and the inadequacies of the "free-market solution" that has been transposed within the UK will become apparent.

The ability of the recovery industry to diversify its operations away from its traditional core competency into plastics recovery or new recycling technologies could potentially mitigate some of the risk that ATFs are been subjected to, but the economic viability of these practices still needs to be proven. One of the strongest barriers to ATFs moving their operations more towards plastic recovery is the inability of the plastic recycling industry to distinguish between household and automotive plastics. Automotive polymers contain a range of additives and fillers that provide beneficial material characteristics for recycled products, yet little or no distinction is made in value. The only solution to this problem is to develop fast, cheap and accurate testing equipment that can be used on EOL plastics to produce an accurate material specification. Only then can a plastics true value be determined, a realistic price be paid for it, and a possible means of offsetting the labour costs incurred through dismantling. This tightening of material specification standards would not only give the recycling industry more validity with product manufacturers but also help promote more sustainable closed-loop application of material.
7.0 Acknowledgements

On behalf the centre for Sustainable Manufacturing and Reuse/Recycling Technologies and the IMCRC project team we would like to thank everybody that took the time to complete the survey, and have found your comments very interesting and enlightening. Also thank to Paul Owen from Rozone Ltd for his assistance, and Alan Greenouff from the BVSF.

8.0 Further information

For information regarding this or any other study work carried out within this project please contact:

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Email: G.Coates@lboro.ac.uk
Tel: 01509-227683
Fax: 01509-227648
www.lboro.ac.uk/smart

9.0 References


<table>
<thead>
<tr>
<th>Operation</th>
<th>Time (Minutes)</th>
<th>Labour cost (£)</th>
<th>Equipment</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>Collection and placement</td>
<td>1 min</td>
<td>£0.15</td>
<td>Forklift</td>
<td>The vehicle is removed from the sack using the forklift and placed below the elevated gantry</td>
</tr>
<tr>
<td>Bonnet popped and battery removed</td>
<td>1 min</td>
<td>£0.15</td>
<td>Spanner</td>
<td>No problem with bonnet opening. Removal quick.</td>
</tr>
<tr>
<td>Caps removed off the fluid containers</td>
<td>20 secs</td>
<td>£0.05</td>
<td>Manual</td>
<td>The caps are removed so a vacuum doesn’t form</td>
</tr>
<tr>
<td>Setup coolant drain</td>
<td>15 secs</td>
<td>£0.04</td>
<td>De-pollution rig</td>
<td>Each vacuum hose is pushed into the required pipe. Vacuum left to suck. Can be done when the vehicle is up in the air, but preferred when on the ground</td>
</tr>
<tr>
<td>Drain coolant</td>
<td>1 min</td>
<td>-</td>
<td>De-pollution rig</td>
<td>-</td>
</tr>
<tr>
<td>Setup washer fluid drain</td>
<td>15 secs</td>
<td>£0.04</td>
<td>De-pollution rig</td>
<td>Vacuum hose placed in the screen wash tank when the vehicle is on the ground.</td>
</tr>
<tr>
<td>Drain washer fluid</td>
<td>1 min</td>
<td>-</td>
<td>De-pollution rig</td>
<td>-</td>
</tr>
<tr>
<td>Setup brake fluid drain</td>
<td>15 secs</td>
<td>£0.04</td>
<td>De-pollution rig</td>
<td>Bleed nipples attached to remove</td>
</tr>
<tr>
<td>Drain brake fluid</td>
<td>1 min</td>
<td>£0.15</td>
<td>De-pollution rig</td>
<td>-</td>
</tr>
<tr>
<td>Oil filter removal</td>
<td>-</td>
<td>-</td>
<td>Spanner, Oil filter removal harness</td>
<td>Equipment is required to remove the fluid from the reservoir and the connecting tube.</td>
</tr>
<tr>
<td>Power steering oil</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Removal of R12 and R134a. Specialist tools required.</td>
</tr>
<tr>
<td>AC drainage</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Air bag removal</td>
<td>-</td>
<td>Air bag deployment tool</td>
<td>Specialist equipment is required to deploy the various bags.</td>
<td></td>
</tr>
<tr>
<td>Wheel bolts removed</td>
<td>1 min 5</td>
<td>£0.16</td>
<td>Air ratchet, compressor pump</td>
<td>Approximately 3 seconds/bolt (12 secs per wheel) Gun picked up / put down. Bolts pulled out, thrown in car. Time to walk around vehicle</td>
</tr>
<tr>
<td>Wheel balancing weights removed</td>
<td>-</td>
<td>Screwdriver</td>
<td>-</td>
<td>Wheel balance weights are attached to vehicle via adhesive bonds. Flat screwdriver can be used to lever the weights off.</td>
</tr>
<tr>
<td>Vehicle picked up and placed on rig</td>
<td>1 min 10</td>
<td>£0.18</td>
<td>Forklift</td>
<td>The forklift moves under the vehicle and shakes the it up and down so the tyres fall off. Lifted and placed on top of rig.</td>
</tr>
<tr>
<td>Sump opened</td>
<td>10 secs</td>
<td>£0.03</td>
<td>Alan key, normally hand drill</td>
<td>The sump is opened very quicking and the catchment drum placed underneath it.</td>
</tr>
<tr>
<td>Engine oil started to drain</td>
<td>4 mins</td>
<td>-</td>
<td>De-pollution rig</td>
<td>-</td>
</tr>
<tr>
<td>Petrol tank is drilled</td>
<td>10 secs</td>
<td>£0.03</td>
<td>Special tool, or standard drill</td>
<td>10 secs to drill the tank.</td>
</tr>
<tr>
<td>Petrol checked</td>
<td>5 secs</td>
<td>£0.01</td>
<td>Pump equipment</td>
<td>-</td>
</tr>
<tr>
<td>Pumping petrol out</td>
<td>6 mins 35 secs</td>
<td>£1.00</td>
<td>De-pollution rig</td>
<td>Fuel pumped out. Plug placed in tank.</td>
</tr>
<tr>
<td>Gearbox drilled</td>
<td>-</td>
<td>-</td>
<td>Special tool, or standard drill</td>
<td>Drilled open, if the vehicle has one.</td>
</tr>
<tr>
<td>Gearbox drained</td>
<td>-</td>
<td>-</td>
<td>De-pollution rig</td>
<td>-</td>
</tr>
<tr>
<td>Rear differential opened</td>
<td>-</td>
<td>-</td>
<td>Special tool, or standard drill</td>
<td>-</td>
</tr>
<tr>
<td>Differential drained</td>
<td>-</td>
<td>-</td>
<td>De-pollution rig</td>
<td>-</td>
</tr>
<tr>
<td>Shock absorber 1</td>
<td>4 mins</td>
<td>£0.61</td>
<td>Special tool; air line</td>
<td>Time per shock absorber</td>
</tr>
<tr>
<td>Shock absorber 2</td>
<td>4 mins</td>
<td>£0.61</td>
<td>Special tool; air line</td>
<td>Time per shock absorber</td>
</tr>
<tr>
<td>Shock absorber 3</td>
<td>4 mins</td>
<td>£0.61</td>
<td>Special tool; air line</td>
<td>Time per shock absorber</td>
</tr>
<tr>
<td>Shock absorber 4</td>
<td>4 mins</td>
<td>£0.61</td>
<td>Special tool; air line</td>
<td>Time per shock absorber</td>
</tr>
<tr>
<td>Catalytic removal</td>
<td>55 secs</td>
<td>£0.14</td>
<td>Hydraulic pickers</td>
<td>-</td>
</tr>
<tr>
<td>Vehicle picked up and placed</td>
<td>1 min</td>
<td>£0.15</td>
<td>Forklift</td>
<td>The vehicle is moved to an area for bailing</td>
</tr>
<tr>
<td>Removal of LPG tanks</td>
<td>-</td>
<td>Handsaw</td>
<td>-</td>
<td>Brief guidance is given by the DTI as to the removal of the LPG tank, but the exact de-pollution of this item is as yet not fully defined.</td>
</tr>
<tr>
<td>Other hazardous materials</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>

Total time = 15.15 mins  £2.32  Labour cost
# De-pollution time study Ford Fiesta (L Reg - No visable damage - Cyriac Tor rig)

<table>
<thead>
<tr>
<th>Operation</th>
<th>Time (Minutes)</th>
<th>Labour cost (£)</th>
<th>Equipment</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>Collection and placement</td>
<td>1 min</td>
<td>£0.13</td>
<td>Forklift</td>
<td>Three vehicles lined up and top side stuff completed on them first before further processing</td>
</tr>
<tr>
<td>Bonnet popped and battery removed</td>
<td>1 min</td>
<td>£0.15</td>
<td>Spanner, Hydraulic pincers.</td>
<td>Of the three vehicles one had problems with getting the bonnet open.</td>
</tr>
<tr>
<td>Caps removed off the fluid containers</td>
<td>20 secs</td>
<td>£0.05</td>
<td>Manual</td>
<td>The caps are removed so a vacuum doesn’t form</td>
</tr>
<tr>
<td>Setup coolant drain</td>
<td>15 secs</td>
<td>£0.03</td>
<td>Hoses within area</td>
<td>Each vacuum hose is pushed into the required pipe. Vacuum left to suck. Always done on the ground</td>
</tr>
<tr>
<td>Drain coolant</td>
<td>1 min 1</td>
<td>£0.15</td>
<td>Hoses within area</td>
<td>Vaccum hose placed in the screen wash tank when the vehicle is on the ground</td>
</tr>
<tr>
<td>Setup washer fluid drain</td>
<td>15 secs</td>
<td>£0.03</td>
<td>Hoses within area</td>
<td>Vaccum hose placed in the screen wash tank when the vehicle is on the ground</td>
</tr>
<tr>
<td>Drain washer fluid</td>
<td>15 secs</td>
<td>£0.03</td>
<td>Hoses within area</td>
<td>The power steering fluid is removed from a reservoir within the engine. None of the natural ELV's have power steering</td>
</tr>
<tr>
<td>Drain brake fluid</td>
<td>15 secs</td>
<td>£0.03</td>
<td>Hoses within area</td>
<td>Removal of R12 and R134a. Specialist tools required. Estimated 5 minutes</td>
</tr>
<tr>
<td>Power steering oil</td>
<td></td>
<td></td>
<td></td>
<td>Removal of R12 and R134a. Specialist tools required. Estimated 5 minutes</td>
</tr>
<tr>
<td>A/C drainages</td>
<td></td>
<td></td>
<td></td>
<td>Removal of R12 and R134a. Specialist tools required. Estimated 5 minutes</td>
</tr>
<tr>
<td>Wheel Backleft</td>
<td>16 secs</td>
<td>£0.04</td>
<td>Air ratchet, compressor pump</td>
<td>Back-left wheel. Four bolts removed using air-ratchet</td>
</tr>
<tr>
<td>Wheel Frontleft</td>
<td>20 secs</td>
<td>£0.05</td>
<td>Air ratchet, compressor pump</td>
<td>Front-left wheel. Four bolts removed using air-ratchet</td>
</tr>
<tr>
<td>Wheel Backlight</td>
<td>65 secs</td>
<td>£0.16</td>
<td>Air ratchet, Spanner, Hammer, chisel</td>
<td>Back-right wheel has some sort of anti-theft nut. Mechanics has real trouble removing it.</td>
</tr>
<tr>
<td>Wheel Frontlight</td>
<td>12 secs</td>
<td>£0.03</td>
<td>Air ratchet, compressor pump</td>
<td>Front-right wheel. Four bolts removed using air-ratchet</td>
</tr>
<tr>
<td>Vehicle picked up and shaken</td>
<td>31 secs</td>
<td>£0.06</td>
<td>Forklift</td>
<td>Four wheels drop.</td>
</tr>
<tr>
<td>Under cage spare tyre support cut</td>
<td>10 secs</td>
<td>£0.03</td>
<td>Manual metal pincers</td>
<td>Two cross-members are cut with sheers to allow the tyre to drop.</td>
</tr>
<tr>
<td>Tyres removed</td>
<td>30 secs</td>
<td>£0.08</td>
<td>Manual labour</td>
<td>De-pollution mechanic picks up the tyres and moves them to a pile adjacent to the rig</td>
</tr>
<tr>
<td>Vehicle picked up and placed in front of</td>
<td>50 secs</td>
<td>£0.13</td>
<td>Forklift</td>
<td>The forklift still has the vehicle in mid-air. The driver then holds the ELV in front of the Cyriac Tor rig to do the shock absorbers</td>
</tr>
<tr>
<td>Buckets placed under absorbers</td>
<td>15 secs</td>
<td>£0.04</td>
<td>Manual labour</td>
<td>The fluid filled shock absorbers have used the pincers to break the bottom of them. The fluid then drains.</td>
</tr>
<tr>
<td>Shock absorber 1</td>
<td>11 secs</td>
<td>£0.03</td>
<td>Hydraulic pincers</td>
<td>The fluid filled shock absorbers have used the pincers to break the bottom of them. The fluid then drains.</td>
</tr>
<tr>
<td>Shock absorber 2</td>
<td>11 secs</td>
<td>£0.03</td>
<td>Hydraulic pincers</td>
<td>The fluid filled shock absorbers have used the pincers to break the bottom of them. The fluid then drains.</td>
</tr>
<tr>
<td>Shock absorber 3</td>
<td>11 secs</td>
<td>£0.03</td>
<td>Hydraulic pincers</td>
<td>The fluid filled shock absorbers have used the pincers to break the bottom of them. The fluid then drains.</td>
</tr>
<tr>
<td>Shock absorber 4</td>
<td>11 secs</td>
<td>£0.03</td>
<td>Hydraulic pincers</td>
<td>The fluid filled shock absorbers have used the pincers to break the bottom of them. The fluid then drains.</td>
</tr>
<tr>
<td>Fluid drain</td>
<td>2 mins 46</td>
<td>£0.42</td>
<td>Plastic buckets</td>
<td>All the fluid drains into the four strategically placed plastic buckets.</td>
</tr>
<tr>
<td>Vehicle picked up and placed on rig</td>
<td>20 secs</td>
<td>£0.05</td>
<td>Forklift</td>
<td>The ELV is placed on the Cyriac Tor rig</td>
</tr>
<tr>
<td>Sump opened</td>
<td>40 secs</td>
<td>£0.10</td>
<td>Allen key, ratchet, hammer, chisel</td>
<td>Allen key does not work as the sump plug is rusted throw. A hammer and chisel is then used to crack the sump.</td>
</tr>
<tr>
<td>Engine oil started to drain</td>
<td>2 mins</td>
<td></td>
<td>De-pollution rig pan</td>
<td>Allen key does not work as the sump plug is rusted throw. A hammer and chisel is then used to crack the sump.</td>
</tr>
<tr>
<td>Oil filter removal</td>
<td>10 secs</td>
<td>£0.03</td>
<td>Oil filter spanner, Oil filter removal harness</td>
<td>Fuel pumped out. Plug placed in tank. All fuel is pushed into outside tank after been passed through the filters</td>
</tr>
<tr>
<td>Petrol tank is punched</td>
<td>15</td>
<td>£0.03</td>
<td>Special tool, or standard drill</td>
<td>Fuel pumped out. Plug placed in tank. All fuel is pushed into outside tank after been passed through the filters</td>
</tr>
<tr>
<td>Pumping petrol out</td>
<td>6 mins</td>
<td>£0.91</td>
<td>De-pollution rig</td>
<td>Gearbox drilling is not done due to Health and safety risk of draining oil. Looms has had experience of drilling gear-boxes and damaged multiple drill-bits after hitting the cogs inside.</td>
</tr>
<tr>
<td>Gearbox drilled</td>
<td></td>
<td></td>
<td>Special tool, or standard drill</td>
<td>Gearbox drilling is not done due to Health and safety risk of draining oil. Looms has had experience of drilling gear-boxes and damaged multiple drill-bits after hitting the cogs inside.</td>
</tr>
<tr>
<td>Gearbox drained</td>
<td></td>
<td></td>
<td>Special tool, or standard drill</td>
<td>Gearbox drilling is not done due to Health and safety risk of draining oil. Looms has had experience of drilling gear-boxes and damaged multiple drill-bits after hitting the cogs inside.</td>
</tr>
<tr>
<td>Rear differential opened</td>
<td></td>
<td></td>
<td>Special tool, or standard drill</td>
<td>Gearbox drilling is not done due to Health and safety risk of draining oil. Looms has had experience of drilling gear-boxes and damaged multiple drill-bits after hitting the cogs inside.</td>
</tr>
<tr>
<td>Differential drained</td>
<td></td>
<td></td>
<td>Special tool, or standard drill</td>
<td>Gearbox drilling is not done due to Health and safety risk of draining oil. Looms has had experience of drilling gear-boxes and damaged multiple drill-bits after hitting the cogs inside.</td>
</tr>
<tr>
<td>Catalytic removal</td>
<td>49 secs</td>
<td>£0.03</td>
<td>Hydraulic pincers</td>
<td>The vehicle is moved back into the main compound. Airbag deployment is then done using a 12 volt battery. (not seen!)</td>
</tr>
<tr>
<td>Vehicle picked up and placed</td>
<td>1 min</td>
<td>£0.15</td>
<td>Forklift</td>
<td>The vehicle is moved back into the main compound. Airbag deployment is then done using a 12 volt battery. (not seen!)</td>
</tr>
<tr>
<td>Removal of LPG tanks</td>
<td></td>
<td></td>
<td>Handsaw</td>
<td>Brief guidance is given by the DTI as to the removal of the LPG tank, but the exact de-pollution of this item is as yet not fully defined.</td>
</tr>
<tr>
<td>Other hazardous materials</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Total time = 19 mins 30 £2.97 labour costs

