THE ENVIRONMENTAL IMPACT OF END-OF-LIFE VEHICLE LEGISLATION AND VEHICLE USE IN EUROPE AND NORTH AMERICA

by

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ABSTRACT

This 'manuscript-based' thesis investigates the environmental consequences of current practices and recent developments in the automotive industry. The first manuscript (Chapter 2) analyses the effect of the European End-of-Life Vehicle Directive (2000) on the environmental performance and level of 'green' innovation in the European automotive industry. The research methodology consists primarily of a review of publicly available academic, governmental and commercial literature. The results show that legislative factors and market forces have led to innovations in recycling, increased hazardous substance removal and improved information dissemination. Such actions may be sufficient to reach ELV Directive targets and could have spill-over benefits to other industries. Carmakers are also taking steps to design for recycling and for disassembly. However, movement towards design for re-use and remanufacturing is limited. This research also highlighted the lack of knowledge about the major economy-wide heavy metal releases resulting from car use. The analysis behind the second manuscript (Chapter 3) was conducted in order to fill this information gap. The methodology involved using Economic Input-Output Life Cycle Assessment in conjunction with other life cycle techniques to assess the economy wide releases of lead, cadmium and their compounds during the life cycle of an average light-duty vehicle in the United States. The results show that lead and cadmium are released into the environment primarily during manufacturing of the original vehicle and replacement parts. Loss of wheel balance-weights during use, battery recycling inefficiencies and lead emitted during disposal of other end-of-life parts also contribute significantly to lead discharges. Consequently, mitigation efforts should focus on minimising releases from metal mining, maximising collection and recycling efficiencies of lead-acid batteries, implementing alternatives to lead wheel weights and minimising the lead content of other components. A significant portion of releases resulted from sources other than the lead and cadmium contained in the car. Thus legislating heavy metals out of vehicles will not eliminate all the lead and cadmium emitted due to automobile use. Accordingly, these manuscripts reinforce the importance of developing environmental management strategies that reflect the economy-wide impacts of vehicle manufacture, use and recovery.
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CO-AUTHORSHIP STATEMENT

Dr. Kandlikar co-authored the manuscripts presented in Chapters 2 and 3 of this thesis. In both papers, Dr. Kandlikar contributed to the research design and to the manuscript review.

My contribution to the manuscripts presented in Chapters 2 and 3 and the overall thesis included:

• identification and design of the research program (with guidance from Milind Kandlikar)
• research
• data analysis
• the literature review for all manuscripts and additional information in this thesis
• manuscript preparation, review and editing
• formatting for journal submission
• Manuscript revision based on comments provided as part of the journal review process
INTRODUCTORY CHAPTER

This thesis investigates the environmental implications of automotive waste. The 'manuscript-based thesis' format (specified by the Faculty of Graduate Studies) was adopted to utilise the two completed journal manuscripts in which the research is documented. The first manuscript, contained in Chapter 2, analyses the implications of the European End-of-Life Vehicle Directive (2000) on the environmental performance of the European automotive industry. The analysis revealed the lack of existing knowledge about life-cycle heavy metal releases resulting from vehicle use. The study uncovered little evidence that the economy-wide implications of vehicle use were well understood, especially with respect to heavy metal discharges. Consequently, this topic became the focus of the second manuscript, included as Chapter 3, which evaluates lead and cadmium releases resulting from vehicle use in the United States. Both papers have been submitted to peer-reviewed journals. The Journal of Cleaner Production has accepted the Chapter 2 manuscript for publication. The Chapter 3 manuscript has been recently submitted to Environmental Science and Technology.

This introduction provides an overview of the research topic and the context for the following chapters. It includes a brief summary of automotive waste, extended producer responsibility legislation and life cycle analysis. This information is supplementary to that which is presented in chapters 2 and 3. A discussion of the research objectives and hypotheses concludes this section.

The Automotive Industry and its Waste

The volume of cars produced annually is substantial and continues to expand quickly. The importance of effectively managing automotive waste will continue to grow as the cars being produced are eventually retired. By 2002, the stock of vehicles worldwide (excluding buses and trucks) was over 500 million vehicles and was growing at roughly 3 percent [1]. This growth is likely to continue well into the future. The number of light-duty vehicles is anticipated to expand by 300-500% over the next 50 years [2]. During 2002, automobile manufacture exceeded 58 million units globally (excluding commercial vehicles such as trucks) [3]. Of these, North America produced over 16 million vehicles (2.6 million in Canada, 12.3 million in the U.S. and 1.8 million in Mexico) [4]. Europe assembled a similar number [3]. Consequently, motor vehicle manufacture, including parts and equipment, accounts for a large proportion of manufacturing value in both Europe and North America (12% of Canadian GDP and over 10% of both EU and US manufacturing value) [3-5].
The treatment of a vehicle at its end-of-life has a substantial impact on its overall environmental impact. Economic and environmental considerations have lead manufacturers to re-use and recover old car parts and thereby reduce waste. However, determining the optimal recovery strategy for end-of-life vehicles ('ELVs') is not an easy task and poses a number of strategic and logistical puzzles for manufacturers. A variety of recovery methods exist. Parts may be re-used, remanufactured, recycled, used as fuel for energy recovery or land filled. Each process has different environmental and economic impacts. The attractiveness of a given recovery option is also dependant upon legislative requirements, the prevailing market dynamics and the characteristics of the product itself.

Product Recovery and Take-back Legislation

Taking responsibility for products at the end of their useful life is becoming a necessity in several industries around the world. This movement is being driven by a broadened understanding of the environmental impacts of waste, which in turn has led to the establishment of numerous legislative measures targeting waste minimisation, including ‘take-back’ legislation. As the name suggests, take-back legislation requires the producer to take back its products at the post-consumer-use stage in order to minimise environmental impact (usually through re-use, recycling or recovery). Such regulations fall under the umbrella concept of Extended Producer Responsibility ('EPR'). Figure 1.1 outlines this schematically.

Figure 1.1: Take-back Legislation and the Product Value Chain

The OECD defines EPR as "a policy concept in which a producer's physical and/or financial responsibility for a product is extended to the post-consumer phase of the product's lifecycle" [6]. The features of this policy are twofold, firstly, to provide an incentive for producers to incorporate environmental considerations into the design of their products and secondly, to shift the responsibility for the end-of-life product (physically and/or financially) upstream to the producer and away from municipalities.
Take-back legislation is already having a direct impact on recovery strategies throughout the EU as well as in Japan, Taiwan and Korea. Many states in North America also have extended producer responsibility legislation in place.

European environmental policy is in transition from waste control to a more comprehensive approach, whereby environmental protection will be integrated broadly into all policy areas [7]. The most recent Environmental Action Plan outlines two main objectives. Firstly, to ensure that the consumption of renewable and non-renewable resources does not exceed the carrying capacity of the environment, and secondly to achieve a de-coupling of resource use from economic growth through significantly improved resource efficiency and the reduction of waste [7]. European take-back legislation in both electrical equipment and automotive industries has stemmed from the objectives of the EU’s Environmental Action Plans.

Due to the volume and toxicity of vehicle waste, end-of-life vehicles remain a key focus of European waste management efforts. As early as 1989, the European Commission’s ‘Community Strategy for Waste Management’ identified ELV as a ‘priority waste stream’. While automobile shredder residue (the roughly 25% of an ELV that is shredded after the car has been dismantled and the metal content recovered) constitutes less than 1% of the total waste generated in the EU, it is estimated to account for 10% of EU hazardous waste [8].

European automotive take-back legislation was enacted by the European Parliament on the 18th of September 2000, in the form of the End-of-Life Vehicle Directive [9]. It aims to prevent waste from end-of-life vehicles and to protect the environment through promoting the collection, re-use and recycling of their components. Central to the Directive are the joint goals of restricting both the increasingly expensive landfill operations and the burning of waste, typically called ‘thermal recovery’ by the industry [10]. Take-back legislation has more recently been imposed across the European Electrical Industry as well. The Waste Electrical and Electronic Equipment (WEEE) Directive was introduced in 2003 and was modelled on the ELV Directive.

Pollution and waste legislation is also starting to appear in North America. The Californian Zero Emission Vehicle mandate (ZEV), which seeks to drive innovation of low emission vehicles, has prompted similar mandates in Vermont, Massachusetts and New York [11]. Regulations in New Jersey, Florida and Minnesota require rechargeable battery manufacturers to take back and manage the batteries they produce. At least ten states have introduced electronics recycling and deposit schemes [12]. In total, extended producer responsibility bills have been introduced in nearly half of the 50 state legislatures in the United States [13].
There is no national ELV legislation in North America (as discussed in Chapter 3). Yet, car production is becoming increasingly global, thus the North American automotive industry is being affected by international regulations. Vehicle sub-systems are often shared among vehicle models and combine to form what is known as a ‘vehicle platform’. Once a design is altered it may be reflected in car production globally. It is estimated that one third of all vehicles produced in 2004 will come from global platforms. A recent report has found that the EU ELV Directive is influencing activities in the US automotive market. In particular, the ELV Directive’s aim to remove toxic and hazardous substances such as lead, mercury and cadmium have resulted in international efforts to eliminate their use in vehicle manufacturing [14].

The Role of LCA in Analysing Automotive Waste

Improving the treatment of ELVs is beneficial, yet a vehicle’s environmental performance is not determined solely by the extent of end-of-life vehicle recovery. Automotive manufacturers and regulators must consider the ecological and human health implications throughout a vehicle’s lifecycle. This includes manufacturing waste, emissions released while the car is on the road as well as waste generated from end-of-life vehicles. Life-cycle Assessment (‘LCA’) is often used to analyse the cumulative impact of these releases.

Life-cycle thinking is grounded in evaluating the environmental effects of a product, service or activity during its life cycle. The life cycle of a product typically includes the extraction and processing of raw materials, followed by manufacturing, transportation and use, and ending with waste management including recycling and final disposal [15]. A life cycle approach prevents a piecemeal attitude to environmental management. It enables a dialogue about the total impacts (usually environmental impacts) of a product over its entire lifespan, rather than looking solely at one step of the production chain.

Life-cycle Assessment has evolved to enable industry to operationalise the life-cycle concept. The United Nations Environment Program defines LCA as an analytical tool for the systematic evaluation of the environmental aspects of a product or service system through all stages of its life cycle [16].

The first studies of waste disposal, energy use and raw material use date back to the late 1960s and early 1970s [17]. One of the first analyses, undertaken by Coca Cola in 1969, investigated resource consumption and environmental releases associated with soft drink containers. Initially companies used LCAs to evaluate product alternatives with respect to specific criteria. LCAs were also utilised to highlight problem complexity in defence against externally enforced environmental requirements [18]. Life Cycle thinking and LCA continue to be used in an
increasing number of applications and contexts [19]. Some surveys indicate that roughly half of
the large companies in Northern Europe and the US conduct LCAs for their products [18].
Significant resources are being dedicated to making LCA accessible and useful in industry and
policy [20].

LCAs typically consist of four components: goal definition and scoping, inventory analysis, impact
assessment, and interpretation [21]. The scoping phase is used to set the boundaries for the
analysis. Following this, an inventory of the inputs and outputs at each stage in the product life
cycle is created – the inventory analysis. Once this data has been acquired, the effect on the
environment of each pollutant is evaluated (the Impact Assessment). The results are then
interpreted in relation to the question at hand in order to guide the actions of decision makers.

Various forms of LCA exist. The most appropriate form depends on the scale of the problem, the
audience and the scope of the analysis being undertaken. Each type of LCA has its benefits and
drawbacks. For instance, Economic Input Output LCA ('EIOLCA') is typically used for industry­
wide analysis, but is less amenable to product design comparisons [22]. Alternately, Screening
LCA (a simplified LCA methodology) is often used in environmental labelling to identify the
environmental “hot spots”, i.e. the criteria for which labelling efforts are assumed to have the
greatest effects [17]. The data and time requirements for an LCA depend on the form and scope
of the analysis, but are usually significant. LCAs can cost thousands to millions of dollars [17].