240
## Ford Escort (L Reg)

<table>
<thead>
<tr>
<th>IDIS No.</th>
<th>IDIS Name</th>
<th>Removal time</th>
<th>Cleaning time</th>
<th>Total processing time</th>
<th>Effort assessment</th>
<th>Gross weight (g)</th>
<th>Stripped weight (if different)</th>
<th>Attachment count (n)</th>
<th>Fixing mechanism</th>
<th>Material marking</th>
<th>Material type</th>
</tr>
</thead>
<tbody>
<tr>
<td>2.1</td>
<td>Indicator lamp</td>
<td>09:40</td>
<td>01:00:00</td>
<td>01:40</td>
<td>3</td>
<td>220</td>
<td></td>
<td>0</td>
<td>Spring attached</td>
<td>YES</td>
<td>ABS</td>
</tr>
<tr>
<td>2.2</td>
<td>Engine under tray</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>7</td>
<td>Alan-key thread</td>
<td>YES</td>
<td>PP</td>
</tr>
<tr>
<td>2.3</td>
<td>Wheel arch finisher</td>
<td>04:00</td>
<td>04:00</td>
<td>2</td>
<td>1000</td>
<td></td>
<td></td>
<td>7</td>
<td>Alan-key thread</td>
<td>YES</td>
<td>PP</td>
</tr>
<tr>
<td>2.4</td>
<td>Road wheel finisher</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>7</td>
<td>Alan-key thread</td>
<td>YES</td>
<td>PP</td>
</tr>
<tr>
<td>2.5</td>
<td>Radiator grille</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>7</td>
<td>Alan-key thread</td>
<td>YES</td>
<td>PP</td>
</tr>
<tr>
<td>2.6</td>
<td>Air inlet finisher</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>7</td>
<td>Alan-key thread</td>
<td>YES</td>
<td>PP</td>
</tr>
<tr>
<td>2.7</td>
<td>Rear lamp</td>
<td>01:17</td>
<td>00:00:31</td>
<td>01:17</td>
<td>4</td>
<td>760</td>
<td></td>
<td>4</td>
<td>Cross-hatch</td>
<td>YES</td>
<td>ABS</td>
</tr>
<tr>
<td>3.1</td>
<td>Instrument pack finisher</td>
<td>01:03</td>
<td>01:14</td>
<td>01:34</td>
<td>3</td>
<td>235</td>
<td></td>
<td>4</td>
<td>Cross-hatch</td>
<td>YES</td>
<td>PPT2</td>
</tr>
<tr>
<td>3.2</td>
<td>Centre console</td>
<td>01:14</td>
<td>02:37</td>
<td>02:37</td>
<td>1</td>
<td>615</td>
<td></td>
<td>4</td>
<td>Cross-hatch</td>
<td>YES</td>
<td>ABS</td>
</tr>
<tr>
<td>3.3</td>
<td>Steering column finisher</td>
<td>02:25</td>
<td>03:39</td>
<td>3</td>
<td>510</td>
<td>320</td>
<td></td>
<td>2</td>
<td>Cross-hatch</td>
<td>YES</td>
<td>ABS</td>
</tr>
<tr>
<td>3.4</td>
<td>Glove box housing</td>
<td>02:40</td>
<td>03:39</td>
<td>3</td>
<td>200</td>
<td>2</td>
<td></td>
<td>2</td>
<td>Cross-hatch</td>
<td>YES</td>
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* Effort level is rated on a scale of 1-4. One requiring no additional stripping, four not able to be separated
## Vauxhall Astra (K Reg)

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<th>Cleaning time</th>
<th>Total processing time</th>
<th>Effort assessment</th>
<th>Gross weight (g)</th>
<th>Stripped weight (if different)</th>
<th>Attachment count (c)</th>
<th>Fixing mechanism</th>
<th>Material marking</th>
<th>Material type</th>
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### Notes

- **Component not included within model**
- **Heater control housing**
  - Control not included within model
- **Dash trim panel (A) / A Pillar finisher**
  - Unable to remove component due to not having the correct tool
- **A Body post finisher**
  - Unable to remove component due to not having the correct tool

### Additional Notes

- **IDIS Name**
- **Removal time**
- **Effort assessment**
- **Gross weight (g)**
- **Stripped weight (if different)**
- **Attachment count (c)**
- **Fixing mechanism**
- **Material marking**
- **Material type**

Effort level is rated on a scale of 1-4. One requiring no additional stripping, four not able to be separated.

---

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## Ford Fiesta (K Reg)

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<td>7.2</td>
<td>Load space finisher</td>
<td>00:10</td>
<td>00:15</td>
<td>00:25</td>
<td>3</td>
<td>203</td>
<td></td>
<td></td>
<td></td>
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<td>PP</td>
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<tr>
<td>7.3</td>
<td>Load space finisher</td>
<td>04:18</td>
<td>04:18</td>
<td>04:18</td>
<td>2</td>
<td>490</td>
<td></td>
<td></td>
<td></td>
<td>NO</td>
<td>No marking</td>
</tr>
</tbody>
</table>

### Notes
- Components not removed due to lack of fluid depollution
- Component not present
- Effort level is rated on a scale of 1-4, one requiring no additional stripping, four not able to be separated
Reliability of the data source

The intended results from the vehicle dismantling study was to provide a pool of reliable data with which to undertake parametric regression analysis, but the exercise also allowed the opportunity to benchmark the current dismantling information system (IDIS). This benchmarking is important for two reasons. Firstly, the proposed costing methodology relies heavily on this data source to provide accurate parameter information for other makes and model of vehicle. Measuring the deviation of actual and catalogued data provides a quantitative way of measuring system confidence, and demonstrates the inaccuracies and limitations of the proposed costing approach. Secondly, benchmarking IDIS is an interesting case-study as to the mentality of vehicle manufacturers towards the ELV directive and the tools they expect the vehicle reclamation industry to use. Only 24% of the 117 components removed had no hidden removal issues, and were correctly catalogued within IDIS. The remaining components were either incorrectly identified or catalogued, physically too difficult to remove, or simply not present. The table below provides a summary:

<table>
<thead>
<tr>
<th>1993 - Ford Escort</th>
<th>37 components identified</th>
<th>12 not removed</th>
<th>2 - unable to gain access</th>
</tr>
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<tbody>
<tr>
<td></td>
<td></td>
<td>2 - already removed</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>2 - unable to identify</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>6 - unable to remove</td>
<td></td>
</tr>
<tr>
<td>25 removed</td>
<td>(3 parameters measured were weight, attachment count and material type)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>5 - had 1-of-3 parameters incorrect</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>8 - had 2-of-3 parameters incorrect</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>0 - had 3-of-3 parameters incorrect</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>12 - had all parameters correct</td>
<td></td>
<td></td>
</tr>
<tr>
<td>1993 - Vauxhall Astra</td>
<td>53 components identified</td>
<td>15 not removed</td>
<td>3 - unable to gain access</td>
</tr>
<tr>
<td></td>
<td></td>
<td>5 - incorrect tools</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>5 - unable to identify</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>5 - unable to remove</td>
<td></td>
</tr>
<tr>
<td>38 removed</td>
<td>(3 parameters measured were weight, attachment count and material type)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>23 - had 1-of-3 parameters incorrect</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>5 - had 2-of-3 parameters incorrect</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>1 - had 3-of-3 parameters incorrect</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>9 - had all parameters correct</td>
<td></td>
<td></td>
</tr>
<tr>
<td>1993 - Ford Fiesta</td>
<td>27 components identified</td>
<td>8 not removed</td>
<td>2 - unable to gain access</td>
</tr>
<tr>
<td></td>
<td></td>
<td>1 - incorrect tools</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>5 - unable to remove</td>
<td></td>
</tr>
<tr>
<td>19 removed</td>
<td>(3 parameters measured were weight, attachment count and material type)</td>
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</tr>
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<td></td>
<td>7 - had 1-of-3 parameters incorrect</td>
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<td>5 - had 2-of-3 parameters incorrect</td>
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<td></td>
<td>0 - had 3-of-3 parameters incorrect</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>7 - had all parameters correct</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Variation in data catalogued within IDIS and actual dismantling times
Appendix 2

Assessing the economics of pre-fragmentation material recovery in the UK

Introduction

This paper has been accepted for publication in Resource, Conservation and Recycling, May 2007.
Assessing the economics of pre-fragmentation material recovery within the UK

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Received 5 January 2007; received in revised form 28 March 2007; accepted 3 April 2007

Abstract

The 2006 end-of-life vehicles (ELVs) directive target for the recycled and reused material content of an ELV has been undertaken using the current recovery infrastructure within the UK. The current expectation is that the conformance for the 2006 recycling target will be mainly achieved using existing post-fragmentation separation technologies rather than manually disassembling vehicles into their constituent materials. With the economic pressure of the current legislative targets weighing heavily on end-of-life stakeholders, and the further increase of recycling levels for 2015, it is important to understand "when" and "if" manual dismantling activities become economically viable within a dramatically changing vehicle recovery industry. This paper describes a method of costing the dismantling of specific makes and models of vehicle due for retirement in 2015, and discusses the economic implications of such practice and possible strategic directions for pre-fragmentation vehicle recovery.

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Keywords: End-of-life vehicles; Dismantling economics; Pre-fragmentation material removal

1. Introduction

Over two million end-of-life vehicles (ELVs) are produced in the UK each year (Kollamthodi et al., 2003), containing a range of metallic, ceramic and polymeric materials.

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The recycling or recovery of these materials at end-of-life has the potential to substantially improve the sustainability of the automobile through resource conservation and waste minimisation. Yet, at present ELV recycling is undertaken by an industry relatively unfamiliar with the various material preparation methods and vehicle manufacturing processes, and one that is unbound by any “direct” producer responsibility. The end-of-life vehicles directive (2000/53/EC), introduced in 2000, has therefore attempted to redress this issue by bringing vehicle manufacturers closer to the recovery of their products via extended producer responsibility (EPR), to facilitate more sustainable closed-loop thinking.