Notwithstanding the profusion of different techniques available, there are ongoing efforts at
national and international levels to establish common databases and methodologies for LCA.
Robust and useful LCA tools and data are provided by the U.S. EPA LCAccess System portal
[21]. Additionally, the ISO14000 Environmental Management framework now includes a trusted
LCA methodology detailed in the standards ISO14040-14049 [23]. However, it is worth noting that
such standards do not regulate every methodological choice, and as a result, allow for the
production of virtually any LCA result [24].

In order to keep LCA analysis manageable, practitioners generally limit the scope of the analysis
to the major inputs at each stage. The process of setting bounds (scoping) is viewed by some as
arbitrary and indefensible, making it impossible to know whether the boundaries that are drawn
encompass all important effects [19]. The subjectivity of defining assessment boundaries can also
make it difficult to compare LCA studies [25]. Notably, some LCA methodologies, such as
EIOLCA (used in Chapter 3) circumvent such problems by manipulating models of the whole
economy [22].
Research Focus and Hypotheses

The manuscripts contained in this thesis focus upon the environmental impact of vehicle use and the manner in which recent European legislative developments have altered ELV processes. The first manuscript presented in Chapter 2 evaluates the extent to which the recent European ELV Directive has influenced vehicle design, the level of ELV recovery, and the extent of 'environmental information' provision. A comprehensive literature review, including journal papers, news articles and government, industry and not-for-profit reports, was conducted in order to test the hypotheses that the ELV Directive has resulted in:

Design changes:
1. Changes in the material composition of new cars
   i. Increased use of recyclable and environmentally beneficial materials
   ii. Increased use of recycled material
   iii. Removal of 'banned' substances (lead, cadmium, mercury and hexavalent chromium)
2. Increased 'design for disassembly, re-use and remanufacture'

Changes in the extent of ELV recovery:
3. Increased levels of re-use and remanufacture
4. Increased levels of recycling of ELV materials

Improved information provision:
5. Provision of the following information:
   a. Part coding standards
   b. Disassembly processes, disposal and recovery of vehicle parts
   c. ELV environmental performance, targeted at vehicle users/purchasers

The above analysis highlighted the need for further research into the interdependencies between the car use and other industrial activities, such as metal mining and infrastructure provision (e.g. road building). It remained unclear to what extent removing heavy metals from cars would reduce the economy-wide heavy metal burden of vehicle use. As a result, the second manuscript evaluates the lead and cadmium emissions resulting from vehicle production, use, maintenance and end-of-life vehicle recovery in the United States. The paper uses life-cycle techniques (with a focus on EIOLCA) to determine the key drivers of these releases.
References


CHAPTER 2: THE IMPACT OF THE EUROPEAN END-OF-LIFE VEHICLE DIRECTIVE ON ‘GREEN’ INNOVATION AND VEHICLE RECOVERY

NOTE:
A version of this paper has been accepted for publication. Gerrard J., Kandlikar M. Is European End-of-Life Vehicle Legislation Living Up to Expectations? Assessing the impact of the ELV Directive on ‘green’ innovation and vehicle recovery, Journal of Cleaner Production, 2006. Awaiting publication.

1. Introduction

Since its introduction five years ago, the global automotive industry and durable goods manufacturers in general have been carefully monitoring the effects of the European Union’s End-of-Life Vehicle (ELV) Directive. The legislation aims to increase recovery of ELVs in order to reduce waste and improve environmental performance.

This analysis presents a framework for assessing the level of environmental performance generated by ELV regulations. It evaluates progress on five expected outcomes of European ELV legislation. While much of the literature has focused on the effects of the ELV Directive on recycling, the directive aims to generate environmental gains through recovery in general. Thus, each recovery option (e.g. recycling, re-use, energy recovery) should be viewed as a means of moving toward environmentally sustainable production, rather than as an end in itself. Accordingly, this paper hopes to broaden the discussion of ELV legislation away from a narrow focus on recycling to one that incorporates green innovation and other vehicle recovery alternatives.

The ELV Directive may have heralded the start of a new era of waste management legislation for durable goods world-wide. The EU has already reinforced its intentions through the introduction of further regulations. Enacted in 2003, the Waste Electrical and Electronic Equipment Directive [1] was modelled on the ELV Directive. Japan, Taiwan and South Korea have instituted similar Extended Producer Responsibility (EPR) legislation over the past three years. EPR legislation is also becoming increasingly prevalent in North America. EPR bills have been introduced in nearly half of the 50 state legislatures in the United States [2]. Thus insights gained from evaluating the early impact of ELV may provide an indication of the future efficacy of EPR regulations worldwide. Additionally, such regulations may have ramifications for vehicle design and production globally as carmakers and regulators learn from the European experience.
2. EU Take-back Legislation and ELV Targets

The ELV Directive [3] came into force on the 18th of September 2000. It aims to prevent waste from end-of-life vehicles and to protect the environment through promoting the collection, re-use and recycling of their components. The directive states that vehicle manufacturers and material and equipment manufacturers must meet the following objectives:

1. Endeavour to reduce the use of hazardous substances when designing vehicles
2. Design and produce vehicles which facilitate the dismantling, re-use, recovery and recycling of end-of-life vehicles
3. Increase the use of recycled materials in vehicle manufacture
4. Ensure that components of vehicles placed on the market after 1 July 2003 do not contain mercury, hexavalent chromium, cadmium or lead (with a few exceptions as listed in Annex II of the Directive).

Currently 75-80% of each end-of-life vehicle is recycled or re-used, the vast majority of which is ferrous metal, as shown in Figure 2.1 [4]. The Directive requires an increase in the rate of re-use and recovery (which includes energy recovery), as outlined in Figure 2.2. Less stringent objectives may be set for vehicles produced before 1980. Professional importers of foreign vehicles are also required to comply with the above. To ensure that the 2015 recovery target is met, the Commission of European Communities has recently proposed that future vehicle approval be contingent on the vehicle's ability to be 95% re-usable or recoverable. Such approval procedures will apply to vehicles put on the market 3 years after the new Directive enters into force (i.e. not before 2007) [5]. Salient dates for European ELV legislation are represented in Figure 2.3.

Figure 2.1: Breakdown of a Passenger Vehicle

<table>
<thead>
<tr>
<th>Material</th>
<th>% by Weight</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ferrous Metal</td>
<td>68.3%</td>
</tr>
<tr>
<td>Plastics</td>
<td>9.1%</td>
</tr>
<tr>
<td>Light Non-Ferrous Metal</td>
<td>6.3%</td>
</tr>
<tr>
<td>Tyres</td>
<td>3.5%</td>
</tr>
<tr>
<td>Glass</td>
<td>2.9%</td>
</tr>
<tr>
<td>Fluids</td>
<td>2.1%</td>
</tr>
<tr>
<td>Rubber</td>
<td>1.6%</td>
</tr>
<tr>
<td>Heavy Non-Ferrous Metal</td>
<td>1.5%</td>
</tr>
<tr>
<td>Other</td>
<td>1.5%</td>
</tr>
<tr>
<td>Battery</td>
<td>1.1%</td>
</tr>
<tr>
<td>Process Polymers</td>
<td>1.1%</td>
</tr>
<tr>
<td>Electrical/Electronics</td>
<td>0.7%</td>
</tr>
<tr>
<td>Carpet</td>
<td>0.4%</td>
</tr>
</tbody>
</table>

Source: [6]
Figure 2.1 shows the breakdown of an average car. Close to 100% of the steel within a vehicle is recycled. Due to their economic value, nearly 100% of car batteries are also already collected and recycled [7]. Additionally, the vast majority of tyres are recovered. The EU is on track to meet the objective to abolish the land filling of old tyres by 2006 [8]. Shredding facilities process crushed ELVs and other scrap metal-rich feedstock, such as white-goods. Seventy percent of shredder output is shredded steel, 25% is ‘shredder fluff’ and the remaining 5% is referred to as ‘heavy media’. Shredder fluff is made up on foam and a range of lightweight non-metallic materials, such as plastic and composite products that are difficult to recycle. This fluff is typically disposed of to landfill, although efforts are underway to develop methods of identification, separation, washing and recycling. The ‘heavy media’ is a mixture of non-ferrous materials and dense non-metallic material including rubber and concrete. This heavy fraction is sent for further processing at heavy media plants where copper, aluminium, magnesium, glass and some plastics are removed.

Figure 2.2: Recovery requirements for vehicles produced after 1980 by weight

<table>
<thead>
<tr>
<th></th>
<th>Current</th>
<th>2006</th>
<th>2015</th>
</tr>
</thead>
<tbody>
<tr>
<td>Re-use and recovery</td>
<td>75-80%*</td>
<td>85%</td>
<td>95%</td>
</tr>
<tr>
<td>Re-use and recycling</td>
<td>80%</td>
<td>85%</td>
<td></td>
</tr>
<tr>
<td>Implied allowable energy recovery</td>
<td>5%</td>
<td>10%</td>
<td></td>
</tr>
</tbody>
</table>

* [4]

NOTE: Energy recovery involves using the waste material to generate energy. This often involves utilising the heat generated from combustion of waste.

The ELV Directive requires a 5 – 10% increase in recovery from current levels by 2006, and a 15-20% increase by 2015. Such improvements need to come from the 20-25% of the vehicle that is not currently recycled. This non-recycled component consists mainly of polymers, rubber, glass and electronic parts (most metals, including batteries, are already recycled). To reach the 2015 targets roughly half of these materials will need to be recoverable or vehicle material composition will need to shift toward materials that are already recyclable. As plastic comprises the largest proportion of the non-recycled component it is the logical focus of much of the current R&D directed toward recycling.

To aid recycling efforts, vehicle and component manufacturers are required to use material coding standards, which allow identification of the various materials during dismantling. Additionally, vehicle manufacturers and importers must provide prospective purchasers of vehicles with
information on the recovery and recycling of vehicle components, the treatment of end-of-life vehicles and progress with regard to re-use, recycling and recovery.

Figure 2.3: ELV Legislation Timeline

2.1 What Should We Expect will Happen?
If legislation has been effective we should see progress toward the objectives outlined above. Under this scenario one could reasonably expect the legislation to have resulted in the following changes (“expectations”):

**Design changes:**

1. Changes in the material composition of new cars
   i. Increased use of recyclable and environmentally beneficial materials
   ii. Increased use of recycled material (“recyclate”)
   iii. Removal of “banned” substances

2. Increased ‘design for disassembly, re-use and remanufacture’

**Changes in the extent of ELV recovery:**

3. Increased levels of re-use and remanufacture
4. Increased levels of recycling of ELV materials
Improved information provision:

5. Provision of the following information:
   a. Part coding standards
   b. Disassembly processes, disposal and recovery of vehicle parts
   c. ELV environmental performance, targeted at vehicle users/purchasers

For each expectation we provide (i) an analysis of the evidence needed to test outcomes, and (ii) an evaluation of the extent to which observed data provides empirical support for that expectation. In section 3 we examine if the available data is sufficient to establish whether change has occurred. In section 4 we assess the extent to which the five expected outcomes have materialized in the aftermath of the ELV directive. Conclusions are provided in section 5.

3. What is the Evidence?

So, what would constitute compelling evidence that the expectations above are being met? In general, transposition of the ELV Directive into member state legislation has occurred too recently to enable statistically rigorous quantitative analysis\(^1\). This partially explains the absence of such analysis to date, notwithstanding the abundance of existing commentary. Yet, an aggregation of publicly available information can provide an emerging picture of future outcomes. Such evidence is most compelling when either a large number of smaller European car manufacturers or a number of larger parent company manufacturers are moving in the same direction. Here 'parent company' refers to a company that owns and operates a number of smaller subsidiary car brands. For instance, Volkswagen is the parent company of the following brands that it manufactures: Volkswagen, Audi, Bentley, Bugatti, Lamborghini, SEAT and Skoda [10].

Our analysis uses data on European carmakers including company reports and websites, assessments made by governments and industry groups, and public media sources\(^2\). The parent company and brand reports reviewed in this analysis account for over 80% of the 2003 new passenger vehicle registrations shown in Figure 2.4. Many car companies, including Toyota, Volkswagen, DaimlerChrysler, Ford, General Motors and the Fiat Group now publish detailed environmental reports. A growing number of carmakers also have dedicated environment/sustainability websites. Each of the 'expectations' above has been evaluated based on the evidence found in these sources and other publicly available reports (as listed in the references). In some cases, industry wide trends facilitate the assessment of environmental

\(^1\) A large number of Member States have now enacted laws relating to the EU ELV Directive, though many did so after April 2002, the date specified for compliance in the ELV Directive itself [9]. ACEA, ELV Country Report Charts. ACEA, 2004.

\(^2\) Extensive searches were conducted on information databases including: Science Citation Index, LexisNexis, ABI/Inform and Business Source Premier.
performance. In others, the available data is insufficient to clearly determine whether a given expectation will be met in the future.

Figure 2.4: New Passenger Car Registrations in Western Europe, 2003

<table>
<thead>
<tr>
<th>Parent Manufacturer</th>
<th>Share of New Registrations, 2003 (%)</th>
<th>Brands included</th>
</tr>
</thead>
<tbody>
<tr>
<td>Volkswagen</td>
<td>18.2%</td>
<td>Audi, SEAT, Skoda, Volkswagen, others</td>
</tr>
<tr>
<td>PSA</td>
<td>14.8%</td>
<td>Citroen, Peugeot</td>
</tr>
<tr>
<td>Japanese Manufacturers</td>
<td>12.7%</td>
<td>Honda, Mazda, Mitsubishi, Nissan, Suzuki, Toyota, others</td>
</tr>
<tr>
<td>Ford</td>
<td>11.0%</td>
<td>Ford, Jaguar, Landrover, Volvo, others</td>
</tr>
<tr>
<td>Renault</td>
<td>10.6%</td>
<td>Dacia, Renault</td>
</tr>
<tr>
<td>GM</td>
<td>9.8%</td>
<td>OPEL, SAAB, others</td>
</tr>
<tr>
<td>FIAT</td>
<td>7.4%</td>
<td>Alfa Romeo, Fiat, Iveco, Lancia, others</td>
</tr>
<tr>
<td>DaimlerChrysler</td>
<td>6.5%</td>
<td>Chrysler, JEEP, Mercedes, Smart, other</td>
</tr>
<tr>
<td>BMW</td>
<td>4.4%</td>
<td>BMW, Mini, others</td>
</tr>
<tr>
<td>Korean Manufacturers</td>
<td>3.3%</td>
<td>Daewoo, Hyundai, Kia, others</td>
</tr>
<tr>
<td>MG Rover</td>
<td>1.0%</td>
<td>Rover</td>
</tr>
<tr>
<td>Other</td>
<td>0.3%</td>
<td></td>
</tr>
</tbody>
</table>

Source:[11]

The attribution of legislation as a driver of observed change also requires that such changes be distinguishable from trends that may occur regardless of regulatory impacts. Numerous factors can influence a company's decision to act. These include cost savings, brand image, regulatory constraints, consumer preferences and competitive pressures. Even without appropriate legislation, the 'invisible hand of the market' may move the automotive industry toward more (or less) sustainable practices. A decrease in the level of waste, energy use and water use can yield economic benefits in addition to being environmentally desirable. Decreasing resource requirements per unit output can mean lower costs of production and higher profits. Determining the level of influence of such factors vis-à-vis regulation is often difficult. While the conclusions of this study provide early evidence of the effectiveness of ELV legislation, the long-term implications of the ELV Directive are only beginning to unfold. A fuller understanding of the impact of ELV legislation will develop over the next decade.

4. Assessing Expected Outcomes

The automotive industry has had advanced notice that ELV legislation was on the agenda since at least 1989 [12]. Industry was also heavily involved in the legislative process that culminated in the ELV Directive. Improvements in ELV recovery have been influenced by national policies since the nineties. ELV regulations and/or voluntary agreements existed in ten European countries prior to 2000 (Austria, Belgium, France, Germany, Italy, the Netherlands, Portugal, Spain, Sweden and the UK). Of these, Austria, France, Italy and the Netherlands had introduced
national policies and agreements prior to the debate over the EU ELV directive proposal, which was put forward in 1997. The other six countries established voluntary agreements and legislation between 1997 and 1999, in parallel with the debate over the ELV Directive. As a result, a number of technological and organisational innovations occurred in the 1990s. These included the creation of ELV treatment infrastructures and efforts to design for dismantling and recycling [12]. Current advances should be seen in the light of such innovations, which might have been stimulated by pending ELV legislation for over a decade.

Increases in the amount of industry-wide Research and Development (R&D) into design for end-of-life would provide strong evidence that ELV legislation is having an effect. R&D into environmental protection can makeup a significant portion of a manufacturer’s total R&D budget. For example, Renault states that around 40% of R&D programs are devoted to environmental protection [13]. In fact, recent information reveals that European automakers and suppliers are investing up to half of their R&D budgets on reducing carbon dioxide emissions [14]. Yet, there is little evidence that R&D expenditure on environmental management (including waste treatment, ELV recovery and emissions technologies) is increasing. Neither did publicly available data show strong trends toward increased levels of environmental investment. For instance, Volkswagen’s operating costs for environmental protection increased by 30% between 1999 and 2003 (from Euro 150m to 195m). Of this 37% was spent on waste management. However Volkswagen’s investment in environmental protection has remained relatively steady over the past 4 years, ranging from Euro 24m to Euro 33m [15]. The environmental protection investments of Mercedes Car Group (part of DaimlerChrysler) jumped significantly from 2002 to 2003. Yet the level of its investments in 2001 and 2002 were smaller than those of 1998 to 2000 [16]. Industry and country-wide figures are also flat. R&D investment in the UK automotive industry has remained steady over the past 3 years at around £1 billion [17]. This represents approximately 2.3% of the total automotive manufacturing turnover, which held relatively constant between 1999 and 2002 at £42-44 billion [17]. Similarly, R&D by European automotive suppliers has remained steady at around 3.5% of revenue [18].

There is a lack of data on ELV specific R&D expenditure. Furthermore, due to the strong drive toward emissions reduction, fuel efficiency and energy consumption there are few clear signs that automotive manufacturers are making ELV recovery an R&D priority. While access to detailed R&D investment data on ELV recovery would have been useful, there are other indicators (discussed in expectations 1 to 5 below) that provide insight into the impact of ELV legislation on ‘green’ innovation and vehicle recovery.

3 For example, the majority of Renault’s efforts are targeted at emissions reduction, fuel efficiency and energy consumption. Similarly, the Fiat Group’s 2003 Environmental Report states five research priorities targeted at environmental stewardship (p.19). These deal with fuel efficiency, emissions, safety and traffic flow improvement. Efforts to increase the level of end-of-life vehicle recovery are not listed as a priority.
Expectation 1: Changes in the Material Composition of New Cars

Legislative, economic, technological and societal factors have all contributed to some distinctive material trends over the past few decades. The use of materials such as plastics and aluminium are increasingly being utilized due to their light weight (resulting in less energy and fewer emissions to power a given vehicle) and desirable mechanical properties. Plastics are becoming central to vehicle production. The use of plastics has increased by 50% over the past 20 years [19]. European cars contained an average of 133kg of plastics parts in 2003 [20]. Polymers are now used in over 1000 parts including bumpers, seats, dashboards, interior trim, fuel systems and upholstery. Plastic has prevailed despite its material predecessors being either easier to recycle (e.g. metals) or produced from renewable sources (e.g. wood).

Aluminium use in cars is also expected to increase dramatically over the coming decade [21]. ELV legislation may provide further incentives for this, as aluminium is easily and cost effectively recyclable to near 100% quality. As a result, ELV regulations may slow the use of composite materials, which could be replaced by metals such as aluminium [20]. Consequently, it is predicted that the average European car will contain 240kg of aluminium by 2010; a 120% increase from 2003 [20]. In addition to decreasing the amount of automotive waste going to landfill, recycling aluminium saves up to 95% of the energy needed to produce primary metal. Recycling one kilogram of aluminium can also save about 8 kilograms of bauxite and four kilograms of chemical products [22]. All of this adds up to sufficient economic savings to make the use of aluminium attractive.

Vehicle design remains driven by cost and functional properties. This is not surprising given that many experts believe that the environmental awareness of consumers is in decline [23]. As a result consumer demand is not a primary driver of environmental change in the automotive industry [23]. Still, some progress in design for end-of-life is being made, driven in part by regulatory pressures. Three ways in which change in a vehicle's material composition can manifest itself are discussed below.

i) Increased Use of Recyclable and Environmentally Beneficial Materials

Increased emphasis on recyclability is leading to a rationalisation of plastic use. Fiat Group's 2003 Environmental Report states that the group's design efforts aim to maximise component recyclability, with hard-to-handle polymers being 'scaled back' in favour of other more easily recyclable plastics [24]. Similarly, Peugeot-Citroën state that efforts are made to a) reduce the variety of materials to facilitate resource recovery after shredding, b) use single family plastics per major function to enable entire sub-assemblies to be recycled without disassembly and c) to reduce the variety of plastics in order to ensure optimal and profitable recovery processes [25]. GM and OPEL also aim to minimise the number of different plastics and to use non-blended compounds where possible [26, 27]. Such efforts will impact a substantial number of car parts.
For instance, Chrysler believes that by 2007, the company will modify up to 1,000 parts per vehicle to ensure compliance with the ELV Directive [28]. A recent report by the SMMT includes numerous additional examples of environmentally friendly design and process improvements [29]. Notwithstanding the above, it is difficult to gauge the true extent of eco-design efforts and of the impact of such efforts (on a vehicle's material composition for example). It is similarly difficult to ascertain whether ELV legislation or cost improvements are driving these changes.

The impetus to minimise ELV waste may also fall counter to a company's desire to reduce the weight of a vehicle. As a rule of thumb a 10% weight reduction can lead to a 3-7% improvement in fuel efficiency and a subsequent reduction in air pollution [30]. A movement toward using recycled components could result in the need for heavier parts if recycled materials have inferior mechanical properties. A recent study by the APME found that such behaviour is counterproductive [31]. Using more recyclable materials that have poor mechanical properties would have the same effect. This has led some commentators to argue that ELV legislation could result in a movement away from plastics (which are light but often hard to recycle) toward metals such as aluminium (which is light and easily recycled) [23].

The drive toward recycling can also be at odds with other design trends. For instance the use of electronics in vehicles is increasing. Such parts typically contain metal-plastic composites and flame-retardant chemicals. This makes them difficult to separate and recover at the end of a vehicle's life. Similarly, products that use recycled content may also compromise recyclability. Composite products that are comprised of plastics reinforced with inorganic fibres such as fibreglass are increasingly being used due to their superior mechanical properties and light weight. As a result, composite use is expected to rise by roughly 5% each year until 2008 [20]. However the fibres themselves and the fillers used in the manufacturing process currently prohibit such materials from being economically recycled [32]. This has driven Ford to work on creating nanocomposite materials that are more recyclable, lighter and have better mechanical properties than regular composites [32].

Materials made of natural fibres have recently been making their way into car production. Carmakers have been testing hemp, flax, purified cellulose and native prairie grasses for automotive uses [33]. While the volumes are still relatively small, such fibres represent valuable alternatives to synthetic fibres. They are renewable, display excellent mechanical properties, are light in weight and can be combined with other materials to form natural-fibre-reinforced composites. For these reasons, the Mercedes E-Class has more than 50 components produced in whole or in part from renewable materials [32]. Renault's Scenic II also contains 12kg of renewable materials [13]. More than 140 auto parts at DaimlerChrysler contain natural fibres [34]. However, concerns have been raised that the ELV Directive may impede the use of raw materials, as recycling natural-fibre-reinforced composites through means other than combustion
is not currently commercially viable. As a result some experts are concerned that designers may switch to less favourable materials in order to meet mandatory recycling quotas [35].

It is also unclear whether ELV legislation will have a beneficial effect on the use of other new materials like bio-plastics. Such plastics are made from plant matter such as sugarcane, corn or soy and can be given biodegradable properties that allow them to be broken down by microorganisms. They may also have environmental benefits such as reduced carbon emissions [36]. For instance, Ford is developing canola and soy based foams as an alternative to polyurethane foams widely used car seats and cushions [33]. However bio-plastics are not highly recyclable to date and as a result are not desirable from an ELV Directive standpoint [32].