The UK transposition of the ELV directive requires vehicle manufacturers to provide free take-back and treatment for all their own vehicles post 2007, and meet stringent recycling and recovery targets of 85% and 95% by 2006 and 2015, respectively. Vehicle manufacturers have opted to conform to the legislation by moving away from actively getting involved and investing in their own recovery facilities, in favour of utilising the existing infrastructure and waste reclamation processes within the UK. This has lead to the establishment of “collection contracts”, whereby the existing vehicle recovery industry has agreed to fulfil the requirements laid down by the ELV directive on the vehicle manufacturer’s behalf. The economic support required to fund such an undertaking is estimated to be in the region of £20–41 million per year in de-pollution and site requirements, and £6.5–10.5 million per year in vehicle de-registration (DTI, 2003). It was widely believed by the recovery sector (based on article 5 of the ELV Directive, and lobbied for by the British Metals Recycling Association during the consultation period; Letsrecycle.com, 2002) that this would be subsidised by the vehicle manufacturers, yet during the establishment of these collection contracts it became apparent that no direct financial support would be given to the vehicle recovery sector, given the substantial intrinsic value that ELVs possessed at the time of the contract negotiations.

The UK transposition of the ELV directive has therefore done little to strengthen the relationships between the vehicle recovery chain and the vehicle manufacturers, and it can be argued that this has been counter productive to the core themes of sustainability. Without a subsidised influence from the vehicle manufacturers, any decisions concerning the end-of-life operations carried out on a vehicle will be based solely on process economics as opposed to any long-term environmental benefits. It is therefore vital for the vehicle recovery industry to begin to understand the economics of its own operation, so that future vehicle salvage is based on economic feasibility as well as environmental benefits.

This paper focuses on the pre-fragmentation element of vehicle salvage, and presents the findings of a data collection study undertaken at a UK Authorised Treatment Facility (ATF). Parametric regression analysis is then used to generate vehicle dismantling equations to cost specific assembly removal, and assess the feasibility of future recycling targets with today’s markets. The aim of this modelling is to not only assess the economic implications of future directive conformance via dismantling, but to highlight further potential value recovery opportunities in light of the current relationships created by the transposition of the ELV directive.
2. Background

The existing vehicle recovery industry is predominantly led by a handful of large metal merchants, that are primarily interested in recovering the metallic fraction from ELVs. Geographically distributed scrap-yards (ATFs) serve as collection hubs for these large operators, and are required to make the vehicle environmentally safe via a process of “de-pollution”, before passing the hulk on to the metal merchant for shredding. Fig. 1 highlights the main actors within the automotive “value chain” (Roy and Whelan, 1992), and the strengths of the markets served by the vehicle recovery sector. More detailed literature exists as to the ELV implementation strategies adopted by each EU member states (Perchard, 2004) and its specific UK transposition (Edwards et al., 2006).

To date, the majority of investment has been made by the ATFs in bringing their facilities up to scratch for the de-pollution requirements of the directive. The financial support required to attain the 2006 and 2015 recycling and recovery targets has prompted many discussions among recovery operators, not only as to the ability of current post-fragmentation technologies to achieve the targets, but also the economic viability of the pre-fragmentation alternative. The general consensus from industry is that the 2006 target will be achieved utilising the existing infrastructure and an assumed recycled metallic fraction of 75% (Weatherhead and Hulse, 2005). However, the attainment of the 2015 target is not as easily assured, provoking discussion in both the UK and EU as to whether the latter target should be reviewed (GHK, 2006; SWG-ELV, 2005). Pro-active European investment would sug-

Fig. 1. Main end-of-life stakeholders within the vehicle value chain.
gest that automated post-fragmentation material recovery is the preferred industry option for achievement of the higher recycling and recovery levels, exemplified by Automotive Recycling Netherlands’ recent announcement to develop a post-shredder technology (PST) plant in Tiel (ARN, 2007). The UK’s largest post-fragmentation material recoverers, European Metals Recycling and Sims Metal that handle approximately 70% of all ELV capacity within the UK (DTI, 2005), have yet to publicly affirm their commitments to the higher 95% target and identify their preferred conformance technologies. It is envisaged that they will use the interim period between now and 2015 to make these decisions. For a more detailed discussion of state-of-art post-fragmentation technologies see Ferrão et al. (2006).

Previous pre-fragmentation data collection exercises have been undertaken in both the UK (Weatherhead, 2005) and the US (Gallmeyer, 2003), and preliminary analysis as to the economic viability of material dismantling carried out. More holistic costing models developed to consider the strategic implementation of the ELV directive have also been considered (Amaral et al., 2006; Ferrão et al., 2006; Johnson and Wang, 2002), in which typical quantities of components to be removed pre-fragmentation were assessed. More generic cost modelling work that has considered the economics of the ELV reclamation as a whole have investigated its optimisation (Reuter et al., 2006; Schaik and Reuter, 2004), its fundamental recycling limits (Reuter et al., 2006) and value analysis of disposal strategies (Gupta and Isaacs, 1997).

With the economic pressure of current legislative targets weighing heavily on end-of-life stakeholders, and the uncertainty as to the stability of future scrap material markets, there will eventually be a need (either due to risk mitigation or business survival) to achieve higher levels of value recovery than that which has been traditionally acceptable. Selective pre-fragmentation material removal could potentially provide this value recovery, but barriers to the widespread adoption of these practices is the level of costing resolution required to give end-of-life operators the confidence to invest and diversify their core competencies. Hence, this paper attempts to provide a vehicle specific costing approach to assess the economics of manual material removal (specifically glass, rubbers and plastics) in the context of value recovery and target attainment.

3. Modelling vehicle dismantling economics

3.1. Pre-fragmentation parts resale

A subset of ATFs within the UK currently remove component sub-assemblies for resale. Despite the economic and sustainable advantages this practice can offer (Coates and Rahimifard, 2006), a survey of ATFs (Coates, 2006) has suggested that component removal cannot make substantial headway into improving the recycling and reuse targets laid down by the ELV directive, as the majority of removed sub-assemblies are metallic and are currently counted within the assumed recycled fraction processed during post-fragmentation (Weatherhead and Hulse, 2005). Therefore, components composed of plastics, rubbers or glass can further support the attainment of the recycling targets, but currently only the headlamps, door mirrors and tyres were listed within the top 10 of most commonly removed assemblies that fulfil this criterion (Coates, 2006). Hence, recycling and reuse target attain-
ment must come from either further manual plastics dismantling at the ATF, or automated plastics recovery post-fragmentation.

It is widely perceived within the vehicle recovery sector that the economics of manual material removal is not viable based on UK labour wage rates. Hence, the only realistic situation in which further vehicle dismantling will be undertaken is if the 2015 target remains the same and post-fragmentation technology is unable to meet the higher recycling target (85%), or if the value received for recycled plastics increases enough to make dismantling economically viable. This highlights the need to establish vehicle specific costing methods which not only help to determine when and if recycling plastics becomes economically feasible, but also assists in supporting selection decisions when targeting the most removable and valuable materials. The following sections discuss the data collection exercises undertaken and the parametric equations developed to calculate theoretical dismantling times, before assessing the cost of the attainment of the 2015 recycling target and opportunities to identify profitable components for a range of top selling ELVs.

3.2. Disassembly costing methodology

Despite the lack of upstream manufacturer data with regard to vehicle dismantling, the in-house dismantling study provides an accurate and consistent data pool with which to consider a more diverse range of costing approaches. Based on the statistical data collated during the vehicle dismantling study it was proposed that a parametric regression approach be adopted. The most beneficial attributes of this cost modelling approach is its ability to generate cost estimate relationships (CERs) that are very quick, and produce a statistically measurable output (providing a good assessment of estimate confidence). CERs can be based on any number of relevant parameters, and can potentially be linked to both upstream design and downstream recovery data sources. Despite the incongruous link between the vehicle manufacturers and the recovery sector (exemplified in the 2006 directives transposition), one such data source that has been made widely available due to legislative requirements is that of the International Dismantling Information System (IDIS). This data source catalogues not only the potentially recoverable materials from each make and model of vehicle, but also provides basic component parameter data for each instance. Given the variation that exists between plastic’s location, quantities and type between different makes and model of vehicle, it is advantageous to develop costing equations that allow dismantling times to be generated based on the specific vehicle considered. Therefore, relating IDIS component parameters to the data obtained from the dismantling study using parametric regression analysis, allows CERs to be determined. In brief, if a component’s attributes can be statistically linked to its removal time, and those attributes can be determined for any other make or model of vehicle (i.e. catalogued within the database), then a dismantling time and labour cost can be generated without physically having to perform the work (see Fig. 2). With over 1069 vehicle variants and 59,000 components this costing approach is highly advantageous.

3.3. Vehicle dismantling studies for ascertaining component dismantling times

The initial stage of any parametric equation generation is the establishment of a large pool of data with which to assess the links between component disassembly time and various
What is recyclable? Cross-reference

What component features do removal times depend on?

Dismantling equations

Disassembly Time: Vehicle type 1

Disassembly Time: Vehicle type 2

Disassembly Time: Vehicle type 3

Disassembly Time: Vehicle type 4

Fig. 2. Costing methodology for developing vehicle specific costing.

component attributes. Unfortunately, destructive plastics dismantling is not a wide-spread practice in the vehicle recovery sector, and apart from manufacturer tear-down data (which is highly proprietary and based on non-destructive dismantling), there is no abundant source of reference data. Therefore, a dismantling study was conducted at a local ATF to generate a range of dismantling times for a number of natural ELVs. These were selected based on the top UK selling vehicles in 1993 (Astra, Escort and Fiesta), which would correspond to the demographic of a natural ELV (13 years) in 2006 (Kollamthodi et al., 2003). IDIS was used to identify and assist the removal of approximately 117 individual components, while separation and stripping times were catalogued for each (see Table 1 for the data collected from the Vauxhall Astra teardown).

Once this data pool was established, an iterative process of testing various component parameters was adopted to investigate if there was a statistical relationship between disassembly time. The methodology used for this iterative process is shown within Fig. 3, and uses an equation development process adapted from Levine et al. (2005).

The starting point for these relationships requires the estimator to hypothesize as to the standard parameters affecting dismantling time (accessibility, fixturing, etc.), and the availability of these parameters within the obtainable data source (i.e. IDIS). Parameters (explanatory variables) must appear statistically independent of one another to be included within the analysis, and must contribute to improving the correlation between the predicted and actual disassembly times. The equation performance metrics (variance inflationary fac-
<table>
<thead>
<tr>
<th>IDIS ref</th>
<th>IDIS name</th>
<th>Removal time (mm:ss)</th>
<th>Cleaning time (mm:ss)</th>
<th>Total time (mm:ss)</th>
<th>Gross weight (g)</th>
<th>Material marking</th>
<th>Material</th>
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<td>01:43</td>
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<td>550</td>
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<td>01:34</td>
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<td>5720</td>
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<td>Sill finisher/a post finisher</td>
<td>01:19</td>
<td>00:40</td>
<td>01:59</td>
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<td>00:40</td>
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<td>A post finisher</td>
<td>01:00</td>
<td>01:00</td>
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<tr>
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<td>00:24</td>
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<td>PA</td>
</tr>
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<td>6.7</td>
<td>Fan</td>
<td>11:04</td>
<td>00:30</td>
<td>11:34</td>
<td>1020</td>
<td>YES</td>
<td>PP</td>
</tr>
<tr>
<td>7.2</td>
<td>Access finisher</td>
<td>01:50</td>
<td>00:10</td>
<td>01:50</td>
<td>380</td>
<td>YES</td>
<td>PP</td>
</tr>
<tr>
<td>7.4</td>
<td>Unlisted component</td>
<td>02:15</td>
<td></td>
<td>02:15</td>
<td>580</td>
<td>YES</td>
<td>PP</td>
</tr>
<tr>
<td>7.5</td>
<td>Unlisted component 2</td>
<td>02:28</td>
<td></td>
<td>02:28</td>
<td>900</td>
<td>YES</td>
<td>PP</td>
</tr>
</tbody>
</table>

See Table 4 for main material names and abbreviations, additional unlisted materials include:

* PPX, Poly paraxylene; PF, phenol-formaldehyde resin; PP/PE, polyethylene/polyethylene.
Fig. 3. Parametric equation development process.

3.4. Costing the 2015 recycling and recovery target and value recovery via dismantling

The aforementioned equations and the additional parameter data located within IDIS has been utilised to predict the expected direct labour costs of meeting the 2015 recycling and recovery target for a range of top selling vehicles, under current market conditions. Given the average natural life of a vehicle is 13 years, vehicles produced in 2002 will be ready for scrapping in 2015. Recent governmental reports estimate the deficit to the 2015 recycling target to be approximately 5.18% of a vehicle’s weight (Weatherhead and Hulse, 2005), hence one possible option to make up this shortfall is to consider and optimise component removal. Assemblies can be selected from a vehicle based on two different metrics, either the mass removal rate as shown in Eq. (1), or the value removal rate as shown in Eq. (2) (Coutler et al., 1998). The use of these metrics to select plastic components should be based on the goal of the dismantler. If target attainment is required, the material removal
Table 2
Zonal dismantling equations and statistical significance of coefficients