Evidence that environmental concerns are being incorporated into the design process, as a first step towards the use of "green" materials is also instructive. The Society for Motor Manufacturers and Traders (SMMT) in the UK notes that 'Design for recycling' principles are gradually being adopted and implemented in the product design process [17]. Carmakers are increasingly using life cycle tools as part of the design process. For instance, all the materials in the Fiat Idea, a new compact car, were selected using Life Cycle Assessment (LCA) [24]. BMW is developing a life-cycle simulation tool for the long-term design and maintenance of an environmentally safe recycling system [37]. Volvo has started to conduct life-cycle analyses for all newly released models. This data is presented on Volvo's website for easy comparison between models [38]. In some cases specific tools have been devised to improve part recyclability. Renault's 'Index of Recyclability by Function (IRF)' was used on the Megane II functions and will be used to set common progress targets for suppliers. Nissan and Renault have also jointly devised the OPERA application (Overseas Project for Economical Recycling Analysis). OPERA is being used to simulate costs and recycling rates in the ELV recycling process [13]. GM's European operations have also adopted 'Design for Recycling' [27].

Overall, we conclude that ELV legislation has contributed to greater consideration of recyclability in the design process. This is already leading to a rationalisation of plastic use. Recyclability and mechanical design considerations might also hasten the trend towards greater use of aluminium. However, the extent of movement toward recyclable materials is difficult to gauge. ELV regulation may also negatively impact the use of novel "green materials" like bio-plastics and natural fibres.

ii) Increased use of Recycled Materials ("Recylcate")
A lack of industry and company-level data on recycle use requires reliance on the aggregation of statements of intent and car-specific information. Evidence suggests that recycle is increasingly being used in car parts [19]; [32]. For instance, Peugeot-Citroën state 'using recycled materials' as a criteria by which polymers are chosen in current designs [25]. BMW has
also stated that it plans to gradually increase the share of recyclates in plastic components for future models [37].

Most ELV metals are relatively easily recycled (as discussed further below). However, the use of non-metallic recycled material remains low. Roughly 9% of the weight of a passenger car is plastic [6] and a common car weighs about 1,100kg [19]. Thus, even 30kg of recycled material would equate to less than a third of the total plastics in a car. Most automobiles contain less than this amount. For example, Renault's Scenic II (an industry leader in terms of recyclability according to their 2003 annual report) contains 16kg of recycled plastics out of a total of 150kg (i.e. just over 10%) of plastic used in the car [13]. The Ford Focus also incorporates 39 recycled plastic parts, accounting for 21 kilograms of the car's weight [39]. Notably, Volvo has been providing externally verified Environmental Product Declarations (EPDs) for a number of their vehicles since 1998. Based on these EPDs, Volvo's 2004 vehicles contain between 7kg of recycled non-metallic materials (the S80 model) and 23kg (the S40 model) [38]. The BMW 3 Series contains 14% recycled plastics by weight [37].

Recyclate tends to be used in parts that do not require high structural/mechanical performance and in parts that are not generally visible to the occupant. For instance, the fuel tank and inner wheel housings contain the largest amount of recycled material in a Volvo [38]. The Volvo Car Corporation estimates that only 30kg of recycled non-metallic materials could be used in a new car, subject to prevailing quality standards and the availability of materials [40]. Thus a figure of 100% non-metallic recycled materials as shown on Volvo's EPDs would signify 30kg of recycled material being used. Similarly, OPEL's 2002 Sustainability Report outlines their goal to increase the share of recycled materials to 20% of the total plastic mass in the vehicle [26]. While it is possible to obtain high quality recyclate, it is rare to find automotive parts made from 100% recycled plastic. It is much more common to use a blend containing 25-50% recycled content [19]. This suggests that automotive plastics are being 'downcycled' rather than 'recycled'.

Recyclate is increasingly being used in car production. In the absence of ELV legislation the incentive to use such material would be significantly reduced. However, there is still a considerable way to go. A number of technological and economic barriers must be overcome before carmakers can replace existing plastics with their recycled counterparts.

### iii) Removal of 'Banned' Substances

The ELV Directive requires that components of vehicles placed on the market after 1 July 2003 do not contain mercury, hexavalent chromium, cadmium or lead. On the 27th of June 2002 the Directive was amended to modify the exceptions in Annex II, yet the target of heavy metal removal remains the same [41]. Evidence indicates that automotive manufacturers and their suppliers are complying with the ELV Directive's requirement [24, 25, 37, 42]. Nor are such
changes restricted to European cars. A recent report has found that the EU ELV Directive is, to some extent, driving activities in the US automotive market. In particular, the ELV Directive’s aim to remove toxic and hazardous substances has resulted in international efforts to eliminate their use in vehicle manufacturing [32].

**Expectation 2: Increased “Design for Disassembly, Re-use and Remanufacture”**

The ELV Directive states that manufacturers must design and produce vehicles which facilitate the dismantling, re-use and recovery (including recycling) of end-of-life vehicles [3]. To this end there are indications that automotive manufacturers are investing resources to improve vehicle disassembly. However, there is little compelling proof that carmakers are designing vehicles to facilitate re-use and remanufacture. Several reasons for this are explored below.

To accurately understand the benefits of re-use and barriers to it in the automotive industry it is useful to place re-use in context with other forms of ELV recovery. Many authors have attested that there is strong evidence that the 3R framework (Reduce, Re-use, Recycle) is robust and generalisable [43]. Implicit within it is the notion that less material and energy use is usually better for the environment. Generally speaking, the higher up the process in the hierarchy the more environmentally friendly it is [44]. Hence re-use is theoretically preferable to recycling (see Figure 2.5). Additionally, studies have shown that traditional recycling saves ten times more energy than performing energy recovery [45].

**Figure 2.5: Theoretical Recovery Hierarchy**

![Theoretical Recovery Hierarchy Diagram]

While the 3R’s provide a useful starting point, a more granular breakdown of ‘re-use’ would include upgrading, reprocessing, remanufacture, refurbishment, reconditioning, revalorisation and repair [46]. However in order to strike a balance between simplicity and comprehensiveness this paper will consider remanufacture in detail. The aim of remanufacturing is to reprocess used products in such a manner that the quality of the products is as good or better than new in terms of appearance, reliability and performance [46]. Remanufacturing can often save more than half the energy and 80% of the material that would otherwise have been used to make a new product from scratch [47]. A recent study found that remanufactured engines could be produced with 68%
to 83% less energy and 26% to 90% less raw materials than the manufacture of a new engine [48]. Figure 2.5 incorporates remanufacturing and energy recovery into the 3R concept.

Vehicle manufacturers realise that there is a lot to be learnt from disassembling vehicles and analysing wear on old vehicle parts. Both Ford and BMW have established recycling and dismantling centres in Europe to integrate learning from end-of-life vehicles into design methods. BMW already has over 100 official dismantling facilities in Germany alone [49]. The information is used to benchmark and improve vehicle recovery (recycling in particular) [2, 37]. DaimlerChrysler are also gaining disassembly and recycling knowledge from dismantling vehicles at Canada's Automotive R&D Center in Windsor [34].

Available information suggests that manufacturers are trying to improve design for disassembly4. For example, PSA Peugeot Citroën claim to have embraced the principles of design for disassembly and reuse, with at least 95% of the average mass of new Peugeot and Citroën vehicles being reusable and recoverable. Renault also states that 95% of the Scenic II is recoverable [13]. Additionally, they profess to be contributing to a ‘dual system’ that incorporates part recovery for the used part trade in addition to material recovery for recycling. Similarly, OPEL engineers are encouraged to design for disassembly by avoiding the use of bonding agents and welded joints where possible and by using easily detachable clips or screws [26].

There is little evidence that car companies are investing in 'design for remanufacture'. Moreover, we have not come across any evidence that car parts are being designed to be remanufactured and then 're-used' in the production of future new vehicles. Such 'closed-loop' remanufacturing typically requires a significant change in design, operations and possibly industry structure. For instance, parts with inbuilt electronic components and design optimisation using finite element analysis have resulted in components becoming less economically feasible to remanufacture [46]. However, remanufacturing can be economically viable and is occurring in other industries. New electronic equipment such as photocopiers already contain a significant proportion of remanufactured parts; enabling the manufacturer to cut costs and increase the environmental performance of their products [50]. Remanufactured parts are also used in safety-critical applications such as aeroplane engines [46].

There may be a lack of remanufacturing because carmakers may not be the main beneficiaries of remanufacturing revenue. Until carmakers commit to remanufacturing their parts, increased profits will accrue to independent remanufacturers. Such a commitment could take the form of

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4 Improving the ease of disassembly is only one part of designing for re-use and remanufacture. The ability to re-use and remanufacture parts can be facilitated by designing them for greater durability. By doing so, a larger proportion of parts will be candidates for re-use when a vehicle reaches the end of its life.
direct involvement in the manufacturing process or through close relationships between Original Equipment Manufacturers (OEMs) and remanufacturers.

At least three additional barriers exist to remanufacturing in a closed loop context. Firstly, the average age of ELVs in Europe is estimated to be approximately 10-12 years (with significant variability among countries) [12]. Yet car designs change regularly; driven by consumer preferences and technological innovation. The combination of a long useful life and rapid technological change can inhibit closed loop remanufacturing, as by the time remanufactured parts are available many of the parts will be outdated. Yet this barrier alone need not prevent remanufacturing. Product leasing (which decreases the time before parts can be returned to the manufacturer) has been utilised effectively in conjunction with closed loop remanufacturing in the photocopier industry. Additionally, vehicle parts that are not forecast to change significantly may be better candidates for closed loop remanufacturing. Second, remanufactured parts are often perceived as being of poor quality and/or being 'old technology'. In high technology industries such as the automotive industry companies appear very cautious about doing anything that may affect their brand image of being an innovative company. It would be useful to understand the extent to which these perceptions continue to reflect customer sentiment.

Finally, design for closed loop remanufacturing must occur in parallel with a change in operations and logistics to be effective. Supply chains and inventory management processes must be altered to integrate remanufactured parts with new parts. ‘Reverse supply chains’ must be established to enable automotive manufacturers to collect used vehicles and then supply remanufactured parts back to the manufacturer. The remanufacturing process itself may require the acquisition of specific (and proprietary) knowledge and skills needed for disassembly and recovery, in addition to investment in customised equipment.

**Expectation 3: Increased Levels of Remanufacture and Re-use**

Reuse and remanufacture remains a small part of automobile recovery, partly due to the high cost of labour. But the industry is growing [51] [52]. If ELV regulation has significantly affected reuse and remanufacture we would expect to see a jump in the volume of parts being used for this purpose. Observable changes in the eagerness of carmakers to participate in the after-use market would also suggest that the ELV Directive has had a direct effect in this area. However such behaviour may be explained by economic motives as well, as discussed below.

Remanufactured parts typically end up in repair and aftermarket industries rather than going into new car production. In the UK, 69% by weight of total disposed vehicles is recycled/recovered, 11% is parts that were able to be re-used and the remaining 20% goes to landfill [29]. There may also be an opportunity to increase automotive remanufacturing in Europe. A comparison with US data suggests that the UK remanufacturing industry may generate revenues of several billion
pounds [46]. However, market penetration for remanufactured products is higher in the United States than in Europe. Americans buy approximately 60 million remanufactured automotive products annually, while Europeans buy only 15 million. Yet, the total stock of vehicles is roughly comparable [51]. This highlights the potential for increased use of remanufactured components in Europe.

OEM participation in remanufacturing remains small, but there are encouraging signs that OEMs are taking a greater interest in remanufacturing and reuse because of the opportunities afforded by remanufacturing to increase profits and to gain feedback on failure modes and durability. In the United States, OEMs account for less than 5% of remanufacturing activity. Independent third parties make up the majority of the 73,000 U.S. remanufacturing firms [53]. The percentage of automotive remanufacturing undertaken by carmakers in Europe is also likely to be small. Nonetheless, remanufacturing by car companies is occurring. Volvo Cars' exchange system for remanufactured parts is one example. Through this system Volvo remanufactures used parts (obtained from dealers) to the same quality as new parts. Over 2,000 different components, from gearboxes to consoles, are remanufactured in this manner and sold to consumers with a full warranty [54]. Remanufactured engines are also used by one OEM as replacements for under-warranty engines, resulting in considerable cost savings [51]. BMW also remanufactures 15,000 engines each year at its Landshut plant [37, 49]. But, while the company supports component reuse in secondary markets, it does not do so in new cars [32]. Companies such as Mercedes-Benz and Ford currently harvest and sell spare parts as well. Notably, Ford has done so by buying salvage yards in North America and Europe [2].

There remain at least four obstacles to the widespread adoption of remanufacturing. Firstly, most products that currently arrive at the end of their life were not designed to be recycled or remanufactured [55]. Secondly, there has been an explosion in the number of car models over the past two decades [51]. This has lead to the production of fewer vehicles of each model. The result is that remanufacturers are less able to take advantages of economies of scale. Their ability to meet stringent supply requirements for just-in-time production processes is also diminished.

Third, there is significant supply-chain uncertainty associated with remanufacturing and re-use, including: the supply of remanufacturable parts, the quality of the returned parts and variable processing times [2, 51]. As a result, remanufacturers are forced to keep large inventories to mitigate against such variability. Yet, some of this uncertainty is likely to decrease in the future. Analysis of experiential data in addition to advances in technology such as electronic data logs will allow a more expedient and accurate assessment of the quality of returned parts. BMW is already moving in this direction. The new BMW 7 Series contains an on-board computer which constantly monitors component wear and alerts the driver when action is required [37].
Finally, alternate recovery methods are becoming more financially attractive. The development of sophisticated post-shredder technology is increasing the economic feasibility of recycling and energy recovery [52]. The ELV Directive could hasten this process.

Thus some carmakers are becoming progressively more involved in the business of collecting and selling used parts. Similarly, a few companies are also taking the next step of rejuvenating these ‘in-house’ and selling them as remanufactured components. But it is too early to tell whether recent actions constitute the beginning of an enduring swing toward greater levels of re-use and remanufacture.

**Expectation 4: Increased Levels of Recycling of ELV Materials**

Ideally, annual ELV recycling rates, published at EU, national or company-wide levels would provide solid evidence as to whether or not ELV recycling has increased as a result of regulation. In the absence of such data, it is possible to gauge progress by assessing the level of R&D and innovation in recycling technologies.

There is mounting evidence that innovation in recycling is occurring. It is being driven by high recycling rate requirements in both the ELV Directive and the Waste Electrical and Electronic Equipment Directive. End-of-pipe recycling solutions are needed as vehicles already in use will reach their end of life by 2006. Hence, BMW has been working on recycling of pyrotechnic components (such as airbags and belt tensioners) in addition to new, automatic sorting techniques for plastics, metals and shredder residues [37]. Volkswagen has designed a new separation and recycling process, known as the VW-SiCon Process [56]. Not surprisingly, recent efforts have focused on extraction and recycling of polypropylene as it comprises the largest fraction of automotive plastics [57]. Yet there are exceptions. For instance, Dupont is developing technology to recycle nylon composites to produce resin that is “essentially equivalent” to virgin nylon [58]. Processes for disposing of plastics with brominated flame retardants are also being tested [59]. A wide variety of additional methods are being developed to sort and recycle plastics as well [60].

The automotive industry need only attain a 5-10% improvement in the rate of re-use and recycling to meet the 2006 recycling target of 80% set out in the ELV Directive. Current data is insufficient to assess whether carmakers are on track to reach this goal. What is clear is that one or both of the following will need to take place in order to reach the 2015 target of 95% recovery: (i) a dramatic increase in the recovery of plastic, rubber, glass and other non-metallic materials, and (ii) a movement away from these materials toward more easily recycled materials such as aluminium. Notably, the 2015 recovery target may be partially achieved through increased rates of energy recovery. This is detailed in Figure 2.2. Still, the variety of recycling innovations taking place point towards an improved ability to recycle current materials on a commercial scale in the
not too distant future. This, in combination with a modest increase in the level of re-use and remanufacturing may make it feasible to reach the 2006 target.

**Expectation 5: Increased Level of Publicly Available Information**

The ELV Directive requires that producers use material coding standards, which allow identification of the various materials during dismantling. It also requires that information on the treatment of end-of-life vehicles and progress with regard to re-use, recycling and recovery be provided to prospective vehicle buyers. Is such information being provided?

Information available to vehicle recovery operators does appear to be improving. The International Dismantling Information System ('IDIS') and the International Material Data System ('IMDS') are two key automotive initiatives aimed at improving data collection and dissemination. The IDIS database and associated software is produced by the IDIS 2 Consortium, which consists of 24 carmakers. IDIS enables the identification of component materials to improve the efficient treatment of end of life vehicles. The database currently lists around 44,000 car components for 888 vehicle models from 24 car manufacturers. The 40 brands referenced in IDIS represent more than 95% of the current automotive European market, as well as all the major manufacturers from Japan, Korea and the United States [61]. Car companies are currently undertaking the significant process of gathering the required information from their suppliers and updating this information for distribution to those involved in end-of-life vehicle recovery. Auto parts suppliers also use another database, the IMDS to catalogue the composition of car parts (including their surface coatings). IMDS was developed by a number of European carmakers, though North American and Asian manufacturers have since embraced the system [62].

Coding standards are also being instituted to enable identification of components that are suitable for recovery, reuse and recycling [24, 27]. The European Council for Automotive R & D (EUCAR) is currently working with its members to collect information on ELV treatment systems as well [24].

Car manufacturers are also making some efforts to communicate a vehicle's environmental performance to the customer. As noted above, Volvo has been providing Environmental Product Declarations (EPDs), verified by Lloyd’s Register Quality Assurance, for a number of their vehicles since 1998. EPDs are seen as a potentially useful tool for communication of a product’s environmental impact and may form part of an integrated European Product Policy according to the European Commission’s Integrated Product Policy (IPP) white paper [63]. However, in order for EPDs to be useful and credible they require verification and standardisation of approach across the industry [64]. This remains to be accomplished.
In general, it appears that car manufacturers are improving industry’s access to information on vehicle disassembly and recovery. Further effort is needed to provide the same level of information to car buyers. Consumers face a deficit of accurate, comprehensive and easily available information pertaining to the ecological impact of their potential purchase. This lack of verified data hinders the ability of a consumer to purchase a vehicle that is environmentally friendly.

5. Conclusion

ELV legislation is having a discernible effect on numerous ‘end-of-pipe’ solutions such as innovation in recycling methods and shredder residue separation techniques. These new technologies are likely to be used to recycle material from a broad range of industries and particularly from white-goods, which are already processed in shredders alongside automobile hulks. However, end-of-life design considerations are not a high priority for car manufacturers. Economic imperatives and a drive toward customisation remain the key motivations in automotive design. Furthermore, eco-design efforts may be restricted by the delayed payback associated with long vehicle lifetimes and the fact that innovations in end-of-pipe recycling technologies will be required to process older cars regardless of design changes. This raises the possibility that car manufacturers might get locked-in to sub-optimal solutions that favour recycling over remanufacture and reuse.

Nonetheless, there are some important impacts of ELV legislation on design, particularly on material choice. There are strong indications that ELV legislation is leading to a reduction in toxic substance use. Numerous life cycle design tools, indicators and processes focused on improved material use are being utilised in the design process. Carmakers are reducing the number of different plastics being used in order to improve recyclability. It is also likely that ELV legislation will increase the use of aluminium, in part due to its ability to be easily recycled. Though the ELV directive does not specify targets for the use of recyclates in vehicles there is evidence that recyclates are increasingly being used in car parts, albeit at low total volumes. There also appears to be a focus on increasing the proportion of materials that can be recycled or downcycled, rather than on the quality of recyclate.

The impact of ELV legislation on design for re-use and remanufacturing is limited. Embracing remanufacturing requires significant changes to organisational processes and an approach to design that incorporates remanufactured parts. Remanufacturing is likely to only be economically attractive to carmakers if they are able to share directly in the profits. To capture such benefits carmakers will need to actively participate in the remanufacture of their own parts or develop close financial and operational ties with existing remanufacturing organisations. At least initially,
most remanufacturing activity will continue to be limited to the provision of replacement parts for existing vehicles.

There is evidence that the ELV Directive is resulting in improved collection and dissemination of data that enables efficient material and part identification. Some initiatives to communicate a vehicle's environmental performance to the customer are also appearing. However, in order for these to be useful to the end consumer such information will need to be available and standardised across models and brands.

Policy instruments can influence the choice of innovation path and "may work as 'selection devices' by constraining some innovative options while providing incentives to pursue other innovation solutions" [12]. In the case of the automotive industry the interplay of legislative and economic factors has led to an increased emphasis on recycling and hazardous substance removal. The resultant innovation may be sufficient to reach ELV Directive targets and may also have spill-over benefits to other industries. The next step toward sustainable vehicle management lies in increasing the levels of re-use and remanufacturing.

References


[38] Environmental Product Declaration. Volvo Car Corporation.
[40] Volvo, Use of Recycled, Non-metallic Materials.

[54] *Clean for Life - Recycling*. Volvo Cars.


CHAPTER 3: ECONOMY-WIDE RELEASES OF LEAD AND CADMIUM RESULTING FROM PRODUCTION, USE AND DISPOSAL OF AUTOMOBILES

NOTE:
A version of this paper has been submitted for publication. Gerrard J., Kandlikar M. Economy-wide Releases of Lead and Cadmium Resulting from Production, Use and Disposal of Automobiles. Environmental Science & Technology. Currently in review.

1. Introduction

Governments have dedicated considerable effort toward reducing industrial heavy metal use through regulation and voluntary programs for decades. The automotive industry remains a focus of this work, with legislative tools continuing to influence vehicle manufacture and operation. Leaded gasoline was first banned in New York City for over 3 years during the 1920s due to health concerns [1]. Yet it wasn’t until the 1970s that most countries began restricting the lead content in gasoline [2]. More recently, the European Union established the End-of-Life Vehicle Directive (2000) in order to increase levels of vehicle recovery and to reduce the use of heavy metals including lead and cadmium [3]. End-of-life vehicle recovery has received less regulatory interest in the US, though regulations pertaining to the disposal of lead-acid batteries, used tire management and the disposal of free liquids to landfills are in force [4]. Some state level restrictions on heavy metal containing devices also exist. The effectiveness of these and future initiatives relies upon a thorough understanding of the dominant lead and cadmium discharges that result from light-duty vehicles. This paper provides insight into the quantity and nature of these life-cycle emissions.

While other heavy metals such as mercury and chromium are also environmentally harmful, we focus on lead and cadmium as they are released throughout the vehicle life-cycle and industrial emissions of these metals now dwarf natural atmospheric fluxes (e.g. from erosion and volcanic activity). It has been estimated that 85% of cadmium emissions and 96% of lead emissions come from industrial activity, while anthropogenic sources of mercury and chromium account for only 59% and 41% respectively of total atmospheric emissions [5]. Lead and Cadmium are also mobilized when automotive components, such as tires and brake pads, wear. In contrast, other heavy metals (e.g. mercury contained in switches, lighting and antilock braking systems) are less likely to be released as a result of car part abrasion and are arguably more easily controlled [4, 6].
Similarly, while mercury content of crude oil is high, removal of mercury during refining results in gasoline, diesel and other oils containing very little mercury (0.7 – 50 ppb) [7].

Both lead and cadmium are metals of significant concern to policy makers, health professionals and environmental groups alike. Infants and young children are often exposed to lead through dust and soil (to which lead binds strongly) and can absorb as much as 50% of their dietary lead intake [8]. In adults, approximately 10% of the dietary lead is absorbed. Lead is readily taken up into the blood stream, is stored in bones and can result in impaired neuropsychological development, kidney damage and death [9]. Unlike lead, cadmium bio-accumulates, is relatively water soluble compared to other metals and is considered carcinogenic [8]. For non-smokers, the major route of cadmium exposure is via food. Plants take up cadmium from the soil. It accumulates in the human body, especially the kidneys, resulting in renal damage. Notably, as much as 50% of inhaled cadmium may be absorbed; while on average only 5% of total oral intake is absorbed [8].