<table>
<thead>
<tr>
<th>DHS zonal area</th>
<th>Dismantling time equation</th>
<th>R²</th>
<th>R² std</th>
<th>S.E.</th>
<th>F-stat</th>
<th>Parameters</th>
<th>Coeff</th>
<th>SE</th>
<th>t-Stat</th>
<th>P-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dashboard (20 observations)</td>
<td>[ Y = 26.76X_1 + 123.47\sqrt{X_2} - 132.27 ]</td>
<td>0.59</td>
<td>0.54</td>
<td>52.89</td>
<td>11.36</td>
<td>No. attachments (X₁)</td>
<td>26.76</td>
<td>7.18</td>
<td>3.73</td>
<td>1.84E-03</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Cleaning effort (X₂)</td>
<td>123.47</td>
<td>32.53</td>
<td>3.80</td>
<td>1.59E-03</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Intercept</td>
<td>-132.27</td>
<td>54.04</td>
<td>-2.45</td>
<td>2.63E-02</td>
</tr>
<tr>
<td>Door and glaze (four observations)</td>
<td></td>
<td>Limited datasets available</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Seats (nine observations)</td>
<td>[ Y = -14.12X_1^2 + 158.26X_1 + 8.81 ]</td>
<td>0.94</td>
<td>0.92</td>
<td>40.54</td>
<td>50.65</td>
<td>Mass (X₁)</td>
<td>158.26</td>
<td>33.52</td>
<td>4.72</td>
<td>3.25E-03</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Mass²</td>
<td>-14.12</td>
<td>5.66</td>
<td>-2.49</td>
<td>4.70E-02</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Intercept</td>
<td>8.81</td>
<td>20.44</td>
<td>0.43</td>
<td>6.81E-01</td>
</tr>
<tr>
<td>Exterior (12 observations)</td>
<td>[ Y = 118.88X_1^2 + 52.08 ]</td>
<td>0.78</td>
<td>0.76</td>
<td>63.76</td>
<td>38.75</td>
<td>Mass (X₁)</td>
<td>118.88</td>
<td>19.10</td>
<td>6.22</td>
<td>6.49E-05</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Intercept</td>
<td>52.08</td>
<td>25.66</td>
<td>2.03</td>
<td>6.74E-02</td>
</tr>
<tr>
<td>Interior (12 observations)</td>
<td>[ Y = 459.94\sqrt{X_1} - 129.73 ]</td>
<td>0.79</td>
<td>0.77</td>
<td>39.63</td>
<td>42.16</td>
<td>Mass (X₁)</td>
<td>459.94</td>
<td>70.83</td>
<td>6.49</td>
<td>4.47E-05</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Intercept</td>
<td>-129.73</td>
<td>37.46</td>
<td>-3.46</td>
<td>5.30E-03</td>
</tr>
<tr>
<td>Engine compartment (seven observations)</td>
<td>[ Y = 879.62\sqrt{X_1} - 448.07 ]</td>
<td>0.65</td>
<td>0.57</td>
<td>92.77</td>
<td>9.10</td>
<td>Mass (X₁)</td>
<td>879.62</td>
<td>291.62</td>
<td>3.02</td>
<td>2.95E-02</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Intercept</td>
<td>-448.07</td>
<td>206.77</td>
<td>-2.17</td>
<td>8.25E-02</td>
</tr>
<tr>
<td>Load space (eight observations)</td>
<td>[ Y = -637.82X_1^2 + 91.72X_1 - 239.94\sqrt{X_2} + 53.35 ]</td>
<td>0.85</td>
<td>0.69</td>
<td>24.83</td>
<td>5.49</td>
<td>No. attachments (X₁)</td>
<td>-6.37</td>
<td>1.95</td>
<td>-3.26</td>
<td>4.72E-02</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Mass (X₂)</td>
<td>-239.94</td>
<td>108.01</td>
<td>-2.22</td>
<td>1.13E-01</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Intercept</td>
<td>53.35</td>
<td>40.66</td>
<td>1.31</td>
<td>2.81E-01</td>
</tr>
</tbody>
</table>

Cleaning effort was a quantitative measure developed during the study that categorises the level of additional post-removal cleaning required. No. attachments refers to the number of mechanically removable fastenings (e.g. clips, screws, bolts).
rate should be used, as this identifies the heaviest and easiest components to remove first, and gives a better mass-versus-labour return. Alternatively, if a dismantler is interested in knowing if there are any components on a vehicle that can return a profit (when compared to a worker’s wage rate (€/s)), then the value removal rate should be used, as this considers the value of the component removed as well as its weight.

\[
\text{material removal rate (kg/s)} = \frac{\text{material (kg)}}{\text{time (s)}} \quad (1)
\]

\[
\text{value removal rate (€/s)} = \frac{(\text{material (kg)} \times \text{value (€/kg)})}{\text{time (s)}} \quad (2)
\]

Table 3 demonstrates both of these scenarios, which include the use of the material removal rate to select components to fulfil the 5.18% deficit to the 2015 recycling target, and the use of the value removal rate on the same vehicles to select components capable of returning a profit. The vehicles selected are the top UK selling vehicles of 2002, representative of typical natural ELVs in 2015.

The worker’s wage rate adopted is taken from ATF interviews undertaken in 2006 and uses a rate of €10.36 per hour (€8.62 per hour in wages and €1.74 per hour in fringe benefits). Fig. 4 provides a sensitivity analysis around this adopted rate to demonstrate its impact on the resulting net revenue (recycling revenue minus removal cost).

An additional cost that must also be considered when assessing the dismantling of components is the logistical cost of transporting the removed materials to a recycling facility. Previous dismantling studies have assumed a flat transport fee of around €74 per tonnes (Weatherhead, 2005), which based on the average mass removed per ELV within both methods (i.e. 51.26 and 7.24 kg, respectively, see Table 3) equates to €3.79 per vehicle for scenario 1 (i.e. material removal rate) and €0.53 per vehicle for scenario 2 (i.e. value removal rate).

![Fig. 4. The impact of hourly wage rates on the average net revenues obtained per ELV for all the vehicles considered.](image)
### Table 3

<table>
<thead>
<tr>
<th>Vehicle</th>
<th>Weight (kg)</th>
<th>No. of components</th>
<th>Dismantling time</th>
<th>Labour cost</th>
<th>Revenue</th>
<th>Value removal rate used to identify components that return a profit when compared to an hourly rate</th>
</tr>
</thead>
<tbody>
<tr>
<td>1st</td>
<td>50.70</td>
<td>33</td>
<td>1 h 40 min</td>
<td>€17.38</td>
<td>€9.56</td>
<td>€12.20</td>
</tr>
<tr>
<td>2nd</td>
<td>50.38</td>
<td>20</td>
<td>1 h 38 min</td>
<td>€17.01</td>
<td>€11.36</td>
<td>€7.41</td>
</tr>
<tr>
<td>3rd</td>
<td>50.38</td>
<td>32</td>
<td>1 h 50 min</td>
<td>€18.97</td>
<td>€9.23</td>
<td>€2.61</td>
</tr>
<tr>
<td>4th</td>
<td>50.38</td>
<td>21</td>
<td>Data unavailable in IDS</td>
<td>€19.34</td>
<td>€9.66</td>
<td></td>
</tr>
<tr>
<td>5th</td>
<td>50.38</td>
<td>24</td>
<td>1 h 52 min</td>
<td>€18.18</td>
<td>€9.55</td>
<td></td>
</tr>
<tr>
<td>Ford focus 1998+</td>
<td>53.49</td>
<td>20</td>
<td>1 h 45 min</td>
<td>€19.34</td>
<td>€9.66</td>
<td></td>
</tr>
<tr>
<td>Vauxhall Astra</td>
<td>51.26</td>
<td>24</td>
<td>1 h 45 min</td>
<td>€19.34</td>
<td>€9.66</td>
<td></td>
</tr>
<tr>
<td>Peugeot 206</td>
<td>53.49</td>
<td>20</td>
<td>1 h 50 min</td>
<td>€19.34</td>
<td>€9.66</td>
<td></td>
</tr>
<tr>
<td>Ford Fiesta</td>
<td>51.26</td>
<td>24</td>
<td>1 h 45 min</td>
<td>€19.34</td>
<td>€9.66</td>
<td></td>
</tr>
<tr>
<td>Averages</td>
<td></td>
<td></td>
<td></td>
<td>€18.18</td>
<td>€9.55</td>
<td></td>
</tr>
</tbody>
</table>

*Labour costs based on an hourly rate of €10.56.
*Material value data taken from recycley.com and interviewed plastics recyclers (January, 2006).
*Average vehicle weight based on 971 kg (Weatherhead and Hulse, 2005).
Table 4
The main types and quantities of material removed during the dismantling study

<table>
<thead>
<tr>
<th>Material</th>
<th>Abbreviation</th>
<th>Mass per vehicle (kg)</th>
<th>Total possible per month (tonnes)</th>
<th>Total possible per year (tonnes)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Polypropylene</td>
<td>PP</td>
<td>11.45</td>
<td>5.8</td>
<td>69.1</td>
</tr>
<tr>
<td>Polyurethane</td>
<td>PUR</td>
<td>8.12</td>
<td>4.1</td>
<td>49.0</td>
</tr>
<tr>
<td>Acrylonitrile-butadiene-styrene</td>
<td>ABS</td>
<td>3.78</td>
<td>1.9</td>
<td>22.8</td>
</tr>
<tr>
<td>Polypropylene-talcum 20%</td>
<td>PP-T20</td>
<td>1.57</td>
<td>0.8</td>
<td>9.5</td>
</tr>
<tr>
<td>Polyamide</td>
<td>PA</td>
<td>0.83</td>
<td>0.4</td>
<td>5.0</td>
</tr>
<tr>
<td>Polypropylene-ethylene-propylene</td>
<td>PP-EPDM</td>
<td>0.82</td>
<td>0.4</td>
<td>4.9</td>
</tr>
<tr>
<td>diene terpolymer</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Poly(ethylene terephthalate)</td>
<td>PET</td>
<td>0.65</td>
<td>0.3</td>
<td>3.9</td>
</tr>
<tr>
<td>Poly(vinyl chloride)</td>
<td>PVC</td>
<td>0.50</td>
<td>0.3</td>
<td>3.0</td>
</tr>
<tr>
<td>Polycarbonate acrylonitrile-</td>
<td>PC + ABS</td>
<td>0.25</td>
<td>0.1</td>
<td>1.5</td>
</tr>
<tr>
<td>butadiene-styrene blend</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total</td>
<td></td>
<td>27.97</td>
<td>14.1</td>
<td>168.7</td>
</tr>
</tbody>
</table>

3.5. Material yield rates via dismantling

A further consideration as to the feasibility of manual material removal is that of achievable material yield rates. The aforementioned value removal rate utilises material value estimates based on minimum recycled quantities. Hence, to realistically consider manual material removal a consideration must be made as to the vehicle throughput required to achieve minimum re-processor specifications. During the study, 22 different material types were removed, with the 9 most abundant materials (>0.25 kg) producing 28 kilos per vehicle. These quantities can then be factored up based on the typical number of vehicles processed at an ATF per day (17 vehicles; Coates, 2006), and are listed within Table 4.

The data catalogued within the above table is representative of those materials found within natural ELVs, but it is envisaged that the material types and quantities will be reduced over time as more “design for recycling” considerations filter through in successive vehicle designs. The current quantities obtained via the study would suggest that only a few key material types (identified in Table 4 as PP, PU and ABS) would produce enough material to satisfy the minimum quantities required by plastic re-processors, and justify their removal.

4. Discussion

The purpose of this paper is to not only describe a means of costing pre-fragmentation vehicle dismantling, but to offer analysis as to the cost of meeting the 2015 recycling target and the potential for using pre-fragmentation dismantling as a means of obtaining value. Table 5 provides an aggregated summary of the costs and revenues associated with the dismantling of vehicles for target attainment (scenario 1), and the dismantling of vehicles for value recovery (scenario 2).

If ELV operators were to meet the 2015 recycling target today through further manual dismantling, it would result in an estimated net cost per ELV of around €12. The additional investment costs of new equipment and storage facilities would also need to be factored into
Table 5
Summary of costs and revenues incurred via different processing scenarios

<table>
<thead>
<tr>
<th>Scenario 1</th>
<th>Scenario 2</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dismantling for meeting the target</td>
<td>Dismantling for value recovery</td>
</tr>
<tr>
<td>Percent of vehicle</td>
<td>Percent of vehicle</td>
</tr>
<tr>
<td>weight removed</td>
<td>weight removed</td>
</tr>
<tr>
<td>(≥5.18%)</td>
<td></td>
</tr>
<tr>
<td>Removal costs</td>
<td>-€18.18</td>
</tr>
<tr>
<td>Recycling revenue</td>
<td>+€9.95</td>
</tr>
<tr>
<td>Logistical costs</td>
<td>-€3.79</td>
</tr>
<tr>
<td>Per ELV</td>
<td>-€12.02</td>
</tr>
<tr>
<td>5.28</td>
<td>0.75</td>
</tr>
</tbody>
</table>

This if an ATF decided to adopt this practice. What this estimate indicates is the substantial cost burden that will be required to meet the future target if the recycling levels remain the same and the investment and technology in UK post-fragmentation facilities is not significantly improved. It must be stated that the likelihood of this eventuality (achievement of the 2015 target via material dismantling) based on the strategic direction of other more proactive EU member states, would suggest that this will not be the case in the UK. Given the achievable throughput rates of automated waste stream technologies, it is envisaged that post-fragmentation processing will provide the most economical viable point at which to recover the additional rubbers, glass and plastics, which are currently sent to landfill. As to whom within the vehicle value chain will ultimately be financially responsible for either of these scenarios is yet to be made clear. Previous investment and de-pollution costs have been offset by extremely strong scrap metal prices, but questions as to this market’s long-term stability may negate the possibility of scrap revenue supporting the vehicle recovery sector in the future.

The previous analysis has also offered an opportunity to assess the feasibility of ATFs disregarding the 2015 target (assuming post-fragmentation conformance), and only dismantling components that can offset the incurred labour costs of their removal. These would be large, heavy sub-assemblies (such as bumpers and internal trim) that are relatively easy to remove. In these instances, the recycling revenue generated (€2.90, see Table 5) is capable of offsetting the direct labour costs incurred (€1.72, see Table 5), producing a net revenue per ELV of around €1.18. This revenue is reduced when the material recovered is transported to a recycling facility for shredding and granulation, resulting in an overall net profit of €0.65 per ELV. This analysis was undertaken using a wage rate of €10.36, Fig. 5 presents the variation in net profit when considering wage rates either side of this value.

Unlike the smooth trend seen within Fig. 4 the resulting net profit seen in Fig. 5 fluctuates substantially. This is due to the number of competing parameters that affect the final cost. A lower wage rate reduces the threshold at which the value removal rate selects components, as more components are capable of offsetting the direct labour incurred. Material type determines the obtainable revenue, while the quantity of material removed increases the logistical costs of transportation.

The variation between Figs. 4 and 5 highlights the significant economic impact of transportation costs, in addition to commonly reported negative environmental impacts associated with reverse logistics. This would perhaps suggest a strong case for a more geographically
concentrated approach to vehicle recovery and material recycling. If ATFs were to diversify their core competencies to incorporate plastics recycling this could potentially allow them to sell reprocessed granulate directly back to the product suppliers and attain high revenues. These recycling activities need not be exclusively focused on recovering just automotive polymers, but could also encompass additional product waste streams (consumer packaging, industrial scrap, plastic from WEEE) that will become increasingly more abundant as end-of-life legislation becomes more established.

Despite the low achievable revenue for manual dismantling of plastic components (based on today's market prices), it should be noted that the quality of the polymer produced is also substantially better than that achieved during post-fragmentation separation, promoting the possibility of more sustainable closed-loop recycling. Questionable drawbacks to this approach are the material yield rates that one ATF alone can achieve ($\approx 7.24$ kg per ELV), suggesting that this option would be more suited to larger ATF operators with a greater throughput. An additional barrier is the lack of distinction made between the beneficial qualities of automotive polymers compared to those currently considered to be recycled plastics (curbside collection consumer packaging). This problem is further aggravated by the lack of cheap and accurate analysis equipment to produce material specifications to form the basis of price negotiations.

5. Conclusion

This paper has demonstrated the substantial cost burden that the 2015 recycling and recovery target will bring to the automotive industry if target attainment is to be achieved via pre-fragmentation material removal. It has also demonstrated that despite the commonly held perception that manual material removal is not economically viable, the targeted removal of a certain components for recycling is. Questionable barriers to the
widespread adoption of these techniques are the achievable material yield rates, and the lack of value distinction made between other plastic recycling sources (consumer plastic wastes).

The costing methods described within this paper have attempted to provide a foundation on which future "what-if" scenarios for vehicle recovery assessment can be undertaken. The current cost drivers affecting the vehicle recovery sector (e.g. scrap steel prices, labour costs, recycled material prices, etc.) are constantly changing, and only at the point at which legislation is fully implemented can the economic viability of pre-fragmentation material recovery be truly assessed and compared to its post-fragmentation alternative. As to whether vehicle dismantling will ever become a part of the standard operations carried out by an ATF either due to necessary target attainment or activity diversification is yet to be seen, but if and when this does occur, accurate methods are required to economically assess and optimise any removal activities.

Acknowledgements

This work has been carried out as part of a collaborative research programme at Loughborough University, funded by the Engineering and Physical Science Research Council (EPSRC), entitled "Cost-oriented Approach to Design and Recovery of Vehicles to meet the requirements for the End-of-Life Vehicles Directive". The authors would like to acknowledge the contributions and support of Mrs. D. Allen and Mr. R. Kirk of Albert Looms Derby, and Mr. P. Owen and staff of Rozone Ltd.

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Appendix 3

Cost models for increased value recovery from end-of-life vehicles

Introduction

This paper was presented in the 13th CIRP Internation Conference on Life Cycle Engineering, held in Leuven-Belgium in 2006.
Cost models for Increased Value Recovery from End-of-Life Vehicles

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Abstract

A sustainable approach to a products End-of-life processing needs to be a balance between the environmental impacts of a particular course of action, and it's economic viability. The research reported in this paper has investigated a structured costing framework to be used in conjunction with improved environmental practises, to provide an economic understanding of varying End-of-Life Vehicle processing routes. The paper presents an holistic end-of-life cost model for the vehicle recovery sector and focuses on the potential applications of this model to support both high and low level decisions, in terms of a processes economic merits and its influence on the ELV Directives recycling and recovery targets.

Keywords
End-of-life Vehicles, Cost modelling, Legislation, Value recovery

1 INTRODUCTION

Environmental legislation, the physical interpretation of the "polluter-pays" principal, has meant that manufacturers and businesses are becoming ever more accountable for their product's environmental effects beyond the traditional boundaries of the product development process. End-of-life disposal and product takeback legislation has taken a proactive stance and has formulated a number of prescriptive European directives encompassing the design, production and end-of-life treatment of a range of products. The automobile, through the End-of-Life Vehicles (ELV) directive [1] has become one of the first products to be actively legislated against, and will undoubtedly act as a reference model to other Original Equipment Manufacturers (OEMs). In its simplest sense the legislation requires vehicle manufacturers to provide free take-back and treatment for all its own vehicles post 2007, and meet recycling and recovery targets of 85% and 95% in 2006 and 2015 respectively. Many vehicle manufacturers have opted to conform to the directive by moving away from actively fulfilling the requirements themselves, in favour of utilising traditional waste reclaimation routes. The vehicle recovery chain comprises of stakeholders that hold their core competency in a particular facet of vehicle salvage, whether it be a type of component reuse or material recovery. Unlike the vehicle supply chain the vehicle recovery chain is a somewhat archaic and reactive industry, and the ideas of lean operation, value improvement and Environmentally Conscious Recovery (ECR) are not well established.

Decisions made throughout a vehicles life-cycle impinge on its level of sustainability, but perhaps one of the most influential factors is where the vehicle recovery chain finally places it within the waste (reuse) hierarchy. Legislative recycling and recovery targets have some influence over this final product routing, but the decision is often based on how well a product retains its economic value and the level of end-of-life (EOL) processing required. Within the recovery chain the investment required to maintain that value can not always be perceivably justified, with little understanding as to the exact economics of varying EOL processing decisions and even less transparency in terms of environmental performance. The research reported within this paper identifies an holistic EOL costing model, to be used in conjunction with environmental best practise, to provide an economic insight into sustainable vehicle recovery.

2 BACKGROUND

The influence of the ELV Directive is apparent throughout the vehicle value chain, from supplier reporting requirements and manufacturer type-approval testing, to the end-of-life operators that actually process the retired vehicles. Each European member state has opted to transpose the Directive into its own laws in a variety of different ways. A more detailed discussion of these difference can be found within the environmental regulations report [2]. The UK has opted for an "own marquee" approach which has seen each vehicle manufacturer establish its own contracted network of Authorised Treatment Facilities (ATFs), where owners can return their vehicles free of charge. At these facilities the vehicle is de-polluted and cannibalised for spare parts, before the hulk is passed to the shredder operator. The shredder then fragments the vehicle and recovers the ferrous content, in advance of passing the remaining residue to the dense media plants to recover any other non-ferrous and non-metallic content. Figure 1 highlights the main actors and material flows within the vehicle value chain.

Figure 1: The recovery chain within the vehicle value chain.
Focusing on the End-of-life processing requirements of the ELV directive, the fulfilment of two key points are driving investment and reform within the recovery sector.

- The establishment of standards for storage, treatment and de-pollution of ELVs (this has been the catalyst for substantial improvements in the whole sectors environmental operating standards).
- Achievement of a recycling and recovery target of 85% (80% recycling) of a vehicles weight by 2006, and 95% (85% recycling) by 2015.

Current ELVs hold a great wealth of intrinsic value given the current scrap steel prices that many EOL operators receive for their vehicle hulks. Vehicle manufacturers have used this to their advantage when negotiating ELV collection contracts with stakeholders within the recovery chain. This has resulted in the formulation of "zero-cost" contracts [3], allowing the manufacturers to achieve the 2006 recycling and recovery target using the existing infrastructure without the need for further direct investment.

Work that has focused on costing the attainment of the 2006 and 2015 targets has concluded that the economics of pre-shredder dismantling are unfavourable compared to that of automated post-shredder separation [4], despite the improvements in purity and secondary applications that can be achieved by utilising dismantled materials [5,6]. The general consensus is that the 2006 target will be achieved utilising the existing stakeholders, but a number of papers [4,7] have highlighted the inability of current post-shredder separation technologies to meet the 2015 target. This has prompted discussion in both the UK and EU as to whether the later target should be reviewed [8].

The economic ramifications of this conformance has left many EOL stakeholders in a uniquely different market to that of which they have been traditionally used to. Their only recent inclusion within the vehicle value chain has meant that the demands of being part of the manufacturers extended enterprise have never been present, and as such the demands of waste reduction and value improvement have never been major industry concerns. With the introduction of the ELV legislation comes an increased need for the recovery chain to better understand the economics of its own operations. Once achieved the sector will more realistically consider environmentally beneficial alternatives.

3 RESEARCH METHODOLOGY

Given the drastic reform and investment that the recovery industry is currently undergoing, there will eventually be a need, either due to risk mitigation or business survival, to achieve higher levels of value recovery than that which has been traditionally accepted. This requires an understanding not only of the costs of processing a particular type of vehicle (pre and post-fragementation), but also of the achievable revenues from the sale of materials and components. Each stakeholder within the recovery chain is different, from the level of investment in their facilities to the variation in their core competencies and value added operations. To establish an economic understanding of their operations and make sustainable recommendations based on it, a method is required that allows the costing of this variation. This has resulted in the development of an EOL costing model, which establishes a base "as-is" model for a particular stakeholder within the vehicle recovery chain before analysing and optimising potential "to-be" scenarios. Figure 2 highlights the main modules used within the framework, and the following sections discuss some of the data collected and techniques used.

![End-of-Life Vehicle Cost Model](image)

Figure 2: Generalised cost model structure

3.1 ELV costing database

The database acts as a central source of reference from which users of the model have access to typical data, this can be used to generate a base model of their operations. Table 1 outlines some of the typical information structures used within the database and the costing modules in which the information is used. The majority of the data has been catalogued from an extensive review of UK EOL operators.

<table>
<thead>
<tr>
<th>Information structure</th>
<th>Model purpose</th>
</tr>
</thead>
<tbody>
<tr>
<td>Capital equipment costs</td>
<td>Typical investment costs, depreciation, power requirements, etc...</td>
</tr>
<tr>
<td>Average material pricing index</td>
<td>Material prices, used to assess the feasibility of material removal at various stages of processing.</td>
</tr>
<tr>
<td>Material property data</td>
<td>Typical material property data (density, conductivity, etc...), used as part of the post-shredder costing module to determine the achievable waste stream separation.</td>
</tr>
<tr>
<td>Machine efficiency values</td>
<td>&quot;Tromp curve&quot; values, used within the post-shredder costing module that describe the ability of a particular machine to separate materials.</td>
</tr>
<tr>
<td>Vehicle information</td>
<td>Average vehicle composition and list of removable materials, used as part of the pre-shredder costing module to identify components to be removed.</td>
</tr>
<tr>
<td>Rates</td>
<td>Labour, exchange, fuel, power, business, etc...</td>
</tr>
</tbody>
</table>

Table 1: Typical information structures within the ELV database.
The database is designed to act not only as a knowledge based repository to assist the user in the generation of an “as-is” base model, but also as a live updatable information source if located on a web server. This would allow parameters within the database (such as the materials price index and waste management costs) that seriously affect the economics of the system to be regularly updated.

3.2 Indirect ELV processing costs

To assess the true cost of processing an ELV, both the direct and indirect costs must be considered. Direct costs are often more visible and easier to attribute, whereas indirect costs are often shared by a number of resources and are not as easily defined. "Traditional cost accounting" has always attributed indirect costs using direct cost-drivers (such as labor). The inequities of such approaches are well documented and have lead to the development of "Activity Based Costing" (ABC) accounting. ABC assumes that activities consume resources, and as such, indirect costs like overheads and equipment depreciation can be directly linked to a machine’s utilisation and throughput. The effective capturing of these links (otherwise known as "cost-drivers") allows the attribution of the total operating cost of an activity to unit, batch or line level quantities. An example of the application of the ABC methodology to the vehicle de-manufacturing process can be found within the work of Bras and Emblemsvag [9] and provides a good example of how in-direct costs can be attributed within the ELV cost model.

3.3 Pre-shredder dismantling costs

Aside from the enforced de-pollution process to remove the fluids and hazardous materials from the vehicle, only reusable components are removed from the vehicle pre-fragmentation. Currently, the removal of materials (plastics in particular) is not a widespread practice by the ATFs, and is not considered feasible given the labour intensive nature of the work. However, this is not to say that this will always be the case. The long-term stability of both scrap steel and global oil prices, combined with an unfavourable downturn in the parts resale business has the potential to make the economic viability of such practices more appealing. Therefore, methods of costing both the removal of reusable component and recyclable plastic trim have been included within the model.

3.3.1 Component removal and resale

Standard sub-assembly removal times have been collected via a questionnaire distributed to over 300 ATFs throughout the UK. The preliminary findings have assisted in determining the most frequently removed sub-assemblies and standard removal times. Table 2 provides an example of this data which will ultimately be utilised to cost component reuse, and recycling target achievement within the model.

3.3.2 Plastics dismantling case study

The recycling targets laid down by the ELV Directive did not assume that the recycling quantities would necessarily come from post-fragmentation separation technologies. There are clauses within the directive that require vehicle manufacturers to provide detailed dismantling information for the plastic components that can be removed from an ELV (included within the International Dismantling Information System, IDIS), should the target be achieved pre-fragmentation. Unfortunately, there is no abundant source of publicly available data that catalogues the destructive dismantling of vehicles with which to develop costing equations. Therefore, a study was undertaken within a UK ATF to generate material removal times for a range of top selling natural and premature ELVs. This data would ultimately be combined with manufacturer "tear-down" data, and be used to develop Cost Estimate Relationships (CERs) using parameters available within the IDIS database (weight, attachment count, location, etc…). By utilising parametric regression analysis, the CERs developed are capable of generating an approximate disassembly time (and hence cost) for any component from any make or model held within the database, without the need to physically carry out the work.