2. Methodology

In this work we provide a comprehensive assessment of releases resulting from light vehicle manufacturing, use, servicing and end-of-life stages. Previous studies of heavy metal emissions fall into one of two categories, general life cycle studies of the automotive industry and specific heavy metal studies of automotive parts (with a few notable exceptions as cited below). To date, much of the literature in the first category has focused on toxic emissions in aggregate, highlighting the need for identification of dominant sources of specific toxic substance releases such as those of heavy metals. Such analysis either does not capture lead and cadmium emissions, or aggregates them as part of an overall toxic waste index [10-12]. Studies in the second category often specify lead and cadmium emissions only for a given part of the life cycle [13-18], again drawing attention to the need for a more systematic and specific source identification and characterization.

To construct a systems view of the life-cycle emissions, we integrated Economic Input Output Life Cycle Assessment ('EIOLCA') with estimates of metal emissions during vehicle use and end-of-life, which are not captured by economic input-output tables. This approach analyses the impact of each life-cycle stage including vehicle production, use, servicing, part replacement and ultimately vehicle recovery (e.g. material recycling). The benefit of EIOLCA is its ability to assess the economy-wide impact of a change in demand for a given commodity or industry. It models industry interdependencies in the production of goods and services [19]. EIOLCA requires the specification of each and every input used in the manufacture of a given unit of output. In doing so, it accounts for both direct and indirect interactions between industry sectors. The method is
based on a linear relationship between the amount of product produced (calculated in dollars) and the corresponding resource usage and environmental burden [19]. The history and assumptions behind Input-Output analysis and the EIOLCA methodology have been comprehensively covered by previous authors [10, 12, 19, 20]. Key limitations of the EIOLCA approach as it pertains to this analysis are discussed in the body of this paper, particularly the final section.

Our EIOLCA analysis uses 1997 Input-Output data (the most recent comprehensive data set) provided by the U.S. Bureau of Economic Analysis ('BEA') [21]. This data disaggregates the U.S. economy into 491 industry sectors, each of which represents one or more 6-digit North American Industrial Classification System ('NAICS') product stream. This is combined with 2002 emissions data from the U.S. Toxic Release Inventory ('TRI') in order to ascertain the lead and cadmium releases attributable to each industry that directly or indirectly contributes to light vehicle manufacture and use [22]. The TRI data (based on Standard Industrial Classification 'SIC' codes) was converted to BEA 1997 data format by using conversion tables based on the U.S. Census Bureau NAICS-SIC Bridge website [23, 24].

The next section provides an overview of lead and cadmium use throughout the life cycle of a vehicle. The sections thereafter expand upon the lead and cadmium emissions arising from each life-cycle stage. The emissions from these stages are then compared. Limitations and implications of this research are presented in the Discussion and Conclusion.

3. Lead and Cadmium in the Vehicle Life Cycle

Both lead and cadmium are emitted throughout the life cycle of a vehicle. In addition to their presence in the vehicle, lead and cadmium may be used in the machinery and/or process materials (e.g. catalysts, solvents, cooling fluids etc.) employed in vehicle and replacement part production [25]. They are also released during the mining and refining of other metals and raw materials used in car manufacture. The production of energy (e.g. electricity) can release lead and cadmium as well [25]. In addition to toxic releases emanating from production, lead and cadmium are released as parts wear and when end-of-life parts are recycled. The key material flows are depicted in Figure 3.1. A full picture of cadmium and lead releases resulting from vehicle use must account for all such material transfers.

Lead appears in a large number of vehicle parts, including: batteries, brake pad linings, vibration dampeners, fuel hoses, soldering and wheel balance weights [26, 27]. It is also found as an alloying element or impurity in steel, zinc coatings, lead-bronze bearing shells and bushes, aluminium and copper alloys used in vehicles [28]. It is a stabilizer in plastics such as polyvinyl chloride (PVC), is found in glass and ceramic matrices in electronic parts and is also used in
piston coatings and spark plugs. However, 90-95% of the total lead used in vehicles resides in the car battery [27]. Cadmium is present in both brake pads and tires and is also used as a pigment in plastic, a stabilizer in PVC and is present in thick film pastes (used in electronic circuit boards) [26, 28, 29].

Additionally, cadmium and lead are by-products of the production of various metal ores [30, 31]. For example, lead is present in zinc, copper, gold, silver and molybdenum ores whose metals are used in the automotive industry [32]. Aluminium alloys and recycled aluminium also contain lead as an impurity [27]. Similarly, cadmium is released in iron and steel operations and is present in trace amounts in iron ore, zinc ore, limestone and coal [25, 33]. One of the main advantages of the EIOlCA method is that it captures these emissions, which may otherwise go undetected.

Figure 3.1: Material Flow in a Vehicle Life Cycle
4. Manufacturing

The manufacturing stage captures all activities involved in the production of the original vehicle, from processing of raw materials up until the final sale of the automobile to the consumer. Figures 3.2 and 3.3 show a breakdown of the sources of lead and cadmium emissions during vehicle production, based on an EIOLCA of the manufacture of an average light-duty vehicle, weighing 1400kg [4]. The data represents onsite releases of lead, cadmium and their compounds. Waste transported offsite and not subsequently recovered appears in the 'Waste management and remediation services' category.

The EIOLCA methodology determines environmental releases based on the industry output required as if all of the commodity were domestically supplied [34]. Implications arising from this treatment of imports and exports are discussed in the final section of this paper. Total metal discharges are calculated by combining metal and metal compound releases provided in the TRI data. Analysis of 2002 data shows that the vast majority (76%) of lead compounds came from 'Copper, Nickel, Lead and Zinc Mining'. More than 65% of cadmium compounds came from the same mining process or the primary refining and smelting of nonferrous metal, except copper and aluminium.

The average metal content of both lead and cadmium compounds provided in the TRI data must be estimated in order to analyse metal releases in aggregate. Due to the high atomic mass of both cadmium and lead (112 and 207 respectively), the metal itself comprises a high proportion of the molecular mass of both cadmium and lead compounds. In what follows we use estimated proportions of cadmium and lead in cadmium and lead compounds (by mass) by assessing the empirical formula of metal compounds that are likely to be the most abundant. Lead sulphide and lead oxides are expected to constitute a significant proportion of lead compounds, due to the oxidation of lead ore, normally galena (lead sulphide) that takes place in the beneficiation, sintering and smelting stages of lead refining [30, 33]. Lead comprises 93% by mass of lead oxide and 87% of lead sulphide. Consequently we have conservatively estimated that 80% of Lead Compounds are lead (by mass). Cadmium represents 78% by mass of the main cadmium mineral, Greenockite (CdS). The production of cadmium is usually a by-product of the production of other metals such as zinc, copper and lead, with Greenockite being nearly always associated with the zinc mineral Sphalerite (ZnS) [33]. It is released in flue dust created during refining and in electric arc furnace baghouse dust. Identifying cadmium's major compounds is more difficult than is the case for lead. However, its large atomic weight (112 amu) means that cadmium is likely to constitute 60% or more, on average of cadmium compounds. We have thus conservatively estimated that 60% of the mass of cadmium compounds is attributable to cadmium.
Using these approximations, the EIOLCA results in Figures 3.2 and 3.3 show that mining of metals contributes by far the majority of lead and cadmium released, 91% and 78% respectively. Primary smelting and refining and secondary metal production (recycling) also contribute significantly. The higher proportion of cadmium releases coming from waste management and remediation may be due to the difficulty in, and cost of, recovering this metal when it appears in very low concentrations in any given part (e.g. as a coating on steel, an impurity in zinc or as a pigment/stabiliser in plastic).

Figures 3.2 and 3.3 highlight the importance of capturing tertiary suppliers in life cycle assessments, especially when assessing heavy metal releases from metal intensive products and processes. Typical LCA methodologies risk missing significant waste releases unless they include emissions from mining and metal production.

Interestingly, our calculations (discussed in the End-of-life Vehicle Recovery section below) show that as little as 6% (77g) of lead released may be due to manufacturing of the car battery. Thus while the battery is the biggest single source of lead in a car, the results indicate that it makes a relatively small contribution to total lead released from car production. This outcome may be explained by both ‘real’ factors and methodological uncertainty.

Several ‘Real factors’ contribute to the non-battery related lead emissions. Firstly, lead may be released as a result of the production and use of manufacturing machinery. Secondly, lead is contained in by-products and waste generated from car manufacture. Thirdly, lead might be released as a result of mining and refining of other metals used in car production (as discussed above), and finally, lead may be emitted from the generation of energy (mainly electricity) that is consumed during vehicle production.

Aggregation in BEA data (which represents economic activity in 491 sectors) introduces uncertainty into input-output (‘I-O’) analysis [19]. For instance, it is difficult to separate the emissions due to the production of copper from those that are due to the production of lead, as both lead and copper mining fall into the same BEA category. Additionally, copper, nickel, lead and zinc ores are often mined from the same mine site. This co-location and co-processing of metal ores inhibits a more detailed attribution of emissions. A more fine-grained attribution would require the use of data at the level of an individual processing facility.

EIOLCA uses the price of a commodity as a proxy for the quantity of production, and hence inputs, for that production process. This in turn determines the level of toxic releases. However, changes in the relative prices of inputs to production can skew EIOLCA results. Consider for example, the price differential (per kg) between lead, copper, nickel and platinum. As at June 8,
2005 the prices of these metals varied greatly and were: lead $1.00, copper $3.53, nickel $17.05 and platinum $31,000 (or $880 per ounce) [35, 36]. Platinum in North America is mined as a co-product of nickel and copper [36]. Thus emissions from facilities that are primarily engaged in nickel production but also produce platinum group metals are classified under the 'Copper, Nickel, Lead and Zinc Mining' category. The difference in unit price between metals may result in a disproportionately high allocation of lead emissions being attributed to metals other than lead that are used in the production of a vehicle (i.e. copper, nickel, zinc and platinum-group metals). If releases were allocated based on mass rather than on the product's value, a greater portion of a facility's environmental discharges would be attributed to heavy yet cheap metals, such as lead. Consequently, the presence of platinum-group metals in vehicle catalytic converters (which remove hydrocarbons, carbon monoxide and nitrogen oxides from vehicle exhaust) may be contributing to non-battery related lead emissions in the EIOLCA results. However, releases stemming from car production are only a part of a vehicle's total lifetime emissions. Discharges from vehicle use, service and repair must also be accounted for and are discussed in the following section.

Figure 3.2: Lead Releases from the Manufacture of an Average Light-Duty Vehicle

<table>
<thead>
<tr>
<th>Industry Sector</th>
<th>Pb Combined (g)</th>
<th>%</th>
</tr>
</thead>
<tbody>
<tr>
<td>Copper, nickel, lead, and zinc mining</td>
<td>999</td>
<td>83%</td>
</tr>
<tr>
<td>Gold, silver, and other metal ore mining</td>
<td>95</td>
<td>8%</td>
</tr>
<tr>
<td>Primary smelting and refining of copper</td>
<td>32</td>
<td>3%</td>
</tr>
<tr>
<td>Primary nonferrous metal, except copper and aluminum</td>
<td>22</td>
<td>2%</td>
</tr>
<tr>
<td>Waste management and remediation services</td>
<td>12</td>
<td>1%</td>
</tr>
<tr>
<td>Ferrous metal foundries</td>
<td>11</td>
<td>1%</td>
</tr>
<tr>
<td>Secondary smelting and alloying of aluminum</td>
<td>6</td>
<td>1%</td>
</tr>
<tr>
<td>Iron and steel mills</td>
<td>5</td>
<td>0%</td>
</tr>
<tr>
<td>Secondary processing of other nonferrous</td>
<td>4</td>
<td>0%</td>
</tr>
<tr>
<td>Power generation and supply</td>
<td>4</td>
<td>0%</td>
</tr>
<tr>
<td><strong>Top 10 Sub-total</strong></td>
<td><strong>1192</strong></td>
<td><strong>99%</strong></td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>1203</strong></td>
<td><strong>100%</strong></td>
</tr>
</tbody>
</table>
Figure 3.3: Cadmium Releases from the Manufacture of an Average Light-Duty Vehicle

<table>
<thead>
<tr>
<th>Industry Sector</th>
<th>Cd Combined (g)</th>
<th>%</th>
</tr>
</thead>
<tbody>
<tr>
<td>Copper, nickel, lead, and zinc mining</td>
<td>6.67</td>
<td>73%</td>
</tr>
<tr>
<td>Waste management and remediation services</td>
<td>1.07</td>
<td>12%</td>
</tr>
<tr>
<td>Gold, silver, and other metal ore mining</td>
<td>0.50</td>
<td>5%</td>
</tr>
<tr>
<td>Primary smelting and refining of copper</td>
<td>0.26</td>
<td>3%</td>
</tr>
<tr>
<td>Alumina refining</td>
<td>0.19</td>
<td>2%</td>
</tr>
<tr>
<td>Secondary smelting and alloying of aluminum</td>
<td>0.11</td>
<td>1%</td>
</tr>
<tr>
<td>Iron and steel mills</td>
<td>0.08</td>
<td>1%</td>
</tr>
<tr>
<td>Primary nonferrous metal, except copper and aluminum</td>
<td>0.07</td>
<td>1%</td>
</tr>
<tr>
<td>Secondary processing of other nonferrous</td>
<td>0.07</td>
<td>1%</td>
</tr>
<tr>
<td>Synthetic dye and pigment manufacturing</td>
<td>0.05</td>
<td>1%</td>
</tr>
<tr>
<td><strong>Top 10 Sub-Total</strong></td>
<td><strong>9.07</strong></td>
<td><strong>99%</strong></td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>9.18</strong></td>
<td><strong>100%</strong></td>
</tr>
</tbody>
</table>

5. Use, Servicing and Replacement Parts

The Use, Servicing and Replacement Parts phase captures emissions from day-to-day car use, from the time the car leaves the showroom until it reaches the end of its life. Estimates for ‘Service and Replacement Parts’, ‘Insurance’ and ‘Road Construction’ contained in Figure 3.4 were obtained using EIOLCA. Other estimates represent direct releases during the use of the part in question. While brake pads, tires and oil all fall under ‘Service and Replacement Parts’ they have been broken down in Figure 3.4 to provide an indication of toxic releases that occur in driving environments (such as urban areas).