3.4 Post-fragmentation costs

Shredders and dense media separation plants are primarily focused at recovering the metallic fractions from the vehicle once it has been fragmented. This is achieved via a series of automated separation technologies that target specific physical and material characteristics within the waste stream that are susceptible to that processes influence. Typical processing equipment used within these facilities include; over-band magnets, floatation tanks, eddy current separators, air cyclones and screening meshes. By identifying the waste stream parameters

<table>
<thead>
<tr>
<th>Component</th>
<th>Average removal time</th>
<th>Labour cost (£)*</th>
<th>Resale price for a Premature ELV (£)**</th>
<th>Resale price for a Natural ELV (£)**</th>
<th>Improvement in recycling and reuse target***</th>
</tr>
</thead>
<tbody>
<tr>
<td>Engine</td>
<td>1 hour 11 minutes</td>
<td>12.35</td>
<td>607</td>
<td>192</td>
<td>10.48%</td>
</tr>
<tr>
<td>Gearbox</td>
<td>52 minutes</td>
<td>9.04</td>
<td>299</td>
<td>163</td>
<td>2.97%</td>
</tr>
<tr>
<td>Alternator</td>
<td>15 minutes</td>
<td>2.61</td>
<td>60</td>
<td>36</td>
<td>0.66%</td>
</tr>
<tr>
<td>Starter motor</td>
<td>17 minutes</td>
<td>2.96</td>
<td>56</td>
<td>44</td>
<td>0.34%</td>
</tr>
<tr>
<td>Distributor</td>
<td>10 minutes</td>
<td>1.74</td>
<td>56</td>
<td>33</td>
<td>0.02%</td>
</tr>
<tr>
<td>Head-lamp assembly</td>
<td>12 minutes</td>
<td>2.09</td>
<td>37</td>
<td>19</td>
<td>0.17%</td>
</tr>
<tr>
<td>Quarter glass</td>
<td>14 minutes</td>
<td>2.43</td>
<td>37</td>
<td>33</td>
<td>0.64%</td>
</tr>
<tr>
<td>Radiator</td>
<td>16 minutes</td>
<td>2.78</td>
<td>54</td>
<td>30</td>
<td>0.42%</td>
</tr>
<tr>
<td>Wing mirror</td>
<td>9 minutes</td>
<td>1.56</td>
<td>43</td>
<td>27</td>
<td>0.13%</td>
</tr>
<tr>
<td>Totals</td>
<td>3 hours 37 minutes</td>
<td>€37.56</td>
<td>€1249</td>
<td>€577</td>
<td>15.85%</td>
</tr>
</tbody>
</table>

* Based on £21,700 per annum mechanic / technicians wage working a 40 hour week.
** Premature refers to a vehicle of 7 years of age (1999), a natural refers to one of 13 years (1993). Top 3 selling vehicles from each respective year researched. (www.carparts-uk.com)
*** Based on the average weight of 1030kg [10]

Table 2: Data used to cost the removal of components for resale and target achievement.
which each of these technologies are trying to target, and utilising process efficiency curves (Tromp/Partition curves) found within the minerals refinement industry, a 'theoretical separation model' can be developed, which can be used to predicted where each material will ultimately end up and its level contamination at that point (see figure 3). This approach allows the modelling of value-added processing operation for each post-fragmentation technology, and results in a grade and recovery percentage for each waste stream constituent. These percentages can then be cross-referenced with estimated "value vs % contamination" curves for each material and a potential recycling revenue or disposal cost generated.

4 PRELIMINARY MODEL DEVELOPMENT

Each module within the ELV cost model has been implemented within Excel spreadsheets to demonstrate the principles before bringing all of the approaches together. Figures 4 and 5 provide examples of this development.

Figure 4: Screenshot from the theoretical separation model, calculating the predicted separation of a trommel.

Figure 5: The use of CERs to cost pre-shredder plastics removal.

5 DISCUSSION

Many businesses have long considered sustainability as focusing too heavily on the environmental performance of their products, and the balance between the economic and social pillars of sustainability have become disassociated with the term. Upstream organisations that promote sustainable practices are often the ones that have tight control over the economic side of their operations, before venturing improvements within their environmental performance. This has seen the adoption of techniques such as Environmentally Conscious Manufacturing (ECM) and waste reduction methodologies within the vehicle supply chain. Yet surprisingly, the stakeholders who have the most active influence over an automobiles level of sustainability are the EOL operators that have made the most investment and understand their processing costs the least.

EOL stakeholders have direct control over how a vehicle is disseminated, and how its components/materials can be reabsorbed into other value chains, whether it be selecting assemblies for reconditioning, through to isolating shredder residue feeds for energy recovery. The
ELV Directive has brought some prescriptive requirements to this process, but the reactive nature of the vehicle recovery industry has meant that many EOL stakeholders have been reluctant to break with traditional practices. Therefore, to expect this industry to move more towards long-term sustainable practices, without first giving them the assistance to understand the economic implications of their operations, will forever mean that an EOL stakeholder’s financial stability will always take priority over any environmental considerations.

The main goal of this research is to model the economics of vehicle recovery as the waste is disseminated by the various EOL stakeholders. The short-term aim of the model is to suggest value improvement opportunities in the wake of the substantial investment made by the industry in conforming to the ELV Directive. The establishment of a tailored, “as-is” base model for a particular EOL stakeholder, gives them a better understanding of how much it costs to process a vehicle in terms of cost traceability and value-added processing.

Given that the recovery of the metallic fraction of a vehicle is based on established separation technologies that have high throughput and good yield rates, the industry is currently focused on trying to recover the plastics fraction from the remaining residue. The debate is centered around whether this should be achieved before or after the vehicle has been shredded. Plastics segregated pre-fragmentation tend to produce higher value materials than if the ATF is re-used, while plastics recovered from post-fragmentation residue are more suited to closed-loop recycling. Looking at this material value from the ATFs perspective, any potential profit. However, some of the limitations of the assumptions used within this analysis must be made clear.

The previous sections have discussed some of the techniques and studies utilised in developing various modules of the EOL model. Although the integration of these approaches into one holistic end-of-life decision support system is still in development, some preliminary analysis can be undertaken looking at the current economics of material recovery pre and post-fragmentation.

Figure 6: Chart comparing the per kilo value of various plastics to that of the price received by the ATF for the vehicles hulk.

\[ VRR = \frac{\text{Material(kg)} \times \text{Value(€/kg)}}{\text{Dismantling Time(sec)}} \] (1)

Figure 7: Graph showing the value vs effort metric VRR for stripped and un-striped components, relative to the cost of direct labour.
there is still a long way to go if the 2015 target remains in
To date, the majority of investment, and the inclusion of
canonheadway in fulfilling the 2006 target, but
down by the directive, the recovery sector has made
within an industry that has traditionally seen little
environmental improvements within the recovery sector,
the ELV directive has been the catalyst for substantial
recovery and recycling targets laid
down by the directive, the recovery sector has made
considerable headway in fulfilling the 2006 target, but
there is still a long way to go if the 2015 target remains in
place.
To date, the majority of investment, and the inclusion of
environmental operating procedures (de-pollution), have
been undertaken by the EOL operators due to
Government legislation. This will ultimately change once
the directive is in full operation, and further prescriptive
parameters that will also affect the viability of these practices, include increases in virgin
polymer costs due to higher oil prices, and the stability of
both the scrap and part resale markets.

6 CONCLUSION
The ELV Directive has proven to be the catalyst for substantial reform within the vehicle recovery sector, and
has clearly brought EOL stakeholders into the vehicle
value chain. The challenges and pressures of being part
of the extended enterprise have required huge investment
within an industry that has traditionally seen little
intervention from either Government or vehicle
manufacturers. Indirectly charged with the responsibility of
measuring the reuse, recovery and recycling targets laid
down by the directive, the recovery sector has made
considerable headway in fulfilling the 2006 target, but
there is still a long way to go if the 2015 target remains in
place.
To date, the majority of investment, and the inclusion of
environmental operating procedures (de-pollution), have
been undertaken by the EOL operators due to
Government legislation. This will ultimately change once
the directive is in full operation, and further prescriptive
legislation has ceased. Therefore, to continually promote
sustainable practices within the vehicle recovery industry the
economic implications of their inclusions must be
understood. Only then will EOL operators realistically
consider them. The research reported within this paper is
attempting to address this by developing a cost model that
provides economic transparency, and a means of
supporting further value recovery under the constraints of
the current and future legislative targets.
Although still in the intermediary stages of implementation the
ELV directive has been the catalyst for substantial
environmental improvements within the recovery sector, yet
at the same time it has been unable to close the
product life-cycle loop and bring manufacturers closer to
the issues regarding the disposal of their products. As a
means of attributing producer responsibility its aim has
become distorted by the recovery sectors economic
needs, and the manufacturers unwillingness to make
vehicle recovery part of their core competency. Therefore,
future sustainable practices within this industry will always
be promoted and supported by the vehicle manufacturers, but
real change will only come from those EOL operators who
can identify genuine opportunities and rewards.

ACKNOWLEDGEMENTS
Many thanks to D. Allen and R. Kirk of Albert Looms
Derby, for there support and assistance during the ATF
dismantling study. Also thank you to P. Owen and staff of
Rozzone Ltd, for providing additional support and
equipment during the study.

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Appendix 4

Implications of the end-of-life vehicle directive on the vehicle recovery sector

Introduction

This paper was accepted for publication in Proceedings of the Institution of Mechanical Engineers - Part B: Journal of Engineering Manufacture
Implications of the End-of-Life Vehicles Directive on the vehicle recovery sector

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The manuscript was received on 27 September 2005 and was accepted after revision for publication on 14 March 2006.

DOI: 10.1243/09544054JEM473SC

Abstract: To cope with the environmental effects of 9 million tonnes of vehicles that reach the end of their useful lives each year in Europe, the EC have created the End-of-Life Vehicles (ELVs) Directive. Two of the most radical measures included in the Directive are to provide free takeback to last owners and to achieve targeted levels for the recycling and recovery of material by set dates. This paper aims to provide a basis for future research by evaluating the potential direction of the recovery industry. This is achieved firstly by assessing the origins of the directive and previous research surrounding the subject. The paper then describes the current recovery infrastructure and practices in the UK, highlighting all the stakeholders involved in the recovery industry. This paper also highlights the issues related to the provision of takeback and the attainment of targets through two stages, namely the implementation and management of takeback, and the use of new technology to achieve the recovery targets. The paper concludes by identifying key aims for future research to support the objectives of the implemented legislation and the financial stability of all stakeholders.

Keywords: end-of-life vehicles, manual disassembly, shredding, plastic recycling

1 INTRODUCTION

Unlike many products, the recovery of a vehicle through the reuse of its parts and the recycling of many of its constituent materials has existed since its inception. The structure of the car has always encouraged parts exchange and the technology of separating and recycling the valuable ferrous content is simple, reliable, and widespread. However, when the value of scrap steel has fallen, the loss of revenue to the recovery industry has usually forced many scrapyards to charge last owners for the disposal of their vehicle. This has previously caused an increase in vehicle abandonment, with the cost of disposal then falling on local government [1]. The recovery industry has also gained an image of un-environmental conduct through the landfilling of the many hazardous substances within a vehicle [2]. The waste sent from the recovery industry to landfill has been estimated to be between 20 and 30 per cent of each processed vehicle's weight, with a survey in 2000 estimating that from the 2.1 million vehicles recovered in the UK that year, approximately 403,000 tonnes of waste in the form of automotive shredder residue (ASR) was landfilled [3]. The emergence of these three factors; abandonment, pollution, and waste has resulted in the creation by the European Commission of the End-of-Life Vehicles (ELVs) Directive which aims 'as a first priority, at the prevention of waste from vehicles and, in addition, at the reuse, recycling and other forms of recovery of end-of-life vehicles, and their components so as to reduce the disposal of waste, as well as at the improvement in the environmental performance of all of the economic operators involved in the life cycle of vehicles and especially the operators directly involved in the treatment of end-of-life vehicles' [4].

This was to be achieved through the implementation of several measures that include:

(a) the setting up of a system for the collection of ELVs by economic operators (producers, dismantlers, and shredders etc.);
Appendix 4

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(b) the assurance that delivery to treatment facilities is at no cost to the last owner by 2007 (unless it does not contain 'the essential components of a vehicle' or contains waste which has been added);

(c) the establishment of standards for storage, treatment, de-pollution, and the regulation of authorised treatment facilities (ATFs);

(d) the recycling and recovery of 85 per cent (80 per cent recycling) of a vehicle's weight by 2006, and 95 per cent (85 per cent recycling) by 2015.

The initial interpretation of the Directive was that the financial burden of implementing these measures would fall on the original manufacturers, making them liable for the disposal of their product and creating a link between themselves and end-of-life (EoL) operators, described by Deutz [5] as a ‘value chain’. Vehicle manufacturers have been instigating environmental awareness for many years with the use of whole life cycle analysis [6] and ‘design for’ programmes [7] increasing the influence of EoL options on the design process. However, this producer responsibility is aimed at giving them a financial interest in recovery, encouraging them further to integrate EoL issues into design as well as incorporate recycled material into new vehicles.

As a result of the Directive, the old style ‘scrapyards’ now require authorized treatment facility (ATF) accreditation, guaranteeing the environmental treatment of vehicles in their care. They are also required to build new relationships with manufacturers to provide takeback and reassess old relationships with other actors in the recovery chain (as shown in Fig. 1) to achieve the recovery targets together. Through a review of the surrounding literature and discussion with many of the stakeholders involved, this paper considers the possible methods by which the ‘value chain’ can achieve the objectives of the Directive. Interviews have been conducted with manufacturers, ATFs, and shredder operators to build a picture of the current recovery industry and develop an understanding of its future direction, based on the two most radical measures included in the directive; free vehicle takeback and recycling/recovery targets. From this, future research activities have been highlighted that could help sustain a free takeback network and reduce landfilled waste.

2 VEHICLE RECOVERY IN THE UK

As the ELV Directive is implemented at a national level, each nation state will take responsibility for both the introduction and achievement of free takeback and the recovery targets. In Germany 10 per cent of the recovery rate is required through dismantling [8], while in the Netherlands a well-established flat-rate disposal levy is added to the price of new vehicles and invested in the recovery industry by Auto Recycling Netherlands (ARN) to ensure compliance for both free takeback and recovery targets [2]. The UK provides a good example of a moderate and common transposition of the Directive, where additional measures have not been attached, and takeback is the responsibility of each manufacturer. The legislation has been transposed into UK law through the EoL Vehicles Regulations 2003 [9] and the EoL Vehicles (Producer Responsibility) Regulations 2005 [10], and therefore the requirements for ATF status and de-pollution standards have been

![Fig. 1 The flow of the vehicle through EoL operations](Image)
Implications of the ELV Directive on the vehicle recovery sector

Traditionally, vehicles have arrived at scrap dealers because of their involvement in an accident or because they have come to the end of their useful lives (as shown in Fig. 2). Dependent on their age and make, these vehicles are then cannibalized for parts by the scrapyard before the remaining vehicle, normally called the 'hulk', is sold on to a shredder operator who recovers the ferrous content. However, there have been improvements in both the processes used and professionalism within an ATF as depollution and ATF status have become a requirement. Those who have achieved ATF status now de-register the vehicle, issue a certificate of destruction to the last owner, and de-pollute the vehicle, which requires the removal of the battery, fluids, tyres, and any other hazardous substances in a certified environment.

Although there is clearly an economic cost that comes with implementing these measures, many have made the successful transition to ATF status. This has been aided by the high value of scrap steel as depicted in Fig. 3 [11], which has brought increased profits from the sale of the hulk and, therefore, money to invest in the necessary equipment. However, the majority of stakeholders interviewed felt that the spare parts market was in decline, citing increased reliability, an expansion in onboard electronics, and frequent component design changes as the reasons for the downturn.

It is estimated that approximately 79 per cent of a vehicle's weight is recovered currently in the UK, as illustrated in Fig. 4 [3], with around 10 per cent of this removed during de-pollution and dismantling at an ATF. However, there is no financial incentive to dismantle pure stream materials for recycling because of high labour costs and the lack of market for low quantities of non-metallic materials. Although the plastic content amounts to approximately 10 per cent of a vehicle's weight [3], the types of plastic used are varied, sometimes unidentifiable, and difficult to separate and clean.

When the vehicle hulk passes on to the shredder operator, the vehicle is shredded using a hammer mill and then, the ferrous metal (approximately 64 per cent) is removed using magnetic separation. The remaining fraction can then be separated further by using eddy current technology followed by dense media separation, which recovers a further 4 per cent of a vehicle's weight in non-ferrous metals. This leaves approximately 21 per cent to be sent to landfill as ASR.

3 THE FUTURE DIRECTION OF THE UK RECOVERY INDUSTRY

The way that the recovery targets in 2006 and the takeback networks in 2007 are measured and developed will have a major impact on the future prosperity of the recovery industry. In this section the future implementation of free takeback provision

Appendix 4

Fig. 2 ELV categories (a) premature fire, theft, vandalism, or accident ELVs; (b) abandoned ELVs that can fall under either; (c) natural ELVs

Fig. 3 The rise of shredded steel prices per tonne delivered (in Euros) between January 1998 and 2005 [11]
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and recovery targets will be discussed from a UK perspective based on knowledge gained from both literature and interviews with UK stakeholders.

3.1 Free takeback

Unlike the Netherlands, the UK is applying an 'own marque' approach to free takeback. This was developed after lobbying from the manufacturers, who felt that providing 'payment-per-car' recovery for all vehicles in the UK was too great a financial liability to appear on their balance sheets. The 'own marque' system adopted asks manufacturers to set up their own network of ATFs to deal with their vehicles. Due to the high value of scrap steel, many ATFs and shredder operators have been unwilling to give up any potential profit to vehicle manufacturers. This has led to the creation of a contractual agreement referred to by many stakeholders as a 'zero cost' contract. These contracts, as the name implies, see no monetary value exchanged between the automotive and recovery sectors. This free-market approach provides a takeback network that gives independence to the recovery chain at a time of high profit, with the manufacturers contributing to the promotion of the network. However, because of the suggested decline in the spare parts market, and with the cost of landfill set to increase from £18 per tonne to a medium-to-long-term rate of £35 per tonne, these factors are expected to impact on the profitability of ATFs. From the advent of 'zero cost' contracts and the financial barriers that the recovery industry faces, the following summations can be made.

1. 'Zero cost' contracts do not provide a direct financial incentive for manufacturers to increase recovery through design. One of the aims of the Directive was to provide producer responsibility so that vehicle manufacturers would have a financial interest in the recovery of their own vehicles. This would provide an incentive to reduce waste through the redesign of their vehicles, therefore promoting reuse and recycling not as an environmental need but as an economic necessity. Although the ELV regulations include fines for non-compliance, the use of zero-cost contracts provides it without any cost to the manufacturer.

2. A free-market system still leaves the recovery industry susceptible to market fluctuations. A drop in the value of scrap steel combined with an increase in landfill and de-pollution costs, would reduce the profitability of both ATFs and shredder operators. This financial burden would fall wholly on the recovery industry, with the manufacturers not obliged to give financial assistance unless it prevents their recovery network from providing free takeback, therefore incurring a fine for non-compliance.

3.2 Recovery targets

To meet the recovery targets for 2006 and 2015, the UK industry must recover an extra 7 per cent
Implications of the ELV Directive on the vehicle recovery sector

and 17 per cent respectively. As mentioned in section 2, dismantling more material at an ATF is not seen as viable by the stakeholders involved. The removal rates are too low and the amount of recyclable material collected too small to make manual dismantling profitable. Although ‘design for disassembly’ methods have been utilized for newer vehicles to make specific parts easier to remove, ATFs still report that this provides neither the removal time nor the quantity of material to make the removal of pure stream plastics worthwhile. This therefore puts much of the onus for recovery on post-shredder operations and ASR recovery methods.

One potential process is skin floatation, which attempts to separate thermoplastics and thermostets from the remaining ASR through their reaction to plasticizers [12]. After several stages of preparation, the material enters a ‘quiet’ tank where heavier engineering plastics sink and lighter olefin plastics and foams float. The light fraction can be separated further or recycled as thermoplastic olefins and thermoplastic elastomers (TPOs and TPEs) while the heavy fraction, composed of 25 per cent plastics (thermoplastic) and 75 per cent rubbers (thermoset), continues to a counter-current rinse tank. Plasticizers are then added to the tank, which induces air bubbles on the surfaces of certain plastics and forces them to float. It has been found that, using the right sequence of plasticizers, ABS, Nylon, PC, and PP can be removed from the ASR with a purity of at least 92 per cent [12]. However, the widespread adoption of this technology has yet to be commercially realized and doubts remain over the required cleanliness of the plastics for the flotation process to take place.

Another possible method is the gasification of waste, which attempts to separate its combustible particles from large inert and metallic particles by heating the waste on an internally circulating fluidized bed to between 500 and 600 °C. This method has been commercialized successfully in Japan through the TwinRec system, which has processed more than 170,000 tonnes of waste in the first three years of its existence [13]. It takes unsorted and uncleared ASR and, through a combination of a gasifier and a cyclone combustion chamber, separates the remaining ferrous and non-ferrous material, while creating energy through a boiler and construction granulate from the remaining slag. The manufacturers claim that from the 20 per cent of vehicle waste they receive, they are able to recover another 2.5 per cent of the metallic content, 5.5 per cent through recycling as construction materials, 10 per cent through energy recovery, and 1 per cent from metal salts, leaving 1 per cent of the vehicle’s weight for landfill.

From the lack of financial incentive to remove more during disassembly and the new technologies available for post-shredder recovery, the following summations can be made.

1. The achievement of the recovery targets is dependent on post-shredder separation. Because the financial burden will fall on the recovery industry, as discussed in the previous section, they now must find the most economic method of achieving the recovery targets. Manual dismantling is not seen as economically viable by the recovery industry and they see no market to create a financial incentive, therefore, post-shredder separation provides the most economic means of reaching the targets.

2. The technology does not exist to recover post-shredder plastics for closed-loop recycling. Although the two technologies presented in this paper could eventually provide compliance, the purity of the materials separated precludes their use in the same application. Cascade recycling, where they are utilized in lower specification applications, is a potential solution. However, several stakeholders reported that there are currently not enough of these applications to provide a market for the quantity of mixed plastic that could be recovered.

4 CONCLUSION

The review of literature and interviews with many of the stakeholders involved has signalled several key indicators to the future direction of the vehicle recovery industry. In terms of free take-back, manufacturers are beginning to establish networks in the UK through several ‘zero-cost’ contracts. The added costs of de-pollution created by the legislation have been absorbed by the extra revenue created by the high value of scrap steel. However, there is no direct financial aid from the manufacturers, which has left many ATFs as vulnerable to changing markets as they were before the Directive’s inception. These ATFs require guidance to maximize their profits through the development of emerging markets. Although many stakeholders believe plastic removal is uneconomic, a market does exist for recycled polymers that remains unexploited by the automotive recovery industry. If detailed information could be gathered on a limited number of parts on specified vehicles along with the potential value of their material content, ATFs would have the ability to base dismantling decisions on real data.

The achievement of the Directive’s recovery and recycling targets is less clear. There is some
confidence within the recovery industry that the 85 per cent target will be met. The research in this paper indicates that the increase in recovery levels required for both 2006 and 2015 will come from post-shredder operations. Although many EU states have different methods of implementation, the recovery targets are the same across the continent and the responsibility of the manufacturers to help achieve them is clear. Future EU type approval regulation [14] will add to manufacturer responsibility by asking them to provide details of how their new vehicles will be recovered. It is therefore essential that the automotive industry is aware of the impact of their product on these processes so that they can be considered during the design process. This could not only aid material selection, but give the manufacturers an impression of the recoverability of their vehicles. Therefore, the authors' future research will focus on aiding ATFs with dismantling decisions through the use of cost models and assisting manufacturers with design decisions based on post-shredder operations.

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Appendix 5

Implications of the end-of-life vehicle directive on the vehicle recovery sector

Introduction

This paper was presented in the 4th International Conference on Design and Manufacture for Sustainable Development, held in Newcastle-UK in 2005.
End-of-life Recovery of Vehicles in the UK

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ABSTRACT

The automotive industry is among the largest manufacturing sectors and is considered
one of the most resource intensive industries in the world. Over the last two decades,
the car has become one of the most important consumer products. The typical lifespan
of a vehicle has significantly shortened in recent years and is now reported to be
between nine and thirteen years. The disposal of vehicles can have a major
environmental impact, both in terms of waste production and in the recovery of
original materials. To cope with the environmental effects of nine million tonnes of
vehicles that reach the end of their useful lives each year in Europe, the European
Union has created the End-of-Life Vehicles Directive to be implemented in all its
member states. The producer responsibility which this directive advocates suggests a
holistic view of vehicles recovery, with materials recovered at the end of a vehicles
life being used based on a closed-loop approach within successive vehicle design and
manufacture. However, the contemporary market drivers under which the current UK
vehicle recovery industry operates do not have the structure to fully support such an
approach, and more importantly the achievement of the targets within the directive.
This paper provides a snapshot of current practices in vehicle recovery within the UK
together with the legislation, stakeholders and markets influencing this industry. The
paper then outlines the factors that must instigate the longer-term changes required to
more readily support the core themes of the End-of-Life Vehicle Directive.

1 INTRODUCTION

The influence of European legislation is becoming progressively more prevalent
within the UK, with many manufacturers and businesses being forced to be more
accountable for their products environmental effects beyond the traditional boundaries
of the product development process. End-of-life disposal and product take-back
legislation has taken a proactive stance in attempting to make manufacturers more
environmentally aware of their producer responsibilities. EU legislation accounts for
an estimated 80% of UK environmental regulations [1], which have resulted in a
number of prescriptive directives encompassing the design, production and treatment
of a range of industrial and consumer products.

One of the most effected products to date is the automobile via the End-of-Life
Vehicle (ELV) Directive. This directive not only relates to the manufacturers, but to
many other stakeholders involved in the car industry. These stakeholders encompass a
wide variety of fields such as material recycling, manufacture, and End-of-Life (EoL) recovery, each with their own specific concerns with one another and the directive. These concerns can sometimes be counter productive and there are many vested interests which need to be considered within each industry if a environmentally friendly and cost effective solution to current recovery problems is to be realised.

This paper provides a general overview of the current infrastructure and through this highlights current and future drivers that will affect the full implementation of the directive. The initial section provides an overview of related research on the areas involved before outlining the current legislative situation and identifying the stakeholders involved. The main section of the paper provides an overview of the current recovery chain (as shown in figure 1), highlighting the implications of the ELV directive on both manufacturers and EoL operators before identifying the contemporary cost drivers. Three key areas that will have ramifications on the successful implementation of the directive are then reviewed. These include the value of the virgin and scrap materials processed by both manufacturers and dismantlers, the available EoL processing options (mainly related to dismantling versus shredding processes) and the effect of other environmental policy on vehicle design.

2 RELATED RESEARCH

In recent years manufacturers of all products have stepped up their use of environmental methodologies by implementing the analysis of whole life cycle [2] and, in more general terms, increasing the influence of all encompassing environmental management strategies. This has in turn promoted the development of trade-off analysis programs to help compare business and ecological factors [3]. More specific developments have been aimed at the design processes influence on EoL options in the form of various ‘design for’ programs. These vary from high level considerations, where environmental issues are contemplated early in the products development, to low level issues where every nut and bolt is considered [4]. These methods bring the manufacturer much closer to the disposal of their product, creating a link between themselves and EoL operators, described by Deutz [5] as a ‘value chain’. Using the link as a commercial advantage through a ‘closed loop’ infrastructure has been the subject of much research including the development of information systems to aid take-back [6].

Fig. 1 The ELV processing jigsaw with the size of the connector representing the material flow

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One of the potential materials/part recovery routes is dismantling, where numerous attempts have been made to improve the process, from the development of equipment like a chain car turner [7] to whole disassembly lines [8]. This does not stop in the workshop with analysis tools created to assess optimum disassembly sequences using recovery cost and revenue data [9]. The options available for ELV disposal are also the focus of numerous studies in recent years as the recovery sector is fragmented and therefore the information required by so many interested parties is difficult to trace. These range from reports on every sector involvement, from manufacturers to recyclers [10], to specific studies on the factors involved in end of life recovery [11].

More explicit surveys have taken place on individual recovery options which includes a report into the current state of vehicle part reuse [12]. The recycling of automotive plastics is also the topic of a great deal of research with Life Cycle Assessment (LCA) studies on the relative traits of closed loop recycling in comparison with cascade recycling [13]. Alternatively many favour the post shredder recovery of Auto Shredder Residue (ASR) instead of the pre shredder dismantling and recycling of parts. Investigations have taken place into the exact composition of shredder waste [14] whilst others have looked at potential techniques to recover ASR like skin floatation [15].