An allocation of road infrastructure has also been attributed to the use phase of a car. The impact of infrastructure development is often omitted from life-cycle assessments, yet it is essential to car use and thus has been accounted for in this analysis. Determining the optimal allocation method for infrastructure costs is not a trivial task. The relationship between the number of cars on the road and road infrastructure requirements is not linear, and numerous allocation methods are plausible. For instance, the proportion of road construction costs born by any given vehicle could be based on distance travelled, passengers carried, weight carried, contribution to road wear or some combined measure. We have allocated annual road construction costs equally to each U.S. registered vehicle (private and commercial) to provide a rough estimate of the contribution of road construction to vehicle use. Based on this measure, each vehicle accounts for approximately $2,600 worth of road construction over the vehicle’s lifetime.
Insurance costs are represented by the insurance carrier sector, and are estimated to be $11,239 (1997$) over a vehicle’s life, based on previous research [10]. As the insurance sector does not report to the TRI, only indirect releases are represented. We agree with previous authors that other fixed costs such as license fees and finance charges have few supplier impacts and thus have been excluded from the analysis [10].

It was not possible to conduct an EIO-LCA on gasoline use as gas stations are aggregated with many other retail outlets in BEA I-O data. Hence, the figures shown below relate only to the lead and cadmium contained in the gasoline, and releases occurring during the production, and distribution of gasoline are not represented. As a result, total emissions from gasoline use will be higher than the figures shown here. However, the difference between the values presented in Figure 3.4 and EIO-LCA estimates is expected to be small, as both the oil and gas extraction industry and petroleum refineries emit lead and cadmium compound at levels that are four orders of magnitude lower than those from the Copper, Nickel, Lead and Zinc mining sector on a ‘per dollar output’ basis.

The loss of wheel balance weights and emissions resulting from vehicle servicing (including the manufacture and installation of replacement parts) dominate the lead released during a vehicle’s use. Road construction and the dust created from brake wear also contribute significantly. ‘Service and replacement parts’, ‘Insurance’ and ‘Road construction’ are the primary contributors to cadmium emitted in this phase. As is the case in vehicle manufacturing, the vast majority of cadmium released as a result of vehicle servicing and part replacement comes from the metal mining and refining processes.

Both cadmium and lead are found in brake pads and tires (cadmium is a contaminant in the zinc oxide used in the tire rubber) [17, 26, 33]. Yet tire wear releases only a very small amount of cadmium to the environment. Cadmium and lead also appear in used oil (through corrosion and wear of alloys contained in vehicles and as a result of impurities in the zinc used to provide wear protection to the engine) [16, 17]. However negligible amounts of lead and cadmium are emitted in this manner. Similarly, the prohibition of lead additives in gasoline (for highway use) in the United States has also removed lead from gasoline [37]. The cadmium released through gasoline is an upper estimate and is based on a minimum detection limit of 0.01 parts per million in a previous study [16].

Unfortunately, releases from repair shops are not captured in the TRI and thus are not accounted for here. However, suppliers to the service and repair sector, including replacement part manufacturers are included in the service and replacement parts estimate. The ‘Service and Replacement Parts’ releases are based on a lifetime service cost of $11,544 (in 1997$), which
has been adjusted to exclude the 14% of insurance premiums that go toward collision repair, to avoid double counting [10].

Figure 3.4: Lead and Cadmium Releases as a Result of Use, Servicing and Part Replacement

<table>
<thead>
<tr>
<th></th>
<th>Cd (g)</th>
<th>Pb (g)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lead Wheel Balance Weight losses</td>
<td></td>
<td>140</td>
</tr>
<tr>
<td>Gasoline</td>
<td>0.15</td>
<td>0</td>
</tr>
<tr>
<td>Road Construction</td>
<td>0.37</td>
<td>39</td>
</tr>
<tr>
<td>Insurance</td>
<td>0.38</td>
<td>26</td>
</tr>
<tr>
<td>Service and Replacement Parts (excl. battery replacement)</td>
<td>2.43</td>
<td>127</td>
</tr>
<tr>
<td>- Brakes</td>
<td>0.02</td>
<td>34</td>
</tr>
<tr>
<td>- Tires</td>
<td>0.04</td>
<td>1</td>
</tr>
<tr>
<td>- Used Oil</td>
<td>0.00</td>
<td>0</td>
</tr>
<tr>
<td>Total</td>
<td>3.34</td>
<td>333</td>
</tr>
</tbody>
</table>

Sources: [14, 16, 17, 26, 38]

6. End-of-Life Vehicle Recovery

At the end of a vehicle's life, parts that can be re-used (e.g. electronics, lights, mirrors) and valuable materials (e.g. large castings, batteries and catalytic converters) are removed by the dismantler and sold. The car hulk is then shredded and the recoverable metals are also typically sold and recycled. The non-metallic components, called automotive shredder residue ('ASR') are not presently recycled due to high plastic recycling costs, and are sent to landfills for disposal [4, 39]. High disassembly, sorting and cleaning costs play a major role in preventing plastic recycling from being economically viable.

Lead and cadmium releases at the end-of-life of a vehicle or part result from automotive material that (a) ends up in landfill, (b) escapes to the environment during processes such as recycling, or (c) does not enter the vehicle salvage process (such as parts left to deteriorate in backyards). Of these, emissions to air and water will dissipate more quickly than releases that are well contained. Yet, waste containment methods are not permanent nor failsafe [25]. Figure 3.5 shows that all three categories contribute significantly to emissions at vehicle/part end-of-life.

Parts containing lead that do not enter the recovery stream (including car batteries that are not recycled) generate the majority of lead releases during the end-of-life phase. We use a battery collection efficiency of 98%, which is consistent with other sources, and lower than the figure of 95% derived from 1998 US Geological Survey data [30, 40]. The impact of this uncertainty can be great – the 450g lead loss from batteries that are not recovered (shown in Figure 3.5) would rise to 1125g if a collection of 95% was used.
Lead losses resulting from recycling of vehicle batteries have been previously estimated at 2% of lead recycled [10]. Figure 3.5 compares our results (using this value) to the lead releases arising from the production of a car battery using EIOLCA. The latter includes losses from secondary lead production (i.e. recycling). BEA data aggregates the production of car batteries with other storage batteries (such as computer and mobile phone batteries). Our analysis adjusts for this by assuming that lead releases from storage batteries are primarily due to the production of vehicle batteries (as other storage batteries are typically based on nickel-cadmium, nickel metal hydride or lithium ion technology) [40]. The value of 156g (in Figure 3.5) accounts for the production of two replacement batteries over the life of a car. This EIOLCA value is very sensitive to the average cost of a replacement car battery. We estimate that in 2002 an average light-duty vehicle battery cost $80, though little reliable data is available to verify this figure. A battery cost of $120 would increase the life-cycle lead emissions to 230g. This value may be more representative of a light truck battery (which tends to be larger and more costly than those used in passenger vehicles).

Roughly two thirds of scrap tires are “recovered” with nearly two thirds of these (i.e. 44% of the total number of tires) being used as fuel in a variety of industrial facilities [4]. Even when tires are recycled we can assume that a portion of the heavy metals that they contain will be released to the environment. For example, pollutants such as fly ash, which result when attempting to recover energy from end-of-life products (often through combustion) are usually disposed of in landfills, which can and often do leak [25]. This analysis captures the full metal content of the used tires. Nonetheless, total metal dissipation from end-of-life tires remains small relative to other sources (this is discussed further in the next section).

Some non-ferrous metal is covered from post-shredder material. However separation of non-ferrous metal is not 100% efficient. Studies have found that ASR contains between 2700 and 7050 mg/kg of lead [4]. A typical ELV weights roughly 1400kg (without tires) of which approximately 20-25% will become ASR [4]. Thus as much as 1200g of lead may be sent to landfill per vehicle, assuming an ASR lead content of 4000 mg/kg. Hence our figures may represent a lower estimate of ELV lead losses. We assume that lead from all parts other than batteries is released to the environment. However our estimates of the total lead contained in an ELV (roughly 8 kg) is significantly lower than that used in a previous study (which estimated that 13kg of lead was present in a 1995 model vehicle) [4]. Similarly, there is evidence to suggest that cadmium is also present in parts other than tires and brake pads, for instance in thick film pastes used in electronics [28]. However no data was available to assess the amount of cadmium contained in a vehicle as a result of such materials. As a result end-of-life cadmium emission estimates may also be higher than stated.
Figure 3.5: Lead and Cadmium Released at Part/Vehicle End-of-Life

<table>
<thead>
<tr>
<th></th>
<th>Cd (g)</th>
<th>Pb (g)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Contained in part</td>
<td>Contained in part</td>
</tr>
<tr>
<td><strong>Losses during recycling:</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Replacement car batteries</td>
<td>-</td>
<td>294</td>
</tr>
<tr>
<td><strong>Not recovered and assumed to dissipate to the environment:</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Batteries that don’t enter the recycling stream</td>
<td>-</td>
<td>450</td>
</tr>
<tr>
<td>Tires</td>
<td>0.03</td>
<td>1</td>
</tr>
<tr>
<td>Brake pads</td>
<td>0.01</td>
<td>11</td>
</tr>
<tr>
<td>Other parts</td>
<td>unknown</td>
<td>597</td>
</tr>
<tr>
<td><strong>Total Metal Released</strong></td>
<td>0.04</td>
<td>1352</td>
</tr>
</tbody>
</table>

7. Comparison of Life Cycle Stages

The majority of cadmium emissions occur during the manufacture of the original vehicle (Figure 3.6). This suggests that there are either high losses associated with the production of parts that include cadmium or that cadmium may be used and released in the manufacture of non-cadmium parts (through the mechanisms discussed above). The majority of the cadmium released during vehicle use comes from servicing and the production and installation of replacement parts. In both vehicle and part production, cadmium releases occur mainly during the metal mining and refining process.