3 ELV DIRECTIVE AND ITS IMPLICATIONS WITHIN THE UK

The ELV directive came into force in October 2000 with member states given till April 2002 to transpose it into national legislation. However, none of the member state completed the implementation by that date mainly due to difficulties with some of the main points of the directive, which are outlined below:

- Owners must be able to have their vehicles accepted free of charge at a registered Authorised Treatment Facilities (ATFs) for vehicles.
- Manufacturers as of 2007 must pay for take-back and recovery of all negative value vehicles.
- By January 2006 at least 85% (by weight) of all ELVs must be reused and recovered, with 80% reused and recycled (i.e. 5% allowed for energy recovery).
- By January 2015 at least 95% (by weight) of all ELVs must be reused and recovered, with 85% reused and recycled. (i.e. 10% allowed for energy recovery).
- The banning and restricting of certain materials used within vehicles.
- The introduction of coding standards to facilitate materials identification and recovery, along with dismantling information being available to ATFs within 6 months of the vehicle being placed on the market.

The introduction of ELV legislation was not the favoured approach amongst vehicle manufacturers during the Directives formulation. For many years industrial bodies such as the Society of Motor Manufacturers and Traders (SMMT) in the UK, lobbied for self-regulation as opposed to direct legislation, with a number of their reports reflecting year-on-year improvements towards attaining the 85% target. The directive allows member states to adapt their own strategies for vehicle recovery and therefore a number of options for financing the proposed legislation were discussed by the UK Government (e.g. a central fund from tax or vehicle registration), but ultimately the industry favoured “own marque” approach was adopted which attempts to place the
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cost burden at the feet of the producers. This will see manufacturers establish their own contracted networks of ATFs which will deal with their own returned vehicles. Hence from 2007, all ELVs returning via a manufacturers contracted network will be a financial liability. As to whether this liability is to be present on a manufacturer’s balance sheet is still an area of contention. This has lead to a number of vehicle manufacturers advocating “zero-cost” contracts with Shredders and ATFs for the return of vehicles based on the value of contemporary market drivers. In this instance, all costs associated with de-pollution and target attainment would be offset by the scrap value of the hulk and the spare parts removed.

At the time of writing, 838 UK facilities are currently registered with the environmental agency as conforming with the site licence requirements of a registered ATF [16], and have made the investment required to de-pollute vehicles. The establishment of an ‘own marque’ contracted network is still under discussion between the vehicle manufacturers and the various ATFs. In terms of the legislation these stakeholders are in a unique position. They retain the tools to carry out the legislation (and assist the producers) but are not financially liable for its successful implementation. They also retain any profit made from the processed ELVs, and in the current climate of high scrap metal prices this has been used to persuade ATFs to sign zero cost contracts, guaranteeing their survival and providing a certain level of vehicle returns. Unlike tiered suppliers within the vehicle manufacturers supply chain, ATFs cannot be as easily influenced to accept future cost burdens. This necessitates a clear deviation from the traditional approach of passing the cost back down the value chain, as the pro’s and con’s of being a contracted ATF still vary depending on the size of the operation.

The process of implementing the ELV directive is a long and arduous task, as evident by considering the fact that not one nation state completed the transposition of the directive by the deadline. The UK government has chosen the ‘own marque’ approach, mainly due to intense lobbying by the manufacturers, which in contrast to the vehicle levy system employed in the Netherlands is unlikely to drastically improve the current recovery structure. It is argued that although this is a good free market solution, it will not change attitudes within many of the stakeholders involved and leaves the manufacturers with enough flexibility to avoid facing the problem head on. This is compounded by the high value of scrap metal, which is encouraging ATFs to sign ‘zero cost’ contracts. However, this could place the entire vehicle recovery industry in jeopardy if the contracts are not connected to the price of scrap steel.

4 THE CURRENT INFRASTRUCTURE AND RECOVERY DRIVERS

4.1 The UK vehicle recovery infrastructure

Before discussing the contemporary drivers within the ELV market it is necessary to provide an overview of the main processing stages and the stakeholders responsible for them. ELVs can be categories into two main groups, natural and premature (see figure 2). As the name suggests premature vehicles have come to the end of their useful life before their average lifespan, either due to fire, theft, flood, vandalism or accident damage. The majority of these vehicles are insurance write-offs and often have a wealth of reusable parts removed before further processing. Natural ELVs however have come to the end of their useful lives (usually 9-13 years) and are either
returned to a treatment facility via a collection merchant, collected by the ATF itself or returned by the vehicles last-owner. Natural ELVs tend to be in a bad state of repair, as prolonged use and “wear and tear” has taken its toll over the years. Parts resale value is therefore at a minimum, and often a number of health and safety issues need to be addressed before de-pollution and further processing.

Fig. 2 ELV Categories,
Left - Premature fire, theft, vandalism or accident ELVs, Right – Natural ELVs, Centre – Abandoned ELVs which can fall under either.

Once the vehicle has arrived at the ATF the Vehicle Identification Number (VIN) is recorded and presented along with its registration document before the vehicle is deregistered and a certificate of destruction issued. It is then de-polluted which requires the removal of the battery, fluids, tyres and any other hazardous substances, and are then collected and processed by a waste management company. Specially designed de-pollution rigs support the vehicle during this exercise, and the process typically takes around 15 minutes per ELV. High value components are then removed via manual disassembly which has also seen a number of smaller facilities separate pure stream plastics (such as bumpers) to sell directly to the recyclers and re-processors. However, the wide spread adoption of these techniques, due to its labour intensive nature, is yet to be implemented within the industry. The majority of natural ELVs are crushed and transported to shredding operations for post-fragmentation recovery. Once the ferrous content has been recovered (approximately 72% [11]) the non-ferrous scrap can be separated using Dense Media Separation processes, and the remaining waste is sent to landfill (see figure 3).

Fig 3 An overview of the current UK vehicle recovery infrastructure
4.2 Contemporary drivers in the UK ELV market

The major contemporary drivers that determine the economics within the UK recovery infrastructure can be summarised as:

- **Scrap metal prices**: Recovery technology for scrap metal is relatively cheap and well established throughout the UK. Ferrous materials still represent the bulk composition of ELVs, hence quantity can be guaranteed based on a small amount of processing. The market value of scrap ferrous combined with the low cost logistics of exporting to countries such as China, has created an over-riding cost driver in the vehicle recovery industry.

- **Spare parts markets**: Premature ELVs are the primary source for parts reuse, with evidence to suggest that newer makes and models provide the greatest sources of revenue [12]. Although spare parts have a long established history in the UK, general trends and interviews would suggest that the market for second hand parts is in decline. Possible reasons for this include the shorter life of the parts/components, reduced compatibility between integrated electronic parts, and a lack of hobbyists.

- **De-pollution costs**: Despite the value of many of the materials removed during pre-treatment (e.g. the lead within batteries), many are not sold directly to re-processors due to lack of the economies of scale. Many ATFs currently pay waste management companies to collect and process these materials, with the cost of disposal being offset by the high value of the scrap.

- **Auto plastic prices**: The removal and segregation of pure stream plastics at the dismantling stage is currently not wide spread given the labour intensive nature of the work and the lack of established secondary markets. The alternative post shredder processing route therefore requires more advanced technologies and investment to ensure the resulting plastics purity and revenue.

- **Landfill taxes**: The standard landfill tax rate is currently £18 per tonne and is set to rise by £3 per year thereafter, moving towards a medium to longer term rate of £35 per tonne. This tax will become an increasingly influential economic instrument over the coming years.

These drivers are clearly influenced by one another, for example the increased regulation of de-pollution and landfill taxes has had a major impact on the working standards of the sector. This financial outlay has been offset by the high value of scrap steel, which continues to be the industries main driver. In addition, with the spares market in natural ELV parts in terminal decline, for the long term stability of the recovery sector, there is a need to establish other key material markets (e.g. plastic, ASR). This is unlikely with the current lack of post-shredder infrastructure, and unless the advancement in technology makes this financially viable, the recovery industries reliance on an unpredictable scrap metal market will continue.

5 ANALYSIS OF THE LONG TERM STRATEGIC VIEW

There are three major factors that have been identified which influence the long term strategic development in the UK ELV infrastructure, these are:
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- Material value and depletion
- End-of-life recovery options
- Environmental design factors

These factors are all interdependent, essential to the future of vehicle recovery and are discussed in more detail in the remaining sections of this paper.

5.1 Material value and depletion

Intrinsic to the long term view of any products manufacture and disposal are the materials used within it. This is especially significant in the car industry where steel and plastic, representing a significant proportion of a European vehicles composition, currently have high global raw material values. In the case of steel, there has been increased consumption worldwide, particularly in China which has driven up the value of both iron ore and scrap steel. This has naturally had a major effect on the vehicle recovery industry which has seen the value of shredded steel rocket, as shown by figure 4 [17]. The price of scrap is generally very volatile because of its finite quantity and the supply/demand balance which is rarely predictable. Despite this instability, world steel production is continuing to grow and there is sustained optimism that this will reflect on the value of scrap well into the future. This may increase the cost of car production and the profitability of vehicle recovery, encouraging manufacturers to make better use of their metal content and become directly involved in end-of-life recovery.

![Average quarterly shredded scrap steel price](image)

**Fig. 4 The rise of shredded steel prices per tonne delivered (in Euros) between January 1998 & 2005**

A similar case can be made for plastics which have also seen an increase in the cost of raw materials. Suppliers of automotive plastics are demanding a higher price for their products and manufacturers are already investigating other options. For example, Toyota have begun to use ‘Eco-Plastics’, derived from raw materials like sugar cane and corn, in low specification application [18]. Since a pilot scheme in 1993, Ford has been recovering their Xenoy resin bumpers in the United States. The material is used in a variety of Ford parts, including bumpers, and is said to save the company approximately $1 million a year [10]. Recycling is a sector not fully exploited by
manufacturers and if the legislative and economic pressures are present, the development of natural plastics and the increased recycling of plastic components could become a reality.

5.2 End-of-life recovery options

Another fundamental problem involves the method with which the recovery targets within the ELV directive will be met. This could either be through an increased efficiency in recovering materials post-shredder (i.e. through more effective separation techniques) or through the increased dismantling of parts pre-shredder. At present the majority of the ATFs in the UK only dismantle parts if there is a legislative or economic reason to do so as the cost of dismantling far outweighs the value of parts. The additional dangers to health and safety when entering an ELV (which could contain shards of glass, needles etc.) make any internal dismantling virtually impossible (see figure 5). Hence, the industry is currently siding with recovery through the shredding process, which creates an unnecessary energy outlay and currently sends ASR to landfill. The reliance on the shredding process is because design is the only factor that would notably change the economic realities of dismantling. Vehicles currently in show rooms will reach the end of their useful lives between 2018 and 2020 and are only marginally easier to dismantle than current ELVs. Unless there are significant economic incentives in place by that time, there is little scope to encourage vehicle manufacturers to significantly change their design to facilitate dismantling.

Fig. 5 The condition of a typical ELV

It is also difficult to account for processed material after shredding as domestic appliances (such as washing machines) are shredded alongside vehicles, forcing the use of protocols based on shredder trials allowing an easier route to target attainment. As ASR has a substantial calorific value there has been considerable investment in improving energy recovery techniques as well as pilot schemes to improve separation techniques. However, there is little argument that dismantling can provide a much higher grade of recycled material to replace virgin material, saving manufacturers money (as shown by Fords bumper program) and providing a ‘closed loop’ in line with the original aims of the ELV directive.

5.3 Environmental design factors

Changes in design and material use are essential to the future success of ELV recovery and these have gradually come into conflict with other crucial environmental factor during the use phase of the vehicle. The use of light weight materials such as composites, though at present potentially hindering material recycling, are seen to be more environmentally beneficial. This is because they reduce the weight of the car which in turn reduces fuel emissions and increases fuel economy. It is therefore both an economic and environmental benefit to increase the use of lightweight materials in
vehicles. In economic terms, improving the vehicles economy can act as a significant marketing ploy for the manufacturers in comparison to improving its EoL recovery which is of little consequence to first owners. Additionally in environmental terms, using LCA studies have indicated that a passenger car will use 83.5% of the energy used throughout its life cycle during the use phase and just 0.1% during the recovery phase (recycling and waste) [19]. Due to both the aforementioned economic and environmental benefits for the use of lightweight material, many expect the recyclability of European cars to gradually decrease in the short term. Even the use of aluminium, that has a higher value at end of life and is more recyclable than steel, could reduce the overall weight percentage of recovery, which is detrimental to the achievement of the ELV targets. It is therefore claimed that in the long term it is in the manufacturers interest to become involved in recovery through the incorporation of environmental design factors which in turn facilitates closed loop recycling, and truly adhere to the aims of the directive.

6 CONCLUDING DISCUSSIONS

The main aim of the ELV directive was for environmental improvement and in the UK this has been partially achieved in the UK, although not to the extent of some other European nations. It has brought uniformity to the existing infrastructure resulting in significant improvements in treatment facilities that were difficult to control before its implementation. However, the producer responsibility contained within the directive has been influenced by the automotive sector, creating an adaptation of the directive purely driven by economic feasibility as opposed to environmental merits. This has satisfied the manufacturers requirements in the short term, but if the economic conditions change they could be left with a significantly diminished contracted network, the implications of which could have serious repercussions on the stability of the sector. This could force manufacturers out of necessity to take a closer involvement in vehicle recovery.

The UK ‘own marque’ approach has not instigated change in post de-pollution recovery routes, with the current system driven by the existing infrastructure and material markets. This has resulted in little financial incentive to improve dismantling or many post-shredder processes, hence creating an obstacle for achieving significant environmental benefits. The research reported in this paper has highlighted a need for further work into the feasibility of both pre and post-shredder recovery options, exploring viable paths that could increase recoverability. Although manufacturers have increased design for disassembly efforts over the past decade, further research is also necessary into new concept of ‘design for separation techniques’(where separation technique may apply to both pre- and post-shredder processes), which constitutes the next phase of the authors work.

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