The removal of tetra-ethyl and tetra-methyl lead from gasoline, which began in the 1970s, dramatically reduced the amount of lead released as a result of car use. Prior to gasoline lead regulations, the lead content in gasoline was as high as 0.6 g/l in Europe [2]. Based on US EPA estimates, an average 2004 model light-duty vehicle has a fuel efficiency of 11.3 litres per 100 km (20.8 miles per gallon) [38]. Over a 12-year vehicle lifetime, driving 15,000 km per year, this would result in 12.2 kg of lead being emitted to the atmosphere. Today, lead releases from vehicle use are roughly a quarter of this. Lead emissions from manufacture and end-of-life processes now contribute the majority of releases to the environment. While the use phase of a vehicle results in only 12% of total lead losses, a significant proportion of these occur in urban environments making the lead from this phase more available to the general population.
Figure 3.6: Comparison of Life Cycle Stages

<table>
<thead>
<tr>
<th></th>
<th>Cd (g)</th>
<th>Pb (g)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Original vehicle manufacture</td>
<td>9.2</td>
<td>1203</td>
</tr>
<tr>
<td>Use, service &amp; replacement parts</td>
<td>3.3</td>
<td>333</td>
</tr>
<tr>
<td>Vehicle/parts recovery and end-of-life</td>
<td>0.04</td>
<td>1352</td>
</tr>
<tr>
<td><strong>Total losses</strong></td>
<td><strong>12.6</strong></td>
<td><strong>2888</strong></td>
</tr>
<tr>
<td><strong>% Contribution:</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Original vehicle manufacture</td>
<td>73%</td>
<td>42%</td>
</tr>
<tr>
<td>Use, service &amp; replacement parts</td>
<td>27%</td>
<td>12%</td>
</tr>
<tr>
<td>Vehicle/parts recovery and end-of-life</td>
<td>0.3%</td>
<td>47%</td>
</tr>
</tbody>
</table>

8. Discussion

The results shown here should be interpreted within the context of several methodological and data based limitations. Model and data constraints associated with the EIOLCA methodology have been discussed by previous authors [10, 12, 19]. Arguably the most limiting of these is that the level of disaggregation may be insufficient in some instances, resulting in I-O sectors that are too heterogeneous to correctly reflect certain industrial processes. The allocation of environmental burdens based on market value can also be problematic, especially during periods of significant technological change.

EIOLCA also assumes that waste generated from the production of imports is the same as that created in the manufacture of U.S. equivalents, per dollar of the good produced. This may introduce errors where product costs or waste generation levels differ significantly from those in the U.S. For example, in the case of products and materials produced in developing countries, prices are likely to be lower and emissions higher than for similar U.S. made products. In such instances, EIOLCA would understate emissions.

Production efficiencies and toxic emissions assumed in EIOLCAs may also differ across North America. A recent report found significant differences in the relative contribution to on-site lead air emissions between US and Canadian facilities [9]. However the overall lead emissions (on land, air, surface water and underground injection) are roughly proportional to the number of firms in each country, suggesting that the EIOLCA assumption of equivalent emissions may be reasonable with respect to Canadian imports. Uncertainties are also associated with the use of TRI data, though these have decreased since the reporting thresholds for lead and its compounds were reduced to 100 pounds in 2001 [32, 41].

Lastly, uncertainty is also associated with converting 2002 TRI data (provided in the SIC format) into a format consistent with BEA Input-Output data (based on the NAICS format). Such
attribution errors may be especially prevalent when assessing releases of individual chemicals, as the allocation of heterogeneous NAICS sectors to SIC sectors may not accurately reflect the nature of the production processes being represented. However, such issues will disappear as TRI reporting moves to the NAICS format over the coming years.

This paper should be viewed in light of both the significant uncertainty introduced by the above limitations and the context of the research. The assumptions and estimates used to represent an ‘average light-duty vehicle manufactured in North America’ may need to be modified when applying this analysis to specific vehicle types or alternate geographical regions. In general, larger vehicles generate greater environmental burdens. Yet, the size and mass of a vehicle are not always proportional to its cost (e.g. sports cars may be small but expensive). Thus care must be taken to avoid inaccurate generalisations of EIOlCA results, as the methodology scales environmental burden based on the cost of the vehicle. In addition, a vehicle’s heavy metal content may vary across different car makes and models. The results of this analysis may also be less pertinent to regions outside of North America. The data and many of the assumptions used in this analysis (e.g. vehicle lifetime, distance travelled per year, vehicle cost, infrastructure requirements, fuel efficiency, recycling rates etc.) are specific to North America, and may not be applicable to other continents. The road conditions, weather patterns, product prices, production processes, legislative requirements and material composition of products may all differ across the globe.

9. Conclusion

Most lead and cadmium releases occur during vehicle/part manufacture and during end-of-life. Thus the quantity of life-cycle lead and cadmium releases is not highly dependent upon driver behaviour. This should be advantageous for policy makers, as changing driver behaviour is difficult and largely unnecessary in this case. Legislation aimed at reducing emissions from car production and end-of-life would attack the dominant drivers of lead and cadmium releases.

Metal mining and processing is the main contributor to releases that result from vehicle and replacement part production. Thus efforts to mitigate emissions from automotive manufacturing should address the importance of minimising releases that occur during ore extraction and metal refining. A high priority should also be placed on implementing substitutes for lead wheel weights and removing lead in brake pads. The loss of lead wheel balance weights constitutes 5% of the total lead released as a result of car use. Brake pad dust results in an additional 1% of the total lead load. The impact on society of these releases may be significant as they are likely to occur in urban environments.
Lead is rarely recovered from end-of-life components that contain lead in small concentrations. Yet in aggregate, such parts represent a considerable amount of lead. Removing the lead from these components would make a strong contribution to the reduction of total end-of-life lead releases. Lead-acid battery production and end-of-life processes continue to play a major role in lead emissions. However, such batteries are likely to remain in vehicles for the foreseeable future. Decreasing lead releases from car batteries will require continued effort to maximise collection rates and minimise losses during lead recycling. Current material trends may also reduce lead and cadmium emissions. The movement away from steel toward greater plastic and aluminium use in vehicles (to minimise vehicle weight and emissions) may have the synergistic effect of decreasing the amount of lead and cadmium released during the mining and refining of 'traditional' metals.

This analysis provides guidance for waste minimisation efforts by indicating the nature and quantity of lead and cadmium released as a result of light-duty vehicle use. It presents an integrated view of direct and indirect impacts that has previously been obscured by an absence of systems-level analysis in this area. Reducing lead and cadmium emissions may rely on designing lead out of automotive components for which it is not essential, reducing the discharges from current mining processes and increasing the collection and recycling efficiency of car batteries. Using non-lead based substitutes for lead wheel-balance weights will also significantly reduce the releases during vehicle use. Further research into emission location, medium, molecular form and transport mechanisms would facilitate an understanding of the impact of current releases on the environment and sub-segments of the population.

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CONCLUDING CHAPTER

The automotive industry is coming under increasing pressure to improve its environmental performance. Legislation such as the ELV Directive is changing the way that car manufacturers design vehicles and recover them at their end-of-life. Yet, the efficacy of environmental management efforts depends on a concrete understanding of the emissions released during car manufacture, use and end-of-life. The economic and regulatory environment influences each of these stages. Understanding how such forces affect the life cycle of a vehicle is essential to the pursuit of environmentally, socially and economically sustainable car production.

By providing an emerging picture of the effects of the ELV Directive, government and industry will able to design improved legislation going forward. Such information may also help inform the re-examination of the ELV Directive's 2015 ELV recoverability and recyclability targets. This process is planned to occur by the end of 2005 (see figure 2.3).

Such information is also of use outside of the European automotive industry. An analysis of the economic and environmental changes brought about by the ELV Directive should be useful when developing product take-back legislation in other countries and for other durable goods. As discussed above, white-goods such as refrigerators and washing machines are already recovered in the same facilities and with the same processes as car hulks. Thus industry dynamics and technological considerations are likely to be similar across such durable goods. If existing take-back legislation is a harbinger of the legislative regime to come, obtaining timely feedback from current initiatives will be essential to designing better legislation in the future.

The ability to learn from experience would be enhanced by combining the research in Chapter 2 with similar studies of other existing take-back regulations (e.g. the EU WEEE Directive, and comparable automotive and electrical equipment legislation in Japan). This would allow the comparison of differing legislative strategies and may facilitate a refined understanding of the economic drivers of 'green' innovation. Further insight may also be gained by contrasting the experiences of individual European member states in implementing both WEEE and ELV regulation. Each country has a unique economic environment and the manner in which the ELV Directive has been transposed into national legislation also varies slightly. Such disparities may result in discernible differences in recovery outcomes. In aggregate, the above research would enable a clearer understanding of the impact of economic aspects (e.g. landfill costs, recovery costs, industry structure etc.) and legislative differences (e.g. allocation of recovery costs between industry, government and the consumer) on product recovery outcomes.
The ongoing globalisation of the automotive industry produces additional opportunities for research. Discerning the interrelationship between economic, legislative and social concerns and the nature of international car manufacturing may yield further insight into the future of car production. For instance, differing costs of production and varied waste management expectations across countries may influence the level of ‘green’ innovation as well as the structure and environmental performance of the industry.

Within Europe, take-back legislation is driving a reduction in the amount of heavy metals in new vehicles (as discussed in Chapter 2). Yet, there is a need to focus on heavy metals released during the full vehicle life cycle, and not just on the metal contained in the vehicle itself. Chapter 3 helps to fill this existing gap in knowledge with respect to lead and cadmium emissions. Importantly, the analysis includes the releases that result from infrastructure use (e.g. road construction emissions). These ancillary requirements for car use have rarely been included in previous research (see Chapter 3), but nonetheless comprise an essential ingredient in the use phase of a vehicle’s life cycle. Including the environmental burden imposed by infrastructure into the analysis enabled a more complete view of the quantity and nature of lead and cadmium releases at each life-cycle stage.

The results show that while manufacturing and end-of-life recovery processes are important, economy-wide ramifications of automotive production (e.g. discharges from metal mining and refining) contribute substantially to overall lead and cadmium releases. These emissions do not necessarily result directly from the inclusion of lead and cadmium in the vehicle. Thus, by itself, legislation that aims to remove heavy metals from vehicles, such as the ELV Directive, will not eliminate all the lead and cadmium emitted due to automobile use. Furthermore, the analysis demonstrates that the volume of lead and cadmium in the vehicle is not the only determinant of overall emission levels. The wear characteristics and end-of-life processes of automotive components are also important. For example, the elimination of lead wheel weights should reduce life-cycle lead releases to a greater extent than decreasing the mass of battery-lead by the same amount. These results reinforce the importance of taking a systems approach to analysing emissions from car use. Regulators are well placed to integrate these results, and life-cycle approaches in general, into the design of policies aimed at hazardous substances minimisation.

While this thesis adds to the body of knowledge on the life cycle lead and cadmium releases due to automobile use, it also raises a number of questions that may be resolved by further research. Firstly, the EIOlCA methodology could be refined by an examination of the impact of metal prices on EIOlCA results. As discussed in Chapter 3, the substantial differences in unit prices between metals such as copper, nickel and platinum may give rise to a disproportionately high allocation of lead emissions being attributed to the automotive use of metals other than lead. Secondly, many mining and processing facilities extract multiple metals simultaneously. Detailed studies of these
operations would enable a more accurate attribution of releases emanating from the production of specific metals. Finally, the accuracy of EIOLCA results could also be improved by refining assumptions surrounding imports and exports (as discussed in Chapter 3).

A number of opportunities exist to build on the results presented in Chapter 3. Now that baseline data on current lead and cadmium emissions generated by car use has been established, it is possible to use life cycle assessment to determine the impact on lead and cadmium dissipation of the material trends outlined in Chapter 2 (e.g. increased aluminium and plastic use). Similarly, one could evaluate the life-cycle impact of the ELV Directive itself. However, doing so would require that changes induced by the Directive be distinguishable from trends that may occur regardless of legislative impacts.

Life cycle impacts of substitute parts could also be analysed and compared to existing practices. In particular, the introduction of hybrid electric cars may present an interesting case study. Such vehicles require larger batteries than ordinary cars and may result in increased heavy metal discharges during both manufacturing and end-of-life. Evaluating the life cycle impact of hybrid vehicles will be a necessary step in anticipating the environmental implications of greater levels of hybrid-vehicle adoption.

Both manuscripts included in this thesis highlight the need to develop environmental management strategies that take account of each industry's unique characteristics. Anticipating the environmental implications of changing patterns of consumption and production requires an appreciation of how economics, product usage patterns, legislative requirements and the nature of innovation interact. This thesis contributes to this field of study by using a multi-disciplinary approach to evaluate the environmental consequences of current practice and recent developments in the automotive industry. The methods used and the knowledge gained may also prove useful in assessing the ecological impact of other durable goods.